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DOE/EIS-0117D

DRAFT
Environmental Impact Statement

Remedial Action at the Weldon Spring Site

February 1987

U. S. Department of Energy

Assistant Secretary for Nuclear Energy

Office of Remedial Action and Waste Technology

DOE/EIS-0117D
Dist. Category UC-70A

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(NOTE: This Draft EIS should be saved because portions that are not significantly revised will not be reprinted in the Final EIS.)

Printed in the United States of America
Available from
National Technical Information Service
U. S. Department of Commerce
5285 Port Royal Road
Springfield, VA 22161

NTIS price codes
Printed copy: A12
Microfiche copy:A01

COVER SHEET
DRAFT ENVIRONMENTAL IMPACT STATEMENT
REMEDIAL ACTION AT THE WELDON SPRING SITE

a) Lead Agency: U.S. Department of Energy (DOE)

b) Proposed Action: Long-term management of wastes associated with remedial action activities at the Weldon Spring site located near Weldon Spring, Missouri.

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d) Designation: Draft EIS (DEIS)

e) Abstract: Several alternatives are considered for long-term management of wastes associated with remedial action activities at the Weldon Spring site, located about 48 km (30 mi) west of St. Louis, Missouri. The site is currently contaminated as the result of processing of uranium, thorium, and other materials previously carried out at the site. The Weldon Spring site consists of four areas: raffinate pits, quarry, chemical plant, and vicinity properties. The alternatives considered are (1) long-term management in the existing raffinate pits with improved containment, (2) long-term management in the raffinate pits area in a new disposal cell (DOE's preferred alternative), (3) long-term management at another site, and (4) no action. Several subalternatives are considered for Alternatives 2 and 3 to more fully explore potential environmental impacts. The potential geological, hydrological, radiological, chemical, ecological, air quality, land-use, and socioeconomic impacts associated with each alternative are assessed and compared for the remedial action period and for long-term management. These analyses indicate that all action alternatives (in particular Alternative 2 -- DOE's preferred alternative) can be implemented without significant environmental consequences provided that appropriate mitigative measures are taken.

f) After consideration of public comments on the Draft EIS, a Final EIS will be prepared. A Record of Decision will be published in the Federal Register no sooner than 30 days after issuance of the Notice of Availability for the Final EIS.

The U.S. Environmental Protection Agency (EPA) will be responsible for preparing and issuing the Record of Decision required by the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). EPA and DOE will coordinate activities associated with preparation of these two Records of Decision to provide appropriate consistency and to minimize duplication of effort. If EPA deems it appropriate, the DOE Record of Decision for the National Environmental Policy Act will also suffice as the EPA Record of Decision for CERCLA.

FOREWORD

This Environmental Impact Statement (EIS) is issued by the U.S. Department of Energy (DOE). It assesses the environmental impacts of various alternatives for the long-term management of wastes associated with remedial action activities at the Weldon Spring site, located about 48 km (30 mi) west of St. Louis, Missouri. The site is currently contaminated as the result of processing of uranium, thorium, and other materials previously carried out at the site. The Weldon Spring site consists of four areas: raffinate pits, quarry, chemical plant, and vicinity properties. An estimated 600,000 m^3 (780,000 yd^3) of contaminated materials are currently located at these four areas.

DOE must decide how to manage the Weldon Spring wastes for the long term and has prepared this EIS to provide environmental input to this decision. The EIS has been prepared in accordance with the National Environmental Policy Act of 1969 (NEPA), as implemented by regulations promulgated by the Council on Environmental Quality (CEQ) (40 CFR Parts 1500-1508, November 29, 1978) and by DOE's implementing guidelines (45 FR 20694, March 28, 1980). A Notice of Intent to prepare this EIS was issued March 2, 1984, and a public scoping process was conducted. The public will have an opportunity to comment on this Draft EIS, both in writing and at public meetings. After considering all comments, DOE will issue a Final EIS. DOE will then issue a Record of Decision not less than 30 days following publication of the Notice of Availability of the Final EIS.

The U.S. Environmental Protection Agency (EPA) will be responsible for preparing and issuing the Record of Decision required by the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). EPA and DOE will coordinate activities associated with preparation of these two Records of Decision to provide appropriate consistency and to minimize duplication of effort. If EPA deems it appropriate, the DOE Record of Decision for NEPA will also suffice as the EPA Record of Decision for CERCLA.

The format of this Draft EIS follows the format suggested in the CEQ regulations. Chapter 1 documents the purpose and need for a decision. Chapter 2 summarizes and compares alternatives and environmental impacts. Chapter 3 summarizes the existing environment at the Weldon Spring site and alternative sites. Chapter 4 provides detailed information on analyses of the environmental consequences of the various alternatives. Chapter 5 presents the names and professional qualifications of the persons responsible for preparing the statement. More detailed information and analyses are provided in appendices.

SUMMARY

The Weldon Spring site is located near Weldon Spring, Missouri, about 48 km (30 mi) west of St. Louis. It is surrounded by large tracts of land owned by the federal government and the state of Missouri. The site was used by the U.S. Department of the Army from 1941 to 1944 for the production of explosives. In 1957, an 89-ha (220-acre) portion of the site was released to the U.S. Atomic Energy Commission, which used the site for processing of uranium and thorium concentrates from 1957 to 1966. The site is comprised of four areas: raffinate pits, chemical plant, quarry, and vicinity properties. The raffinate pits and chemical plant are on adjoining land about 3.2 km (2 mi) southwest of the junction of Missouri Route 94 and U.S. Route 40/61. The raffinate pits and chemical plant areas are accessed from Route 94. The quarry is located in a comparatively remote area about 6.4 km (4 mi) south-southwest of the raffinate pits area; the quarry is also accessed from State Route 94. The contaminated vicinity properties are located within a few kilometers of these three areas.

Most of the contaminated materials are located at the raffinate pits and quarry. St. Charles County operates a well field near the quarry, and there has been public concern that contaminants leaching from the quarry wastes might be transported in the groundwater toward the well field. Contamination on the vicinity properties is located mainly along ditches, roads, and railroads. The raffinate pits, chemical plant, and quarry areas are fenced, and the U.S. Department of Energy (DOE) owns and restricts access to these areas. Some of the vicinity properties are on land that is open to the public and used for recreational purposes.

It is estimated that a total of 600,000 m^3 (780,000 yd^3) of contaminated materials are currently located at the Weldon Spring site. The raffinate sludge (processing residue) and the quarry sludge, which comprise about one-third of the total volume of contaminated materials, contain most of the radioactive contaminants. About two-thirds of the total volume of contaminated materials consists of soil and rubble. These materials are contaminated with naturally occurring radionuclides of the uranium-238 and thorium-232 decay series. The wastes also have chemical contaminants.

DOE prepared this Environmental Impact Statement (EIS) to support the decisions it must make regarding remedial action activities associated with the Weldon Spring site, including long-term management of the contaminated materials in compliance with appropriate DOE Orders and other federal regulations. A scoping process was conducted to determine the alternatives to be analyzed, the significant issues to be analyzed in depth, and the issues to be eliminated from further detailed study. Based on public and technical scoping input, DOE decided to take the "tiered" approach recommended by the Council on

Environmental Quality under its regulations for implementing the National Environmental Policy Act (NEPA). This EIS is intended to support the major decisions on cleanup and long-term management of the contaminated materials, including long-term management of the chemical plant wastes. However, many of the specific issues associated with decontamination and decommissioning of the chemical plant are not yet ready for a decision. Therefore, a separate NEPA document, tiered to this EIS, will be prepared at a later date to support decontamination and decommissioning decisions for the chemical plant.

The following alternatives were defined for analysis in this EIS:

Alternative 1: Long-Term Management in the Existing Raffinate Pits with Improved Containment. Under Alternative 1, all the wastes from the chemical plant, quarry, and vicinity properties as well as the raffinate sludge will be contained in a disposal cell in the existing raffinate pits on the Weldon Spring site. A multilayered cover -- consisting of clay, riprap (rock), sand/gravel, and topsoil -- will be constructed over the wastes.

Alternative 2: Long-Term Management in the Raffinate Pits Area in a New Disposal Cell (DOE's Preferred Alternative). DOE is considering two subalternatives for Alternative 2.

Alternative 2a: Partially Above Grade. Under Alternative 2a, a new disposal cell, partially above grade, will be constructed on the Weldon Spring site. The new cell will contain the same wastes as Alternative 1 plus contaminated soils from beneath the raffinate pits area. The new cell will have a multilayered cover identical to Alternative 1, but it will also have a leachate monitoring system to monitor the performance of the cover.

Alternative 2b: Completely Above Grade. Under Alternative 2b, a new disposal cell, completely above grade, will be constructed on the Weldon Spring site. It will contain all the wastes and will have a multilayered cover and a leachate monitoring system. It will also have a layer of lead in the cover.

Alternative 3: Long-Term Management at Another Site. Under Alternative 3, DOE will transport all or part of the wastes to other sites. Three subalternatives are considered.

Alternative 3a: Hanford Site. Under Alternative 3a, all the Weldon Spring wastes will be transported to the DOE Hanford site near Richland, Washington, for disposal in trenches near an existing disposal area. The wastes will be covered with native soils (excavated from the trenches), riprap, and another layer of soil.

Alternative 3b: "Nearby Site". Under Alternative 3b, the Weldon Spring wastes will be transported to a "Nearby Site" for disposal in a containment cell similar to Alternative 2a. The "Nearby Site", which will be owned and operated by DOE, is assumed to be located in Missouri within 160 km (100 mi) of the Weldon Spring site and would be chosen to have more favorable conditions (e.g., thicker clay, lower hydraulic conductivity, deeper groundwater table, and/or higher sorption capacity) than the Weldon Spring site.

Alternative 3c: Uranium Processing Site. Under Alternative 3c, only the raffinate and quarry sludge will be transported to an existing uranium processing facility in the Four Corners area of the southwestern United States for reprocessing to extract the uranium in the sludge. The wastes remaining at the Weldon Spring site will be placed in a disposal cell, similar to Alternative 1, in the existing raffinate pits.

Alternative 4: No Action. Under Alternative 4, the raffinate sludge will continue to be stored in the pits, the quarry wastes will be left in the quarry, and the chemical plant area and vicinity properties will also be left in their current conditions.

On-site disposal of the Weldon Spring wastes is considered by DOE to be the environmentally preferable means for long-term management of these wastes. Alternative 2 is preferred over Alternative 1 because Alternative 2 involves construction of a new disposal cell rather than use of the existing pits. It would likely be more difficult to modify the existing pits in a manner that will ensure adequate confinement. The design requirements for the new disposal cell will be determined by DOE and will incorporate appropriate features to ensure adequate confinement.

DOE will limit access to the disposal area for all action alternatives; the raffinate pits, chemical plant, and quarry areas will also have limited access under the no-action alternative. For all alternatives, including no action, DOE will maintain containment structures, monitor any releases of contaminants to the environment, and take any necessary corrective actions. Cleanup of the chemical plant, quarry, and vicinity properties is common to

all action alternatives. For the two alternatives in which all the wastes will be transported off-site (Alternatives 3a and 3b), the raffinate pits area will also be decontaminated. After these areas have been decontaminated, they will be surveyed to ensure that residual contamination levels do not exceed DOE guidelines and will then be released for appropriate use. The DOE guidelines are based on limiting the dose to a hypothetical individual under a worst-case plausible scenario to 100 mrem/yr committed effective dose equivalent above background under potential future land-use conditions. DOE policy requires that all exposures to radiation be limited to levels that are as low as reasonably achievable (ALARA). This policy requires that residual radioactivity be reduced to levels that are as far below those that would result in a 100-mrem/yr dose as is reasonably achievable considering technical, economic, and social factors.

Implementation of any of the alternatives would permanently commit some land to waste management. The smallest land commitment would be 11 ha (28 acres) for Alternative 3c at Weldon Spring, neglecting the land commitment at the uranium processing site for storage of the mill tailings; the highest land commitment would be 120 ha (300 acres) for Alternative 3a at the Hanford site. Implementation of any of the action alternatives would lead to increased risk of injury and death associated with transportation of wastes and fill materials, ranging from 0.096 deaths and 1.6 injuries for Alternatives 1 and 2a to 2.5 deaths and 34 injuries for Alternative 3a (transport of all wastes to the Hanford site).

Radiological impacts (health effects -- primarily increased risk of death from cancer) would be insignificant for all alternatives. During the action period, radiation doses to the general public from implementing the action alternatives would range from 31 to 250 person-rem (0.0053 to 0.043 health effects). Workers would incur doses of 110 to 230 person-rem (0.019 to 0.039 health effects). During the long term, estimated cumulative radiological impacts over 1,000 years range from 130 to 520 person-rem (0.022 to 0.088 health effects) for the action alternatives and 11,000 person-rem (1.9 health effects) for the no-action alternative. For comparison, over 1,000 years the exposed population near the Weldon Spring site would receive 230,000,000 person-rem from background radiation and 4,200,000 cancer deaths would normally be expected from other causes.

If action is taken and the wastes are removed from the quarry, the maximum concentration contribution of uranium at the county well field is estimated to be 0.033 pCi/L in 800 years. This estimated concentration contribution is very small in comparison to the reported background uranium concentration of <1.5 to <3.6 pCi/L. If no action is taken and the wastes are

not removed, the maximum concentration contribution is estimated to be 0.2 pCi/L in 1,700 years.

For all action alternatives, radionuclides are not expected to reach the groundwater table under the disposal areas within 1,000 years. Several chemical species are expected to reach the groundwater table under the disposal areas within 1,000 years. However, maximum concentration contributions under the disposal areas and at the site boundaries are expected to be below regulatory limits.

CONTENTS

FIGURES

TABLES

1. PURPOSE AND NEED FOR DECISION

1.1 INTRODUCTION AND BRIEF HISTORY

As part of its Surplus Facilities Management Program (SFMP), the U.S. Department of Energy (DOE) has prepared this Environmental Impact Statement (EIS) to assess the environmental impacts of alternatives for the long-term management of contaminated materials associated with remedial action activities at the Weldon Spring site. The Weldon Spring site is located about 48 km (30 mi) west of St. Louis and 23 km (14 mi) southwest of St. Charles, Missouri (Figure 1.1). The site is comprised of four areas: raffinate pits, chemical plant, quarry, and vicinity properties (see Figures 1.2 through 1.4). Most of the contaminated materials are located at the raffinate pits and quarry. The vicinity properties are areas in the vicinity of the raffinate pits, chemical plant, and quarry areas -- but outside of the current boundaries -- that are radioactively contaminated above current criteria for unrestricted use. Contamination in the vicinity properties is located mainly along ditches, roads, and railroads.

From 1941 to 1944, the U.S. Department of the Army operated the Weldon Spring Ordnance Works for production of trinitrotoluene (TNT) and dinitrotoluene (DNT). The Army used the quarry for disposal of rubble contaminated with TNT. In the mid-1950s, 89 ha (220 acres) of the ordnance works property was transferred to the U.S. Atomic Energy Commission (AEC); this area is now the main site consisting of the raffinate pits and chemical plant. From 1957 to 1966, the AEC operated a uranium processing facility at the site. Impure ore concentrates and some scrap metal were processed at the plant, and products that included pure uranium metal were then shipped to other sites. Some of the concentrates were only sampled and then sent to other sites. Thorium-containing materials were processed on an intermittent basis. Radioactive sludge residues (raffinates) resulting from the processing were placed in four on-site pits (the raffinate pits). Other radioactive wastes were disposed of in the quarry by the AEC.

After closure by the AEC, the chemical plant was reacquired by the Army in 1967. The Army partially decontaminated the buildings, dismantled some of the equipment, and began converting the facilities to produce herbicides. In 1969, prior to becoming operational, the herbicide project was canceled.

In 1971, the Army returned the 21-ha (52-acre) portion of the site containing the raffinate pits to the AEC. As successor to the AEC, DOE took over responsibility for the raffinate pits. In 1984, the Army repaired

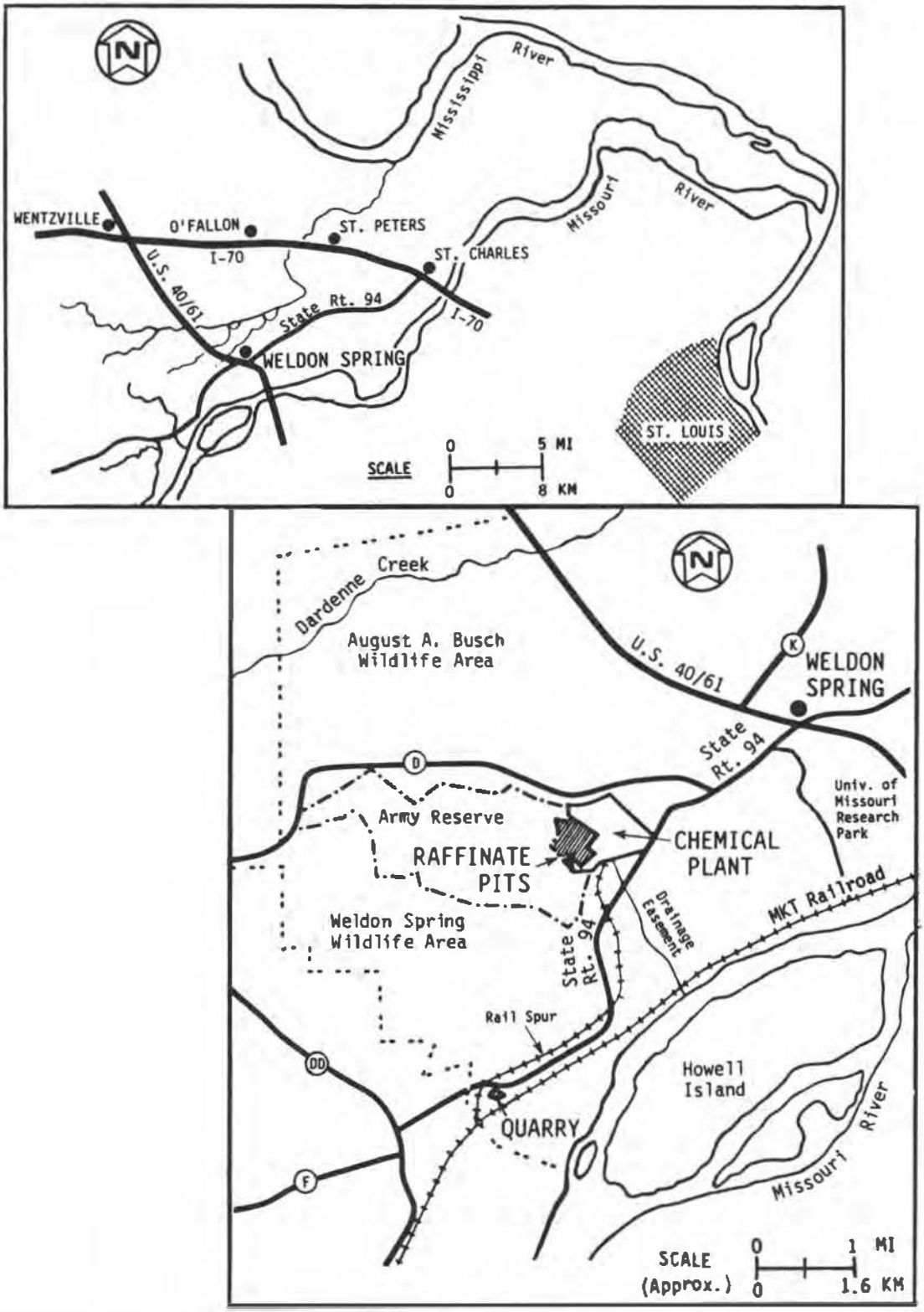

Figure 1.1. Area and Vicinity Map of the Weldon Spring Site, Weldon Spring, Missouri. Source: Modified from Bechtel National (1985c).

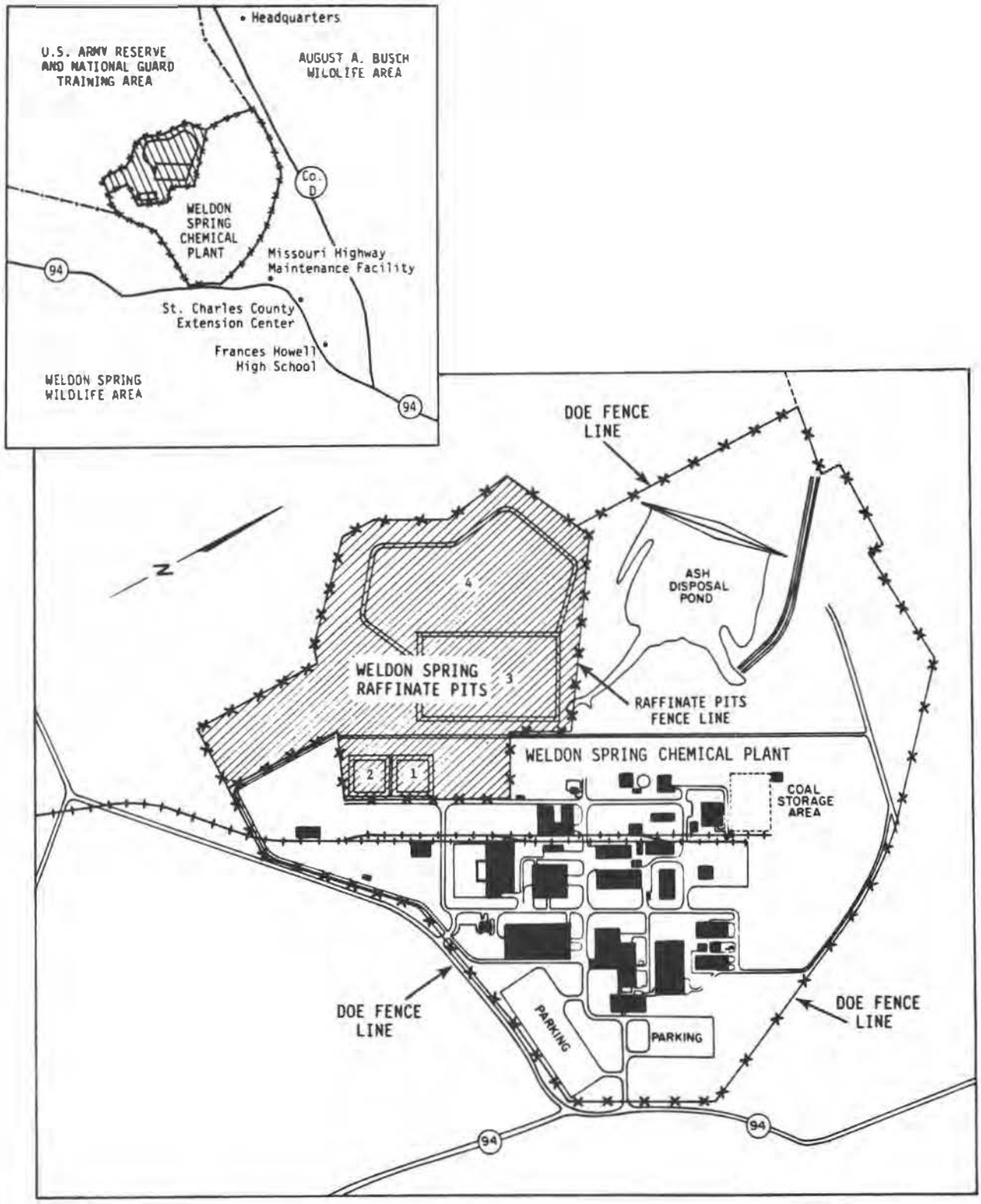

Figure 1.2. Location and Layout of the Weldon Spring Raffinate Pits and Chemical Plant. Source: Modified from National Lead Company of Ohio (1977).

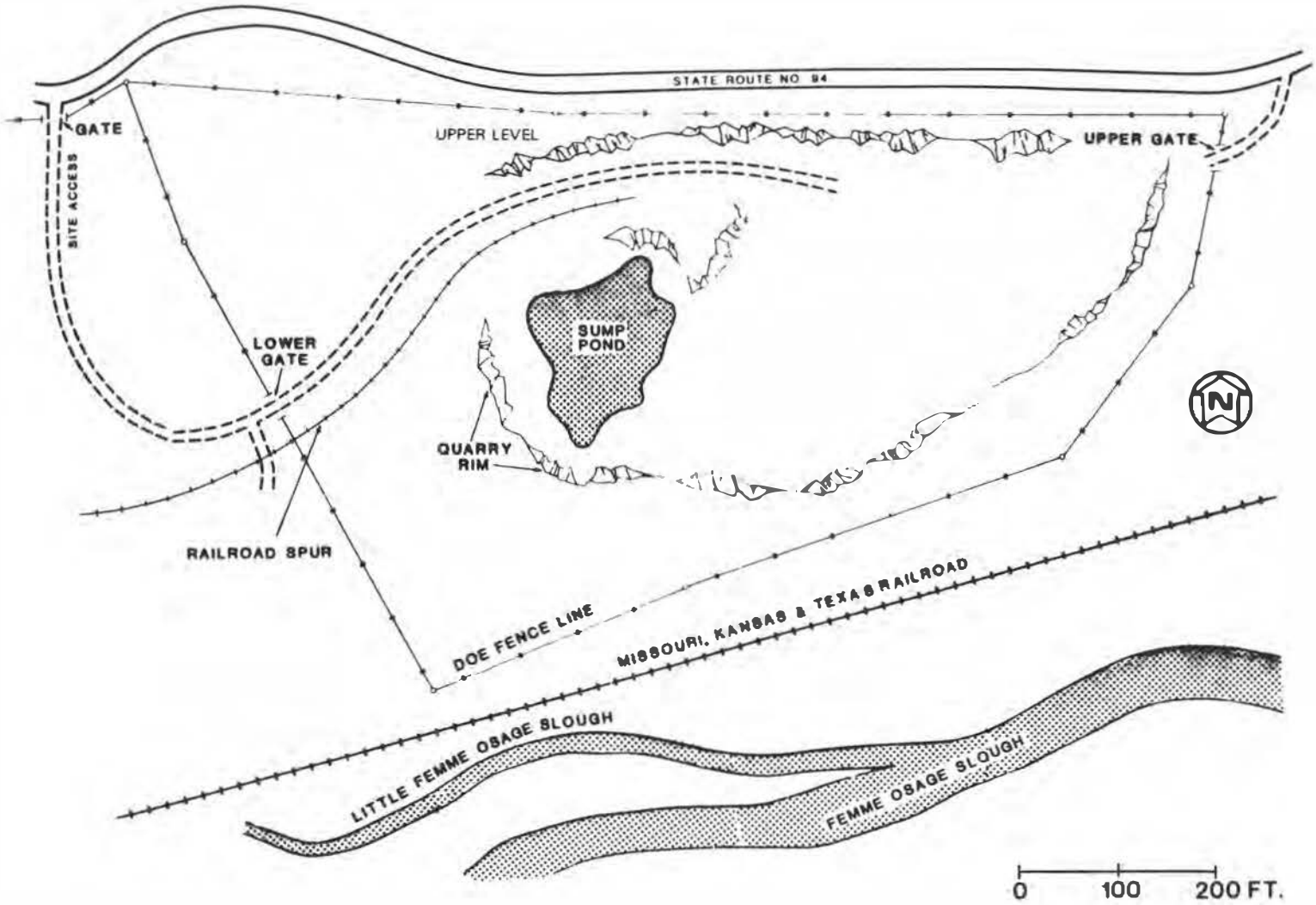

Figure 1.3. Plan View of the Weldon Spring Quarry. Conversion Factor: To convert feet (ft) to meters (m), multiply by 0.3048. Source: Modified from Bechtel National (1985b).

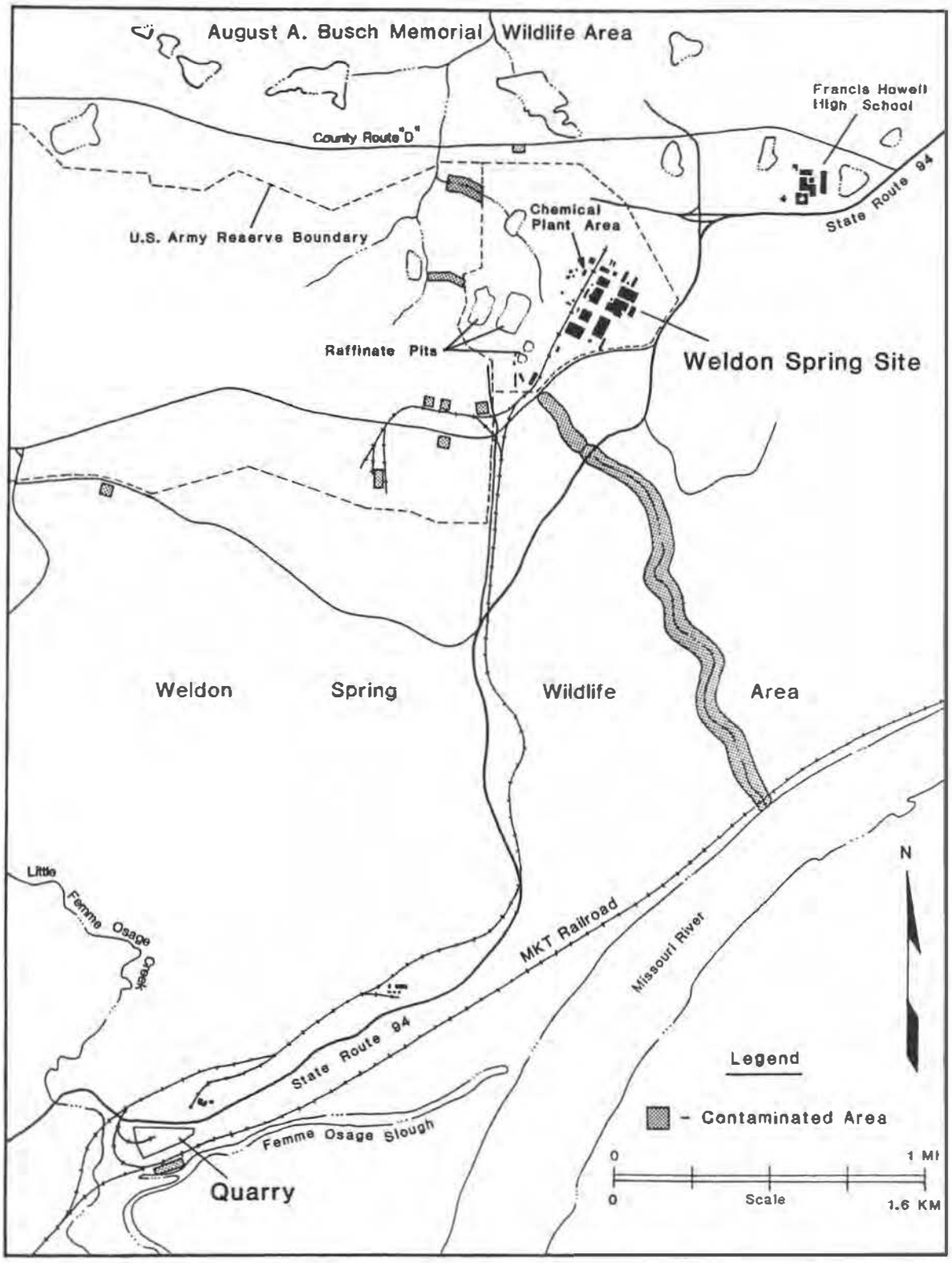

Figure 1.4. Location of Contaminated Vicinity Properties in the Area of the Raffinate Pits, Chemical Plant, and Quarry.

several of the buildings at the chemical plant; decontaminated some of the floors, walls, and ceilings; and isolated some contaminated equipment. In October 1985, custody of the chemical plant area was transferred to DOE. The main site and quarry are fenced and closed to the public. The vicinity properties occur on land owned by the Army and on land that is open to the public (Figure 1.4). Access to the land owned by the Army is restricted.

1.2 SUMMARY DESCRIPTION OF MATERIALS AT THE WELDON SPRING SITE

It is estimated that a total of 600,000 m^3 (780,000 yd^3) of contaminated materials are currently located at the Weldon Spring site (Table 1.1). The raffinate sludge (processing residue) and the quarry sludge, which comprise about one-third of the total volume of contaminated materials, contain most of the radioactive contaminants. About two-thirds of the total volume of contaminated materials consists of soil and rubble. These materials are contaminated with naturally occurring radionuclides of the uranium-238 and thorium-232 decay series. The wastes also contain chemical contaminants. The radiological characteristics of these materials are summarized in Table 1.2 and Appendix H; the chemical characteristics are discussed in Appendix H.

The raffinate pits have a total capacity of 500,000 m^3 (650,000 yd^3) and contain about 170,000 m^3 (220,00 yd^3) of raffinate sludge (Bechtel Natl. 1984, 1985a). The estimated volume of contaminated soils beneath the pits is 98,000 m^3 (130,000 yd^3) (Bechtel Natl. 1984, 1985a). About 216,000 m^3

Table 1.1. Estimated Volumes of Existing Contaminated Materials at the Weldon Spring Site

	Volume[a]	
	m^3	yd^3
Raffinate pits		
Raffinate sludge	170,000	220,000
Soils beneath pits	98,000	130,000
Quarry	73,000	95,000
Chemical plant	240,000	310,000
Vicinity properties	21,000	27,000
Total	600,000	780,000

[a] Rounded to two significant figures.

Table 1.2. Summary of Radiological Concentrations and Inventories of the Weldon Spring Wastes

	Average Concentrations (pCi/g) and Inventories (Ci)[a]			
	Raffinate Sludge[b]		Quarry	
Species	pCi/g	Ci	pCi/g	Ci
Uranium-238[c]	150	30	170	30
Thorium-232	32	6	16	3
Thorium-230	3,500	700	540	90
Radium-226	97	20	63	10

	Average Concentrations (pCi/g) and Inventories (Ci)[a]			
	Chemical Plant		Vicinity Properties	
Species	pCi/g	Ci	pCi/g	Ci
Uranium-238[c]	20	7	120	4
Thorium-232	3	1	6.4	0.2
Thorium-230	6	3	780[d]	30[e]
Radium-226	3	1	14	0.5

[a] Inventory values for all wastes and average concentrations for the chemical plant wastes are estimated to one significant figure.

[b] Concentrations are given in terms of the wet sludge. Concentrations of radionuclides in the soils beneath the pits have not been measured. For purposes of analysis in this EIS, it was assumed that the inventory of radionuclides in these soils is 1% of the inventory in the raffinates.

[c] The amounts of uranium-238, uranium-235, and uranium-234 are assumed to be present in their natural activity ratio, 238:235:234 = 1:0.046:1.

[d] This concentration was estimated by dividing the inventory of 30 Ci by the mass of the contaminated materials in the vicinity properties.

[e] Inventory in the main drainage ditch from the chemical plant to the Missouri River.

(57,000,000 gal) of contaminated water in the raffinate pits will require treatment and disposal. The pits were constructed by excavating into the existing clay soils and using the soils for construction of dikes around the pits. The residues in Pits 1, 2, and 3 consist mainly of neutralized raffinates from uranium-refining operations and washed slag residue from uranium metal production. Pit 4 contains the same types of residues as well as raffinate solids from the processing of thorium; it also contains drums and rubble resulting from the Army's partial decontamination of the chemical plant. The majority of the radionuclide inventory at the Weldon Spring site is located at the raffinate pits area. Radionuclides in both the uranium-238 and thorium-232 decay series are present, with thorium-230 being the principal radionuclide. The raffinate sludge has high concentrations (above 5,000 ppm wet weight) of fluorides and nitrates; and arsenic, chromium, copper, and lead are present at concentrations of 10 to 25 ppm (wet weight).

The quarry is located in limestone and covers about 3.6 ha (9 acres). The deepest part is filled with water and covers about 0.2 ha (0.5 acres). It is estimated that the total volume of contaminated materials that will result from cleanup of the quarry is 73,000 m^3 (95,000 yd^3) (Bechtel Natl. 1985b). The total volume of contaminated water in the quarry that will require treatment and disposal is estimated to be 11,000 m^3 (3,000,000 gal). Concentrations of most radiological species are lower in the quarry wastes than in the raffinate sludge (Table 1.2).

The estimated total volume of radioactive materials that will result from decontamination and demolition of the chemical plant is 240,000 m^3 (310,000 yd^3), consisting of contaminated rubble, soil materials, and process equipment (Rockwell Int. 1979). The radiological contamination at the chemical plant (Table 1.2) is estimated to account for only a few percent of the total inventory at the Weldon Spring site.

Radiological surveys have indicated that there are contaminated areas along ditches, roads, and railroads (Boerner 1986; Deming 1986). It is estimated that cleanup of these vicinity properties will result in 21,000 m^3 (27,000 yd^3) of contaminated materials.

1.3 DECISION TO BE MADE

As specified in the regulations of the Council on Environmental Quality (CEQ) for implementation of the National Environmental Policy Act (NEPA), this EIS is being prepared early in DOE's decision-making process for the Weldon Spring site. In addition to engineering, cost, and other considerations, environmental impacts are being considered. The planned sequence of decisions

is shown in Figure 1.5. DOE is taking the CEQ "tiered" approach, and this EIS is the document on which subsequent assessments will be tiered.* The set of tiered documents will constitute the complete environmental impact assessment for all actions at the Weldon Spring site.

This EIS is intended to support the major decisions on cleanup and long-term management of the contaminated materials from the four areas at the Weldon Spring site: raffinate pits, chemical plant, quarry, and vicinity properties. However, many of the specific issues associated with decontamination and decommissioning (D&D) of the chemical plant are not yet ready for a decision. Therefore a separate NEPA document, tiered to this EIS, will be prepared at a later date to support D&D decisions for the chemical plant.

Decontamination of the quarry is common to all action alternatives. After removal of the wastes from the quarry to a disposal cell, DOE will evaluate the need for any further actions at the quarry such as groundwater restoration. DOE also intends to evaluate the need for groundwater restoration in the raffinate pits area following implementation of one of the action alternatives. If groundwater restoration is not needed at either area, a decision will be made regarding whether the quarry or portions of the raffinate pits area can be released for unrestricted use or whether appropriate restrictions must be imposed for a period of time. If groundwater restoration is needed at either or both areas, DOE will perform a groundwater restoration feasibility study and select an appropriate remedial action alternative. After implementation of the groundwater restoration alternative, DOE will determine if the quarry area or portions of the raffinate pits area can be released for unrestricted or other appropriate use.

1.4 EIS SCOPING

DOE issued a Notice of Intent (NOI) in the Federal Register on March 2, 1984 (U.S. Dept. Energy 1984), to prepare an EIS to assess the environmental impacts of alternatives for the long-term management of the radioactive materials at the Weldon Spring site. The NOI was also mailed to federal,

*The concept of tiering relates to the preparation of additional NEPA documentation on specific project actions not currently developed to a level of detail that would allow for their assessment in this EIS. Specific examples include NEPA documentation for (1) decommissioning of the chemical plant and (2) site-specific impacts of long-term management at another site in Missouri within 160 km (100 mi) of Weldon Spring if this alternative (Alternative 3b in this EIS) were selected. This concept is described in 40 CFR Part 1508.28.

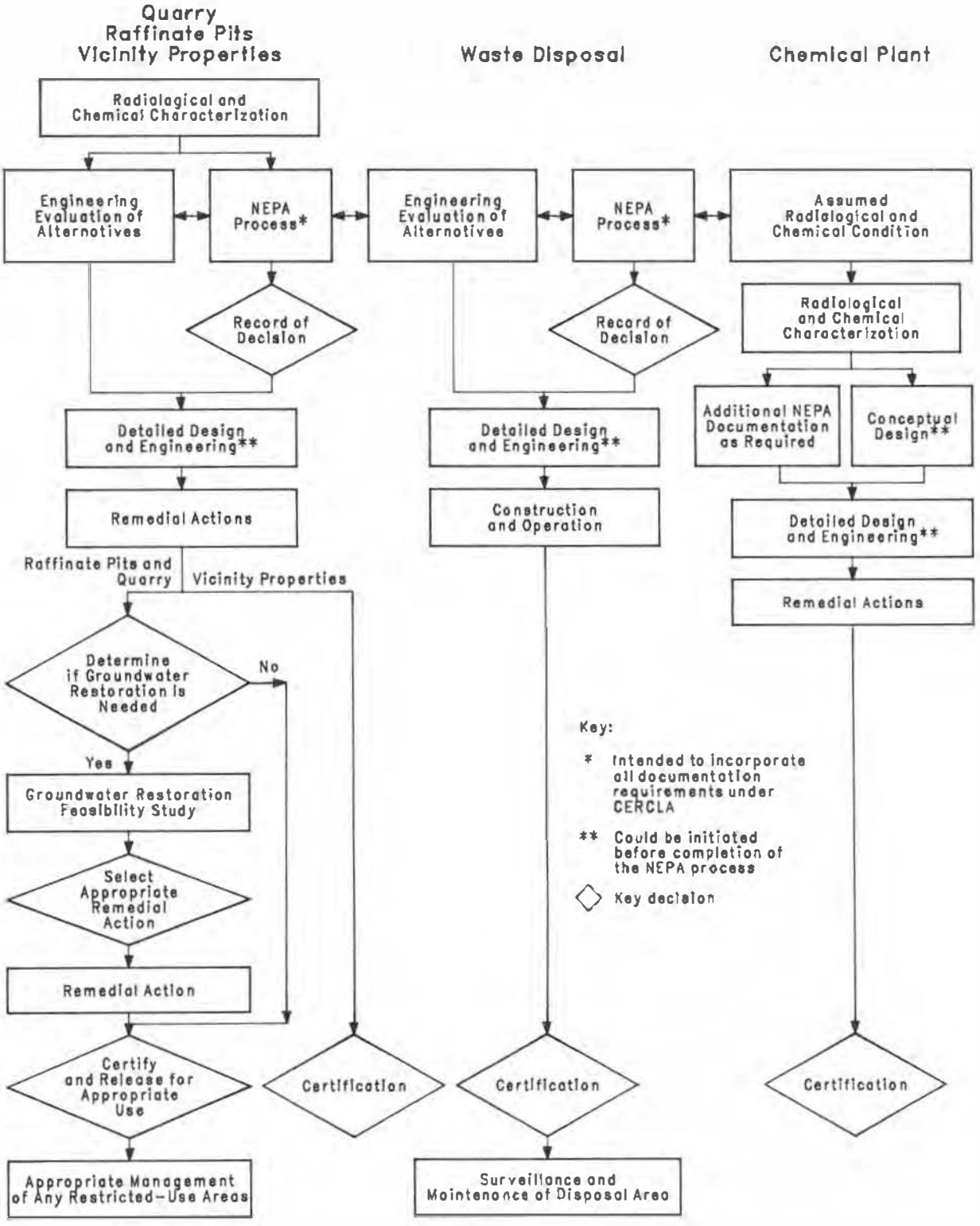

Figure 1.5. DOE Decision-Making Process for the Weldon Spring Site.

state (Missouri and Washington), and local governments and agencies, and to members of the general public. The Department conducted a scoping process to determine the significant issues, the alternatives to be analyzed in depth, and the issues to be eliminated from further detailed study. A report on the scoping process is given in Appendix B.

Based on public and technical input to the scoping process, the following alternatives were identified for analysis in this EIS:

Alternative 1: Long-Term Management in the Existing Raffinate Pits with Improved Containment. Under Alternative 1, all the wastes from the chemical plant, quarry, and vicinity properties as well as the raffinate sludge will be contained in a disposal cell in the existing raffinate pits on the Weldon Spring site. A multilayered cover -- consisting of clay, riprap (rock), sand/gravel, and topsoil -- will be constructed over the wastes.

Alternative 2: Long-Term Management in the Raffinate Pits Area in a New Disposal Cell (DOE's Preferred Alternative). DOE is considering two subalternatives for Alternative 2.

Alternative 2a: Partially Above Grade. Under Alternative 2a, a new disposal cell, partially above grade, will be constructed on the Weldon Spring site. The new cell will contain the same wastes as Alternative 1 plus contaminated soils from beneath the raffinate pits area. The new cell will have a multilayered cover identical to Alternative 1, but it will also have a leachate monitoring system to monitor the performance of the cover.

Alternative 2b: Completely Above Grade. Under Alternative 2b, a new disposal cell, completely above grade, will be constructed on the Weldon Spring site. It will contain all the wastes and will have a multilayered cover and a leachate monitoring system. It will also have a layer of lead in the cover.

Alternative 3: Long-Term Management at Another Site. Under Alternative 3, DOE will transport all or part of the wastes to other sites. Three subalternatives are considered.

Alternative 3a: Hanford Site. Under Alternative 3a, all the Weldon Spring wastes will be transported to the DOE Hanford site near Richland, Washington, for disposal in trenches near an existing disposal area. The wastes will be covered with native soils (excavated from the trenches), riprap, and another layer of soil.

Alternative 3b: "Nearby Site". Under Alternative 3b, the Weldon Spring wastes will be transported to a "Nearby Site" for disposal in a containment cell similar to Alternative 2a. The "Nearby Site", which will be owned and operated by DOE, is assumed to be located in Missouri within 160 km (100 mi) of the Weldon Spring site and would be chosen to have more favorable conditions (e.g., thicker clay, lower hydraulic conductivity, deeper groundwater table, and/or higher sorption capacity) than the Weldon Spring site.

Alternative 3c: Uranium Processing Site. Under Alternative 3c, only the raffinate and quarry sludge will be transported to an existing uranium processing facility in the Four Corners area of the southwestern United States for reprocessing. The sludge will be reprocessed to extract the uranium that remains in it; the other wastes will be placed in a disposal cell, similar to Alternative 1, in the existing raffinate pits.

Alternative 4: No Action. Under Alternative 4, the raffinate sludge will continue to be stored in the pits, the quarry wastes will be left in the quarry, and the chemical plant area and vicinity properties will also be left in their current conditions.

The phrase "long-term management" is used in this EIS because the parent radionuclides (i.e., uranium, thorium, and radium) in the Weldon Spring wastes have long half-lives, and the hazard will not diminish appreciably for thousands of years. The potential environmental impacts under conditions of continuing management (maintenance, monitoring, and corrective actions as necessary) and under conditions of potential loss of management are assessed in this EIS. The time frames for analysis and the assumptions about management controls for each time frame are given in Chapter 4.

1.5 RELATED FEDERAL PROJECTS

DOE has recently prepared environmental impact statements for other programs and other sites under its Remedial Action Program, including:

U.S. Department of Energy. 1983. Final Environmental Impact Statement, Remedial Actions at the Former Vitro Rare Metals Plant Site, Canonsburg, Washington County, Pennsylvania. DOE/EIS-0096-F (2 vol.). July 1983.

U.S. Department of Energy. 1984. Final Environmental Impact Statement, Remedial Actions at the Former Vitro Chemical Company Site, South Salt Lake, Salt Lake County, Utah. DOE/EIS-0099-F (2 vol.) July 1984.

U.S. Department of Energy. 1985. Final Environmental Impact Statement. Remedial Actions at the Former Vanadium Corporation of America Uranium Mill Site, Durango, La Plata County, Colorado. DOE/EIS-0111F (2 vol.). October 1985.

U.S. Department of Energy. 1986. Draft Environmental Impact Statement, Remedial Actions at the Former Climax Uranium Company Uranium Mill Site, Grand Junction, Mesa County, Colorado. DOE/EIS-0126-D. March 1986.

U.S. Department of Energy. 1986. Final Environmental Impact Statement, Long-Term Management of the Existing Radioactive Wastes and Residues at the Niagara Falls Storage Site. DOE/EIS-0109F. April 1986.

In addition, the U.S. Nuclear Regulatory Commission and U.S. Environmental Protection Agency have prepared EIS's on various related programs, proposed standards, and specific sites, including:

U.S. Environmental Protection Agency. 1982. Final Environmental Impact Statement for Remedial Action Standards for Inactive Uranium Processing Sites (40 CFR 192). Vols. 1 and 2; EPA 520/4/82-013-1, -2. October 1982.

U.S. Nuclear Regulatory Commission. 1983. Final Environmental Impact Statement Related to the Decommissioning of the Rare Earths Facility, West Chicago, Illinois. Docket No. 40-2061, Kerr-McGee Chemical Corporation. NUREG-0904. May 1983.

U.S. Environmental Protection Agency. 1986. Environmental Impact Statement. Proposed Wastewater Treatment Facilities for Eastern St. Charles County, Missouri, Including: Duckett Creek Sewer District, St. Peters Sewer District, St. Charles Sewer District, Portage de Sioux Sewer District. EPA 907/9-86-003. May 1986.

1.6 CONSULTATION WITH OTHER AGENCIES

The remedial actions to be carried out by DOE at the Weldon Spring site are subject to EPA oversight under the Comprehensive Environmental Response,

Compensation, and Liability Act (CERCLA) and Superfund Amendments and Reauthorization Act (SARA). For this project, the oversight function will be performed by EPA Region VII. The EPA/DOE Federal Facility Agreement for the Weldon Spring Remedial Action Project was signed in August 1986. This agreement defines the procedures and actions that DOE and EPA must carry out in order to discharge the responsibilities placed upon them by CERCLA, NEPA, and the Atomic Energy Act. The agreement provides for exchange of information and expertise between EPA and DOE. The agreement also establishes a basis for delisting the Weldon Spring site from the National Priorities List (if listed) at the completion of the project.

In preparation for this remedial action, DOE has also been exchanging information with the Missouri Department of Natural Resources, the St. Charles County Board, and federal legislators from Missouri.

1.7 POTENTIALLY APPLICABLE LAWS AND REGULATIONS

Major federal laws and regulations potentially applicable to the various alternatives are presented in Appendix C. Copies of DOE Orders and other supporting documents referred to in the EIS are available for public inspection. DOE will comply with all applicable state statutes for air quality, water quality, and historic preservation.

1.8 REFERENCES

Bechtel National, Inc. 1984. Engineering Evaluation of Alternatives for the Disposition of the Weldon Spring Raffinate Pits Site, Weldon Spring, Missouri. DOE/OR/20722-5. Prepared for U.S. Department of Energy, Oak Ridge Operations Office, Oak Ridge, Tenn. April 1984.

Bechtel National, Inc. 1985a. Addendum to the Engineering Evaluation of Alternatives for the Disposition of the Weldon Spring Raffinate Pits Site, Weldon Spring, Missouri. DOE/OR/20722-5 (Add. 1.). Prepared for U.S. Department of Energy, Oak Ridge Operations Office, Oak Ridge, Tenn. April 1985.

Bechtel National, Inc. 1985b. Radiological Survey Report for the Weldon Spring Quarry. DOE/OR/20722-70. Prepared for U.S. Department of Energy, Oak Ridge Operations Office, Oak Ridge, Tenn. September 1985.

Bechtel National, Inc. 1985c. Weldon Spring Site Environmental Monitoring Report, Calendar Year 1984. DOE/OR/20722-58. Prepared for U.S. Department of Energy, Oak Ridge Operations Office, Oak Ridge, Tenn. July 1985.

Boerner, A.J. 1986. Radiological Survey of the August A. Busch and Weldon Spring Wildlife Areas, Weldon Spring Site, St. Charles County, Missouri. Prepared by Oak Ridge Associated Universities, Oak Ridge, Tenn., for U.S. Department of Energy. April 1986.

Deming, E.J. 1986. Radiological Survey, U.S. Army Reserve Property, Weldon Spring Site, St. Charles County, Missouri. Prepared by Oak Ridge Associated Universities, Oak Ridge, Tenn., for U.S. Department of Energy. January 1986.

National Lead Company of Ohio. 1977. Study of Radioactive Waste Storage Areas at ERDA-Weldon Spring Site. NLCO-1141 (Special). Prepared for U.S. Energy Research and Development Administration, Oak Ridge Operations Office. April 25, 1977 (Republished August 1981).

Rockwell International. 1979. Program Plan for the Decontamination and Disposition of Weldon Spring Chemical Plant. ES-79-31. Prepared by Energy Systems Group, Atomics International Division for U.S. Army Toxic and Hazardous Materials Agency, Aberdeen Proving Ground, Md. September 1979.

U.S. Department of Energy. 1984. Compliance with the National Environmental Policy Act; Intent to Prepare an Environmental Impact Statement and Conduct a Public Scoping Meeting on Long-Term Management of Existing Radioactive Materials in the Vicinity of Weldon Spring, Missouri. Fed. Regist. 49(43):7851-7854 (March 2, 1984).

2. DESCRIPTION OF ALTERNATIVES AND SUMMARY OF ENVIRONMENTAL IMPACTS AND PROCESS AND CONTAINMENT OPTIONS

Descriptions of the alternatives* for long-term management of the Weldon Spring wastes and a summary of the environmental impacts and process and containment options are presented in this chapter. Descriptions of the existing environment at each site and environmental impact analyses are presented in Chapters 3 and 4. Additional details are presented in several appendices, including Appendix E (engineering options), Appendix F (transportation), Appendix H (radiological and chemical characterization), Appendix I (groundwater models, input parameters, and calibration of contaminant transport), Appendix J (estimation of airborne radioactive releases), and Appendix K (physical and biological effects on containment systems).

The analysis of environmental impacts covers two time frames: (1) action period, approximately 10 years during which activities such as excavation, transportation, and disposal will take place, and (2) long-term management, time during which human access to disposal areas will be limited, containment structures will be maintained, any releases to the environment will be monitored, and corrective remedial actions will be taken as necessary. Cumulative impacts over 1,000 years are assessed for each alternative. The 1,000-year time frame was selected to be consistent with the U.S. Environmental Protection Agency (EPA) regulations for management of inactive uranium mill tailings (40 CFR Part 192), which require that "control measures be carried out in a manner that provides reasonable assurance that they will last, to the extent reasonably achievable, up to 1,000 years, and, in any case, for a minimum of 200 years" (U.S. Environ. Prot. Agency 1983). The Weldon Spring wastes are similar to uranium mill tailings. (See the introduction to Chapter 4 for further discussion of the time frames for analysis.)

The major federal laws, regulations, and executive orders potentially applicable to this project are listed in Appendix C. Wastes that are radiologically contaminated only, with no associated chemical contamination hazard, will be disposed of in accordance with the EPA regulations for uranium mill tailings (40 CFR Part 192). Wastes that are chemically contaminated only, and are determined to be hazardous with no associated radiological contamination hazard, will be disposed of off-site at a licensed hazardous-waste-disposal facility in accordance with the requirements of the Resource Conservation and Recovery Act (RCRA). Substances that are both chemically and

*Technical descriptions of alternatives are discussed using the verb "will". This should be interpreted to mean that something will occur if the alternative is implemented as described.

radiologically contaminated will be managed in accordance with the best technical approach available considering the requirements given in 40 CFR Part 192 and RCRA to ensure maximum protection of public health, welfare, and the environment. In addition, all substances will be managed consistent with the requirements of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) as amended by the Superfund Amendments and Reauthorization Act (SARA).

2.1 DESCRIPTION OF ALTERNATIVES

The alternatives assessed in this EIS are

Alternative 1: Long-Term Management in the Existing Raffinate Pits with Improved Containment

Alternative 2: Long-Term Management in the Raffinate Pits Area in a New Disposal Cell (DOE's Preferred Alternative)

- 2a: Partially Above Grade
- 2b: Completely Above Grade

Alternative 3: Long-Term Management at Another Site

- 3a: Hanford Site
- 3b: "Nearby Site"
- 3c: Uranium Processing Site

Alternative 4: No Action

On-site disposal of the Weldon Spring wastes is considered by DOE to be the environmentally preferable means for long-term management of these wastes. Alternative 2 is preferred over Alternative 1 because Alternative 2 involves construction of a new disposal cell rather than use of the existing pits. It would likely be more difficult to modify the existing pits in a manner that will ensure adequate confinement. The design requirements for the new disposal cell will be determined by DOE and will incorporate appropriate features to ensure adequate confinement (e.g., features such as clay liner, riprap, and leachate monitoring system). Tests will be conducted during construction of the disposal cell to verify that the design requirements are met.

Decontamination of the chemical plant, quarry, and vicinity properties is common to all action alternatives. The DOE guidelines for residual radioactive contamination are presented in Appendix D. The residual radionuclide concentrations are based on limiting the dose to a hypothetical individual

under a worst-case plausible scenario to 100 mrem/yr committed effective dose equivalent above background under potential future land-use conditions. DOE policy requires that all exposures to radiation be limited to levels that are as low as reasonably achievable (ALARA). This policy requires that residual radioactivity be reduced to levels that are as far below those that would result in a 100-mrem/yr dose as is reasonably achievable considering technical, economic, and social factors.

For all action alternatives, water from the raffinate pits, chemical plant, and quarry areas will be collected and treated, as necessary, prior to disposal. According to conceptual plans, disposal will be by spray irrigation on a 45-ha (110-acre) area on the federally owned U.S. Army Reserve Property near the Weldon Spring site. DOE will need to obtain the Army's permission to use this property. As an alternative to spray irrigation, the treated water could be released to the Missouri River under controlled conditions and in compliance with Missouri state requirements. However, spray irrigation is considered to be the most feasible approach for disposal of this water. After completion of the actions, the spray irrigation area will be surveyed, decontaminated if necessary, restored, and released for appropriate use.

Decontamination will be accomplished by using conventional construction equipment such as backhoes, bulldozers, and trucks. DOE and its contractors have been involved in numerous decontamination projects under DOE's Remedial Action Program. Standard control measures are used to minimize the spread of radioactive or chemical contaminants. After a site is decontaminated, a separate contractor verifies that the cleanup guidelines have been met before DOE releases the site for appropriate use.

The areas that have been decontaminated (including the quarry) will be backfilled and restored. DOE will consult with officials from the August A. Busch and Weldon Spring wildlife areas to ensure that disruption is minimized and that restoration plans are compatible with recreational or wildlife management plans.

2.1.1 Alternative 1: Disposal in the Existing Raffinate Pits with Improved Containment

Under Alternative 1, sludge from the raffinate pits and all wastes from the chemical plant, quarry, and vicinity properties will be contained in a disposal cell in the existing raffinate pits at the Weldon Spring site. The contaminated soils underneath the raffinate pits will remain in place. The raffinate sludge in the pits will be temporarily moved from one pit (or portion of one pit) to another pit in order to inspect the existing clayey soil under the pits to ensure its adequacy to contain the wastes. Testing of

the clay (e.g., density, moisture, permeability) will be performed as required. The thickness of the clay separating the wastes from the underlying bedrock is about 6 to 9 m (20 to 30 ft) (Bechtel Natl. 1984). If necessary, the clay liner may be removed and reinstalled to verifiable specifications. Additional clay will be transported to the raffinate pits area as needed and will be installed and compacted to appropriate permeability characteristics in the bottoms of the existing raffinate pits to ensure adequate confinement.

Based on conceptual plans, the raffinate sludge and quarry sludge will be stabilized by adding a mixture of 80% fly ash and 20% cement (by weight) at a rate of 1 kg (2.2 lb) for each 1.2 L (0.33 gal) of sludge. The purpose of the stabilizer is to improve the physical properties of the sludge so that it can be moved by standard earth-moving equipment and will be resistant to compaction and subsidence. Stabilization will increase the sludge volume by about 37% (Bechtel Natl. 1984). DOE will take into account the physical and chemical properties of the sludge, additives, and stabilized product in determining the specific stabilization formula.

The wastes will be compacted, and empty spaces in the rubble will be filled with grout and sand (Bechtel Natl. 1984, 1985). The total quantity of wastes to be contained within the disposal cell is estimated to be about 560,000 m^3 (730,000 yd^3) (see Appendix E, Table E.1). This value includes an estimated 64,000 m^3 (84,000 yd^3) volume increase due to the addition of stabilizer to the sludge and excludes an estimated 98,000 m^3 (130,000 yd^3) of contaminated clay already present in the bottoms and sides of the pits. The net effect is a 35,000 m^3 (46,000 yd^3) decrease from the waste volume shown in Table 1.1.

The volume of wastes requiring disposal, 560,000 m^3 (730,000 yd^3), is greater than the capacity of the raffinate pits for storage of liquid raffinates, 500,000 m^3 (650,000 yd^3). All wastes in the disposal cell will be solid (i.e., the raffinates will be stabilized) and will be mounded above the current pit tops (as needed) in an engineered facility (see Figure 2.1).

A multilayered cover will be constructed over the wastes. In the conceptual design, the cover consists of layers of 1.5 m (5 ft) clay, 0.15 m (6 in.) sand/gravel, 0.9 m (3 ft) riprap, 0.15 m (6 in.) sand/gravel, and 0.46 m (18 in.) topsoil (Figure 2.1). A synthetic membrane is not included in this conceptual cover design because its expected service life is only 25 to 30 years at the Weldon Spring site (Bechtel Natl. 1986). The purposes of these layers are as follows: the clay will minimize the escape of radioactive radon gas and the infiltration of precipitation; the sand/gravel layer over the clay will act as a drain to divert infiltrated water out to the periphery of the disposal cell; the riprap will inhibit erosion and intrusion by plants,

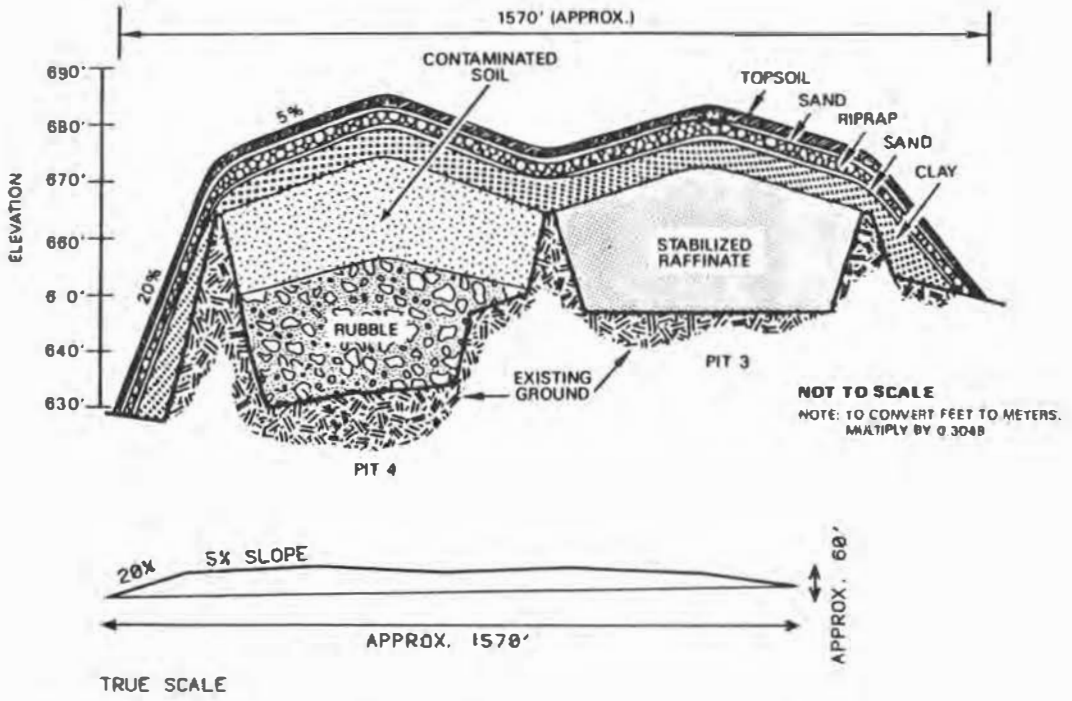

Figure 2.1. Conceptual Design for Alternative 1: Improved Containment in the Existing Raffinate Pits. Source: Modified from Bechtel National (1985).

animals, and humans; the upper layer of sand/gravel will retard the movement of the topsoil into the voids in the riprap layer; and the topsoil layer will provide a growth medium for vegetation. About 480,000 m^3 (630,000 yd^3) of earthen materials will be required for construction of the cover.

During the action period, water from the raffinate pits, chemical plant, and quarry areas will be collected and treated as necessary prior to disposal. DOE will develop a long-term maintenance and monitoring plan for the disposal cell. The containment system will be monitored and inspected on a periodic basis to ensure that it is performing as planned. If necessary, DOE will take corrective actions.

2.1.2 Alternative 2a: New Cell, Partially Above Grade

Under Alternative 2a, a new disposal cell -- partially above grade (Figure 2.2) -- will be constructed on the Weldon Spring site. DOE is currently conducting geological studies on the site to determine an appropriate location for a new disposal cell. If the existing amount of clay is not adequate, DOE intends to import and install clay of sufficient thickness and compact the clay to appropriate permeability characteristics to

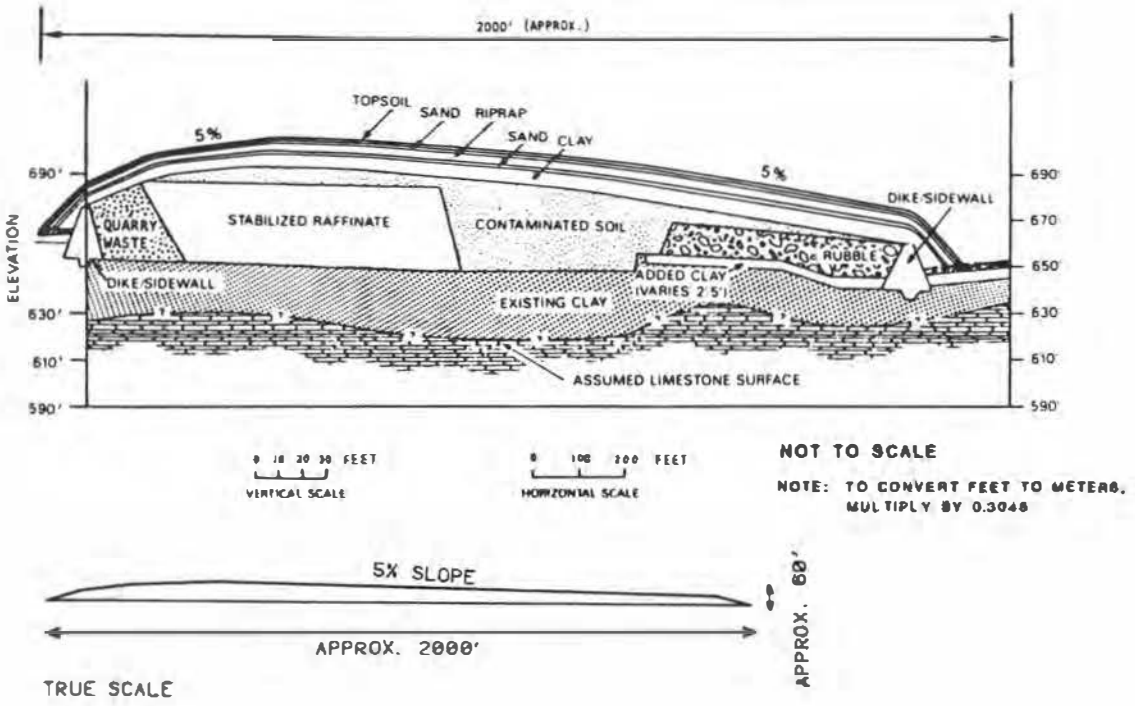

Figure 2.2. Conceptual Design for Alternative 2a: New Cell, Partially Above Grade. Source: Modified from Bechtel National (1985).

ensure adequate confinement. The new disposal cell will contain all the wastes from the chemical plant, quarry, and vicinity properties as well as the raffinate sludge and contaminated soils from beneath the raffinate pits. The raffinate and quarry sludge will be stabilized, thus increasing the volume of sludge to be disposed of. The wastes will be compacted and the spaces in the rubble filled with grout as in Alternative 1. The total quantity of wastes to be contained in the disposal cell is estimated to be about 660,000 m^3 (860,000 yd^3) (see Appendix E, Table E.2). This value is larger than that given in Table 1.1 by 64,000 m^3 (84,000 yd^3) because of the addition of stabilizer to the sludge.

A leachate monitoring system will be constructed for Alternative 2a (Figure 2.3). The floor of the disposal cell will be sloped and covered with a layer of sand to collect any water that infiltrates through the multilayered cover and underlying wastes. The leachate monitoring system will be designed to minimize the release of any radon gas that might be dissolved in the leachate. Any collected leachate will be pumped from the sump, monitored, and treated (if necessary). The purpose of the leachate monitoring system is to monitor the near-term performance of the multilayered cover over the wastes.

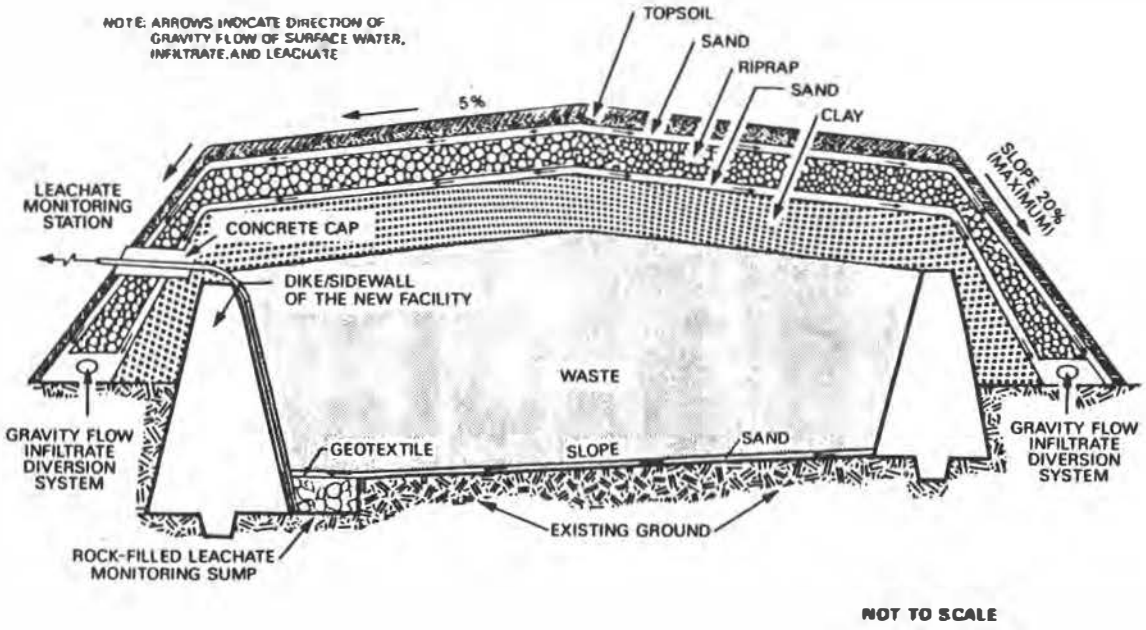

Figure 2.3. Conceptual Design for a Leachate Monitoring System for Alternative 2a: New Cell, Partially Above Grade. Source: Modified from Bechtel National (1985).

The system can be designed to last approximately 25 years. It will probably not be operable for the 200- to 1,000-year design life of the containment system. The uranium mill tailings regulations that DOE is using for guidance require that containment systems be designed for "passive" controls. The leachate system will therefore not be relied upon to collect leachate for 200 to 1,000 years. (For analysis of the long-term transport of contaminants into groundwater under the disposal cell, no credit was taken for collection and removal of leachate.)

For Alternative 2a, a multilayered cover identical to that in Alternative 1 will be constructed over the wastes. More earthen materials will be required to construct the new disposal cell (550,000 m^3 [720,000 yd^3]) because of the larger volume of material to be contained in the cell. Contaminated water collected during the remedial actions will be treated as necessary prior to disposal. After the remedial actions, DOE will maintain and monitor the disposal area.

2.1.3 Alternative 2b: New Cell, Completely Above Grade

Under Alternative 2b, a new cell -- completely above grade -- will be constructed on the Weldon Spring site. As in Alternative 2a, the exact

location will be determined after the results of ongoing geological studies are available. If the existing amount of clay is not adequate, DOE intends to import and install clay of sufficient thickness and compact the clay to appropriate permeability characteristics to ensure adequate confinement. The new disposal cell will contain about 660,000 m^3 (860,000 yd^3) of wastes, including wastes from the chemical plant, quarry, and vicinity properties as well as stabilized raffinate sludge and contaminated soils from beneath the raffinate pits (see Appendix E, Table E.3). This value is larger than that given in Table 1.1 by 64,000 m^3 (84,000 yd^3) because of the addition of stabilizer to the sludge. The raffinate and quarry sludge will be stabilized and compacted, and spaces between the rubble will be grouted as in Alternative 1.

As in Alternative 2a, a leachate monitoring system will also be constructed (Figure 2.4) for the purpose of monitoring the near-term performance of the multilayered cover over the wastes. The multilayered cover will be similar to that for Alternative 1, except that it will have a layer of lead (Figure 2.5). The lead is intended to act as an additional barrier to both infiltrating precipitation and radon gas emission. The addition of a lead sheet to the cover was requested during the scoping process. Construction of a large lead sheet over a containment area has not been done before (Jones 1986).

Construction of the containment cell for Alternative 2b will require considerably more earthen materials than either Alternative 1 or 2a, a total of 960,000 m^3 (1,300,000 yd^3). As in Alternative 1, contaminated water collected during the remedial actions will be treated as necessary prior to disposal. After the remedial actions, DOE will monitor and maintain the disposal area.

2.1.4 Alternative 3a: Hanford Site

Under Alternative 3a, all the Weldon Spring wastes will be transported by train to the Hanford site near Richland, Washington. The raffinate and quarry sludge will be dried and placed in metal containers prior to shipping. Drying will reduce the sludge volume to about 30% of its current wet volume (Bechtel Natl. 1984). The other wastes will be transported in bulk form (unpackaged). A new railroad spur will be constructed at the Weldon Spring site on the existing right-of-way. Loading and unloading facilities will be constructed at Weldon Spring and Hanford, as needed. The total volume of wastes to be transported to the Hanford site (i.e., wastes from the chemical plant, quarry, and vicinity properties; and dried sludge and contaminated soils from beneath the raffinate pits) is estimated to be about 480,000 m^3 (630,000 yd^3) (see Appendix E, Table E.4). This value is lower than that

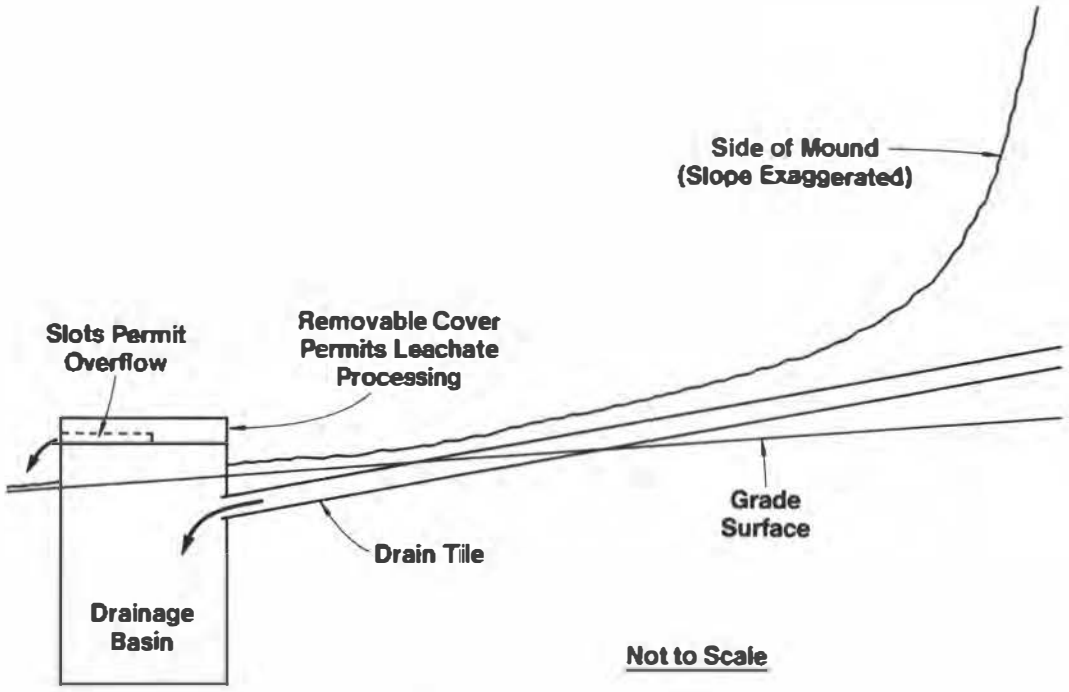

Figure 2.4. Conceptual Design for a Leachate Monitoring System for Alternative 2b: New Cell, Completely Above Grade.

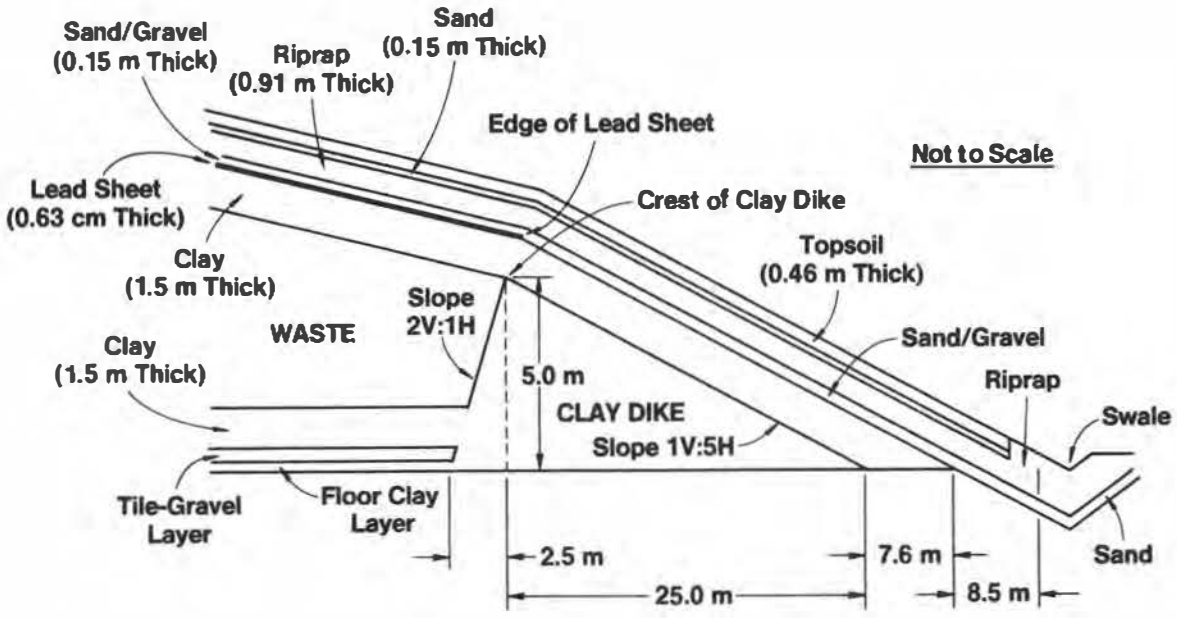

Figure 2.5. Conceptual Design for Cover for Alternative 2b: New Cell, Completely Above Grade.

given in Table 1.1 by 120,000 m^3 (160,000 yd^3) because of the removal of water by heat drying of the unstabilized sludge.

At the Hanford site, the wastes will be placed in trenches and covered with layers of 0.9 m (3 ft) of native soils (originally excavated from the trench), 0.9 m (3 ft) of riprap, and another 1.2-m (4-ft) layer of native soils (Figure 2.6). About 1,300,000 m^3 (1,700,000 yd^3) of soil will have to be excavated and backfilled, and 310,000 m^3 (400,000 yd^3) of riprap will be required. Contaminated water collected during the remedial actions will be treated as necessary prior to disposal. DOE will maintain and monitor the containment area at the Hanford site. The entire Weldon Spring site will be released as appropriate for future use.

2.1.5 Alternative 3b: "Nearby Site"

Under Alternative 3b, all of the Weldon Spring wastes will be transported by truck to a "Nearby Site" in Missouri within 160 km (100 mi) of the Weldon Spring site. The "Nearby Site" will have more favorable conditions (e.g., thicker clay, lower hydraulic conductivity, deeper groundwater table, and/or higher sorption capacities) than the Weldon Spring site and will be owned and operated by DOE. The raffinate and quarry sludge will be dried and placed in metal containers prior to shipping; the other wastes will be transported in bulk form (unpackaged). The amount of wastes to be transported is the same as that for Alternative 3a, about 480,000 m^3 (630,000 yd^3) (see Appendix E, Table E.5).

At the "Nearby Site", the wastes will be placed in a disposal cell similar to Alternative 2a -- i.e., partially above grade with a multilayered cover and a leachate monitoring system. About 450,000 m^3 (590,000 yd^3) of earthen materials will be required to construct the disposal cell. Contaminated water collected during the remedial actions will be treated as necessary prior to disposal. DOE will maintain and monitor the containment area at the "Nearby Site." The entire Weldon Spring site will be released as appropriate for future use.

2.1.6 Alternative 3c: Uranium Processing Site

Under Alternative 3c, only the raffinate and quarry sludge will be transported to an existing uranium processing facility in the Four Corners area of the southwestern United States; at the processing facility, the uranium that remains in the sludge will be extracted. Prior to transport, the sludge will be dried and placed in containers. Only 51,000 m^3 (67,000 yd^3) will have to be transported, which is about 10% of the volume of wastes that would be transported off-site in Alternatives 3a and 3b (see Appendix E,

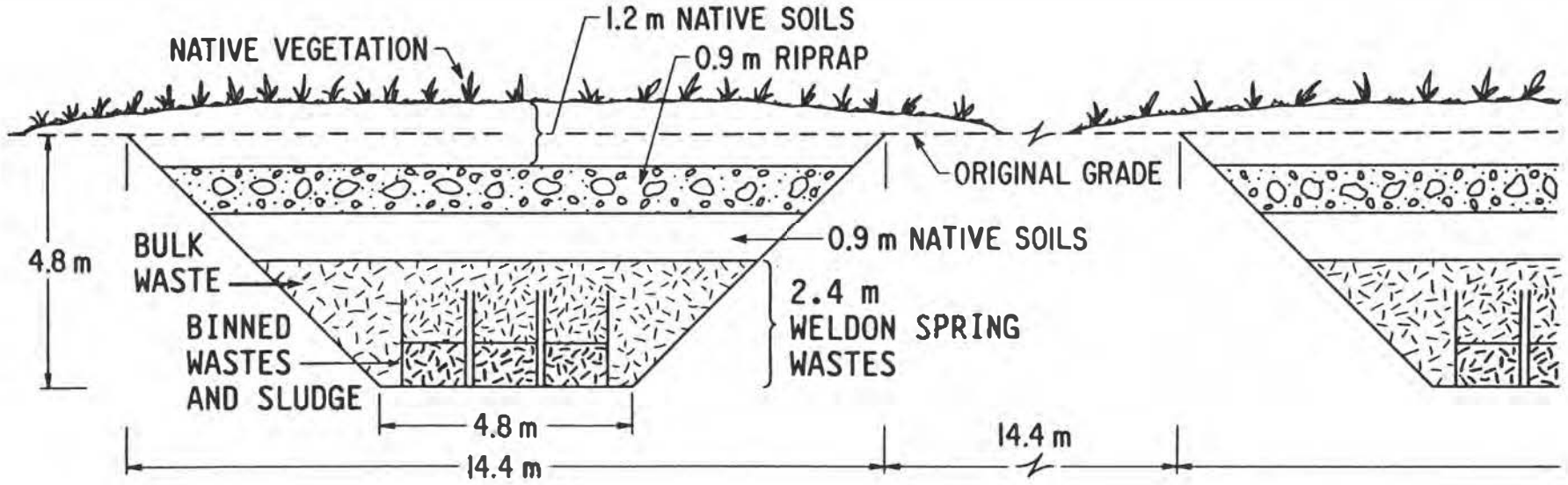

Figure 2.6. Conceptual Design for Alternative 3a: Hanford Site.
Source: Modified from U.S. Department of Energy (1986).

Table E.6). The sludge contains most of the radioactivity in the Weldon Spring wastes.

The remaining wastes, 370,000 m^3 (480,000 yd^3) (see Appendix E, Table E.6), will be placed in the existing raffinate pits at the Weldon Spring site. The disposal cell will be similar to that for Alternative 1. This waste volume includes 42,000 m^3 (55,000 yd^3) of contaminated clay from the sides and bottoms of Pits 1, 2, and 3. The contaminated clay in the sides and bottom of Pit 4 will be left in place because Pit 4 will be used as the disposal cell in this alternative. Measures similar to those used for Alternative 1 will be taken to ensure the integrity of the clay in the sides and bottom of Pit 4 (see Section 2.1.1). Because the cell will be smaller, a smaller amount of earthen materials will be needed for the cover, approximately 290,000 m^3 (380,000 yd^3). Contaminated water collected during the remedial actions will be treated as necessary prior to disposal. DOE will monitor and maintain the disposal area at the Weldon Spring site. The wastes resulting from reprocessing the sludge at the uranium processing site will be a small addition to the existing tailings pile at that site.

2.1.7 Alternative 4: No Action

In the no-action Alternative 4, the raffinate sludge will continue to be stored in the raffinate pits, the quarry wastes will be left in the quarry, and the chemical plant area and vicinity properties will also be left in their current conditions. Ongoing maintenance and monitoring activities will be continued at the raffinate pits, chemical plant, and quarry. The federal government will continue to own and restrict access to the raffinate pits, quarry, and chemical plant areas. Alternative 4 is included in this EIS because CEQ guidelines for preparation of an EIS require inclusion of a no-action alternative for comparative purposes.

2.2 SUMMARY COMPARISON OF ENVIRONMENTAL IMPACTS

A summary comparison of the environmental impacts for the various alternatives is given in Table 2.1. The baseline for the comparison is the no-action Alternative 4. Alternatives listed to the right of Alternative 4 denote a positive environmental impact relative to Alternative 4, and alternatives listed to the left of Alternative 4 denote a negative environmental impact relative to Alternative 4. This comparison is a qualitative, not quantitative, comparison of the alternatives. The environmental impacts of implementing the various alternatives are described in detail in Chapter 4. For the alternatives requiring activities at two sites (Alternatives 3a, 3b, and 3c), the impact rankings for each environmental factor and time period account for activities at both sites.

Table 2.1. Qualitative Ranking of the Alternatives by Environmental Factors[a]

Environmental Factor	Worse ←								No Action								Better →
Action Period																	
Groundwater									4	<	1 2a 3b	<	2b 3a 3c				
Surface water	3b	<	1 2a 2b	<	3c	<	3a	<	4								
Radiological					3a 3b	<	1 2a 2b 3c	<	4								
Ecological			3a 3b	<	2a 2b	<	1 3c	<	4								
Air quality					1 2a 2b 3b 3c	<	3a	<	4								
Socioeconomic			3a 3b	<	2b	<	1 2a 3c	<	4								
Long-Term Management																	
Groundwater									4	<	1	<	2a	<	3b 3c	< 2b <	3a
Surface water									4	<	2b	<	1 2a 3b 3c	<	3a		
Radiological									4	<	3a	<	1 2a 2b 3b 3c				
Ecological									4	<	1 2a 2b 3a 3b 3c						
Air quality									No Change								
Socioeconomic									4	<	1 2a 2b	<	3a 3b	<	3c		

[a] The alternatives are ranked in this table according to their relative impact. The baseline for comparison is the no-action Alternative 4. Alternatives listed to the right of Alternative 4 have a positive environmental impact relative to Alternative 4; alternatives listed to the left of Alternative 4 have a negative environmental impact relative to Alternative 4.

The rankings given in Table 2.1 apply within each row only; they are not intended to be used to compare different environmental factors or time periods to generate a single overall ranking. Details describing the procedures used to determine these rankings are given in the following sections. This analysis was done separately for the action period and for long-term management because the types of activities during these two time frames are quite different from one another.

2.2.1 Groundwater

2.2.1.1 Action Period

For most chemical and radioactive species, there would be no groundwater impacts for any of the action alternatives at the raffinate pits area during the action period because the time for the action period (10 years) is short compared to the time required for the contaminants to reach the groundwater. Mobile chemical species are exceptions in that they would reach the groundwater in the limestone at the site boundary in 2 to 3 years. For these species a positive impact would be expected to occur for the action alternatives at the raffinate pits area because the raffinate sludge would be stabilized (Alternatives 1, 2a, 2b,) or removed (Alternatives 3a, 3b, 3c). However, the magnitude of the benefit would not be large because the predicted peak concentration contribution* of these species in the groundwater for Alternative 4 is a small fraction (7% or less) of the Missouri state groundwater limit. During the action period, the groundwater impacts in the quarry area for the action alternatives would be about the same as for the no-action Alternative 4.

Alternatives 2b, 3a, and 3c would have a higher positive impact than Alternatives 1, 2a, and 3b for the following reasons. The presence of the lead sheet in Alternative 2b would prevent leaching of the wastes during the action period. No impacts to groundwater are expected at the Hanford site (Alternative 3a) during the action period because no chemical or radioactive species would reach the groundwater during this period. In Alternative 3c,

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*"Concentration contribution" is defined in this EIS as the incremental concentration resulting from leaching of the Weldon Spring wastes. Direct comparison between concentration contributions and water quality limits is not applicable because water quality limits include contributions from all sources (including background). Because DOE does not have authority to control activities conducted on nearby properties in the affected area, DOE is not able to determine if applicable water quality limits may be exceeded at a specific location in the future. DOE intends to ensure that all applicable state and federal release limits are not exceeded and that all releases are maintained as low as reasonably achievable.

the substances with the highest concentrations of radioactive and chemical contaminants (i.e., the raffinate and quarry sludge) would be removed to a uranium processing site for reprocessing. Alternatives 2b, 3a, and 3c were therefore ranked higher than Alternatives 1, 2a, and 3b for this environmental factor.

2.2.1.2 Long-Term Management

Positive long-term groundwater impacts would be expected in the raffinate pits area for Alternatives 1, 2a, and 2b because stabilizing the sludge and placing the wastes in a covered disposal cell would reduce the leaching of chemicals and radiological species. Positive long-term groundwater impacts would also be expected at the raffinate pits area for Alternatives 3a and 3b because all the wastes would be removed and for Alternative 3c because the raffinate sludge would be removed and the other wastes placed in a covered disposal cell.

A positive long-term impact would occur for all action alternatives in the quarry area because, for most chemical species, the areas of groundwater near the quarry that are predicted to be contaminated above Missouri groundwater limits would be larger and would last longer for Alternative 4 than for the action alternatives. This positive impact would be the same for all action alternatives because the quarry wastes would be removed. The impact of the action alternatives at the county well field would be positive because peak concentration contributions of radiological and chemical species in the water at any of the wells are predicted to be smaller for the action alternatives than for the no-action Alternative 4. The impact would be small because only one measured radiological species, uranium, has peak concentration contributions in the water of any of the wells that are predicted to be about equal to background for the no-action Alternative 4. The predicted values are lower by a factor of about seven for all action alternatives. For all alternatives, peak concentration contributions for chemical species in water of any of the wells are predicted to be far below background values and regulatory limits.

The long-term groundwater impact for Alternative 3a at the Hanford site would be very small because only the most mobile species would reach the groundwater and move off-site. Peak concentration contributions in groundwater under the Hanford site are predicted to be below background (sulfate) or within the range of background values (nitrate). The long-term groundwater impact at the "Nearby Site" for Alternative 3b would be less than the impact for Alternative 2a at the raffinate pits area because the "Nearby Site" would be chosen to have more favorable conditions for waste disposal than the raffinate pits area. A negligible long-term impact would occur at

the uranium processing site for Alternative 3c because the raffinate and quarry sludge would contribute only a small increment to the wastes already present at that site.

The long-term groundwater impacts at the various locations were combined to give an overall ranking for the alternatives. Alternative 3a would have the highest positive impact because the wastes would be removed from the Weldon Spring site and little groundwater contamination would occur at the Hanford site. Alternative 2b would have next highest positive impact because the lead sheet would prevent leaching of the wastes for much of the 1,000 years and corroded lead would not reach the groundwater in 1,000 years. Alternatives 3b and 3c would have about the same positive impact, lower than that for Alternative 2b. The positive impacts for Alternatives 3b and 3c would be about the same because of approximately compensating effects; Alternative 3b ("Nearby Site") would have higher source term concentrations and more favorable hydrogeologic conditions, whereas Alternative 3c (uranium processing site) would have lower source term concentrations and less favorable hydrogeologic conditions (raffinate pits area). Alternative 2a would be next and Alternative 1 would have the lowest positive groundwater impact because model calculations show that, of all the action alternatives, Alternative 1 would have the highest concentrations of contaminants in groundwater under the raffinate pits area.

2.2.2 Surface Water

2.2.2.1 Action Period

During the action period, waste-handling activities (e.g., excavation and transportation) for all action alternatives at the Weldon Spring site would be expected to generate increased sediment and contaminant loading of surface runoff over that expected for the no-action Alternative 4. Good housekeeping procedures, such as collecting runoff in settling ponds, would be expected to minimize the impact. Some runoff might also result from disposal by spray irrigation of the 314,000 m^3 (83,000,000 gal) of treated water generated at the Weldon Spring site, but this impact would be minimized by controlling the rate of spray irrigation.

For Alternatives 3a and 3b involving off-site disposal, some sediment and contaminant loading of surface runoff would be expected at the respective disposal sites. Because the Hanford site (Alternative 3a) is in a more arid environment than the "Nearby Site" (Alternative 3b), the impact on surface runoff should be less at the Hanford site than at the "Nearby Site". Contamination of surface runoff along the transportation routes due to spillage or accidents would probably be greater for Alternative 3a than

Alternative 3b because of the longer transport distance. The impact on surface runoff along the transportation route for Alternative 3c should be less than that for Alternative 3a because the transport distance is shorter and the volume of wastes transported is much less. In addition, all of the wastes will be packaged in Alternative 3c. The impact on surface runoff at the uranium processing site would be small because the site is located in an arid environment.

All action alternatives would result in negative impacts relative to the no-action Alternative 4. Alternative 3b would have the greatest surface water negative impact because sediment and contaminant loading of runoff would occur at two sites (Weldon Spring site and "Nearby Site"). Alternatives 1, 2a, and 2b would be next. Alternative 3c would have a lower negative impact because little runoff would occur at the arid uranium processing site and fewer activities would occur than at the Weldon Spring site. Alternative 3a would have the lowest negative impact because little runoff would occur at the arid disposal site (Hanford) and fewer activities would occur at the Weldon Spring site than for Alternative 3c.

2.2.2.2 Long-Term Management

Over the long term, a positive surface water impact would occur for all action alternatives because the disposal cells would be actively maintained and monitored to minimize erosion. Also, unlike Alternative 4, maintenance of the cell cover means that any sediment loading of surface runoff would be expected to contain uncontaminated materials only. The existing contamination in the lakes, springs, and sloughs in the vicinity properties would be expected to be gradually reduced during the long term.

The positive impact for Alternative 3a would be slightly higher than that for the other action alternatives because the wastes would be buried below grade and the extension of the covered disposal cell above grade would be the smallest of all the action alternatives (Figures 2.1 through 2.6). Because of this and the arid climate at the Hanford site, very little (if any) sediment loading of surface runoff would be expected. The positive impact would be least for Alternative 2b because of the potential for transport of corroded lead off the cover of the disposal cell, which would result in contamination of surface runoff. The positive impacts would be similar for the other action alternatives.

2.2.3 Radiological

The net radiological impacts for the action alternatives were determined from the estimated cumulative radiological doses to the general public and workers for the action period and long-term management. For each alternative,

the cumulative dose is the sum of the estimated doses to the general public and to workers during the respective time frame. The results are summarized in Table 2.2.

2.2.3.1 Action Period

All action alternatives would have a negative radiological impact relative to the no-action Alternative 4 during the action period, as shown in Table 2.2. This negative impact would be similar for Alternatives 1, 2a, 2b, and 3c; and similar but greater for Alternatives 3a and 3b.

2.2.3.2 Long-Term Management

During the long term, a positive radiological impact would occur for all action alternatives. This positive impact would be similar for Alternatives 1, 2a, 2b, 3b, and 3c and lower for Alternative 3a. The positive impact would be lowest for Alternative 3a due to the higher dose to the general public from radon gas emissions at the Hanford site (Section 4.2.1.1). The radon gas emissions at the Hanford site would be larger because the drier soil used in the cover at the Hanford site is more porous.

2.2.4 Ecology

2.2.4.1 Action Period

All action alternatives would incur a negative ecological impact during the action period because animal and plant habitats would be lost in the raffinate pits, chemical plant, and quarry areas. Also, aquatic habitat would be lost due to the removal of water from the raffinate pits and quarry pond, and habitat in the area receiving spray irrigation would be disturbed. Aquatic and terrestrial habitats would also be impacted in the areas of the vicinity properties that are decontaminated. A larger habitat loss might occur for Alternatives 2a and 2b than for Alternative 1 because additional land, 18 ha (45 acres) for Alternative 2a and 23 ha (58 acres) for Alternative 2b, must be cleared for construction of a new disposal cell (Section 4.3.1.1). Alternatives 3a and 3b are expected to have the largest negative impact because habitat would be lost in the new areas chosen for waste disposal; 120 ha (300 acres) of arid habitat already designated as a waste-management area would be lost at Hanford and about 15 ha (37 acres) of old-field/pasture or upland forest habitat would be lost at the "Nearby Site". Habitat loss for Alternative 3c would be similar to that for Alternative 1 because receipt and processing of the Weldon Spring sludge at the uranium processing site would not result in loss of additional habitat.

Table 2.2, Summary of Radiological Doses to Workers and the General Public[a]

Alternative	Dose during Action Period (person-rem)			Dose during Long-Term Management (person-rem)		
	Workers	General Public	Total[b]	Workers	General Public	Total[b,d]
1	110	31	140	-[c]	180	180
2a	120	31	150	-	160	160
2b	120	31	150	-	180	180
3a	130	250	380	-	720	720
3b	230	120	350	-	160	160
3c	120	39	160	-	130	130
4	5.1	45	50	-	11,000	11,000

[a] Values rounded to two significant figures.

[b] The sum of the cumulative radiological doses to workers and the general public.

[c] Less than 0.1 person-rem/yr.

[d] Equal to the cumulative doses to the general public because the worker dose would be small during the long term (Section 4.2.2.3).

Source: Tables 4.7 and 4.8.

2.2.4.2 Long-Term Management

Over the long term, a small positive ecological impact would occur for all action alternatives because the newly constructed disposal cell should be more stable and less subject to earth slides and erosion (which affects habitat) than the current dikes and walls of the raffinate pits. However, under the long-term maintenance and monitoring programs for the action alternatives, site maintenance might limit wildlife diversity and use of the areas containing the disposal cells. This positive ecological impact would be similar for all action alternatives.

2.2.5 Air Quality

2.2.5.1 Action Period

During the action period, waste-handling and earth-moving activities would generate dust for all action alternatives. This dust could be carried off-site by wind and would then affect the air quality in areas surrounding

the activities. Model calculations of the maximum 24-hour average concentrations of total suspended particulates (TSP) to be expected at the site boundary were carried out for the action alternatives. These results are summarized in Table 2.3.

As expected, there would be a negative impact on air quality during the action period for all action alternatives. The estimated impacts are higher, but similar, for Alternatives 1, 2a, 2b, 3b and 3c; and lower for Alternative 3a. The most significant activities affecting air quality are transportation of the wastes and construction of the earthen cover for the disposal cell.

2.2.5.2 Long-Term Management

No significant impact to air quality is expected for any of the action alternatives over the long term. Maintenance of the covers of the disposal cells would minimize fugitive dust generation and would keep it at the background levels expected to exist for the no-action Alternative 4.

Table 2.3. Summary of Estimated Maximum 24-Hour Average Concentrations of Total Suspended Particulates

Alternative	Site	Maximum Average 24-Hour TSP Concentration at Site Boundary ($\mu g/m^3$)
1	Weldon Spring site	95
2a	Weldon Spring site	95
2b	Weldon Spring site	100
3a	Weldon Spring site	47
	Hanford site	Background
3b	Weldon Spring site	47
	"Nearby Site"	95
3c	Weldon Spring site	95
	Uranium processing site	Background
4	Weldon Spring site	40

[a] Background value for the Weldon Spring area.

Source: Table 4.14.

2.2.6 Socioeconomics

2.2.6.1 Action Period

During the action period, several socioeconomic factors would be affected by activities carried out for the action alternatives. The factors considered in this analysis are land commitment, effect on local economy, effect on local traffic, and esthetic effects.

All action alternatives would have a negative impact with regard to land commitment during the action period because additional land would be used for spray irrigation. Additional land would also be committed for off-site disposal in Alternatives 3a and 3b. Additional land commitment at the uranium processing site for Alternative 3c would be minimal. Land impacts in terms of the area of land committed for waste disposal would be highest for Alternative 3a and lowest for Alternatives 1, 2a, 2b, and 3c. Alternative 3b would have a negative land commitment impact intermediate to these.

All action alternatives would have a positive impact on the local economy during the action period because more jobs would be available and materials would be purchased locally. This impact is expected to be greatest for Alternatives 2b and 3b, less for Alternatives 1 and 2a, and least for Alternatives 3a and 3c. Alternative 2b would have larger work force and materials requirements (due to purchase and installation of the lead sheet) than Alternative 2a. Alternative 3b would require activity at two sites, and transportation of the wastes by truck might benefit the local economy. The extra activity required at the Hanford site or the uranium processing site would be insignificant compared to the activity already occurring at these sites (the money expended for rail transportation of the wastes would benefit the national economy, but it would not be expected to benefit the local economy). The work force and materials required at the Weldon Spring site would be less for Alternative 3a, 3b, or 3c than for Alternative 1.

Action-period activities would have a negative impact in terms of increased congestion and disruption of traffic at the site areas. A convenient measure of this impact is the number of truck trips per day required to move materials at the active sites for the action alternatives (Section 4.5.3). On this basis, Alternative 2b would have the largest negative traffic impact followed by Alternative 3b. The negative impact for Alternatives 1 and 2a would be lower than that for Alternative 3b. Alternative 3a would be next with an even lower negative impact, and Alternative 3c would have the smallest negative impact.

There would be a negative esthetic impact during the action period because dust would be generated, temporary piles of wastes and construction

materials would be created, and noisy equipment would be operating. This impact would be highest for Alternatives 3a, 3b, and 3c (which require off-site transportation to other disposal sites) and less for Alternatives 1, 2a, and 2b.

These components were combined to give an overall ranking of the action-period socioeconomic impacts. Alternatives 1, 2a, and 3c would have the smallest negative action-period socioeconomic impact, Alternative 2b the next, and Alternatives 3a and 3b the largest negative socioeconomic impact.

2.2.6.2 Long-Term Management

The long-term socioeconomic impacts were also analyzed in terms of several components -- land resources, land use, and esthetics. Land area is a convenient measure of the land resources committed permanently to waste disposal (Section 4.5.2). The impact for an action alternative was obtained by comparing the area of land committed permanently to waste disposal with the area of land committed for the no-action Alternative 4. On this basis, Alternatives 3b and 3c would have essentially the same positive land resource impact; Alternatives 1, 2a, and 2b would have essentially the same smaller positive impact; and Alternative 3a would have a negative land resource impact.

Positive land-use impacts would occur for all action alternatives over the long term. Land-use impacts include use of state lands surrounding the Weldon Spring site for recreation and increased population growth in the area of the Weldon Spring site. Alternative 3a would have the largest positive impact because all the wastes would be removed to a site that already contains wastes. Alternative 3c would be next because some of the wastes would be removed to a uranium processing site that already contains wastes, and Alternatives 1, 2a, 2b, and 3b would all have the same smallest positive impact. Alternative 3b would have the same positive impact as Alternative 2a because it is assumed that the combination of land-use benefits for a decontaminated Weldon Spring site with the negative land-use impacts for a new disposal site would be about the same as the positive land-use impact for Alternative 2a.

Decontamination of the Weldon Spring site and long-term maintenance of the disposal cell should have a positive long-term esthetic impact. This impact would be larger for Alternative 3a because the wastes would be buried below grade and the covered disposal cell would have a low profile above grade. The benefit would be less for the other action alternatives because of a higher profile for the disposal cell.

These components were combined in the same manner for the long term as for the action period. Alternative 3c would have the largest socioeconomic impact, and Alternatives 3a and 3b would be next. Alternatives 1, 2a, and 2b would have about the same lowest positive socioeconomic impact.

2.3 MITIGATIVE MEASURES

Implementation of any of the alternatives would result in impacts to the environment. The impacts can be reduced by continuing a monitoring program and by taking any necessary corrective actions. DOE will consult with federal, state, and local officials in developing specific mitigative measures for the alternative selected. During the action period, such plans will include procedures for minimizing contaminant releases to the environment, controlling traffic, and keeping the public updated on project activities. For the long term, the plans will include monitoring the releases of contaminants to the environment and monitoring the performance of the containment system. In addition to routine maintenance, DOE will take any necessary corrective actions. A summary of the monitoring and mitigation plans for the chosen alternative will be published in the Federal Register as part of the Record of Decision.

2.4 SUMMARY OF PROCESS AND CONTAINMENT OPTIONS

The alternatives analyzed in this EIS were chosen to cover the range of environmental impacts of reasonable actions. They are not meant to preclude a management action that is some combination of particular features of the alternatives. For example, if the use of a lead sheet in the disposal cell cover is deemed to be appropriate, this lead sheet could be added to whatever alternative is selected for implementation. The environmental impacts for use of a lead sheet have been addressed in this EIS (in Alternative 2b). The same rationale holds for many other features included in the alternatives considered. It is not feasible to address all the permutations associated with remedial actions at the Weldon Spring site. The advantages and disadvantages of the major process and containment options for long-term management of the Weldon Spring wastes are summarized in Table 2.4 and are briefly discussed in the following sections. Additional discussion and details are given in appropriate sections of this document.

2.4.1 Wastewater Disposal

The amount of contaminated water that is estimated to be generated during the action period is 314,000 m^3 (83,000,000 gal). This includes 11,000 m^3 (2,900,000 gal) from the quarry pond, 216,000 m^3 (57,000,000 gal) from the

Table 2.4. Advantages and Disadvantages of the Major Process and Containment Options for Long-Term Management of the Weldon Spring Wastes

Option	Advantages	Disadvantages
	WASTEWATER DISPOSAL	
Missouri River	Has negligible impact on water resources.	May require an NPDES permit and extensive treatment of water. Requires construction along discharge route.
Spray irrigation	May require less treatment of water.	May require controlled rate of disposal for acceptable impacts. Could extend length of remedial action period.
	RAFFINATE SLUDGE STABILIZATION AND DISPOSAL	
Stabilization Methods		
Addition of stabilizer	Is relatively inexpensive.	Results in increased weight and volume, increased injuries and accidents, and increased transportation costs. Potential contaminants in stabilizer may result in increased concentrations of contaminants in leachate.
Heat drying	Results in large reduction in volume. Has fewer accidents associated with handling smaller volume. Transportation costs are lower.	Requires large amounts of energy (16,200 m^3 of fuel oil) and has large energy costs. Requires packaging for dried sludge from all four pits.

Table 2.4. Continued

Option	Advantages	Disadvantages
Stabilization Methods (Continued)		
Processing for uranium removal	Removes a contaminant (uranium) from the sludge that can be used for other purposes.	Requires transport to uranium processing facility with resultant increased costs, accidents, and doses. Has increased costs for reprocessing (reduced by the value of recovered uranium). Raffinate sludge may not be accepted at existing uranium processing facilities.
Disposal		
Above-grade vs. below-grade disposal		
Above-grade	Avoids some disposal cell excavation and associated costs and accidents. Maximizes soil thickness between wastes and groundwater.	Results in increased erosion of cell cover and sides. Requires cell walls or berms.
Below-grade	Avoids construction of cell walls or berms. Results in minimal erosion of cell cover.	Requires excavation of disposal cell. Results in decreased soil thickness between wastes and groundwater.

Table 2.4. Continued

Option	Advantages	Disadvantages
	DISPOSAL CELL COMPONENTS	
Leachate Monitoring and Collection System		
Leachate monitoring system	Allows timely detection of leachate buildup.	Requires presence of holes or pipes to cell bottom to provide for leachate collection, which could result in pathways for leachate escape and radon emission. Service life may be short compared to 1,000 years.
Leachate collection system	Allows timely detection of leachate buildup as well as collection and removal of leachate.	Same as for leachate monitoring system.
Cell Cover		
Lead sheet	Prevents entry of water and emission of radon gas.	Differential settling likely to tear sheet. Corrosion may result in buildup of lead in sediments and runoff. Workers may be exposed to lead fumes and dust.
	TRANSPORTATION MODES	
Truck	Is flexible and avoids waste transfers. More routes are possible and may be more economical for short distances.	Results in higher occupational dose. May be more costly for long distances.
Rail	Results in lower occupational dose. May be more economical for long distances.	Is less flexible and requires waste transfers. May be less economical for short distances.

raffinate pits, and an additional 87,000 m^3 (23,000,000 gal) from additional sources such as rainwater or process water. Two options are considered for disposal of this water: discharge to the Missouri River and disposal by spray irrigation.

Discharge to the Missouri River may require an NPDES permit from the state of Missouri, and the water would have to be treated to satisfy the conditions of that permit. Because the Missouri River is a source of drinking water for several communities, treatment could be extensive if stringent concentration limits were included in the permit (e.g., the Missouri drinking water limits--Appendix H, Table H.12). Also, some construction (e.g., pipeline, ditch, road) along the discharge route would probably be required. The main advantage of discharging to the Missouri River is that, once the treated water is in the river, it will mix with and be carried away by large volumes of water and would thus have a negligible impact on water resources in the area.

Disposal by spray irrigation would require the water to be treated to comply with Missouri regulatory limits for irrigation (Appendix H, Table H.12). Water disposed of in this manner would not directly enter drinking water supplies, but the potential exists for other adverse impacts resulting from drifting spray, surface water runoff, and/or groundwater contamination. These potential impacts could be controlled by appropriate choice of such factors as the size and location of the irrigated area and the rate of water discharge by spray irrigation. Spray irrigation could increase the length of the project because of the need to limit the rate at which water is disposed of.

2.4.2 Raffinate Sludge Stabilization and Disposal

2.4.2.1 Stabilization Methods

Stabilization by mixing in a stabilizing agent has the advantage that it is relatively inexpensive; however, both the volume and weight of the stabilized sludge are larger than for the original wet sludge. The method considered in this EIS of using a mixture of fly ash and portland cement as a stabilizer would increase the volume by 37% and the weight by 69%. This has the disadvantage of increasing the effort and cost required to place the wastes in a disposal cell. Another potential disadvantage of using a stabilizing agent is that contaminants may be present in sufficient concentrations in the stabilizer to result in increased concentrations of contaminants in the stabilized sludge compared to those in the original wet sludge (Section 4.1.2.4). This problem could be minimized or even avoided by careful choice of the stabilizing agents.

Stabilization of the excavated raffinate sludge by heat drying has the advantage of a projected 70% volume reduction (Bechtel Natl. 1984), which would result in a lower disposal volume and fewer accidents. This would be especially significant for the alternatives requiring off-site disposal because transportation costs and accidents would be expected to be much lower than those for transporting the original wet sludge for the no-treatment option. Disadvantages include the large energy requirements and costs associated with the heat-drying operation (Appendix E, Section E.4.1.1). One conceptual design would consume 16,200 m^3 (102,000 bbl) of fuel oil, would require 300 kW of electrical power to operate the process, and would take 4 years to carry out (Bechtel Natl. 1984).

Processing the raffinate sludge to remove uranium under Alternative 3c has the advantage that the uranium removed from the raffinate sludge could be used for other purposes. A disadvantage of this alternative is that it requires transport of the raffinate sludge to the uranium processing site with the attendant monetary costs, accidents, and radiological doses. The wastes remaining after the uranium was removed would be disposed of along with other radioactive wastes from operations at that facility and would not require any special treatment. The cost of processing the raffinate wastes would be reduced by the commercial value of the recovered uranium. The dried raffinate sludge may not be accepted at any of the existing uranium processing facilities because of such factors as high pH, higher specific activity, and possible classification of the raffinate sludge as out-of-state waste (Bechtel Natl. 1985).

2.4.2.2 Disposal

Above-grade disposal of the Weldon Spring wastes (Alternative 2b) has the advantage that a minimal amount of soil must be removed to construct a disposal cell and the total thickness of soil from surface grade to the water table is thus available to retard the movement of percolating waste leachate. As a result, the time required for waste leachate to reach the groundwater would be larger than for below-grade disposal at the same site. Also, costs and accidents associated with excavation of a disposal cell would be avoided. One disadvantage is the fact that, because the top of the cell would be elevated above the surrounding grade, erosion of the cover by wind and water and sediment loading of surface runoff would be greater. These impacts could be minimized (but not eliminated) by design and active maintenance and monitoring of the disposal cell. Another disadvantage is that cell walls or berms would have to be built, which would result in increased earth-working activities (e.g., excavation, hauling, compacting) with resulting increases in costs and accidents.

Below-grade disposal of the Weldon Spring wastes (Alternative 3a) has the advantage that, because the top of the covered waste pile would be only slightly above or at grade level, erosion of the cover and sediment loading of surface runoff from the cover would be minimized. However, excavation of a disposal cell would reduce the distance between the buried wastes and the groundwater table and thus reduce the time required for waste leachate to reach the groundwater. Another disadvantage would be the costs and accidents associated with excavation of the disposal cell.

Disposal of the wastes partially below grade (Alternatives 1, 2a, 3b, and 3c) has some advantages and disadvantages of both above-grade and below-grade disposal. Because some excavation of a disposal cell would be needed, the time required for leachate to reach the groundwater table would be more for partially below-grade disposal than for completely below-grade disposal and less than that for above-grade disposal. Some cell wall construction would have to be done, but the amount needed would be less than that needed for above-grade disposal. Wind and water erosion and sediment loading of surface runoff would be greater for partially below-grade disposal than for completely below-grade disposal but less than would occur for above-grade disposal.

2.4.3 Disposal Cell Components

2.4.3.1 Leachate Monitoring and Collection System

Use of a leachate monitoring system has the advantage that monitoring of leachate in the cell bottom allows the timely detection of developing problems. These include such things as cracks in the cover, which could allow the infiltration of rainwater or snowmelt and result in buildup of leachate in the cell. The disadvantages are related to the fact that such a system requires the presence of entry holes or pipes into the bottom of the cell through which the monitoring is done, and such holes or pipes provide pathways for leachate escape and radon gas emission. Over the long term, it is unlikely that such a system would function for 1,000 years, and water could seep into the cell along or through the monitoring pipes or holes that breach the containment (Bechtel Natl. 1984).

Use of a leachate collection system has similar advantages to the leachate monitoring system. It would also provide for the collection of leachate (in sumps for example) and its possible removal from the system. This would minimize percolation into soils underlying the cell and prevent any possible breaching of cell containment due to leachate buildup. The disadvantages of the monitoring system also apply to the collection system. The possible emission of radon gas and infiltration of water along or through pipes that breach the containment cell must be considered. (Water

infiltration is not a problem for systems designed with gravity flow to sumps outside the cell.) Also, differential settling in parts of the floor could allow local leachate ponding to occur without the system being able to detect or remove the accumulated leachate.

2.4.3.2 Cell Cover

The major option addressed in the cell cover is the use of a lead sheet. This was considered because it was brought up during the scoping process. Use of a lead sheet in the cover (Alternative 2b) would result in further prevents water infiltration and radon gas emission. The additional shielding provided by the lead sheet against gamma radiation would be a negligible advantage because the thick cover would provide sufficient shielding to result in a negligible dose even to persons who would be carrying out maintenance activities on the cover (Section 4.2.2.3). Disadvantages include the possibility of tears resulting from differential settling. Also, corrosion of the lead sheet could result in lead buildup in sediments in the ditch surrounding the waste cell and in lead contamination of surface runoff (Section 4.1.3.2). Other disadvantages are the fact that the construction of such a large lead sheet has never been done before (Jones 1986) and the potential would exist for exposure of workers to lead fumes and dust (even though good housekeeping techniques would be used).

2.4.4 Transportation Modes

Two modes of transportation, truck and rail, are considered in this EIS. Transport of the wastes by barge has been analyzed, but it was not considered feasible because it is quite expensive and transfers of the wastes are required enroute (Bechtel Natl. 1984--Appendix G).

2.4.4.1 Transportation by Truck

An advantage of truck transportation is that it is quite flexible. Trucks can load the wastes on-site, transport them to the disposal site, and unload them at the disposal site with no transfers. Thus, exposure to contaminated dust and radiological doses and accidents associated with waste transfers would be avoided. Also, truck transport for a short distance (Alternative 3b) would likely be more economical. Total one-way distances of 3,000 km (1,900 mi) to the Hanford site (Alternative 3a) and 1,800 km (1,100 mi) to the uranium processing site (Alternative 3c) are slightly shorter for truck travel than the distances for rail transport to these sites (Appendix F). However, truck transport has the disadvantage that the occupational dose to workers would be higher than for rail transport (Appendix F, Section F.4.3) because the truck drivers would be closer to the

wastes in the truck. The total amount of material spilled by accidents is estimated to be about the same for truck transport as for rail transport.

2.4.4.2 Transportation by Rail

Rail transport has the advantages that it is more economical than truck transport for long distances and the radiological doses to workers would be lower. Rail transport has the disadvantage that it is less flexible than truck transport. Wastes must be off-loaded at a suitable siding and then trucked to the disposal site for placement in the disposal cell. Facilities for loading and unloading the wastes must be built, which adds to the expense of rail transport.

2.5 REFERENCES*

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*In this reference list, the term "personal communication" is used to indicate either a telephone conversation or a face-to-face conversation.

3. AFFECTED ENVIRONMENT

3.1 WELDON SPRING

3.1.1 Topography, Geology, Soils, Seismology, and Mineral Resources

The Weldon Spring site -- consisting of the raffinate pits, quarry, chemical plant, and vicinity properties -- is located in Townships 45N and 46N, Ranges 2E and 3E, St. Charles County, Missouri. The site is located near the boundary of the Central Lowlands Physiographic Province and Ozark Plateaus Physiographic Province (Figure 3.1).

The raffinate pits and chemical plant are located just north of a ridge dividing the Missouri and Mississippi river valleys. Gently rolling topography characterizes the drainage systems to the north and west of the site. The terrain to the south and east is rugged, ravined, and heavily wooded (Figure 3.2). The elevation of the raffinate pits area ranges from about 202 m (664 ft) above mean sea level (MSL) at the top of the dikes surrounding the raffinate pits to about 191 m (628 ft) MSL near the bottom of Pit 4.

The quarry area is approximately 7 km (4 mi) southwest of the chemical plant near the Missouri River floodplain. With the exception of the Missouri River floodplain to the south of the quarry, the surrounding topography is rugged, heavily wooded, and characterized by deeply dissected hills and deep ravines (Pennak et al. 1975) (Figure 3.2) Ground surface elevations range from about 140 m (450 ft) MSL at the Missouri River floodplain to 170 m (550 ft) MSL at the quarry. The quarry floor is at an elevation of about 140 m (450 ft) MSL.

The raffinate pits, chemical plant, and vicinity properties are underlain by Quaternary age unconsolidated sediments and Paleozoic age bedrock formations. Geological investigations conducted in 1982 through 1983 (Bechtel Natl. 1984b) determined that these areas are underlain by six unconsolidated sedimentary units (Table 3.1). The predominant soil type belongs to the Harvest-Urban land complex group and has a low permeability (U.S. Soil Conserv. Serv. 1982; Bechtel Natl. 1984a). More than 20 m (64 ft) of alluvial deposits blanket the bedrock in the Missouri River valley (Allen and Ward 1977).

Underlying the unconsolidated deposits is a thick sequence of limestones and sandstone bedrock formations of Paleozoic age (Table 3.2). The uppermost bedrock formation, the Burlington-Keokuk limestone, is fractured and contains many immature karst features such as solution-enlarged cavities and voids that developed along bedding planes and northeast-trending joints (Bechtel Natl.

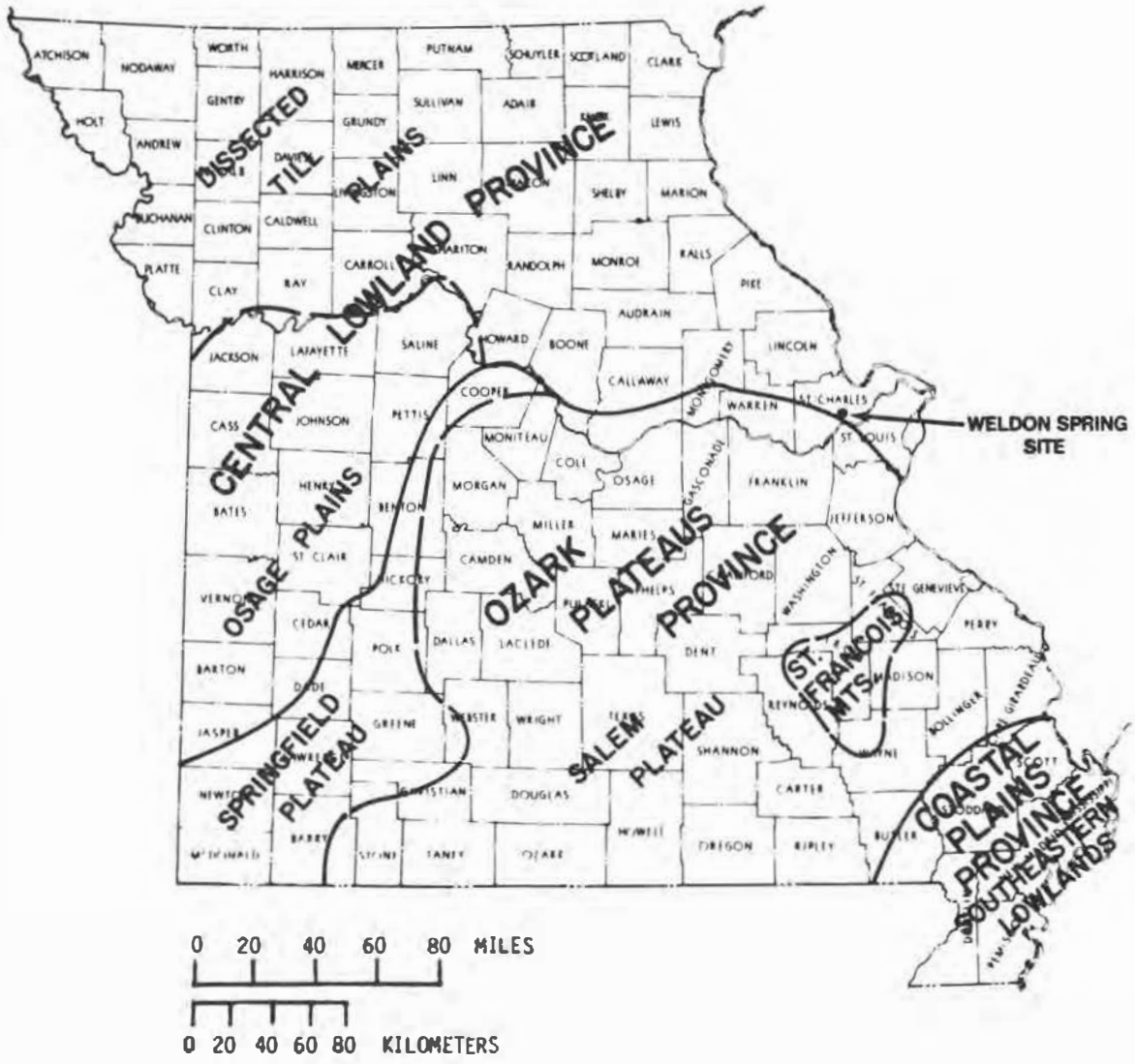

Figure 3.1. Physiographic Provinces of Missouri. Source: Modified from U.S. Geological Survey et al. (1967).

1984b). Other karst features such as springs, losing streams, caves, and sinkholes are found in St. Charles County; as many as 14 caves are known to exist in the county (Duley 1983). Limestone bedrock exposed on the quarry walls and on the steep bluffs along the Missouri River is predominantly Ordovician limestone, shale, dolomite, and sandstone (Figures 3.3, 3.4, and 3.5). The uppermost stratum at the quarry area is Kimmswick limestone, and the quarry floor is Decorah shale.

Primarily as a result of Paleozoic structural activity, the bedrock formations of the region have been formed into arches, basins, and other

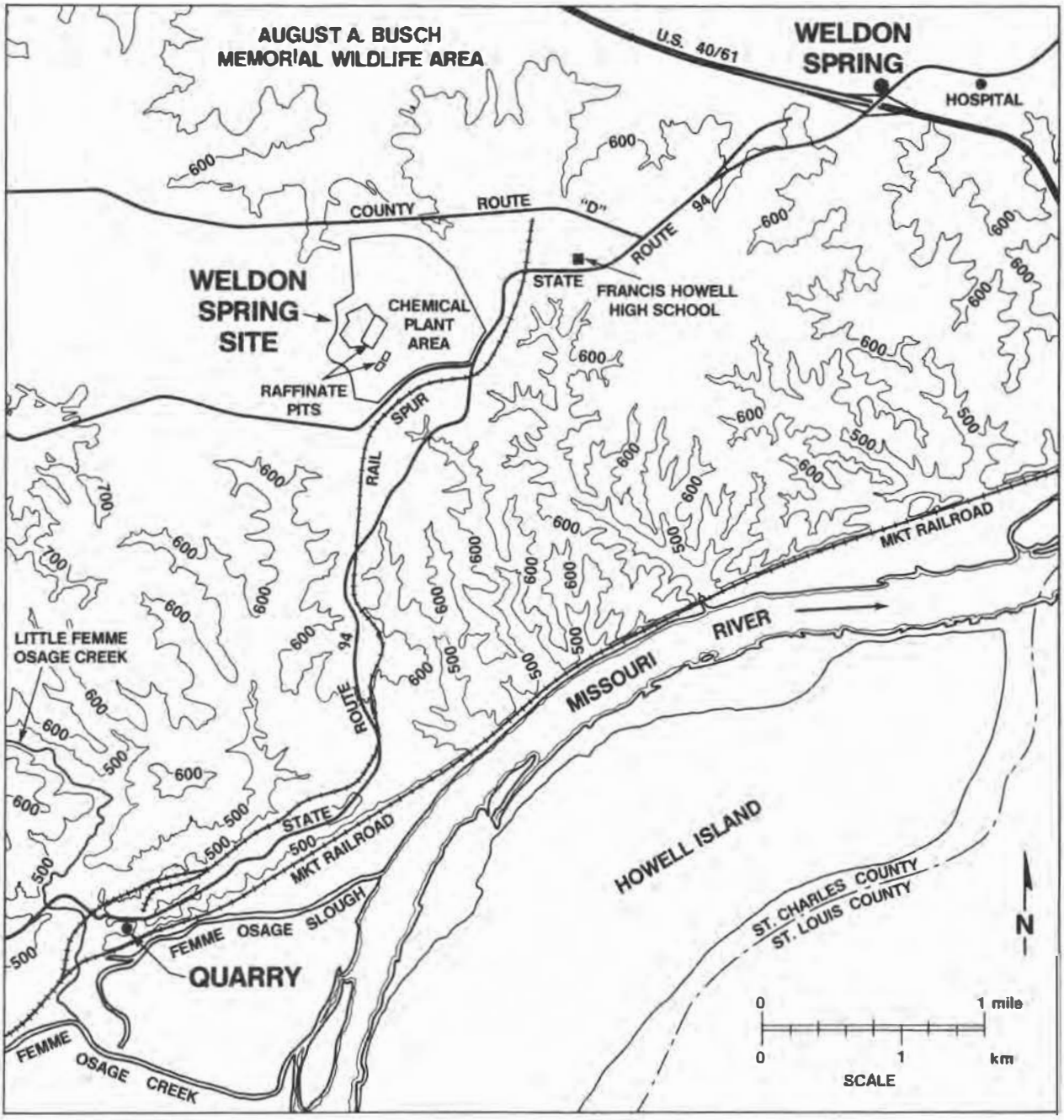

Figure 3.2. Topographic Map of the Weldon Spring Area (elevations are feet above MSL). Conversion Factor: To convert feet to meters, multiply by 0.3048. Source: Base map adapted from 7.5-minute U.S. Geological Survey maps for the Weldon Spring and Defiance quadrangles.

Table 3.1. Unconsolidated Overburden Units in the Area of the Raffinate Pits, Chemical Plant, and Vicinity Properties

Unit	Characteristics	Thickness (m)
Topsoil	Sandy clay, blackish-brown, organic-rich	0.15-1.5
Modified loess	Clayey silt, mottled gray-dark yellowish-orange, becomes dense and plastic with depth, manganese stained. The loess is modified in the sense that it contains higher than average clay content for loess	0.3-4.6
Clay (Ferrelview Formation)	Clay, mottled gray-dark yellowish-orange, plastic, dense, manganese stained, contains weathered iron nodules.	0-4.6
Clay till	Clay, yellowish-brown, plastic, dense, manganese stained, blocky fractures, contains sand- to pebble-sized quartz, granitic rock, and chert dispersed throughout the clay matrix.	0.3-11.3
Basal till	Sandy, clayey silt, yellowish-brown, abundant in broken chert nodules, loosely bound by matrix.	0.3-2.4
Cherty clay	Multicolored brown, red, orange, and yellow, very dense, clay matrix with tightly bound abundant granular- to cobble-sized chert.	0-4.6

Source: Bechtel National (1984b).

structures (Eardley 1962). The Weldon Spring site is located on the gently dipping east flank of the northwest-trending House Springs-Eureka anticline (Bechtel Natl. 1984a).

The Weldon Spring site lies within the tectonically quiet, central stable seismic region. A few scattered seismic events have been recorded throughout Missouri and Illinois (Zoback and Zoback 1981; Johnston 1982; Bechtel Natl. 1984b), but these have been of small magnitude and do not define a currently active fault or faults. No evidence has been found of tectonic surface ruptures related to historic earthquakes in the area. The New Madrid seismic zone at the northern end of the Mississippi Embayment, located about 260 km (160 mi) south of the site, is the nearest zone of major seismic activity (Bechtel Natl. 1983a). Isoseismal maps compiled for the 1811, 1843, 1895, and

Table 3.2. Generalized Stratigraphic Column for the Weldon Spring Region

System	Formation	Thickness (m)	Description
Cenozoic Quaternary	Overburden	5-15	See Table 3.1.
Paleozoic Mississippian	Burlington-Keokuk Limestone	45-61	Massive, bedded, cherty light brown to light gray limestone; contains limestone clay-filled pockets and several solution voids along fractures and bedding planes.
	Fern Glen Limestone	23	Yellowish-brown, finely crystalline, cherty limestone. Not distinguished in the field, but has been identified in this area.
	Chouteau Limestone	8	Finely crystalline, massive, gray dolomitic limestone.
Devonian	Bushberg Sandstone	1-3	Fine- to medium-grained, green, white or reddish-brown sandstone. Often contains sand concretions. Its age is unassigned Devonian-Mississippian.
Ordovician	Kimmswick Limestone	30	Medium- to coarse-grained, fossiliferous crystalline, white to light gray limestone; contains large voids due to solution effects along fractures and bedding planes.
	Decorah Shale	9	Green or brown shales with numerous, thin, interbedded limestone layers in lower part that grade upward into a medium to thinly bedded limestone containing thin shale partings.
	Plattin Limestone	30-61	Gray to dark gray, fine-to medium-grained, thin-bedded fossiliferous limestone.
	Joachim Dolomite	30	Yellowish-brown to brown, contains a few thin shales, thin to massively bedded dolomite, and some sand grains near the base.
	St. Peter Sandstone	24-60	Yellowish white to white, fine- to medium-grained, friable, massive to cross-bedded quartz sandstone.

Sources: Krummel (1956); Koenig (1961); Miller et al. (1977); Satterfield (1977); Bechtel National (1984b).

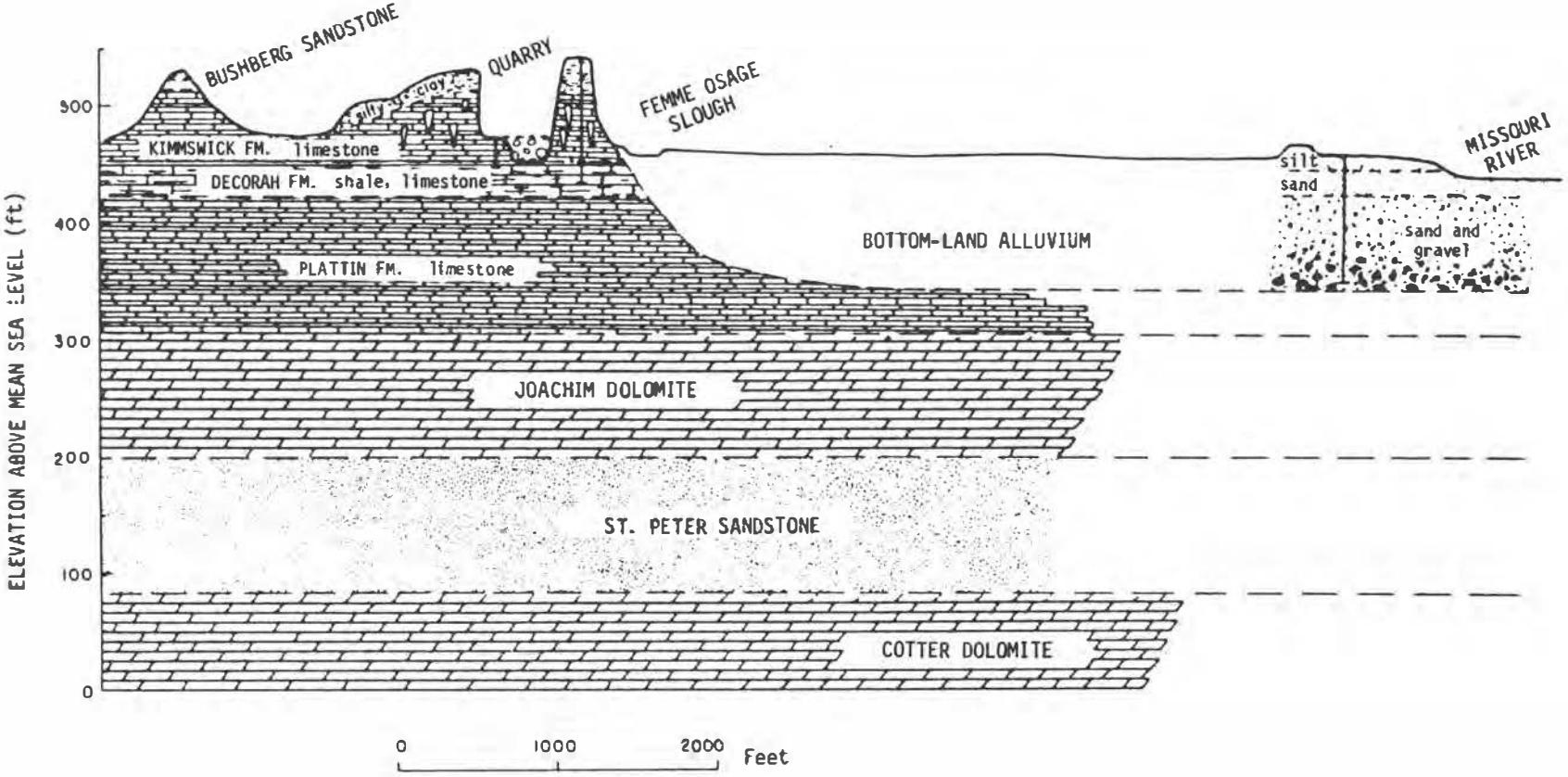

Figure 3.3. Idealized North-South Cross Section through the Weldon Spring Quarry Area. Conversion Factor: To convert feet to meters, multiply by 0.3048. Source: Modified from Berkeley Geosciences Associates (1984).

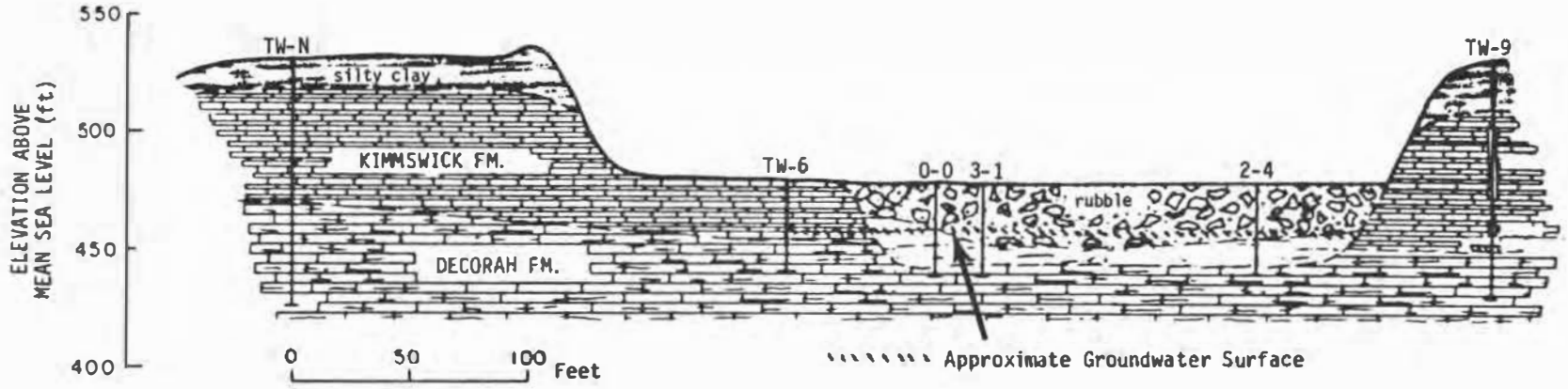

Figure 3.4. North-South Profile through the Weldon Spring Quarry Area. Conversion Factor: To convert feet to meters, multiply by 0.3048. Source: Modified from Berkeley Geosciences Associates (1984).

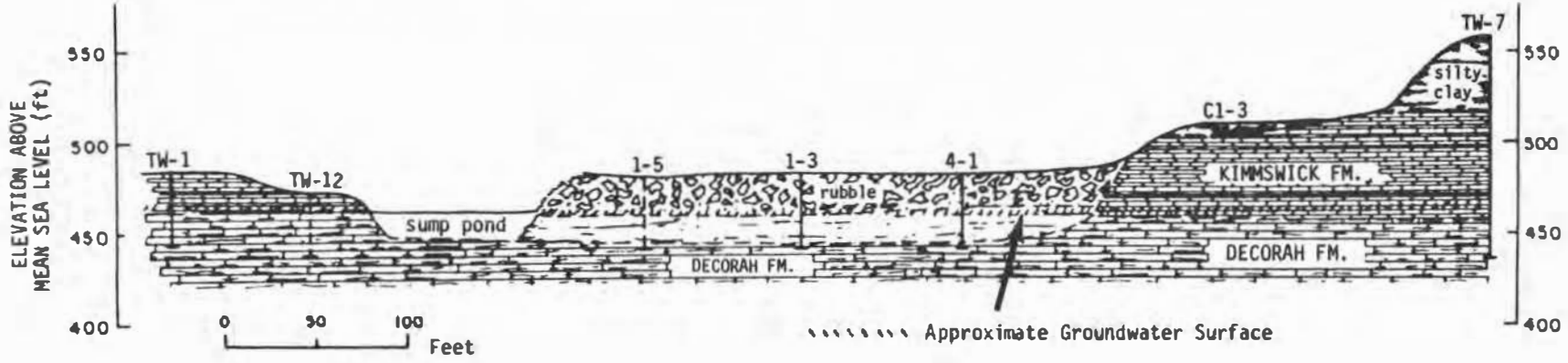

Figure 3.5. East-West Profile through the Weldon Spring Quarry Area. Conversion Factor: To convert feet to meters, multiply by 0.3048. Source: Modified from Berkeley Geosciences Associates (1984).

1968 earthquakes by Hopper et al. (1983) show that the Weldon Spring area has experienced Modified Mercalli earthquake intensities ranging from V to VII.

The mineral resources of St. Charles County, Missouri, include limestone, sand, gravel, clay, shale, and coal. Of these, only limestone and industrial sand have economic potential, and they are extensively mined in surrounding areas (U.S. Geol. Survey et al. 1967). With the exception of quarried limestone, no mineral resources have been identified within the Weldon Spring site.

3.1.2 Hydrology, Water Use, and Water Quality

3.1.2.1 Surface Water

The Mississippi River is north and the Missouri River south of the Weldon Spring site (Figure 1.1). All runoff from land surfaces in the area eventually reaches the Mississippi or Missouri River. The combined flow of these two rivers at St. Louis averages about 5,100 m^3/s (180,000 cfs) (Waite et al. 1985).

The raffinate pits, chemical plant, and some of the vicinity properties are located on the Mississippi River (northern) side of the drainage divide in the headwater of Schote Creek (Figure 3.6). Surface runoff from the area flows into nearby intermittent streams or Ash Pond on the chemical plant property (Figure 3.7). Discharges from the streams and Ash Pond combine near County Route "D" and flow northward to Lake 35, an impoundment on Schote Creek in the Busch Wildlife Area just southwest of U.S. Routes 40 and 61 (Figures 3.6 and 3.7). Schote Creek enters Dardenne Creek about 6 km (3.7 mi) northeast of the raffinate pits and chemical plant areas and has a drainage area of about 13 km^2 (5 mi^2). Water in Dardenne Creek eventually reaches the Mississippi River near Seeburger, Missouri, about 32 km (20 mi) northwest of St. Louis.

Rainwater and snowmelt runoff and percolation enter various drains at the chemical plant area. The drains collect the water into the chemical plant process sewer, which exits on the southern slope of the drainage divide. Effluent from this exit flows to the Missouri River through a drainage ditch (Figure 3.7).

Preliminary estimates of flood peak discharges have been prepared by the U.S. Army Corps of Engineers, St. Louis District, for Schote Creek at several reaches and for different recurrence intervals. At the site, the 100-year and 500-year flood peak discharges at the main stem of Schote Creek are expected

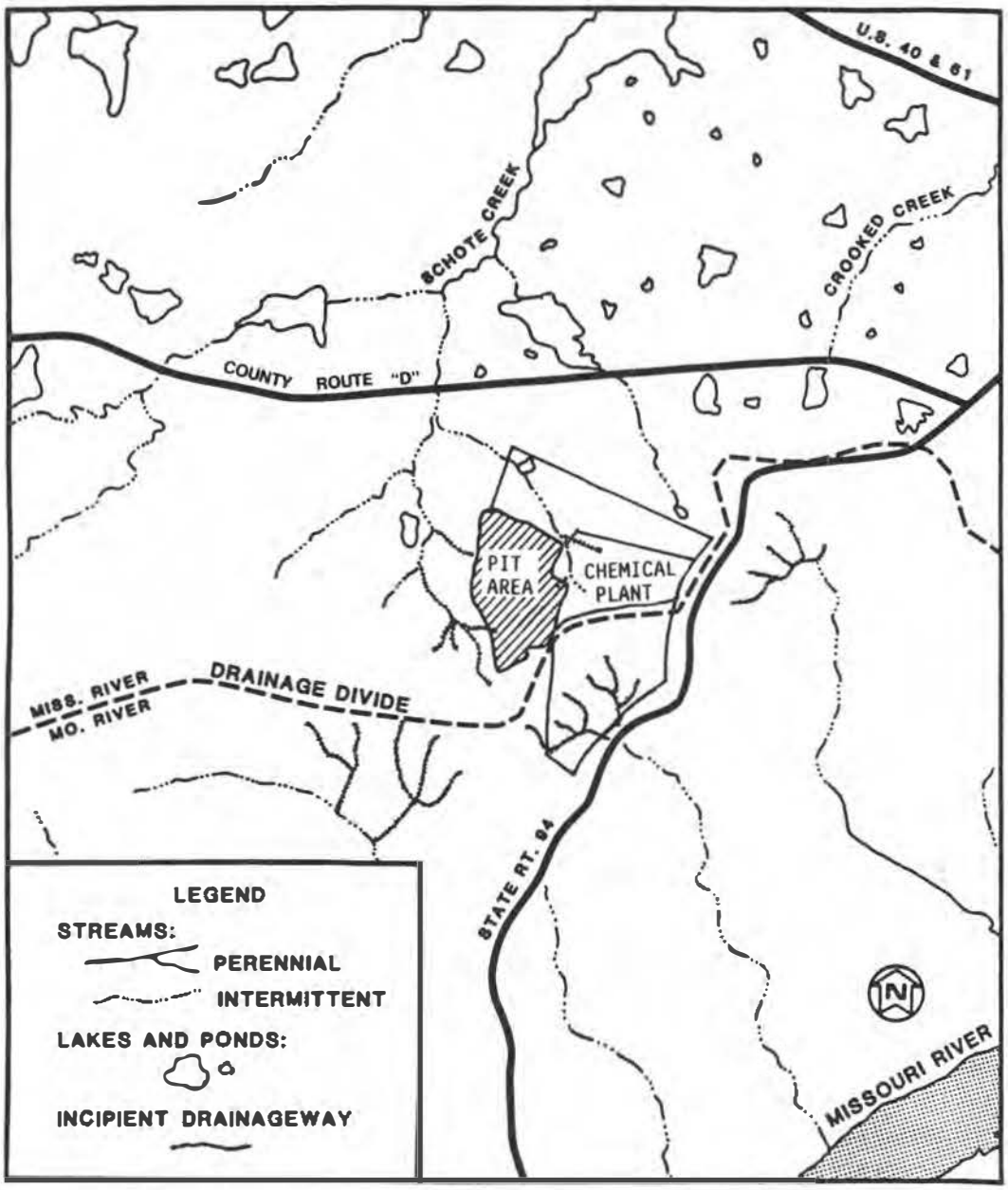

Figure 3.6. Surface Hydrologic Features near the Raffinate Pits Area. Source: Modified from Bechtel National (1984b).

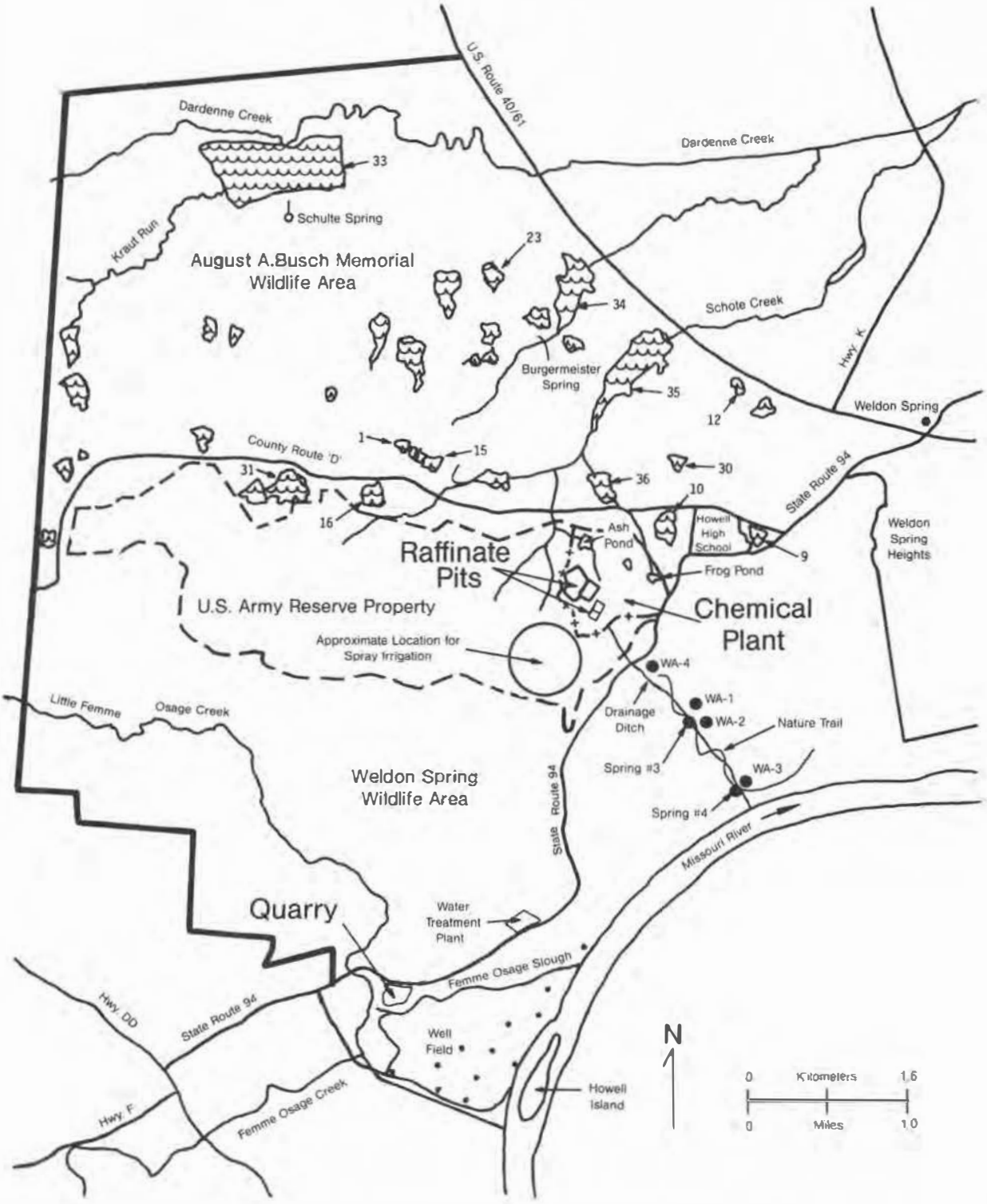

Figure 3.7. Map of the Weldon Spring Site and Vicinity. Source: Modified from deRoos (1984) and Stevens (1984).

to be about 60 and 76 m^3/s (2,100 and 2,700 cfs) (Corbin 1984). The 500-year flood elevation near the raffinate pits area would be about 160 m (530 ft) MSL. Because the average ground elevation is about 200 m (650 ft) MSL, the raffinate pits area would not be affected by either a 100-year or a 500-year flood occurring in the main stem of Schote Creek.

The quarry area and some of the vicinity properties are located on the Missouri River (southern) side of the drainage divide (Figures 3.6 and 3.7). Surface streams in the vicinity of the quarry area include Femme Osage Creek, Little Femme Osage Creek, an unnamed tributary to Little Femme Osage Creek, and Femme Osage Slough (Figure 3.7). Femme Osage Slough is a 2.4-km (1.5-mi) reach of Femme Osage Creek that was dammed at both ends in 1960 by the University of Missouri (Bechtel Natl. 1984a).

The Missouri River bottom at the quarry (river mile 49 from the confluence with the Mississippi River) is at an elevation of about 129 m (422 ft) MSL (U.S. Army Corps Eng. 1980). Although the floodplain area below the quarry is partially behind a levee, the area floods every 3 to 5 years to a depth of about 1 m (3-4 ft) and takes 1 to 2 months to dry up (it is drained by a 41-cm [16-in.] diameter pipe through the levee) (Hovatter 1986a).

Water in the Missouri River near St. Charles is a calcium bicarbonate type and contains significant amounts of calcium, magnesium, sodium, bicarbonate, and sulfate. The water is hard, and turbidity is relatively high (Miller 1977). Water impoundments on the Missouri main stem and tributaries have resulted in a significant decrease in turbidity over the past 20 years. Water in the Mississippi River upstream from the mouth of the Missouri River is also a calcium bicarbonate type and contains significant amounts of magnesium and sulfate in the dissolved solids. Turbidity upstream from the Missouri River is relatively low, and the water is very hard. Tributary streams in St. Charles County represent a small part of the total volume of surface water available to the area. The tributary streams receive surface runoff as well as treated and potentially untreated domestic wastewater effluents. The public water supply comes mainly from the alluvial aquifers along the Mississippi and Missouri rivers.

3.1.2.2 Groundwater

A large amount of water is available from the alluvium and underlying bedrock in the St. Charles County area (Miller 1977). Water occurs in the bedrock along fractures and bedding planes, in solution openings in the limestone and dolomite, and in voids between the grains in sandstone. In the alluvium, water can be found in the openings between the individual sand and

gravel particles. The availability of water from various units depends on the degree of weathering of the rocks and the connections with surface-water sources.

The major alluvial aquifers in the St. Charles County area are the sands and gravels in the basal part of the alluvium that underlies the floodplains of the Mississippi and Missouri rivers. The value of the alluvium as a source of water depends on its thickness and on the size and sorting of the materials. The alluvium in the St. Charles County area ranges in thickness from 27 to 45 m (89 to 150 ft) along the Mississippi River and from 8 to 35 m (27 to 120 ft) along the Missouri River (Miller 1977). The alluvium along other streams and rivers in this area ranges in thickness from 1.2 to 27 m (4 to 87 ft).

The bedrock units in the area have been classified into five aquifer groups based on similar lithologic characteristics, geographic distribution, and overall similarity of water characteristics (Miller 1977). The distribution of these aquifer groups is shown in Figure 3.8. The major bedrock aquifers in the region are St. Peter sandstone and underlying sandstone and dolomite formations more than 210 m (700 ft) below the surface (Table 3.2). The typical yield of a well drawing from these aquifers is about 0.6 to 9 L/s (10 to 140 gpm) (Miller 1977).

In the St. Charles County area, groundwater is the main source of public water supplies. The cities and public water-supply districts served by a central water supply are listed in Table 3.3. Groundwater in the area also provides a source of water for rural use and irrigation. Water for irrigation is withdrawn from wells in the Mississippi River alluvium; the amount used varies with rainfall.

Groundwater users in the area also draw water from the Bushberg sandstone and the Burlington-Keokuk limestone. These aquifers yield small amounts of water to wells, about 0.3 to 3 L/s (5 to 50 gpm). The Bushberg sandstone is approximately 100 m (320 ft) below ground surface. The water in this formation is naturally high in nitrates and chlorides and is therefore not used extensively. The Burlington-Keokuk limestone aquifer is also not generally used as a water supply because of its limited yields and poor water quality. Based on a well survey performed by the Missouri Geological Survey for the St. Charles Planning and Zoning Commission (published in 1977), the Burlington-Keokuk aquifer in St. Charles County is not used as a potable water supply (Bechtel Natl. 1984a).

The chemical quality of groundwater in the Weldon Spring area is quite variable. It ranges in dissolved-solids content from about 120 to 18,000 mg/L, and it varies from a calcium-magnesium bicarbonate (at low

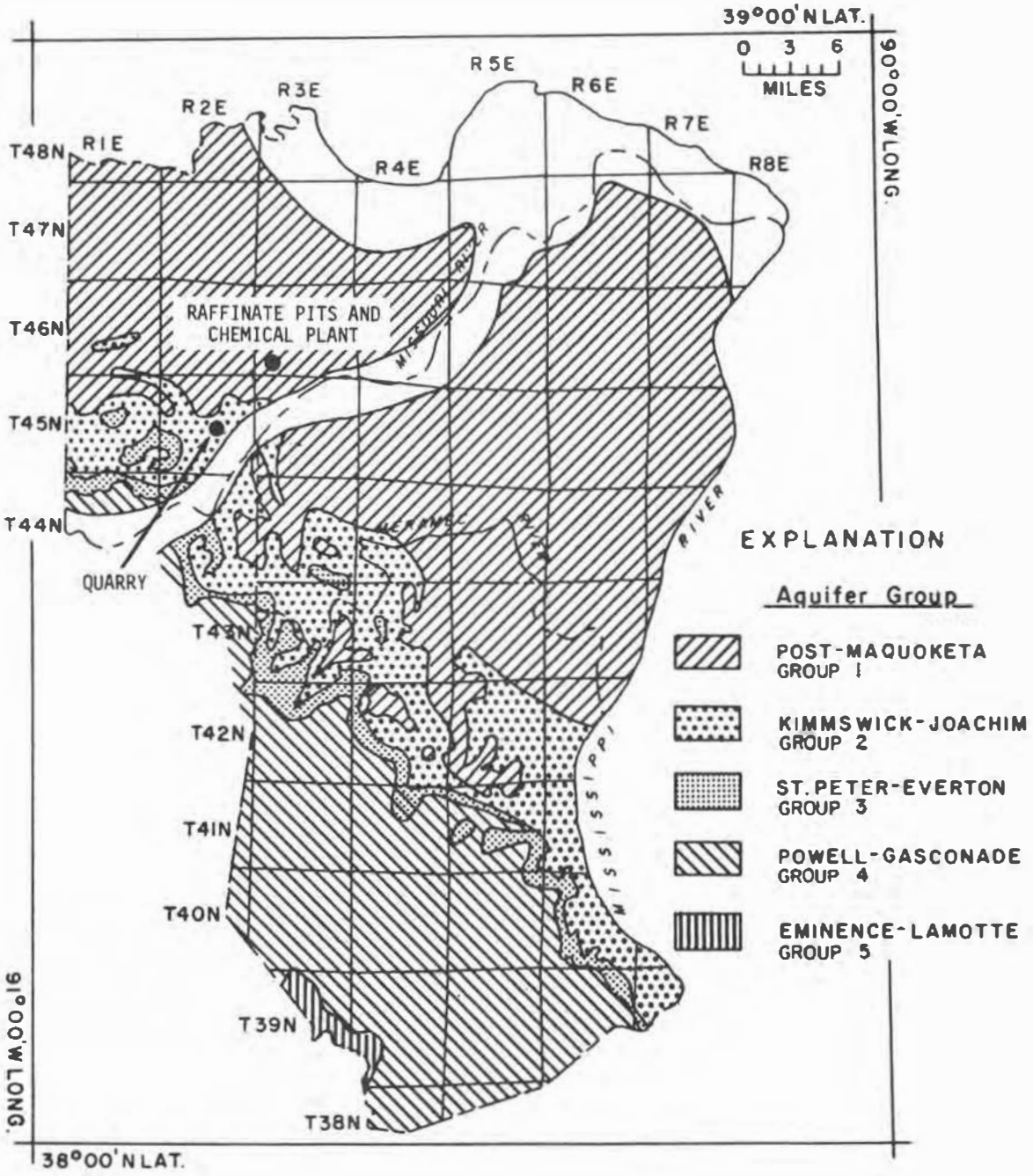

Figure 3.8. Geologic Map Showing Areal Distribution of Aquifer Groups. Conversion Factor: To convert miles to kilometers, multiply by 1.609. Source: Modified from Miller (1977).

Table 3.3. Water-Supply Facilities in St. Charles County

Municipality or Operating Agency	Source of Supply	Average Daily Consumption (10^6 gal/d)
Portage Des Sioux	1 alluvial well	0.022
St. Peters	5 deep wells	1.500
O'Fallon	4 wells	0.972
Wentzville	4 deep wells	0.825
St. Charles PWSD No. 1	1 deep well	0.045
St. Charles County Water Division:		
Elm Point plant	4 shallow wells	4.000
Missouri River plant	Missouri River	0.400
Weldon Spring[a]	4 alluvial wells	2.728
St. Charles PWSD No. 2, East	4 alluvial wells	0.051
St. Charles PWSD No. 2, North	4 deep wells	0.677
St. Charles PWSD No. 2, West	1 deep well	0.014

[a]Referred to as the St. Charles County well field in this EIS.

Conversion Factor: To convert gal/d to m^3/d, multiply by 3.785×10^{-3}.

Source: Missouri Department of Natural Resources (1982).

concentrations of dissolved solids) to a sodium chloride, sodium sulfate, or sodium bicarbonate type (high concentrations of dissolved solids). The principal factors affecting the quality of groundwater in the bedrock are (a) the chemistry and permeability of the rock units, (b) the length of time the water has been in the aquifer, and (c) the distance the water has moved from the recharge area.

Water from alluvial deposits along the river valleys has fairly uniform chemical characteristics, except that it varies widely in dissolved-solids content. The water generally is a calcium-magnesium bicarbonate type and, locally, may contain significant quantities of sulfate. Iron and manganese concentrations are usually high, and the water is very hard (Miller 1977).

Raffinate Pits Area. Groundwater investigations near the raffinate pits and chemical plant areas have been conducted since the mid-1940s by the U.S. Geological Survey (Fishel and Williams 1944). These investigations have determined that a groundwater divide exists south of the raffinate pits area and that the groundwater table in the vicinity of the raffinate pits area

generally reflects the land surface topography (Roberts and Theis 1951). Groundwater flow has recently been reported to be in a northerly direction, with an average gradient of about 0.0095 (Bechtel Natl. 1984b). This gradient is the average value over the raffinate pits area and is used for analysis in this EIS; however, local variations do occur. Seasonal variations in water-level elevations have also been reported (Bechtel Natl. 1984b).

The groundwater table in the raffinate pits area is reported to be in Mississippian limestones, 18 m (60 ft) below the raffinate pits (Task Force 1967). [Surface elevations across the area range from 190 to 200 m (620 to 660 ft) MSL.] A recent study indicates that the depth to groundwater may be somewhat less; the groundwater elevation was measured at 177 to 187 m (580 to 617 ft) MSL (Bechtel Natl. 1984b), about 7.5 to 18 m (25 to 60 ft) below ground surface, and is estimated to be about 5.5 to 6.1 m (18 to 20 ft) below the lowest excavated location of Pit 4. A groundwater depth of 6 m (20 ft) beneath the bottom of the disposal cell is used in this EIS for analysis of potential groundwater contamination impacts for Alternatives 1, 2a, and 2b.

Groundwater in the vicinity of the raffinate pits may also occur under localized perched conditions in the various unconsolidated geologic units. Information on the character, size, and water-producing capability of such lenses is important in establishing the basis for predicting migration of pollutants located in the pits. It is also important in determining the best location for a disposal cell for Alternative 2, which involves disposal of the wastes at the raffinate pits area somewhere other than in the existing raffinate pits. More information on such lenses will be obtained by additional geohydrologic characterization studies to be carried to support detailed engineering design activities (Section 3.1.9).

The clays underlying the pits have low hydraulic conductivities, with laboratory-measured values ranging from 2.9×10^{-6} cm/s (1.1×10^{-6} in./s) (Task Force 1967) to 3.2×10^{-7} cm/s (1.3×10^{-7} in./s) (Natl. Lead Co. Ohio 1977) to 1×10^{-8} or 1×10^{-9} cm/s (4×10^{-9} or 4×10^{-10} in./s) (Lomenick 1982). A recent study indicated that the hydraulic conductivities of the overburden range from 1.6×10^{-9} cm/s (6.3×10^{-10} in./s) to 3×10^{-6} cm/s (1.2×10^{-6} in./s) and the moisture content ranges from 15 to 30% (Bechtel Natl. 1984b). The study also indicated that the Ferrelview Formation (see Section 3.1.1) generally fractures conchoidally and occasionally has a blocky fracture pattern. Where the blocky fractures occur, they continue down through the underlying clay till (Bechtel Natl. 1984b). The residuum underlying the clayey silt is more permeable than the clay materials, and the underlying Mississippian limestones have high conductivities associated with their solution cavities and fractures (Task Force 1967).

Water that is recharged to the groundwater moves downgradient in the direction of the slope of the potentiometric surface and is discharged by natural or artificial means. Discharge is accomplished by evaporation, plant transpiration, discharge by springs, seepage into streams, or pumpage from wells.

Missouri state geologists conducted a tracer study in 1983 in an attempt to determine the flow direction of lost drill water when drilling some boreholes through the weathered limestone at the raffinate pits area (Dean 1983; Bechtel Natl. 1984b). Dye placed in two of the boreholes was never detected in streams and springs within several kilometers of the site on either side of the groundwater divide (Figure 3.6) near State Route 94 (Dean 1983; Bechtel Natl. 1984b). In June 1984, dye was injected in the outfall flow just south of the outfall sewer discharge point located about 350 m (1,100 ft) south of the raffinate pits (drainage ditch in Figure 3.7). The results showed that surface water enters the subsurface on the U.S. Army Reserve Property and emerges in at least two springs (springs 3 and 4 in Figure 3.7) to the south (Dean 1984). In March 1985, dye was placed in the Ash Pond discharge, 27 m (90 ft) west of the DOE fence line where the entire flow goes underground (Dean 1985). Dye was detected in Burgermeister Spring, which suggests that a subsurface connection may exist between the stream drainage from Ash Pond and Burgermeister Spring.

A recent water balance study was performed to examine parameters contributing to the variation in water volumes in the raffinate pits (Bechtel Natl. 1986a). This study concluded that the water loss from these pits could be attributed to natural seepage into the clays lining the raffinate pits. The amount of seepage from these pits is typical of that expected from pits excavated into and lined with clay. Reported soil permeabilities for the underlying clays relate closely to the loss rate. Saturation of soils was detected under Pit 3 and adjacent to Pits 1, 2, and 4. However, water samples from test boreholes and trenches had only background levels of radioactivity. No leakage paths other than subsurface seepage were identified.

Quarry Area. Groundwater in the vicinity of the quarry occurs in the alluvium, in the limestone bedrock, and in deeper layers of the dolomite and sandstone (Figure 3.3). Groundwater in the upper layers of the limestone moves in a complex system of solution channels, joints, and fractures.

Several groundwater studies have recently been performed for the quarry area (Berkeley Geosci. Assoc. 1984; Layne Western 1986; Kleeschulte and Emmett 1986). Berkeley's study (1984) was conducted in the fractured limestone near the quarry as well as in the alluvium near Femme Osage Slough. The study in the fractured limestone included pump testing for transmissivity and

storativity, tracer testing for effective porosity, point dilution testing for natural groundwater velocity, and fracture mapping to identify potential conduits for fluid flow. The testing in the alluvium included similar types of tests as were performed in the limestone. Layne Western (1986) installed 16 monitoring wells in the alluvium between the Missouri River and the Missouri-Kansas-Texas (MKT) Railroad tracks and conducted an aquifer pumping test using County Well 8. Data on water levels were collected at the monitoring wells and analyzed to determine aquifer transmissivity, specific yield, and hydraulic conductivity. The U.S. Geological Survey's study (Kleeschulte and Emmett 1986) presents some preliminary results of their three-year groundwater investigation at the raffinate pits and quarry areas. For the quarry investigation, groundwater levels were measured in the county wells and in 13 monitoring wells around the quarry. No pumping test was performed in this study.

Berkeley's study measured the groundwater flow velocity in test wells between Femme Osage Slough and the Missouri River (Wells OBS-11 and OBS-16 in Figure 3.9); the results indicate no detectable flow in the alluvium near the slough. However, at a well farther away from the slough toward the Missouri River (OBS-19 in Figure 3.9), a finite groundwater velocity was measured in the alluvium (Berkeley Geosci. Assoc. 1984). Measurements at another well drilled into the underlying limestone (OBS-13 in Figure 3.9) also suggest a finite groundwater velocity (Berkeley Geosci. Assoc. 1984). These measurements on the four test wells indicate the magnitude of the average pore water velocity using the point dilution method (Berkeley Geosci. Assoc. 1984). Groundwater movement in the fractured limestone is complex, with flow around and under the quarry, and either under the slough into the limestone under the river alluvium or directly into the finer-grained alluvium between the slough and quarry. The tightness of the alluvium between the slough and quarry and the possible presence of a clay layer beneath the slough, which may extend down to bedrock, may be the reasons that groundwater flow in the alluvium near the slough was not detected (Berkeley Geosci. Assoc. 1984).

Groundwater elevations based on measurements made at several test wells drilled between the quarry and the Missouri River (Kleeschulte and Emmett 1986; Layne Western 1986) are shown in Figure 3.10. Groundwater flow in the limestone occurs primarily through the fractures (Berkeley Geosci. Assoc. 1984). The alluvium located between the quarry and Femme Osage Slough is relatively thin (3-8 m [10-26 ft]) and, based on pump tests, appears to be tight (transmissivities were not calculated due to the very slow rate of water-level recovery) (Berkeley Geosci. Assoc. 1984).

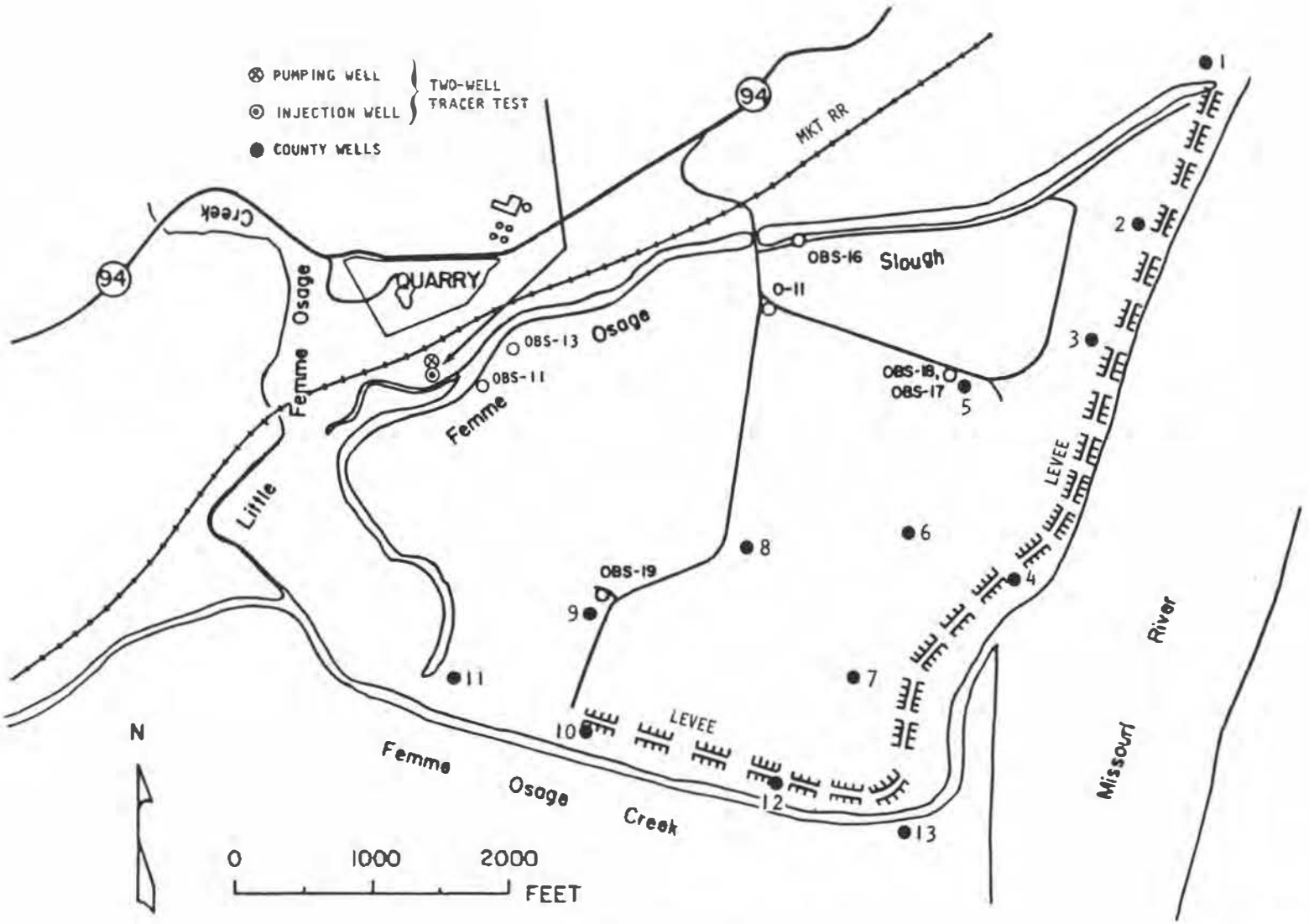

Figure 3.9. Approximate Locations of Some Test Wells and the County Wells in the Well Field Near the Quarry. Conversion Factor: To convert feet to meters, multiply by 0.3048. Source: Based on maps from Berkeley Geosciences Associates (1984), St. Charles Countians Against Hazardous Waste (1985), and U.S. Geological Survey (Defiance and Weldon Spring quadrangles).

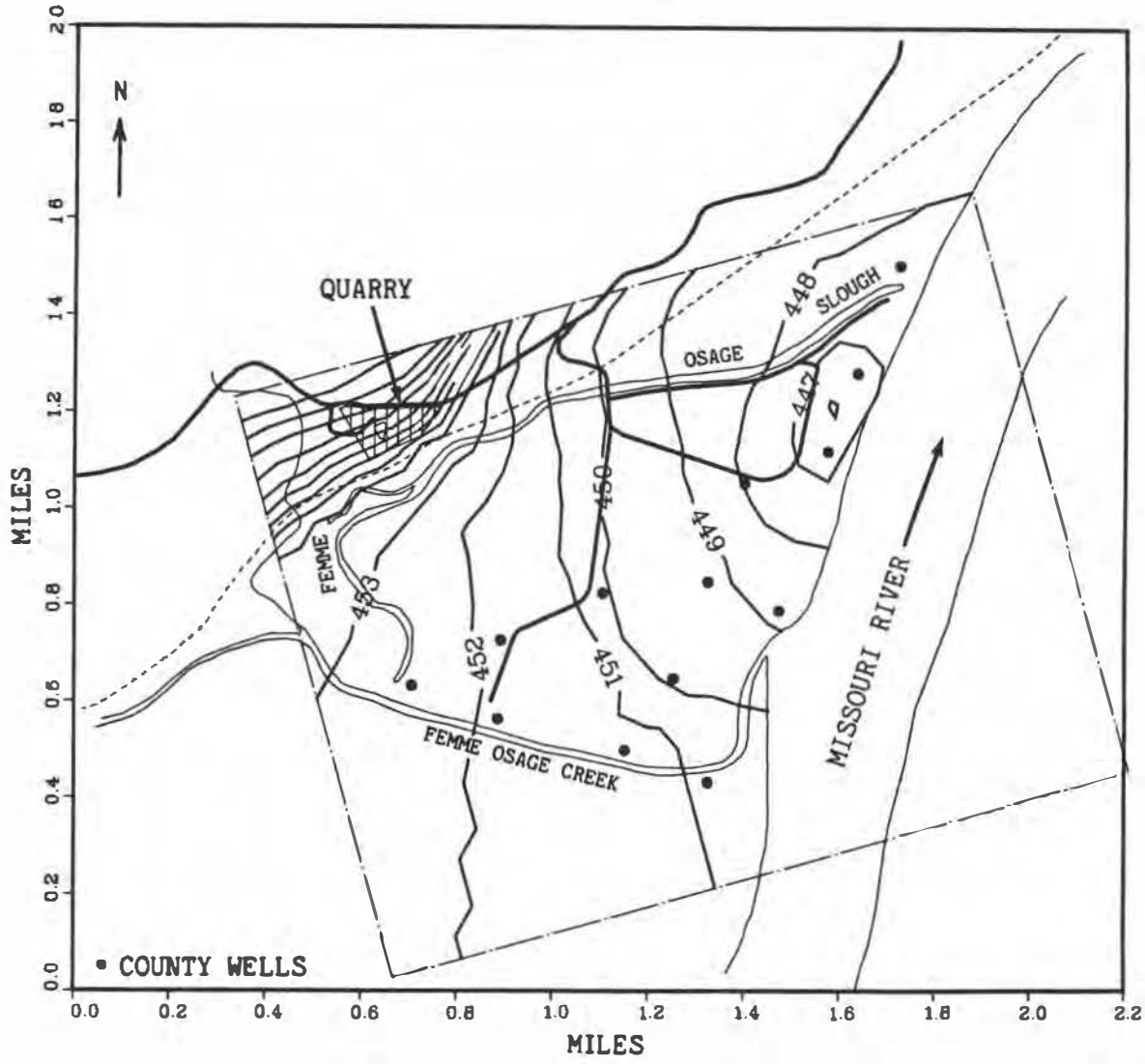

Figure 3.10. Measured Groundwater Elevations (feet above MSL) in the Well Field Area. Contour interval = 1 foot. Conversion Factors: To convert feet to meters, multiply by 0.3048; to convert miles to kilometers, multiply by 1.609.

The values of transmissivity and storativity in the fractured limestone were estimated by Berkeley Geoscience Associates (1984) based on pump tests and tracer tests. The estimated storativity ranges from 6.7×10^{-5} to 3.8×10^{-3} and averages 6.4×10^{-4}; estimated transmissivity ranges from 3.7×10^{-6} to 2.0×10^{-4} m^2/s (4.0×10^{-5} to 2.2×10^{-3} ft^2/s) and averages 4.3×10^{-5} m^2/s (4.6×10^{-4} ft^2/s). These results indicate that the matrix of limestone is relatively impermeable and that fractures of differing degrees of interconnection occur. The average effective porosity for the limestone is

0.0015 based on values obtained from a two-well tracer test that showed a double-peaked breakthrough curve (Berkeley Geosci. Assoc. 1984).

The transmissivity at various locations in the alluvium, estimated using pump tests (Layne Western 1986), ranges from 0.0011 to 0.066 m^2/s (0.012 to 0.71 ft^2/s) and averages 0.031 m^2/s (0.33 ft^2/s). The effective porosity of the alluvium, estimated using a two-well tracer test (Berkeley Geosci. Assoc. 1984), ranges from 0.27 to 0.32.

A major source of potable groundwater in the St. Charles County area is the county well field, which is located about 1.6 km (1 mi) southeast of the quarry and consists of 13 wells (Figures 3.9 and 3.10). Four wells (Wells 1, 2, 3, and 5) were in service until the middle of 1986. As of September 1986, Wells 3, 4, 6, and 7 were in service and Wells 1, 2, 5, 8, and 9 were either being repaired or were available for backup. It is not known when Wells 10 through 13 will be placed in service (Hovatter 1986b). Each well is drilled about 30 m (100 ft) deep in the alluvium to the underlying bedrock. Water is withdrawn from the bottom 12 m (40 ft) of the well through a 41-cm (16-in.) diameter screen. The space between the screen and the outer wall of the well (diameter about 1.5 m [5 ft]) is filled with gravel. Each active well is being pumped at the rate of 7.6 m^3/min (2,000 gpm) (Hovatter 1985). The approximate locations of the county and test wells are shown in Figure 3.9.

3.1.3 Climate and Meteorology

The climate in the area of the Weldon Spring site is continental in nature, with moderately cold winters and warm summers. The site's location in the middle latitudes results in alternating warm/cold, wet/dry air masses converging and passing eastwardly through the area on an almost daily basis (Bechtel Natl. 1986b).

Normal annual precipitation in the area is about 94 cm (37 in.), based on climatological data from St. Louis, Missouri. The heaviest rainfalls occur in spring and early summer. Summer rains are frequently in the form of thunderstorms, often with hail and high winds. Locally, rains can be very heavy, with 25 cm (10 in.) having been recorded in 24 hours (Ficker 1981). The three winter months are the driest, with an average of about 15 cm (6 in.) of precipitation; the spring months of April to June are normally the wettest, and normal total precipitation is nearly 30 cm (12 in.) (Ficker 1981).

The variability of the weather is shown by the record high and low temperatures for the state; the lowest recorded temperature was -40°C (-40°F) in 1905, and the highest was 48°C (118°F) in 1936 (Ficker 1981). The monthly

average temperature is 13°C (56°F), the average daily minimum temperature is 7°C (45°F), and the average daily maximum temperature is 19°C (66°F).

The prevailing winds are from the south and west, with an average speed of 19 km/h (12 mph). The winds in the area of the quarry are generally from the south during the summer and from the west-northwest during the winter and early spring. The mean wind velocity on a yearly basis is 15 km/h (9.5 mph) from the south, and the highest recorded wind was 97 km/h (60 mph) from the southeast. These data were used in calculations involving dispersion of particulates and radon gas in air at the Weldon Spring site.

The Weldon Spring site is located in the St. Louis Air Quality Control Region. The background concentration of airborne particulates is about 40 $\mu g/m^3$ based upon data collected at Queeny Park located about 23 km (14 mi) southeast of the site (Shissler 1986).

Tornadoes occur in Missouri most often in April and May. Tornadoes may occur in the St. Louis area once or twice per year, but they usually have a narrow path and often disintegrate after a few kilometers. In fact, during the most recent 40-year period of record for the St. Louis area, there have been only four tornadoes that produced extensive damage and loss of life (Sci. Appl. 1979).

3.1.4 Ecology

The Weldon Spring site is located within the Bluestem Prairie, Oak-Hickory Forest Mosaic (northern) subsection of the Prairie Parkland province. The Oak-Hickory Forest (northern) subsection also occurs within the Weldon Spring area (Bailey 1978; Galvin 1979). Much of the area surrounding the Weldon Spring site is state-owned wildlife areas containing secondary growth forest (August A. Busch Memorial Wildlife Area, Weldon Spring Wildlife Area, and Howell Island Wildlife Area). Nonforested areas occur over about 75% of St. Charles County and are largely used for crop production and pastures or are old-field habitat (Mo. Bot. Gard. 1975).

Habitat types within the vicinity of the Weldon Spring site include open fields and pastures, forests (upland, slope, and bottomland), and cultivated fields. These habitats have been characterized by the Missouri Botanical Garden (1975) and references cited therein. The upland forests are primarily oak and oak-hickory forests. Slope forests are similarly dominated by oak and hickory but also include sugar maple, American elm, and black walnut among the predominant species. Bottomland forests occur within the Missouri River floodplain and along stream and lake banks. Tree species can include willow, eastern cottonwood, silver maple, American elm, boxelder, red mulberry, pecan, oak (pin and bur), hackberry, and persimmon. Old-field habitat includes

species such as Indian mallow, crabgrass, ragweed, aster, Canada thistle, mustard, fleabane, and goldenrod. Cultivated fields contain harvestable crops whereas pastures contain herbaceous plants for grazing. The raffinate pits and chemical plant areas are essentially old-field habitat; however, mowing maintains much of these areas in a pasture-like condition. The quarry consists of slope and bottomland forests with eastern cottonwood predominating much of the quarry area. The vicinity properties are largely old-field/upland forest habitat near the raffinate pits, chemical plant, and quarry areas. Most of the habitat along the creeks and drainages in the vicinity properties is bottomland forest.

Based on habitat preferences and ranges of Missouri mammals (Schwartz and Schwartz 1959), over 30 species could be common to abundant in the area. These would include eastern cottontail rabbit, opossum, raccoon, white-tailed deer, and several species of mouse, vole, shrew, squirrel, bat, and fox. Several mammal species -- most notably the woodchuck, eastern mole, and plains pocket gopher -- dig burrows into habitat similar to that occurring at the raffinate pits area (Table 3.4).

In the Busch Wildlife Area immediately north of the raffinate pits area, 277 species of birds have been observed. Of these, 103 nest in the area whereas 43 species are common to abundant throughout at least three seasons of the year (Mo. Dept. Conserv. 1981). About 10 waterfowl species are common to abundant during the spring and fall migration, and a few species such as Canada goose, mallard, and wood duck nest and/or overwinter in the area. The raffinate pits and quarry provide habitat suitable for waterfowl. St. Charles County is within the range of over 50 reptile and amphibian species (Conant 1975). Some of these species occur at the Weldon Spring site due to the variety of terrestrial and aquatic habitats that are present.

Aquatic habitats in the vicinity of the Weldon Spring site include intermittent and permanent streams that drain the site (see Section 3.1.2), various sized ponds and lakes, and the Mississippi and Missouri rivers that ultimately receive drainage from St. Charles County. The Busch Wildlife Area contains 32 lakes and ponds ranging from 0.4 to 74 ha (1 to 183 acres) for a total of 200 ha (500 acres) (Mo. Dept. Conserv. 1981). The four raffinate pits (when containing standing water) and the quarry pond can also be classified as ponds.

The lakes and ponds in the Weldon Spring vicinity contain a warmwater fish community. Common species include carp, black bullhead, bluntnose minnow, fathead minnow, bluegill, black and white crappie, and gizzard shad (Mo. Bot. Gard. 1975). Lakes 35 and 36 of the Busch Wildlife Area support a warmwater fishery for largemouth bass, bluegill, and channel catfish, the

Table 3.4. Burrowing Mammals in the Vicinity of Weldon Spring

Species	Habitat	Burrowing Habits	Comments
Short-tailed shrew (*Blarina brevicauda*)	Forests, grasslands, marshes, brushy areas.	Will tunnel as deep as 60 cm.	Occurs less often in grassy cover. In loose soil, may tunnel 0.3 m/min.
Least shrew (*Cryptotis parva*)	Open grass, brush, dry fallow fields.	Will tunnel as deep as 20 cm.	Tunnels are 25 cm to 1.5 m in length.
Eastern mole (*Scalopus aquaticus*)	Meadows, pastures, lawns, open woodlands.	Permanent tunnels 25 to 46 cm underground.	Constructs series of tunnels. Can dig at rate of 0.3 m/min near surface and 3.7 to 4.6 m/h in deeper excavations.
Woodchuck (*Marmota monax*)	Open woods, brushy and rocky ravines, fencerows.	Nest chamber 0.9 to 1.8 m deep.	One woodchuck may have several burrows.
Eastern chipmunk (*Tamias striatus*)	Timbered borderland, brushy areas.	Enlarged chamber of tunnel built 0.75 m underground.	Digs many burrows.
Plains pocket gopher (*Geomys bursarius*)	Prairie grasslands, pastures, meadows, cultivated fields of alfalfa and clover.	Enlarged nest chamber 0.75 to 0.9 m to, rarely, 1.8 m underground. Main tunnel 15 to 23 cm below surface.	Main tunnel may be 150 m long, with 20- to 50-cm long side tunnels dug at intervals. One animal's tunnel system may occupy more than 0.4 ha.
Pine vole (*Microtus pinetorum*)	Deciduous forests, sometimes mixed hardwood-pine forests. Also fields adjacent to forests, orchards, gardens, shrub areas.	Tunnels 10 to 30 cm or more deep.	Numerous holes open at intervals from their tunnels.
Norway rat (*Rattus norvegicus*)	Around human inhabitations, fields in rural areas.	Commonly tunnels in ground to 75 cm.	Tunnels average 0.9 m long, with one or more main entrances.
Red fox (*Vulpes fulva*)	Borders of forested areas and adjacent open lands.	Chamber about 1.2 m underground.	Often uses modified woodchuck burrow or former fox den, but chamber may be dug by female fox.
Striped skunk (*Mephitis mephitis*)	Forest borders, brushy fields, fencerows, open grassy fields broken by wooded ravines or rocky outcrops.	Den about 1.2 m underground.	Usually uses den discarded by other mammals, but may dig its own. Den has one to five openings that lead to tunnels that are 1.8 to 17 m in length.

Conversion Factors: To convert meters (m) to feet (ft), multiply by 0.3048; to convert centimeters (cm) to inches (in.), multiply by 0.3937; to convert hectares (ha) to acres, multiply by 2.471.

Sources: Burt and Grossenheider (1964); Schwartz and Schwartz (1959).

latter of which is regularly stocked (Bechtel Natl. 1982). These two lakes receive drainage from the Weldon Spring site. No biological surveys of the raffinate pits or quarry areas have been recorded.

Based on habitats and distributions of Missouri fishes (Pflieger 1975), the most abundant fish species that would occur in the streams in the site vicinity include carp, creek chub, redfin shiner, bigmouth shiner, fathead minnow, white sucker, green sunfish, orangespotted sunfish, johnny darter, and fantail darter. The major species in the Missouri and Mississippi rivers include gar, gizzard shad, carp, river carpsucker, buffalo, channel catfish, freshwater drum, white bass, sturgeon, paddlefish, blue catfish, and blue sucker. Largemouth bass, bluegill, and crappie are also abundant in backwaters and oxbows (Pflieger 1975).

Three endangered species may be present in the Weldon Spring vicinity: bald eagle (Haliaeetus leucocephalus), fat pocketbook pearly mussel (Potamilus capax), and Higgins' eye pearly mussel (Lampsilis higginsi). There is currently no designated critical habitat in the project area (Nash 1984). Additional species, both state and federally listed, may occur within the area, based on both habitat requirements and known distributions. These species are listed in Appendix G, Table G.1. However, specialized habitat requirements and/or habitat preferences of these species are generally not met by the Weldon Spring site. Nevertheless, it is possible that the more mobile species, such as the bald eagle, could intermittently occupy the site.

3.1.5 Land Use, Visual and Cultural Resources

3.1.5.1 Land Use

The Weldon Spring site is located in the southwest portion of St. Charles County, Missouri. The county, roughly triangular in shape, is bounded by the Mississippi River on the north and east and the Missouri River on the south. Approximately half the county land is floodplain and half is uplands characterized by gently rolling topography (Dean 1977). The southwest uplands, which contain the site, are dissected by small stream valleys (Dean 1977).

Urban areas occupy 6.2% and nonurban areas 90.9% of the county land (St. Charles Co. Plan. Dept. 1983). The remaining area is dedicated to transportation and water uses. The St. Charles County Planning Department (1983) estimates that approximately 4% of the county's nonurban land will be converted to urban uses during 1980 to 2000. Development in the county has been dynamic in the past, and strong residential and commercial/industrial demands are anticipated to continue. The cities of St. Charles, St. Peters, O'Fallon, Lake St. Louis, and Wentzville are located along I-70 where major development has been occurring. The area south of I-70 from St. Charles City

to Wentzville and bounded by U.S. 40/61 to the west and the MKT Railroad to the south (Figure 1.1) is locally referred to as the "Golden Triangle". This area is considered likely to experience the most growth in the coming decades (East-West Gateway Coord. Counc. 1983). The Golden Triangle includes the cities of St. Charles, St. Peters, O'Fallon, Lake St. Louis, Wentzville, Weldon Spring, Cottleville, Harvester, Weldon Spring Heights, Dardenne, and All Saints Village. In addition to development within the Golden Triangle, there is substantial development potential in other areas of the county. A development of 400-500 parcels of land has recently been approved along U.S. 40/61 (Nichols 1984).

The August A. Busch Memorial Wildlife Area is located to the north of the raffinate pits and chemical plant areas, and the Weldon Spring Wildlife Area is situated to the south and east of the raffinate pits and chemical plant areas (Figure 1.1). Both of these areas are park-like tracts administered by the Missouri Department of Conservation and are dedicated to various kinds of recreational uses.

The University of Missouri operates the St. Charles County Extension Center (Figure 1.2) and owns 300 ha (740 acres) of land to the east of the Weldon Spring site that is currently used by the Center for pasture (Bechtel Natl. 1984a). A portion of this land (about 100 ha [250 acres]) is being developed as a high-technology research park, and this land will remain under the ownership of the University of Missouri. The purpose of the research park is to help stimulate the development of high-technology industries in the St. Louis area (Wagman 1982; Haggans 1984). A state of Missouri highway maintenance facility and Francis Howell High School are also located east of the Weldon Spring site along State Route 94 (Figure 1.2).

The county-assessed values for private property are strongly controlled by current and projected land uses. Land-use patterns are determined in part by ongoing socioeconomic trends (see Section 3.1.6), specifically, the dynamic growth in St. Charles County that is concentrated in the Golden Triangle. It appears that the area north of the Weldon Spring site and along U.S. 40/61 is experiencing no slowdown in property development or sales (Welch 1985). Expensive commercial tracts are for sale in the area, and over the last several years a number of commercial projects have been built -- including a bank, service stations, supermarkets, and shopping centers. New subdivisions are selling at a fast pace; homes are valued at over $100,000 and condominiums are valued at $75,000 to $100,000 per unit. South of the Weldon Spring site, very little has changed and the area remains as farmland.

3.1.5.2 Visual Resources

The raffinate pits and chemical plant areas comprise a managed environment where the vegetation is mowed and the roads are maintained. Man-made structures are present at both areas. Architecturally, the chemical plant reflects the post World War II period. The quarry is surrounded by a forest-covered natural area that has strong visual appeal. The bottom of the quarry contains a small pond. A roadway into the quarry area is maintained, and a mowed path is located inside the quarry fence. The areas surrounding the Weldon Spring site appear to have been farmed, but they subsequently reverted to a forested environment. The gently rolling terrain, permanent and intermittent streams, and forest provide an attractive setting, particularly in the area along Schote Creek.

Public access is prohibited at the raffinate pits, chemical plant, and quarry areas; however, these areas are visible through the perimeter fences. With the exception of a small part of the upper quarry, the raffinate pits and quarry areas are situated back from major highways and are screened from highway travelers by dense vegetation. Consequently, the number of individuals observing the areas is small. The chemical plant area is readily visible from State Route 94. Some of the contaminated vicinity properties are on land that is accessible to the public. Trails along small streams encourage recreational uses of this area.

3.1.5.3 Cultural Resources

The Region. The cultural history of east central Missouri and St. Charles County is complex. Paleo-Indian materials have been reported in St. Charles County, and a recently discovered site farther south represents the most easterly location of a Paleo-Indian occupation in association with extinct fauna (Graham 1979, 1980; Haas 1978). The Archaic, Woodland, and Mississippian cultural traditions are well represented at sites found throughout east-central Missouri, including Little Femme Osage Creek which passes through St. Charles County and the Weldon Spring site (Haas 1978).

The Proto-Historic Period was represented by numerous eastern and midwestern Indian tribes that were reported in the lower Missouri region (Ray et. al. 1984). Tribes that were present in the general area include the Osage, Missouri, Kaskaskia, Kickapoo, Delaware, Shawnee, Sauk, and Fox (Ray et al. 1984).

The Historic Period began with the entry of European explorers into the state. During the 1700s, French trappers traveled up the Missouri River (Foley 1971; March 1967). Although Spain controlled this area during the late 1700s, French cultural characteristics and architectural styles continued (Ray

et. al. 1984). Several settlements, including St. Louis and St. Charles, were well established by 1803. After 1815, the area west of St. Charles experienced rapid growth and many settlers passed through St. Charles on their way west along the Boones Lick Trail (Ray et. al. 1984).

These prehistoric and historic epochs are well represented by the architectural resources and prehistoric sites that are present in St. Charles County. The structures and archaeological sites listed in the National Register of Historic Places for St. Charles County are presented in Table 3.5. None of these sites are within 1.6 km (1 mi) of the raffinate pits, chemical plant, quarry, or vicinity properties.

The Weldon Spring Site. A survey to locate cultural resource sites has not been conducted at the raffinate pits, chemical plant, quarry, or vicinity properties. However, Haas (1978) has conducted a sample field survey on 3,200 ha (8,000 acres) of land in the immediate vicinity. During this survey, 216 cultural resource sites were located in St. Charles County and these sites represent occupations for all of the chronological periods discussed above. Sites are reported along State Route 94 and the MKT Railroad tracks within 1.6 km (1 mi) of the raffinate pits and chemical plant, on the north edge of the quarry, and near the vicinity properties (Haas 1978). None of the sites located during this survey have been evaluated for their potential eligibility for nomination to the National Register. Other prehistoric and historic sites may exist in the unsurveyed areas of the Weldon Spring site and include potentially eligible National Register properties as well.

The state of Missouri has determined that there is no need to conduct archeological investigations on the areas directly involved with implementation of this project at the Weldon Spring site because these areas have either been disturbed by construction-related activities or exhibit a low potential for archeological resources. However, archeological investigations may be required for areas that are currently uncontaminated or have not been subject to substantial previous disturbance, such as new borrow sites (Weichman 1986).

3.1.6 Population and Socioeconomics

3.1.6.1 Population

The Weldon Spring site is located within the St. Louis metropolitan area in St. Charles County. The population trends for the city of St. Louis and the counties that comprise the St. Louis Standard Metropolitan Statistical Area (SMSA) are presented in Table 3.6. The city of St. Louis has been losing population since 1960 whereas several of the surrounding counties have been

Table 3.5. National Register Properties in St. Charles County, Missouri[a]

AFRICAN CHURCH, (A.M.E. CHURCH OF ST. CHARLES), 554 Madison St., St. Charles.

BOONE, DANIEL, HOUSE, (NATHAN BOONE HOUSE), Hwy. F., Defiance vic., (S13, T45N, R1E).

CITY HALL, OLD, (MARKET HOUSE), 101 S. Main St., St. Charles.

FIRST MISSOURI STATE CAPITOL BUILDING, 208-216 S. Main St., St. Charles (STATE HISTORIC SITE).

HAYS, DANIEL BOONE, HOUSE, (HAYS FARM), Defiance vic., (S29, T45N, R2E).

LINDENWOOD HALL, (SIBLEY HALL), Lindenwood College Campus, St. Charles.

MARTEN-BECKER HOUSE, 837 First Capitol Dr., St. Charles.

NEWBILL-MCELHINEY HOUSE, 625 S. Main St., St. Charles.

ST. CHARLES HISTORIC DISTRICT, roughly bounded by the Missouri River and Madison, Chauncey and 2nd Sts., St. Charles.

ST. PAUL'S CHURCH, (ST. PAUL LUTHERAN CHURCH), SR D, New Melle.

STONE ROW, 314, 316, 318, 324, 328, 330 S. Main St., St. Charles.

STUMBERG, DR. JOHN H., HOUSE, 100 S. 3rd St., St. Charles.

WATSON, SAMUEL STEWART, HOUSE, (ERMELING HOUSE), 205 S. Duchesne Dr., St. Charles.

WOLF-RUEBELING HOUSE, Hwy. 94, Defiance vic., (T45N, R3E).

ARCHAEOLOGICAL SITE 23SC556, Peruque Creek, (S9, T47N, R3E).

ARCHAEOLOGICAL SITE 23SC592, (S6, T46N, R4E).

ARCHAEOLOGICAL SITE 23SC592, (LITTLE HILLS EXPRESSWAY SITE), (S20, T47N, R5E).

OLD MONROE ARCHAEOLOGICAL DISTRICT (23SC528, Survey 578, T48N, R2W), (also in Lincoln Co. - 23LN2, 23LN5, and 23LN104).

[a] Base data derived from information supplied by the Missouri Department of Natural Resources, August 1984.

Table 3.6. Population of the St. Louis Standard Metropolitan Statistical Area

County/City	1960	1970	% Change 1960-70	1980	% Change 1970-80
Missouri					
St. Louis (City)	750,026	622,236	-17.0	453,085	-27.2
St. Louis County (less St. Louis City)	703,532	951,353	+35.2	973,896	+2.4
Franklin County	44,566	55,116	+23.7	71,233	+29.2
Jefferson County	66,377	105,248	+58.6	146,183	+38.9
St. Charles County	52,970	92,954	+75.5	144,107	+55.0
Illinois					
Madison County	224,689	250,934	+11.7	247,691	-1.3
St. Clair County	262,509	285,176	+8.6	267,531	-6.2
Clinton County[a]	-	-	-	32,617	-
Monroe County[a]	-	-	-	20,117	-
St. Louis SMSA	2,104,669	2,410,884	+14.6	2,356,460	-2.3

[a] Not included in the St. Louis Standard Metropolitan Statistical Area in 1960 and 1970.

Source: U.S. Bureau of the Census (1970, 1980a).

gaining population. In particular, St. Charles County, with six decades of continuous population growth, has experienced the greatest percentage increases in population in comparison to the other counties within the St. Louis SMSA (East-West Gateway Coord. Counc. 1983).

Population trends for communities in the region are indicated in Table 3.7. The city of St. Charles has the largest population, St. Peters experienced the most growth from 1970 to 1980 (increasing from 486 to 15,700), and O'Fallon and Wentzville had the next largest populations in 1980.

Population projections for St. Charles County estimate a growth rate of about 70% from 1980 to the year 2000. According to projections by the East-West Gateway Coordinating Council (1983), the 1980 population of about 144,000 (U.S. Bur. Census 1980a) is expected to grow to 165,000 in 1985, 190,000 in 1990, 217,000 in 1995, and 245,000 in 2000 under the high-growth scenario. These projections are currently being used by the county for planning purposes (Nichols 1984).

Table 3.7. Population of the Region Surrounding the Weldon Spring Site, 1960-1980

City	1960	1970	% Change 1960-70	1980	% Change 1970-80
All Saints Village	NA[a]	NA	NA	NA	NA
Defiance	NA	NA	NA	NA	NA
Harvester	NA	NA	NA	NA	NA
Cottleville	NA	230	-	184	-20.0
New Melle	NA	NA	-	168	-
O'Fallon	3,770	7,018	+86.2	8,677	+23.6
St. Charles	21,189	31,834	+50.2	37,379	+17.4
St. Peters	404	486	+20.3	15,700	+3,130.0
Weldon Spring	NA	NA	-	70[b]	-
Weldon Spring Heights	NA	135[c]	-	144	+6.7[c]
Wentzville	2,742	3,223	+17.5	3,193	-1.0

[a] NA indicates data not available.
[b] Bechtel National (Undated).
[c] National Lead Company of Ohio (1977).
Source: U.S. Bureau of the Census (1970, 1980a, 1980b), except as noted.

3.1.6.2 Economic Development and Employment

St. Charles County has been growing primarily because people and businesses have migrated from St. Louis County to St. Charles County. A study of migration in the St. Louis metropolitan area showed that 81% of the migration originated and terminated within the St. Louis region zip code areas (East-West Gateway Coord. Counc. 1982); the primary destinations were the cities of St. Charles, St. Peters, and O'Fallon. In addition to migration of population and businesses from the St. Louis area, major firms are attracted to St. Charles County because of its central location in the country and its access to midwestern markets. General Motors has opened a plant in Wentzville that employs about 5,500 people (Grix 1985). Other large firms have located or are considering St. Charles County for relocation.

Employment increased in St. Charles County from about 18,000 in 1960 to 35,000 in 1970 to 67,000 in 1980. Although the major employing industrial sector is manufacturing, this sector decreased in size relative to other industrial sectors between 1960 and 1980. The trade and service sectors have steadily increased (U.S. Bur. Econ. Anal. 1984). This relative shifting of employment from manufacturing to trade and services probably represents a maturing of the economic base in the area and parallels the national trend toward service sector development.

3.1.6.3 Local Institutions and Services

St. Charles County Water Distribution. The sources of municipal water in the area surrounding the Weldon Spring site are given in Table 3.8. The larger communities of St. Charles, O'Fallon, and St. Peters rely primarily on their own water sources. Public Water District #2, Missouri Cities Water Company, Francis Howell High School, the Army, and a portion of the county all rely on the county well field for all or part of their water supply. The county well field is an important issue to the local residents because of the well field's proximity to the quarry.

Table 3.8. Water Sources in St. Charles County

User	Community Water System	Suppliers Relying on County Well Field[a]		
		Water District #2	Missouri Cities Water Co.	St. Charles County
St. Charles	X	-	X_1	-
O'Fallon	X	-	X_1	-
St. Peters	X	-	X_1	-
Lake St. Louis	X	b	-	-
Weldon Spring Heights	X	-	-	-
Defiance	-	X	-	-
Wentzville	-	X_2	-	-
Dardenne	-	X	-	-
New Melle	-	X	-	-
Cottleville	-	-	X	-
Harvester Area	-	-	X	-
Weldon Spring	-	-	-	X
Francis Howell High School	-	-	-	X
Army	-	-	-	X
All Saints Village[c]	-	-	-	-

[a] X, Source of water.
X_1, Certain subdivisions within city limits served by Missouri Cities Water Company.
X_2, Currently Public Water District #2 supplies water to the General Motors plant and will eventually supply water to the city.
-, Not applicable.

[b] Water originates from well in Lake St. Louis but is administered by Public Water District #2.

[c] Water supplied by private wells.

Missouri Cities Water Company is the private water company that services portions of St. Charles County. The company also provides contracted expertise and personnel for the operation of the county well field and the water treatment facilities of St. Charles County. Missouri Cities Water Company was at the center of a recent controversy concerning expansion of the county well field. A water agreement allowing Wentzville to obtain county water (supplied through Public Water District #2) precipitated the controversy. The concern that additional pumping at the well field would cause migration of radionuclides from the quarry to the well field prompted the Missouri Cities Water Company to declare a moratorium on additional hookups. The moratorium was declared to give the company and the concerned governmental entities time to conduct a study of the well field area, including areas upriver and downriver 3.2 km (2 mi) in each direction. The moratorium on additional hookups was short-lived and ended with the only restriction on water use being alternate-day grass watering (Sullivan 1986).

Francis Howell School District. Francis Howell High School is located approximately 1 km (0.6 mi) from the raffinate pits and chemical plant areas on State Route 94 (Figure 3.2). Enrollments are steadily increasing and current enrollment is approximately 2,000 students (Rebore 1984). In addition to regular classes, there is a preschool program (150-200 students) and a parenting class (80 parents). It is estimated that there are 2,300 persons on campus daily (Rebore 1984). Students are bused to school from all over the district; 50 buses come and go on a daily basis. Primary times for bus traffic are 6:45 to 7:20 a.m. and 2:15 to 2:45 p.m.

The school district is concerned about the health and safety of all people that use the high school campus, especially considering the proximity of the Francis Howell High School to the Weldon Spring site. The U.S. Army has monitored for radon gas on the school property and found no evidence of levels above background. A consultant was hired by the school district to test for gross alpha/beta contamination in the soil. The findings indicated no dangerous levels of contamination (Rebore 1984).

Some parents have indicated concern over potential health effects associated with the Weldon Spring site and others have refused to enroll their children in the preschool program because of that concern (Rebore 1984). School officials are planning approximately 10 million dollars worth of renovation at the site over the next 5 years (Rebore 1986).

Home Builders Association. The Home Builders Association of Greater St. Louis is a trade association representing approximately 1,000 builders in the area. The association was concerned that developers in the association would be adversely affected by a moratorium by the Missouri Cities Water Company on water hookups (Sullivan 1984). The moratorium meant that there

would be no additional water contracts for area builders until a study of the St. Charles County well field area was completed. The water issue with the well field had the potential for redirecting builders toward other areas where there were no constraints on water (Sullivan 1984). Developers with property adjacent to towns with municipal water service were requesting annexation to guarantee a supply of water. The well field study has been completed (January 1986), and the moratorium on hookups has been lifted. The period of uncertainty concerning water hookups was brief and thus minimized disruption to building activities in the area (Sullivan 1986).

Missouri Conservation Commission. The Missouri Conservation Commission operates three wildlife areas in the vicinity of Weldon Spring: the August A. Busch Memorial Wildlife Area (2,800 ha [7,000 acres]), the Weldon Spring Wildlife Area (3,000 ha [7,500 acres]), and the Howell Island Wildlife Area (1,100 ha [2,600 acres]). The Busch and Weldon Spring wildlife areas are located immediately adjacent to the raffinate pits, chemical plant, and quarry, and some radioactive contamination is present (Figure 1.4 and Appendix H, Section H.1.4). In the Weldon Spring Wildlife Area, this contamination necessitated the relocation of a nature (hiking) trail (the areas along the nature trail having elevated levels of radioactivity are designated WA-1 through WA-4 in Figure 3.7). Elevated levels of radioactivity have also been detected in Lakes 34, 35, and 36 and in Burgermeister Spring in the Busch Wildlife Area (Figure 3.7 and Appendix H, Table H.8). The Missouri Conservation Commission has been conducting sampling of both surface water and fish for radionuclides. The monitoring results indicate that no restrictions on fishing are necessary.

The majority of users of the wildlife areas come from the St. Louis metropolitan area (Cassidy 1984). The Busch Wildlife Area is used most intensely in the spring and summer, when monthly attendance ranges from about 100,000 to 130,000 (Cassidy 1984). Yearly attendance for the last 10 years has averaged about 710,000 (Cassidy 1984; Calvert 1986).

Attendance at the August A. Busch Memorial Wildlife Area fluctuates by as much as 12% (increase or decrease) from year to year. Some perceive that the fluctuations in attendance may be related to public concern over radioactive contamination in these areas (Bollmeier 1984), but the manager of the Busch Wildlife Area believes that the fluctuations are due to weather conditions (Cassidy 1984) -- specifically, higher than normal precipitation and/or very hot temperatures in summer as well as cold temperatures in winter are related to variations in attendance. This relationship between attendance and temperature has been confirmed by regression analysis for 1976-1984, which indicated that the monthly average temperature accounted for 64% of the variance in monthly attendance.

3.1.6.4 Interest Groups

The St. Charles Countians Against Hazardous Waste (SCCAHW) is the primary citizen's group voicing concern about the Weldon Spring site. The group was formed in 1982 and has become a clearinghouse of technical information. The group publishes a newsletter, presents a slide show on the Weldon Spring site to interested parties, and monitors private wells in the area (with the Missouri Department of Natural Resources). In addition, the organization helped organize data for an epidemiological study conducted by the Missouri Department of Health (Bollmeier 1984).

The SCCAHW is a county chapter affiliate of Missourians Against Hazardous Waste. The group is also affiliated with the Coalition for the Environment and interacts with national environmental groups. Membership of the SCCAHW is approximately 230. Pressure from this group on all entities involved has had the effect of increasing public awareness and public input into the DOE planning process and increasing cooperation among the various interested parties (Williams and Payne 1984).

3.1.7 Radiological Characterization

The Weldon Spring wastes are located in four areas: the raffinate pits, quarry, chemical plant, and vicinity properties. Radionuclides in both the uranium-238 and thorium-232 decay series are present (see Appendix H, Section H.1). Because of their short half-lives, radionuclides in the thorium-232 decay series are assumed to be in secular equilibrium with thorium-232. The radionuclides in the uranium-238 series are not in secular equilibrium, and the amount of radium-226 and its decay products will increase over time as secular equilibrium is gradually reestablished. Following are summaries of radiological conditions at each of the four areas. More information is given in Appendix H, Section H.1.

3.1.7.1 Raffinate Pits

Raffinate Pits 1, 2, and 3 contain raffinate sludge and slag resulting from the refining of uranium ore concentrates and the recycling of scrap metal carried out at the chemical plant. Pit 4 contains similar slag and sludge as well as wastes from the processing of thorium-containing materials and drums and rubble from the partial decontamination of the chemical plant (Natl. Lead Co. Ohio 1977; Bechtel Natl. 1984a). It has been estimated (Bechtel Natl. 1984a) that there are 170,000 m^3 (220,000 yd^3) of wastes in the pits (Table 1.1).

The wastes in the pits are reported to be stratified and heterogeneous (Task Force 1967; Natl. Lead Co. Ohio 1977). The amount of surface water

covering the wastes varies during the year. In summer, all surface water may evaporate from Pits 1 and 2 (although, according to the staff at the site, Pits 1 and 2 have not been dry in the past several years). Surface water is always present in Pits 3 and 4. Pit 3 is designed to overflow into Pit 4 through a pipe in the dike wall common to both pits (Natl. Lead Co. Ohio 1977; Bechtel Natl. 1984a). Pit 3 contains most of the wastes. The physical characteristics of the wastes in the pits are summarized in Appendix H, Table H.1.

Concentrations of radionuclides in the raffinate sludge have been reported in various studies (Task Force 1967; National Lead Co. Ohio 1977; Bechtel Natl. 1984a); these results are presented in Appendix H, Table H.2. Thorium-230 is the predominant isotope. The average concentration of thorium-230 in the raffinate sludge is estimated to be 3,500 pCi/g (wet wt.), and the total inventory is estimated to be 700 Ci. For radium-226, the average concentration and total inventory in the wet sludge are 97 pCi/g and 20 Ci, respectively (Table 1.2). Because of ingrowth of radium-226 from the radioactive decay of thorium-230 (which has a half-life of 77,000 years), the concentration of radium-226 in the raffinate sludge (averaged over the four pits) will increase to a peak concentration of 3,200 pCi/g (wet wt.) in 9,100 years. After this, the radium-226 concentration will decrease with the 77,000-year half-life of the parent thorium-230 (see Appendix H, Section H.1.1).

It has been estimated that there are 98,000 m^3 (130,000 yd^3) of contaminated soils beneath the raffinate pits (Bechtel Natl. 1984a). For purposes of analysis, it is assumed that the radionuclide inventory in these soils is 1% of that in the raffinate sludge, i.e., 7 Ci of thorium-230 (Table 1.2).

The presence of gamma-emitting radionuclides in this area results in elevated exposure rates. The background exposure rate for the Weldon Spring area is 11 μR/h, based upon data reported in the 1985 annual monitoring report for the Weldon Spring site (Bechtel Natl. 1986b), and the average reading in the raffinate pits area is 23 μR/h (Bechtel Natl. 1984c). There are two areas of higher exposure rates in the raffinate pits area, one along the northern edge of Pit 3 (66 μR/h) and the other around the southern perimeter of the pit area near the entrance (234 μR/h) (Bechtel Natl. 1984c). Access to the raffinate pits area is restricted.

Radon gas concentrations measured at six locations in the raffinate pits area are within the range of background values. Measured concentrations range from an annual average low of 0.2 pCi/L to an annual average high of 0.5 pCi/L; the average concentration at these six locations is 0.35 pCi/L (Bechtel Natl. 1986b). These values are comparable to the background radon

gas concentration in this area of 0.5 pCi/L and are below the DOE maximum permissible value of 3 pCi/L for uncontrolled areas (DOE Order 5480.1A--Attachment XI-1).

Although background air quality data are not currently available for the raffinate pits area, a program is being implemented to develop air quality baseline data. A groundwater monitoring program is in place around the raffinate pits to monitor for potential migration of radioactive contaminants from the raffinate pits; these results are reported annually (Bechtel Natl. 1986b).

3.1.7.2 Quarry

Subsurface samples were recently taken from boreholes drilled into the quarry wastes (Bechtel Natl. 1985), and the concentrations of radioactive species in the samples varied greatly, as a function of both depth within a borehole and borehole location (see Appendix H, Section H.1.2). It has been estimated that there are 73,000 m^3 (95,000 yd^3) of contaminated wastes in the quarry (Bechtel Natl. 1985). The concentration and inventory of thorium-230 are lower in the quarry wastes (540 pCi/g and 90 Ci) than in the raffinate sludge (3,500 pCi/g and 700 Ci); however, the uranium-238 concentration and inventory are similar (170 pCi/g and 30 Ci for the quarry and 150 pCi/g and 30 Ci for the raffinate sludge) (Table 1.2).

Concentrations of uranium, radium, and thorium in water samples taken in the quarry pond are discussed in detail in Appendix H, Section H.1.2. These data suggest that concentrations of total uranium in the quarry pond increased from 1 pCi/L in 1960 to a peak value of 17,000 pCi/L in 1967 and then decreased to 620 pCi/L in 1985. The buildup from 1960 to 1967 may represent increasing amounts of uranium available for leaching as wastes were dumped in the quarry during that period. The decline after 1967 may represent a slow leachout and depletion of uranium inventory from the quarry wastes or it may result from sampling at different depths in the stratified water of the quarry pond.

Groundwater in the quarry area is also contaminated with uranium, up to 8,800 pCi/L. Background concentrations in this area range from 6 to 27 pCi/L (see Appendix H, Section H.1.2). There are large differences in uranium concentrations at nearby monitoring wells, which may reflect the fact that groundwater movement in the limestone occurs mainly in solution channels and fractures and thus different solution channels are sampled by the boreholes. A groundwater monitoring program is in place to assess the potential migration of radioactive contaminants from the quarry; these results are reported annually (Bechtel Natl. 1986b).

The presence of these gamma-emitting radionuclides results in elevated exposure rates. Measured exposure rates at the surface of the quarry range from 1.5 to 625 μR/h, indicating that the distribution of radioactive materials in the quarry is very uneven (Berkeley Geosci. Assoc. 1984). The background exposure rate for the Weldon Spring area is 11 μR/h.

Radon gas concentrations in the atmosphere in the quarry range from 0.8 to 18 pCi/L and average about 14 pCi/L (Berkeley Geosci. Assoc. 1984) (see Appendix H, Section H.1.2). Other concentrations measured at different points along the fence around the quarry range from an annual average low of 0.2 pCi/L to an annual average high of 1.3 pCi/L (Bechtel Natl. 1986b). These values are comparable to the background radon gas concentration in this area of 0.5 pCi/L and are below the DOE maximum permissible value of 3 pCi/L for uncontrolled areas (DOE Order 5480.1A--Attachment XI-1).

3.1.7.3 Chemical Plant

Extensive contamination of the chemical plant is known to have occurred (Niedermeyer 1976; Ryckman & Assoc. 1978; Bechtel Natl. 1984a). From June 1957 to December 1966, the chemical plant was used to process uranium concentrates in the form of sodium diuranate containing 70% uranium. Small amounts of materials containing depleted and slightly enriched uranium were also processed, and thorium concentrates were processed in 1965 and 1966 (Task Force 1967; Harris 1986).

Uranium is the main contaminant at the chemical plant. Estimated concentrations of uranium-238 range from 3.9 to 50,000 pCi/g, with a total estimated inventory of 7 Ci. Estimated concentrations and inventories of radium-226, thorium-230, and thorium-232 are much lower, i.e., average concentrations of 3 pCi/g and 1 Ci for radium-226 and thorium-232, and 6 pCi/g and 3 Ci for thorium-230 (Table 1.2). An estimated 240,000 m^3 (310,000 yd^3) of contaminated soils and rubble must be removed prior to releasing the plant area for appropriate use (Rockwell Int. 1979).

Concentrations of uranium, radium, and thorium in water measured at various locations in the chemical plant area range from 14 to 10,000 pCi/L for natural uranium,* <0.022 to 0.73 pCi/L for thorium-232, 0.09 to 1,100 pCi/L for radium-226, and 0.45 to 42 pCi/L for radium-228. For comparison, average background values are 3.3 pCi/L for natural uranium, 0.33 pCi/L for radium-226, and <5 pCi/L for radium-228 (Bechtel Natl. 1986b). Based on comparison of measured values with background values, all the water samples

*Uranium-234, -235, and -238 present in the activity ratio of 1:0.046:1, which is the ratio for uranium as it exists in nature.

are contaminated with natural uranium (see Appendix H, Section H.1.3, for more details).

The measurements of concentrations of radioactive species in water and in soils in the chemical plant area vary over wide ranges and are based on a limited number of samples. DOE is planning to carry out more extensive measurements in order to obtain better estimates of the extent of contamination in the chemical plant area.

3.1.7.4 Vicinity Properties

Concentrations of radioactive species in soils and sediments have been measured at many locations in the vicinity properties (Ryckman & Assoc. 1978; deRoos 1984; Berkeley Geosci. Assoc. 1984; Deming 1986; Boerner 1986). These data are discussed in detail in Appendix H, Section H.1.4.

Contamination occurs in the main drainage ditch from the chemical plant along the entire 2.1 km (1.3 mi) length from the U.S. Army Reserve Property fence to the Missouri River. Thorium-230 is the main radionuclide in the ditch materials (Deming 1986). Elevated concentrations of uranium-238 have also been found in the alluvium area between the quarry and Femme Osage Slough (Berkeley Geosci. Assoc. 1984; Boerner 1986).

There are an estimated 21,000 m^3 (27,000 yd^3) of contaminated soils on the vicinity properties (Deming 1986; Boerner 1986). Thorium-230, which is present in the main drainage ditch in concentrations up to 10,000 pCi/g, has a total inventory of 30 Ci. Uranium-238 is present in most of the contaminated soils, with an average concentration of 120 pCi/g and a total inventory of 4 Ci (Table 1.2 and Appendix H, Section H.1.4).

Essentially no radium-226 or thorium-232 contamination has been measured in sediment samples collected from 34 lakes in the Busch Wildlife Area (Boerner 1986). Uranium-238 concentrations slightly higher than background were found in sediments in several lakes; concentrations of uranium-238 in sediments of other lakes (34, 35, and 36) are considerably above background values but below cleanup guidelines (Boerner 1986).

Elevated gross alpha activity occurs in water in and around Lakes 34, 35, and 36; Nature Trail Valley; Burgermeister Spring; and Femme Osage Slough. Uranium concentrations that are above background (up to 220 pCi/L) occur in water in Little Femme Osage Slough, Femme Osage Slough, Burgermeister Spring, and Lakes 34, 35, and 36. Radium-226 and radium-228 concentrations are all low and close to background values. The concentrations of radionuclides are below applicable DOE guidelines for uncontrolled areas (U.S. Dept. Energy 1986d). Details on contamination in water on the vicinity properties are given in Appendix H, Section H.1.4.

Possible sources of the uranium contamination in the slough include (1) pumping of quarry pond water during 1960-1963 into Little Femme Osage Creek, which flowed into Femme Osage Slough until the slough was isolated by levee construction in late May 1961 (Berkeley Geosci. Assoc. 1984), (2) past dumping of contaminated materials, and (3) groundwater movement from the quarry through fractures in the quarry limestone into the alluvium in the slough area.

Concentrations of radon gas in air at the vicinity properties are comparable to the annual average background of 0.5 pCi/L in this area. The exposure rates are essentially the same as the normal background value in the Weldon Spring area, i.e., 11 μR/h (Bechtel Natl. 1986b).

3.1.8 Nonradiological Characterization

Following is a summary of the nonradiological characterization of wastes, surface water, and groundwater at the Weldon Spring site. Details are given in Appendix H, Section H.2.

3.1.8.1 Raffinate Pits

The results of measurements of chemical species in the raffinate pits solids for 1967 (Task Force 1967) and 1983 (Bechtel Natl. 1984a) are given in Appendix H, Section H.2.1. Large differences in concentrations exist in the different sets of data, particularly for the major constituents. These differences may be partially explained by the stratification and heterogeneity of the pit sludge (Natl. Lead Co. Ohio 1977). The raffinate sludge has high concentrations of several metals, including iron, lead, magnesium, molybdenum, and zirconium. The principal anions are nitrate and fluoride.

An analysis for organic priority and nonpriority pollutants was carried out on a composite sample of raffinate sludge (Haywood 1984). The sample was analyzed for 82 priority pollutants (19 pesticides, 7 PCBs, and 56 acid and base/neutral compounds) and 13 nonpriority pollutants (including PCB 1262, which is not listed as a priority pollutant [Keith and Telliard 1979]). All concentrations were reported as detection limits only, which varied from 0.1 to 1 ppm for the different individual compounds. No positive results (above detection limits) were reported for any organics. However, organic compounds would be expected to be present as minor components because the processing of the uranium materials included a solvent extraction step using tributyl phosphate in hexane (Niedermeyer 1976).

Concentrations of nonradiological chemical species in the water standing in the pits -- as determined from samples taken in several different years from 1967 through 1984 -- vary greatly for different years (up to factors of

10 to 100 in some cases). Such variations in concentrations with time may be associated with meteorological conditions. For example, samples collected after a heavy rainfall will have lower concentrations, due to dilution and stratification factors, than will samples collected in a period of drought. Also, water in the pits exists in two phases: free water above the sludge and water in intimate contact and bound to the raffinate materials making a sludge or gel. The water in intimate contact with the raffinate materials is quite likely to have higher concentrations of dissolved solids than free water standing over the sludge.

Comparisons of concentrations of chemical species in the pit waters with state of Missouri irrigation limits show that, with the exception of arsenic, concentrations of these chemical species are below the limits. Current plans are to dispose of the water by treating it, as necessary, to reduce contaminant concentrations and then using it for spray irrigation of a restricted area (Section 4.1.3.1). As an alternative to spray irrigation, the water could be released to the Missouri River under controlled conditions and in compliance with Missouri state requirements.

3.1.8.2 Quarry

Based on the known history of materials dumped in the quarry (Task Force 1967; Pennak 1975; Berkeley Geosci. Assoc. 1984), materials contaminated with residues from TNT manufacture could be present. Barium sulfate residues from the vicinity of the St. Louis Airport may also have been dumped in the quarry (Berkeley Geosci. Assoc. 1984). Other chemical species present are those in the various types of building rubble and contaminated materials that were also dumped in the quarry.

The ranges and average concentrations of metals and organic compounds in the quarry wastes as determined from samples taken in 1984 from six boreholes are presented in Appendix H, Section H.2.2. The concentrations of arsenic, copper, lead, nickel, and zinc in the quarry wastes are 100 ppm or greater. The concentrations of some metals at one or more locations are higher than those in the dried raffinate sludge. Most organic pollutants are below detection levels. Organic compounds that were found in one or more samples at concentrations above detection limits include lindane (0.0013 ppm), PCB 1254 (up to 46 ppm), polycyclic hydrocarbons (up to 75 ppm), and diacetone alcohol (up to 14 ppm). The presence of PCB 1254 prevented detection of most pesticides (Bechtel Natl. 1985). A complete list of compounds that were found at concentrations above detection limits is presented in Appendix H, Table H.13.

Concentrations of several chemical species were measured in surface water and groundwater at the quarry. Arsenic and cadmium are the only elements whose concentrations in one or more samples of quarry pond water equaled or

exceeded the Missouri state irrigation limits. Average concentrations of arsenic, lead, manganese, sulfate, and zinc in groundwater in boreholes in the yellow zone* of the quarry or in wells along the quarry fence exceed the Missouri subsurface water limits (see Appendix H, Section H.2.2, for more details).

3.1.8.3 Chemical Plant

Few data are available on nonradiological contamination of soils and materials in and around the chemical plant. The plant is contaminated with about 4 MT (4.4 tons) of uranium compounds, some of which is "green salt" (uranium tetrafluoride) that is visible around the plant (Niedermeyer 1976; Ryckman & Assoc. 1978). This gives an indication of the extent of fluoride contamination at the chemical plant.

As of 1978, two stainless steel tanks at the plant were partially filled with tributyl phosphate, an organic solvent used to extract uranium (Ryckman & Assoc. 1978). There were also some tanks still present from an on-site refinery tank farm (Henry 1986); possible contents included nitric and sulfuric acids, caustic soda solution, ether, and hexane. The tank farm area is reported to be heavily contaminated (Rockwell Int. 1979).

Limited soil analyses have been carried out in and around the plant for TNT and related compounds. The measured values range from "none detected" to 290 ppb of TNT (Niedermeyer 1976). In general, measured subsurface soil concentrations are higher than surface soil concentrations. Presumably, this is due to the protection of subsurface materials from direct solar radiation, which helps break down some of the organic compounds (Niedermeyer 1976).

Concentrations of DNT and TNT were found to be above detection limits in only a few water samples. One sample taken from standing water had concentrations of 40 µg/L of 2,4-DNT, 7.1 µg/L of 2,6-DNT, and 1.5 µg/L of TNT. The results also indicate that some of the nitrated toluenes have been leached from the soil (Niedermeyer 1976).

3.1.8.4 Vicinity Properties

No data are available on concentrations of nonradiological parameters in the vicinity properties soils. Measurements of nonradiological chemical parameters in groundwater from observation wells between the quarry and Femme Osage Slough indicate that, for several elements -- arsenic, lead, manganese,

*The yellow and red zones in the quarry are areas of relatively high surface alpha activity that were so designated for protection of personnel during surveys of the quarry in 1979-1981 (Berkeley Geosci. Assoc. 1984).

and selenium -- the average concentrations measured in groundwater between the quarry and slough in 1979 through 1981 are at or above Missouri groundwater limits. For other species -- barium, copper, zinc, nitrate, and sulfate -- the averages are below the groundwater limits, but the upper limits of the concentration ranges are above the groundwater limits. For elements such as arsenic and selenium, the results are not too meaningful because the reported detection limits are greater than the state groundwater limits. Concentrations of arsenic in water samples from the slough exceed limits for groundwater, drinking water, and irrigation water.

According to 1985 data for groundwater in the alluvium between Femme Osage Slough and the quarry, average concentrations of copper, lead, manganese, and zinc are above the Missouri groundwater limits. Concentrations of arsenic and selenium are below the Missouri groundwater limits. However, the 1985 data are for samples collected from only three observation wells and are thus less representative of the near alluvium area than are the 1979-1981 data.

Based on the 1985 data for groundwater in the alluvium between Femme Osage Slough and the Missouri River, the average concentrations of manganese and zinc are also higher than the state groundwater limits. For arsenic and copper, the average concentrations are below the groundwater limits, but the upper limits of the concentration ranges are at or above the groundwater limits. For other elements, the ranges and averages are below state groundwater limits (see Appendix H, Section H.2.4).

For the groundwater contaminants that are reported to be at or above groundwater limits for the slough and the alluvium between the quarry and Femme Osage Slough, it is not clear at present whether these are contaminants that are being actively leached from the quarry, are background values, or are a result of the high detection limits (arsenic and selenium). The elevated manganese and zinc concentrations (above the groundwater limit) probably represent background values because the groundwater in the river alluvium has elevated concentrations of manganese and zinc. One well to the north (upgradient) of the quarry has high concentrations of barium. Also, the same area south of the quarry that has elevated groundwater concentrations of uranium has elevated concentrations of chloride, sulfate, nitrate, sodium, and manganese (Berkeley Geosci. Assoc. 1984). These results suggest that other elements besides uranium are being actively leached and transported from the quarry. However, the complexity of the chemical transport of elements in the fractured limestone and adjacent alluvium is indicated by the fact that two adjacent boreholes have quite different chemical concentrations (Berkeley Geosci. Assoc. 1984).

Measurements of TNT concentrations in water from two wells between Femme Osage Slough and the limestone cliffs near the quarry indicate that the TNT concentration of 380 µg/L in one well is above an interim environmental criterion of 44 µg/L for protection of public health (U.S. Dept. Army 1980). It is also documented that two other off-site locations (i.e., Schote and Dardenne creeks) were contaminated during operation of the TNT plant (Niedermeyer 1976). Total organic halides in water from the same two wells were measured at concentrations of 36 and 71 µg chloride per liter (see Appendix H, Section H.2.4).

Some measurements of nitrate and chloride concentrations in surface water have been made at several off-site locations. Elevated concentrations of nitrate have been found in the chemical plant process sewer outfall stream at the head of the main drainage ditch to the Missouri River (Weidner and Boback 1982; Bechtel Natl. 1983b). Concentrations of nonradiological species in groundwater at other locations are below the Missouri groundwater limits, with the exception of manganese in the water of Well 5 in the county well field and nitrate in Burgermeister Spring (see Appendix H, Tables H.12 and H.16). Dye-tracing studies suggest the existence of a subsurface connection between the stream drainage out of Ash Pond and the Burgermeister Spring area (Dean 1985) (see Section 3.1.2.2).

3.1.9 Plans for Additional Characterization of the Weldon Spring Site

The assessments in this document are based on the best available information. DOE is planning to carry out additional site characterization analyses to support detailed engineering design. As new information becomes available, the assessments in this EIS will be modified as necessary. These investigations are expected to include the installation of additional groundwater monitoring wells and the identification of areas most suitable for construction of the on-site disposal cell for Alternative 2 (DOE's preferred alternative). DOE is continuing to gather additional data characterizing the physical, radiological, and chemical properties of these wastes. These additional data will be factored into detailed engineering design activities to ensure that the most appropriate design features are incorporated into the disposal cell. DOE's decisions (see Figure 1.5) will be supported by the characterization studies.

3.2 HANFORD SITE

The Hanford site is a large DOE-owned site of approximately 1500 km^2 (570 mi^2). It is a candidate site for disposal of the Weldon Spring wastes because it offers a large land area that is remote from population centers.

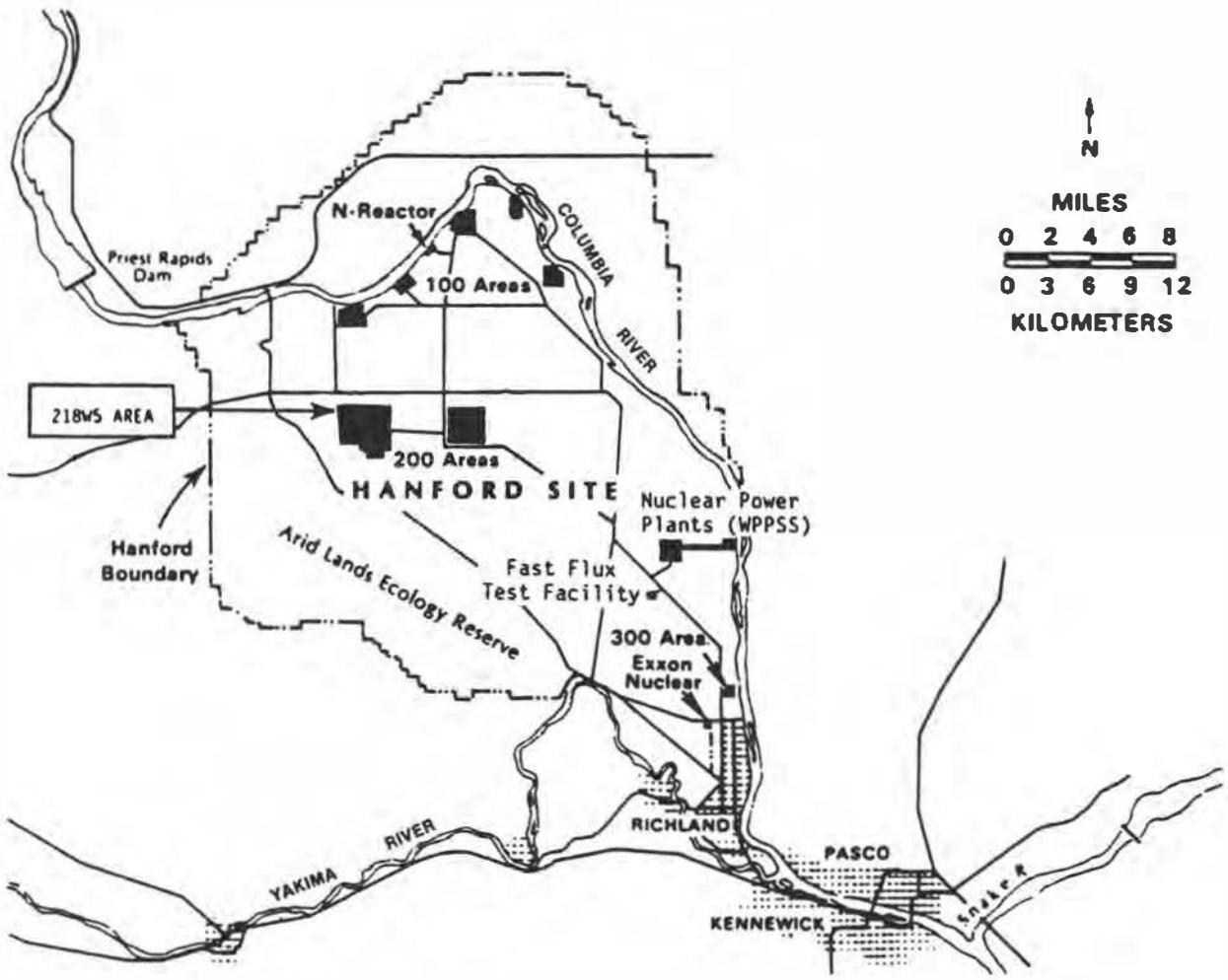

Figure 3.11. Nuclear-Related Installations on the Hanford Site. Source: Modified from Sula et al. (1982).

Also, the climate is semiarid to arid, and commercial and federal waste disposal sites are already located there (Figure 3.11).

3.2.1 Topography, Geology, Soils, Seismology, and Mineral Resources

The Hanford site is located near Richland, Washington, on the Columbia River alluvial plain within the central part of the Pasco Basin in the Columbia Basin Geologic Province. The site is bounded to the south, west, and north by large anticlinal ridges (Rattlesnake Mountain, Saddle Mountains, and Umtanum Ridge); and to the east by the Columbia River. The 218W5 Area, the area identified for possible disposal of the Weldon Spring wastes (Figure 3.11), is located near the west central portion of the site on an ancestral Columbia River bar. Elevations in this area range from 200 to 220 m (650 to 715 ft) MSL.

Most of the site is underlain by coarse-grained alluvial sediments. Because of the semiarid to arid climate in this region for nearly 12,000 years, the surficial sediments have been only slightly weathered. Sediments at or near the ground surface range from coarse boulder and cobble gravel in the extreme northern section of the site to coarse sand in the southern section. The entire site has been blanketed by as much as 15 m (50 ft) of windblown deposits ranging from very fine sands and silts to coarse sands. Numerous live dunes and relic dune features also exist within the Hanford site. On the west side of the 200-West Area, adjacent to the Yakima and Rattlesnake hills, the sediments grade into silts and fine sands (U.S. Energy Res. Dev. Admin. 1975).

The Columbia River Plateau Physiographic Province consists of a 3,700-m (12,000-ft) thick sequence of basaltic lava flows. Deformation of the accumulated lava created the Pasco Basin (U.S. Energy Res. Dev. Admin. 1975; Brown and Isaacson 1977). The major stratigraphic units underlying the Hanford site, in ascending order, are (1) Columbia River Basalts, which exist to depths greater than 3,200 m (10,400 ft), (2) Ringold Formation, consisting of semiconsolidated ancestral Columbia River alluvial sediments up to 370 m (1,200 ft) thick, (3) Hanford Formation, consisting of unconsolidated sands, silts, and gravels, and buried former river channels that were carried into the area by glacial floodwaters, and (4) surficial deposits of alluvium, dune sand, and loess (windblown silt), in part weathered to clay, and fine sand that overlie part of the eroded surface of the Ringold Formation (U.S. Energy Res. Dev. Admin. 1975; Brown and Isaacson 1977). The major stratigraphic units in the unsaturated zone (above the water table) underlying the 200-West Area are (1) surficial deposits, (2) the Hanford Formation, and (3) the upper portion of the Ringold Formation (Figure 3.12). In some areas, loess and alluvium may be overlain by the Hanford Formation. The thickness of the unsaturated sediments ranges from 55 to 82 m (180 to 270 ft) beneath the 200-West Area.

Eastern Washington is in a region of low to moderate seismicity that lies between the western Washington and eastern Montana zones of much greater seismicity. On the basis of the worst damage that has occurred since 1840 (as designated by the U.S. Coast and Geodetic Survey), the Hanford site falls within a region having the potential for moderate earthquake damage. The Corfu quake of 1918, located along the Saddle Mountains fault, caused the maximum historical ground motion on the Hanford site, with a Modified Mercalli intensity of IV-V (U.S. Energy Res. Dev. Admin. 1975).

No mineral resources have been identified within the 200 Areas of the Hanford site.

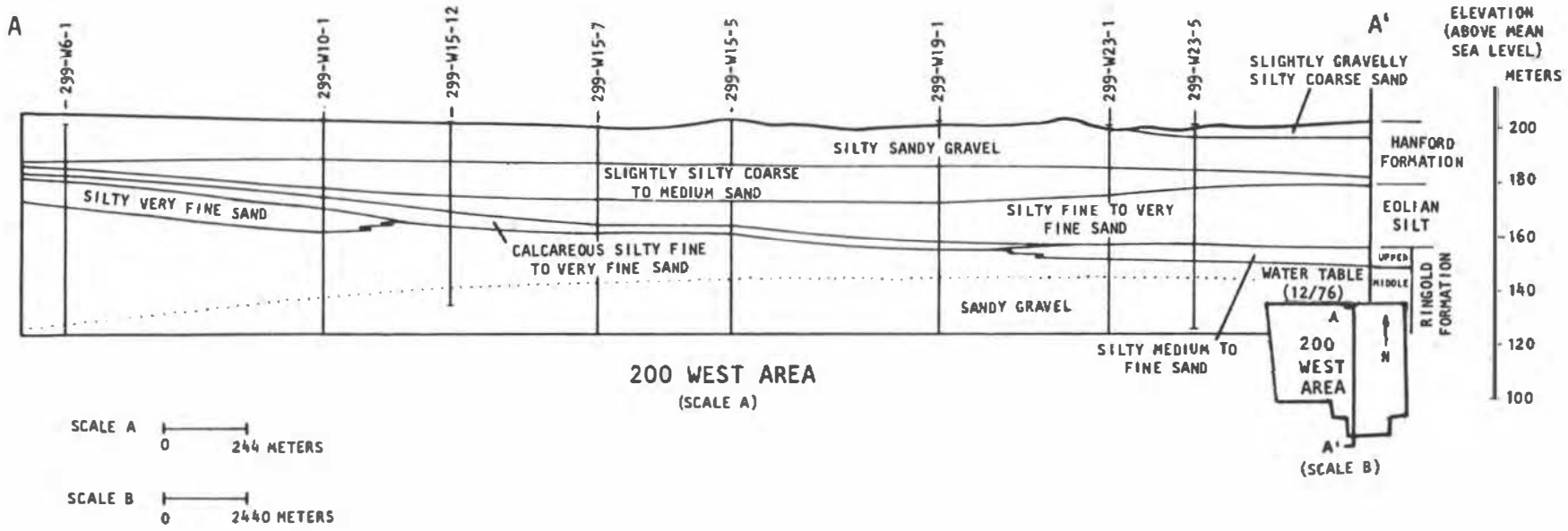

Figure 3.12. Geologic Cross Section of the 200-West Area on the Hanford Site. Source: Modified from Brown and Isaacson (1977).

3.2.2 Hydrology, Water Use, and Water Quality

3.2.2.1 Surface Water

The Columbia River is north and east of the site (Figure 3.13) and is the dominant hydrologic feature in the Hanford area. Flow in the Columbia River is influenced by water usage, including upstream reservoir projects. In the Hanford reach of the Columbia River, the water is of excellent quality and is used for municipal drinking water at Richland and Pasco. Municipal water intakes are located both above and below the confluence of the Yakima and Columbia rivers.

Cold Creek, an ephemeral stream that drains the 200-West Area (Figure 3.13), has a relatively short reach (about 1 km in length). On rare occasions, flow is sufficient to increase the reach to 10 km (6 mi). Cold Creek valley ends at the Yakima River about 48 km (30 mi) southeast of the 200-West Area.

The potential for flooding is generally confined to areas near the Columbia River, which is normally about 75 to 90 m (250 to 300 ft) below the 200-West Area. The 200 Areas are more than 20 m (70 ft) above the probable maximum flood level (i.e., the flood discharge that may be expected from the most severe combination of meteorological and hydraulic conditions reasonably possible in the region). The potential for flash flooding from the Cold Creek drainage has been examined, and a maximum flood depth of 2.3 m (7.5 ft) was estimated along the southwestern portion of the 200 Areas plateau and extending to the 200-West Area. A 100-year peak stage flood would not reach the 200 Areas (U.S. Dept. Energy 1986a).

3.2.2.2 Groundwater

In the Pasco Basin, groundwater is found in both the sedimentary deposits and basalt bedrock. Groundwater occurs unconfined in the sedimentary deposits, although locally confined zones exist. Water in the basalt bedrock occurs mainly under confined conditions. The general elevation and configuration of the unconfined aquifer are shown in Figure 3.14. The water table under the 200 Areas is about 56 to 100 m (180 to 330 ft) beneath the ground surface (U.S. Dept. Energy 1986a). More moisture is evaporated from the ground than is recharged by precipitation, and the unsaturated zone is generally very dry (Brown and Isaacson 1977). The bottom of the unconfined aquifer is not a continuous surface; in some areas, the bottom is the basalt bedrock and in other areas it is the silt/clay zones of the Ringold Formation.

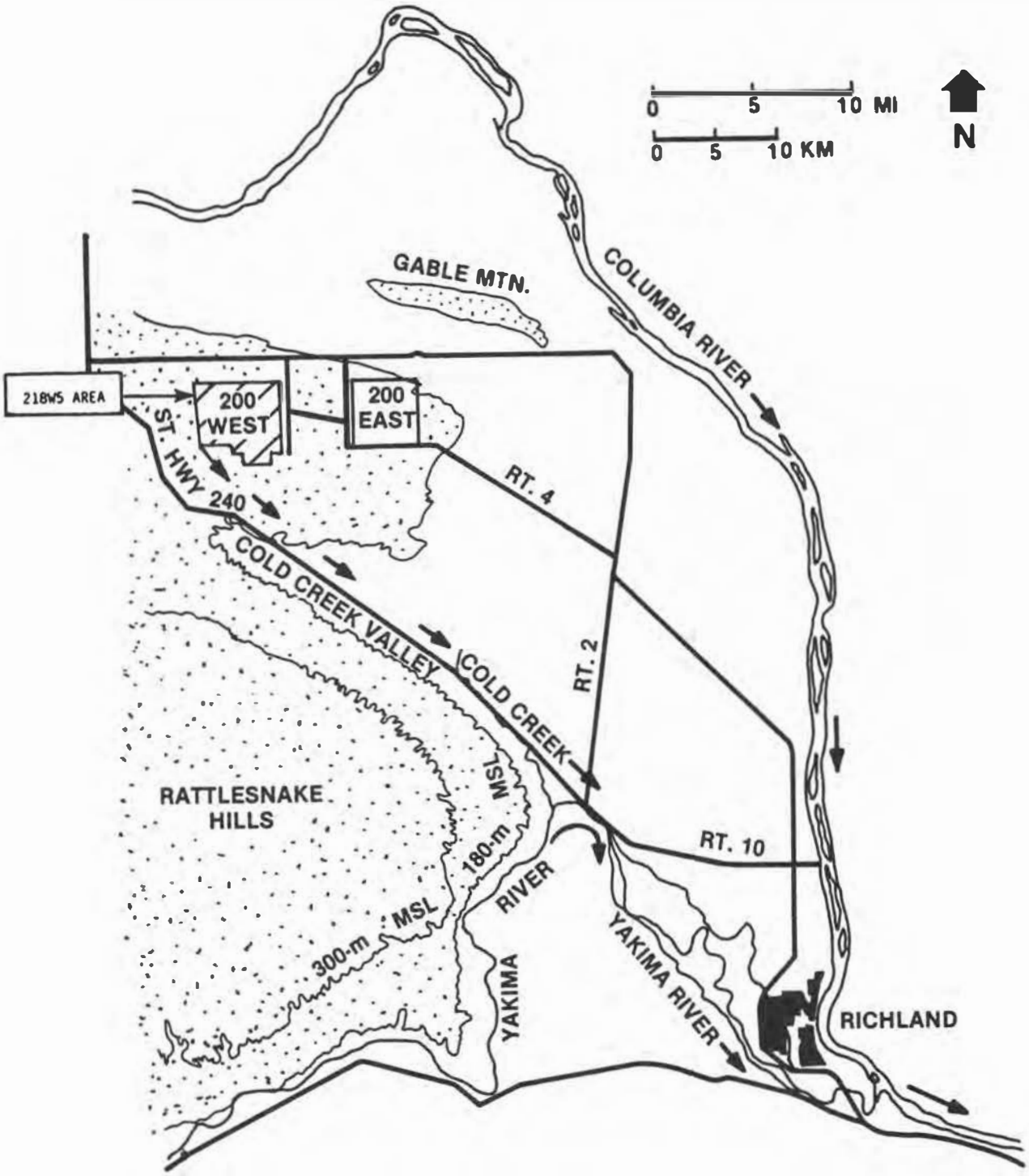

Figure 3.13. Surface Water Drainage from the 200-West Area on the Hanford Site. Source: Modified from U.S. Department of Energy (1986b).

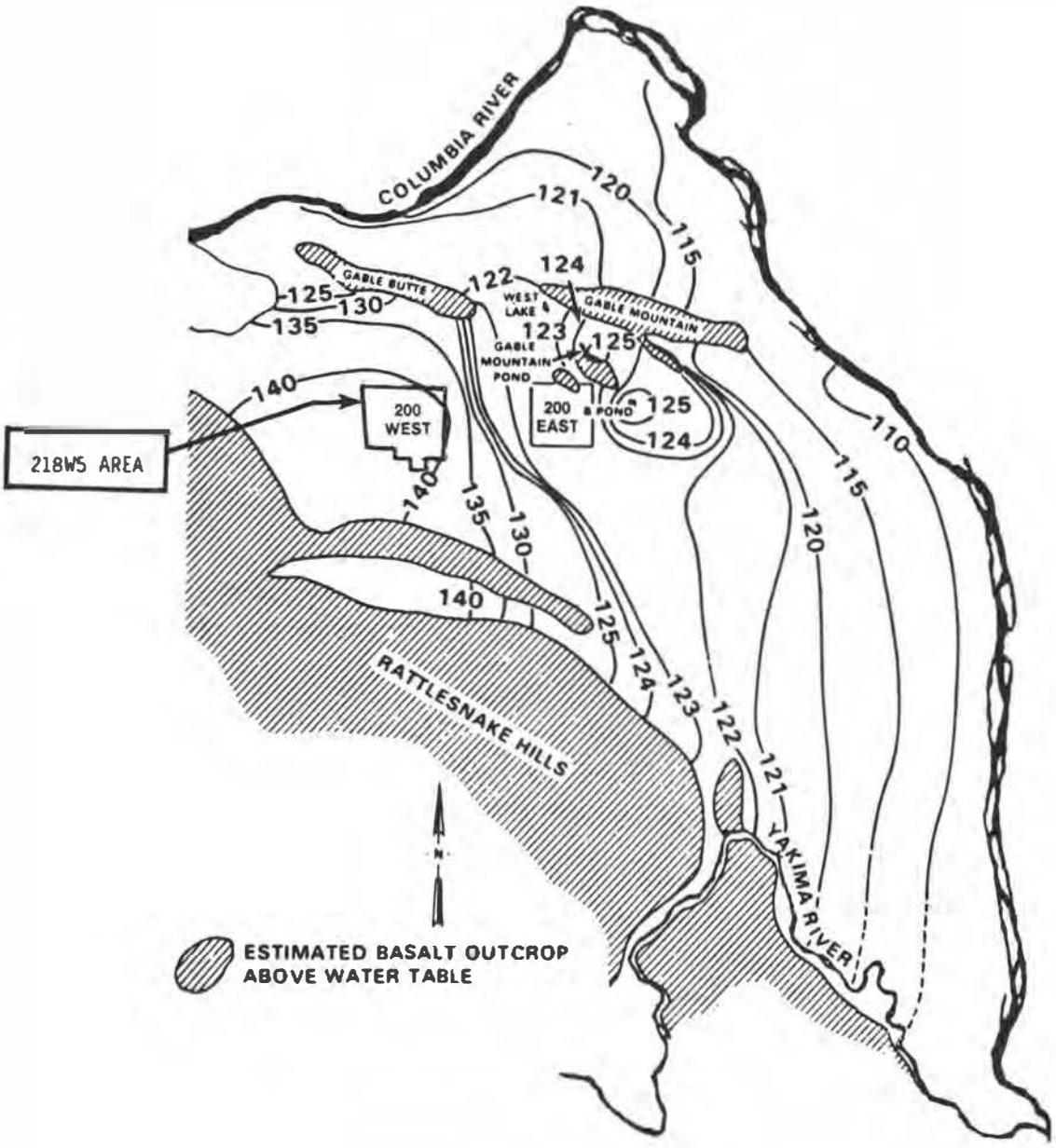

Figure 3.14. Water Table Contour Map for the Hanford Site, December 1982 (contours in meters above MSL). Source: Modified from U.S. Department of Energy (1986a).

Precipitation that recharges the water table ranges from 0 to 5 cm (2 in.) per year (U.S. Dept. Energy 1986a). The aquifer receives natural recharge from creek valleys west of the 200 Areas and from runoff along Rattlesnake Hills (Figure 3.13). Artificial recharge enters the aquifer from waste-processing and waste-disposal activities in the 200 Areas. The principal direction of groundwater flow is eastward from the recharge areas to the Columbia River, which is a sink for groundwater discharge from the water table aquifer.

The hydraulic characteristics of the unconfined aquifer are quite variable (U.S. Energy Res. Dev. Admin. 1977). The hydraulic conductivity for the unconfined aquifer ranges from 0.3 to 3,000 m/d (1 to 10,000 ft/d) (U.S. Energy Res. Dev. Admin. 1975). Values for the storage coefficient ranging from 0.0008 to 0.2 have been estimated for the Hanford area from field tests (U.S. Energy Res. Dev. Admin. 1977), but the typical range of storage coefficient values for unconsolidated sediments is 0.05 to 0.4. Hydraulic properties of the upper confined aquifers are presented in several reports (U.S. Energy Res. Dev. Admin. 1975, 1977; Brown and Isaacson 1977; Gephart et al. 1979).

Groundwater in the Hanford area also exists in the interflow zones of the basalt flows and in sedimentary interbeds. The uppermost confined aquifers appear to be hydraulically connected to the overlying unconfined aquifer (Brown and Isaacson 1977). Recharge to the confined aquifers results from precipitation and streamflow in the mountains west of Hanford, and the groundwater in these confined aquifers has the same general west-to-east movement toward the Columbia River.

3.2.3 Climate and Meteorology

The climate of the Hanford region may be characterized as semiarid to arid (U.S. Dept. Energy 1986b). Precipitation averages 16 cm/yr (6.3 in./yr) (period of record, 1912-1970). January has the highest average amount of rainfall, 2.4 cm (0.93 in.); and July has the lowest, 0.36 cm (0.14 in.). The highest 24-hour rainfall was 4.9 cm (1.9 in.) in October 1950. Tornadoes are rare, averaging less than one per year for the entire state. Fourteen tornadoes have occurred within 160 km (100 mi) of the Hanford site since 1916; no loss of life or major damage was associated with any of them. Thunder and lightning storms occur on the average of 12 days per year, mostly in the summer. Hail has been observed at the Hanford Meteorological Station on 16 days in 12 years of record.

At the Hanford Meteorological Station (located between the 200-West and 200-East Areas), prevailing winds are from north-northwest through northwest every month of the year (U.S. Dept. Energy 1986b). These winds are caused in

part by channeling of the air by topographic features. Elevations in this area range from 200 to 220 m (650 to 715 ft) MSL. The strongest winds tend to be from the southwest, and gusts often exceed 64 km/h (40 mph). The average annual wind speed is 12 km/h (7.5 mph), varying from 15 km/h (9.0 mph) in June to 9.7 km/h (6.0 mph) in November. This unusual annual cycle of wind speeds is caused by strong drainage winds from the Cascade Mountains during summer evenings and nights. Because of the nearby mountains, wind speed and direction at any one time are known to vary from place to place across the Hanford site (U.S. At. Energy Comm. 1972). The average summer temperature is 23°C (73°F), and the average winter temperature is 0°C (32°F). The hottest and coldest temperatures ever recorded were 46°C (115°F, July 1939) and -33°C (-27°F, December 1919).

3.2.4 Ecology

The sagebrush/cheatgrass or Sandberg's bluegrass community is the most broadly distributed plant community of the Hanford site (U.S. Energy Res. Dev. Admin. 1975) and characterizes the 200-West Area (Uresk et al. 1977). This plant community is dominated by low, widely spaced shrubs along with various grasses and forbs. Between 45% and 75% of the ground surface is covered by vegetation (Uresk et al. 1977), and wildfires are common. Recovery of vegetation following fires is slow because sagebrush is killed by burning and large perennial grasses that can survive fire are absent (U.S. Energy Res. Dev. Admin. 1975).

Small mammals, especially the Great Basin pocket mouse, are numerous on the Hanford site; deer mouse, Townsend's ground squirrel, and northern pocket gopher are locally abundant. Other mammals on the site include mule deer (most abundant big game species), mountain cottontail rabbit (most abundant small game species), raccoon (most abundant furbearer), coyote (more abundant on than around the site), badger (present in low numbers), porcupine (widely distributed), bobcat (present in low numbers), and whitetail jackrabbit (widely distributed) (U.S. Energy Res. Dev. Admin. 1975). It is believed that badgers burrowed into wastes in a crib disposal area during the 1950s and early 1960s (Uresk et al. 1977), allowing jackrabbits access to radioactive materials. These materials were subsequently dispersed to other portions of the site via rabbit feces and wind or water dispersion of loose contaminated materials brought to the surface by the rabbits. Badger burrows (during the breeding season) can be 1.5 to 9.1 m (5 to 30 ft) in length and 0.6 to 0.9 m (2 to 3 ft) in depth (Schwartz and Schwartz 1959). Coyotes may also dig dens during their breeding season, but frequently use former dens of other animals such as badgers (Schwartz and Schwartz 1959). The Great Basin pocket mouse, northern pocket gopher, and Townsend's ground squirrel are also active burrowers (Burt and Grossenheider 1964).

About 27 bird species are known to breed in the shrub-steppe ecosystem occurring on the Hanford site (Uresk et al. 1977). However, only a few species are abundant, including the western meadowlark, horned lark, sage sparrow, and black-billed magpie. Raptors occurring on the site include the red-tailed hawk, Swainson's hawk, great horned owl, golden eagle, and bald eagle; the latter two are only winter visitors (U.S. Energy Res. Dev. Admin. 1975). The Canada goose is the most important nesting waterfowl in the vicinity, inhabiting the islands in the free-flowing reach of the Columbia River. Great numbers of other waterfowl, mostly mallards, use the Columbia River near the site for resting during migration (U.S. Energy Res. Dev. Admin. 1975).

Invertebrates are important components of the shrub-steppe ecosystem. Beetles and ants are often the major herbivores, and they perform important roles in the soil community such as soil aeration and detrital processing.

No aquatic habitat occurs in the immediate 200-West Area. An ephemeral stream (Cold Creek) drains the area and, during periods of flow, probably supports biota characteristic of intermittent desert streams -- e.g., hemipterans, ostracods, and algae (Cole 1968). Several ponds and springs also occur on the site. The Columbia River, which is the major aquatic habitat in the area, contains 39 species of fish in the vicinity of the Hanford site, including salmon (sockeye, chinook, and coho), steelhead trout, whitefish, and sturgeon (U.S. Energy Dev. Res. Admin. 1975).

No endangered or threatened species of vascular plants are known to occur in the 200-West Area (U.S. Energy Res. Dev. Admin. 1975; Uresk et al. 1977). Federally endangered raptors that migrate across the area are the peregrine falcon (Falco peregrinus) and bald eagle (Haliaeetus leucocephalus). Several rare bird species, which do not have a designated status, also occur on the site: the ferruginous hawk (Buteo regalia) (nests in low numbers), American osprey (Pandion haliaetus) (only a visitor), western burrowing owl (Speotyto cunicularia hypugaea) (nests in low numbers), and long-billed curlew (Numenius americanus) (nests in low numbers) (U.S. Energy Res. Dev. Admin. 1975).

3.2.5 Land Use, Visual and Cultural Resources

The Hanford site is divided into six major subareas: (1) the 100 Areas where plutonium production reactors are located, (2) the 200 Areas (East and West) where nuclear fuel processing, waste processing, and waste management installations are situated, (3) the 300 Area where there are laboratories and a fuel-fabrication installation, (4) the 400 Area which contains the Fast Flux Test Reactor, (5) other multipurpose areas, and (6) the Arid Lands Ecology Reserve, a 310-km^2 (120-mi^2) area set aside by the Atomic Energy Commission in 1968 for ecological studies (U.S. Energy Res. Dev. Admin. 1975, 1977;

U.S. Dept. Energy 1986c). The site also contains a wildlife refuge north of the Columbia River that is operated by the U.S. Fish and Wildlife Service and a controlled hunting area that is operated by the Washington State Department of Game (U.S. At. Energy Comm. 1972). The proposed waste disposal area, the 218W5 Area, is located in the western portion of the 200-West Area (U.S. Dept. Energy 1986b).

The land-use and zoning status of areas surrounding the Hanford site is varied (U.S. Energy Res. Dev. Admin. 1977). Land use includes residential, commercial, industrial, agricultural, recreational, and undeveloped areas. Agricultural land use predominates, reflecting its economic importance for the region.

West of the Hanford site is the Yakima Indian Reservation on which several bands and tribes reside (Earth Co. 1976). From prehistoric to contemporary times, cultural and natural resources -- including plants, animals, and water -- have had importance for the Confederated Tribes and the Yakima Indian Nation (Yakima Indian Nation 1985). Environmental resources on and near the Hanford site may have cultural and religious importance for the Yakima Indian people, and environmental degradation could have serious cultural consequences (Jim 1980; Carrel 1984). The Yakima Nation and other tribes and nations in the Hanford area have expressed concerns about nuclear waste disposal at the Hanford site (Tomaskin 1979; Totus, undated).

The Hanford site is sparsely vegetated, contains little topographic relief, and has numerous man-made structures such as roads and buildings. Aesthetic features of this area include the Hanford reach of the Columbia River; the White Bluffs, rising 100 to 165 m (340 to 540 ft) above the eastern shoreline of the river; and Rattlesnake Mountain at the southwestern edge of the site (U.S. Dept. Energy 1986c). However, some of this area is restricted from public use.

Prehistoric, historic, and ethnohistoric sites are reported throughout the environment of the Columbia Plateau. Site locations include riverine valleys as well as inland locations (Warren et al. 1963; Willey 1966; Rice 1968b; Dancy 1973; Ames and Marshall 1980). Some of these sites may have importance to the Yakima Indian Nation (Yakima Indian Nation 1985).

According to a report of the U.S. Department of Energy (1986a), there are 10 major archeological sites on or adjoining the Hanford site, and most of these are located on the islands or shoreline of the Columbia River. There are a total of 115 archeological sites on or adjoining the Hanford site, including open camps, fishing stations, house pit sites, cemeteries, and flaking floors. Two sites are located north of the 200 Areas near Gable Mountain and Gable Butte, and two others lie on the western part of the

Hanford site at Rattlesnake Springs and Snively Canyon; however, no known sites are located within the 200 Areas. A detailed description of the location and character of a number of these sites is presented in reports of Rice (1968a, 1968b) and the U.S. Energy Research and Development Administration (1975).

An intensive cultural resource survey has not been made in the proposed disposal area (Wash. Off. Archaeol. Hist. Preserv. 1983). The DOE Richland Office is currently reviewing the cultural resource management of this area.

3.2.6 Population and Socioeconomics

The Hanford site is located primarily in Benton County, Washington, with portions in Franklin and Grant counties. The closest population center is the city of Richland, which is located about 48 km (30 mi) southeast of the 200-West Area. From 1970 to 1980, the population in all three counties increased about 15% to 20%, but Benton County increased by about 62%. The city of Richland experienced a 28% increase over this same time period. Between 1981 and 1984, population declined in both Benton and Franklin counties -- at 7% and 4%, respectively. The cause of the population change was the phasedown of construction of the Washington Public Power Supply System nuclear power plants (U.S. Dept. Energy 1986c).

Because the Hanford site is a federal installation dedicated to nuclear energy research and production as well as a natural reserve, there are no population centers or public facilities such as schools and hospitals within an 8-km (5-mi) radius of the 200-West Area. Recent population declines in nearby Benton and Franklin counties have relieved pressure on public services such as schools, public parks, police and fire protection, water supply, and sewage treatment (U.S. Dept. Energy 1986c). The economy of the counties near the Hanford site is based primarily on agriculture and on energy research and nuclear material production.

3.2.7 Existing Radiological Environment

The Hanford site and its vicinity contains several nuclear-related installations that have been operating since 1943. At one time, nine federal government plutonium production reactors were in operation, but only one remains active. Other government facilities include a spent nuclear fuel reprocessing plant, nuclear fuel manufacturing facilities, Pacific Northwest Laboratories (research and development laboratories), and the Fast Flux Test Facility. Radioactive wastes are stored or disposed of in the 100, 200, and 300 Areas. Private or leased facilities include the Washington Public Power Supply System nuclear generating station and a low-level radioactive waste

burial site (located between the 200-West and 200-East Areas). The Exxon Nuclear fuel fabrication plant is located on private land south of the Hanford site.

Liquid radioactive wastes are stored on the Hanford site in large underground tanks. Process liquids containing dilute low-level liquid wastes have been disposed of in cribs and evaporation ponds (U.S. Energy Res. Dev. Admin. 1975). Solid low-level radioactive wastes are buried in trenches, and solid transuranic radioactive wastes are stored in a retrievable manner. The radioactive waste inventory at Hanford is 233,000 m^3 (420 million Ci) of high-level radioactive wastes in the form of liquid sludge and salt cake stored in underground tanks, 340,000 m^3 of low-level radioactive wastes (5.9 million Ci), 91,700 m^3 of buried transuranic radioactive wastes, and about 13,700 m^3 of retrievably stored transuranic radioactive wastes (Oak Ridge Natl. Lab. 1985).

The Hanford area has an extensive environmental monitoring program to determine the contribution of Hanford activities to radiation levels (Price 1986). Existing nuclear activities on the Hanford site have been estimated to result in a 7 person-rem/yr dose to the surrounding population. This dose is only a small fraction of the 34,000 person-rem/yr dose received by this same population from natural background sources of radiation. The dose to the maximally exposed individual as a result of nuclear activities at the Hanford site is estimated to be 0.1 mrem/yr; such an individual would receive a dose of approximately 100 mrem/yr from background radiation sources (Price 1986).

3.3 "NEARBY SITE"

The general description of the "Nearby Site" as an alternative disposal site is based on its assumed location in Missouri within a 160-km (100-mi) radius of the Weldon Spring site (referred to as the study area). No specific candidate location has been identified. It is assumed that the "Nearby Site", which would be owned and operated by DOE, would have more favorable conditions (e.g., thicker clay, lower hydraulic conductivity, deeper groundwater table, and/or higher sorption capacity) than the Weldon Spring site, and these conditions would delay the migration of radionuclides into groundwater even longer than at the raffinate pits area.

3.3.1 Topography, Geology, Soils, Seismology, and Mineral Resources

The topography of Missouri within a 160-km (100-mi) radius of the Weldon Spring site can be characterized by two distinct physiographic regions: the Central Lowland Province in the north and west, and the Ozark Plateaus Province in the south (Figure 3.1). The plains or prairies north of the Missouri River are gently undulating and have altitudes ranging from about 150

to 210 m (500 to 700 ft) MSL. This area was glaciated twice during the Pleistocene, and the till deposits are thin and dissected. The Ozark Plateaus Province -- which contains the St. Francois Mountains -- is an elevated, ruggedly dissected plateau of lower Paleozoic rocks that has undergone mild uplifts since Paleozoic time. Upland elevations range from 300 to 520 m (1,000 to 1,700 ft) MSL, and relief in some areas is as high as 210 m (700 ft) (U.S. Geol. Surv. et al. 1967; Miller et al. 1974; Vineyard and Feder 1982).

Exposed Precambrian felsitic volcanic rocks, granitic rocks, and basaltic and diabasic dikes and sills occur in the St. Francois Mountains. The bedrock geology of the area is composed of generally flat-lying, interbedded sequences of Paleozoic age dolomite, limestone, and shale. Unconsolidated Pleistocene age alluvial and eolian deposits, ranging from 0 to 33 m (0 to 110 ft) in thickness, overlie the bedrock formations along the Mississippi, Missouri, and other river valleys in the study area (Duley 1983). In general, the alluvium becomes coarser-grained with depth. A generalized stratigraphic column for the "Nearby Site" is given in Table 3.9.

In eastern Missouri, a karst topography has developed on the carbonate bedrock of the area (Figure 3.15). This topography -- characterized by such features as sinkholes, caves, and underground drainage -- is formed by dissolving of limestone, dolomite, or gypsum.

Several major fault systems associated with the Ozark uplift and the Mississippi Embayment are present about 260 km (160 mi) south and southeast of the Weldon Spring site, the largest of which is the New Madrid seismic zone (McKeown 1978). According to Hopper et al. (1983), if the epicenter of an 1811-sized earthquake is located anywhere along the New Madrid seismic zone, the study area could experience Modified Mercalli intensities ranging from VII to IX.

Missouri is a leading producer of lead, zinc, and barite and an important supplier of stone, cement, and iron ore (U.S. Geol. Surv. et al. 1967). The known metallic deposits in the study area are found in the southern part and mostly occur in the Precambrian rocks and the overlying upper Cambrian strata (U.S. Geol. Surv. et al. 1967). Most of the lead ores in the southeastern portion of the study area -- together with associated amounts of copper, cobalt, nickel, silver, and cadmium -- occur in the dolomite of the Bonneterre Formation. Virtually all the active metal mines are in or near the St. Francois Mountains and the surrounding area of the center of the Ozark dome. Some of the nonmetallic minerals, e.g., most of the dolomite, occur in the Cambrian and lower Ordovician rocks. Silica sandstone suitable for industrial use crops out in a single belt between Cape Girardeau and Montgomery counties. The study area contains large reserves of lead ore and other potential nonmetal resources. According to Nash (1977), the Precambrian

Table 3.9. Generalized Stratigraphic Column for the "Nearby Site"

System	Formation	Thickness (ft)	Dominant Lithology
Cenozoic			
Quaternary	Alluvium[a]	0-150	Sand, gravel, silt, and clay
	Loess	0-110	Silt
	Glacial till	0-55	Pebbly clay and silt
Paleozoic			
Pennsylvanian	Undifferentiated	0-75	Shales, siltstones, "dirty" sandstones, coal beds and thin limestone beds
	Undifferentiated	0-90	
	Undifferentiated	0-200	
	Undifferentiated		
Mississippian	Ste. Genevieve Formation	0-160	Argillaceous to arenaceous limestone
	St. Louis Limestone	0-180	
	Salem Formation	0-180	
	Warsaw Formation	0-110	
	Burlington-Keokuk Limestone	0-240	Cherty limestone
	Fern Glen Formation	0-105	Red limestone and shale
	Undifferentiated	0-122	Limestone, dolomitic limestone, shale, and siltstone
Devonian	Bushberg Sandstone	0-60	Limestone and sandstone
	Glen Park Limestone		
	Grassy Creek Shale	0-50	Fissile, carbonaceous shale
Silurian	Undifferentiated	0-200	Cherty limestone

Table 3.9. Continued

System	Formation	Thickness (ft)	Dominant Lithology
Paleozoic			
Ordovician	Maquoketa Shale	0-163	Silty, calcareous or dolomitic shale
	Cape Limestone	0-5	Argillaceous limestone
	Kimmswick Formation	0-145	Massive limestone
	Decorah Formation	0-50	Shale with interbedded limestone
	Plattin Formation	0-240	Finely crystalline limestone
	Rock Levee Formation	0-93	Dolomite and limestone, some shale
	Joachim Dolomite	0-135	Primarily argillaceous dolomite
	St. Peter Sandstone	0-160	Silty sandstone, cherty lime stone grading upward into quartzone sandstone
	Everton Formation	0-130	
	Powell Dolomite	0-150	Sandy and cherty dolomites and sandstone
	Cotter Dolomite	0-320	
	Jefferson City Dolomite	0-225	
	Roubidoux Formation	0-177	
	Gasconade Dolomite Gunter Sandstone Member	0-280	
Cambrian	Eminence Dolomite	0-172	Cherty dolomites, siltstones, sandstones, and shale
	Potosi Dolomite	0-325	
	Derby-Doerun Dolomite	0-165	
	Davis Formation	0-150	
	Bonneterre Formation	245-385	
	Lamotte Sandstone	235+	
Precambrian			Igneous and metamorphic rocks

[a] Basal part may be of Pleistocene age.

Conversion Factor: To convert feet to meters, multiply by 0.3048.

Sources: Miller et al. (1974); Anderson et al. (1979).

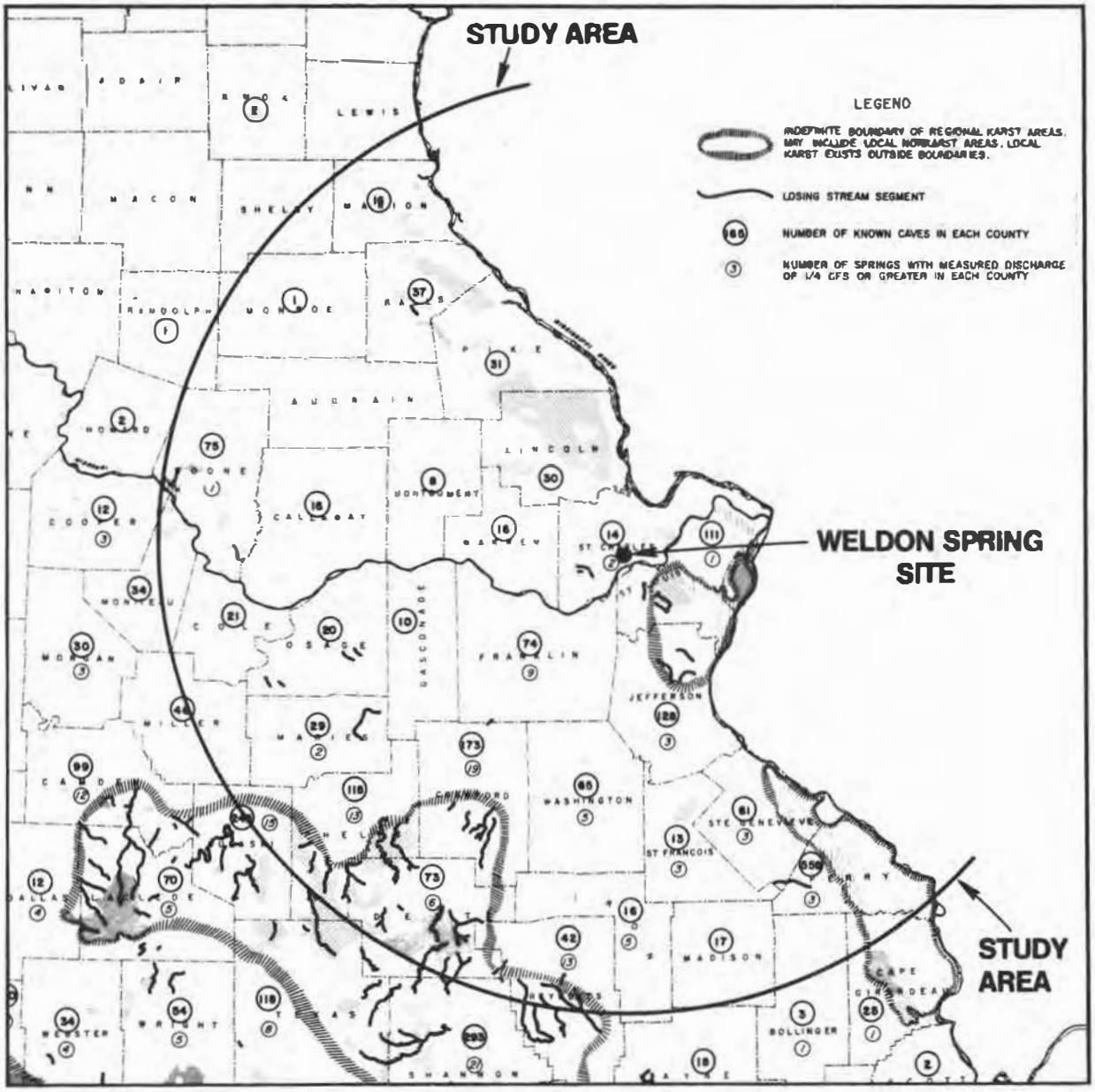

Figure 3.15. Karst Areas of the "Nearby Site".
Source: Modified from Duley (1983).

granites exposed in the St. Francois Mountains are among the most uraniferous igneous rocks in the United States, with the most radioactive being the Graniteville granites that contain an average of 16.9 ppm (12 pCi/g) natural uranium.* The Butler Hill and Breadtray granites are also enriched in uranium, averaging 6.2 and 5.6 ppm (4.5 and 4.0 pCi/g) natural uranium, respectively (Nash 1977).

Stohr et al. (1981) evaluated and defined the geologic requirements necessary for the exploration, development, and operation of hazardous-waste sites in Missouri. According to their evaluation, a small area in the northwest part of the study area has slight to moderate limitations and the majority of the study area has moderate to severe limitations for the disposal of hazardous waste (Figure 3.16).

3.3.2 Hydrology, Water Use, and Water Quality

3.3.2.1 Surface Water

The principal surface-water hydrologic features in the study area can be characterized by three river basins: the Meramec, Missouri, and Mississippi rivers. These rivers have a large amount of available surface water resources and provide most of the water used in the study area. Developments that require large water supplies and/or waste dilution naturally tend to concentrate along these rivers. The level of the Missouri River is controlled by an extensive reservoir system in the headwater areas, and the flooding potential has been greatly reduced by this system. Summer flows are maintained at levels that ensure adequate depths for navigational purposes. The Mississippi River, on the other hand, is not significantly controlled at medium and high stages above the confluence with the Missouri. Navigation depths are maintained by a system of locks and dams that alter mean and high flows very little; thus, flooding is a more frequent problem on the Mississippi River (Miller et al. 1974). The Meramec River rises in the Salem Plateau, flows generally northeasterly, and enters the Mississippi River about 19 km (12 mi) south of St. Louis. Surface elevations range from 460 m (1,500 ft) MSL at its headwaters to about 140 m (450 ft) at its junction with the Mississippi River. There are many sinkholes in the Meramec River basin. Many springs in the basin contribute to the high, sustained base flow of the Meramec and many of its tributaries (Vineyard and Feder 1982).

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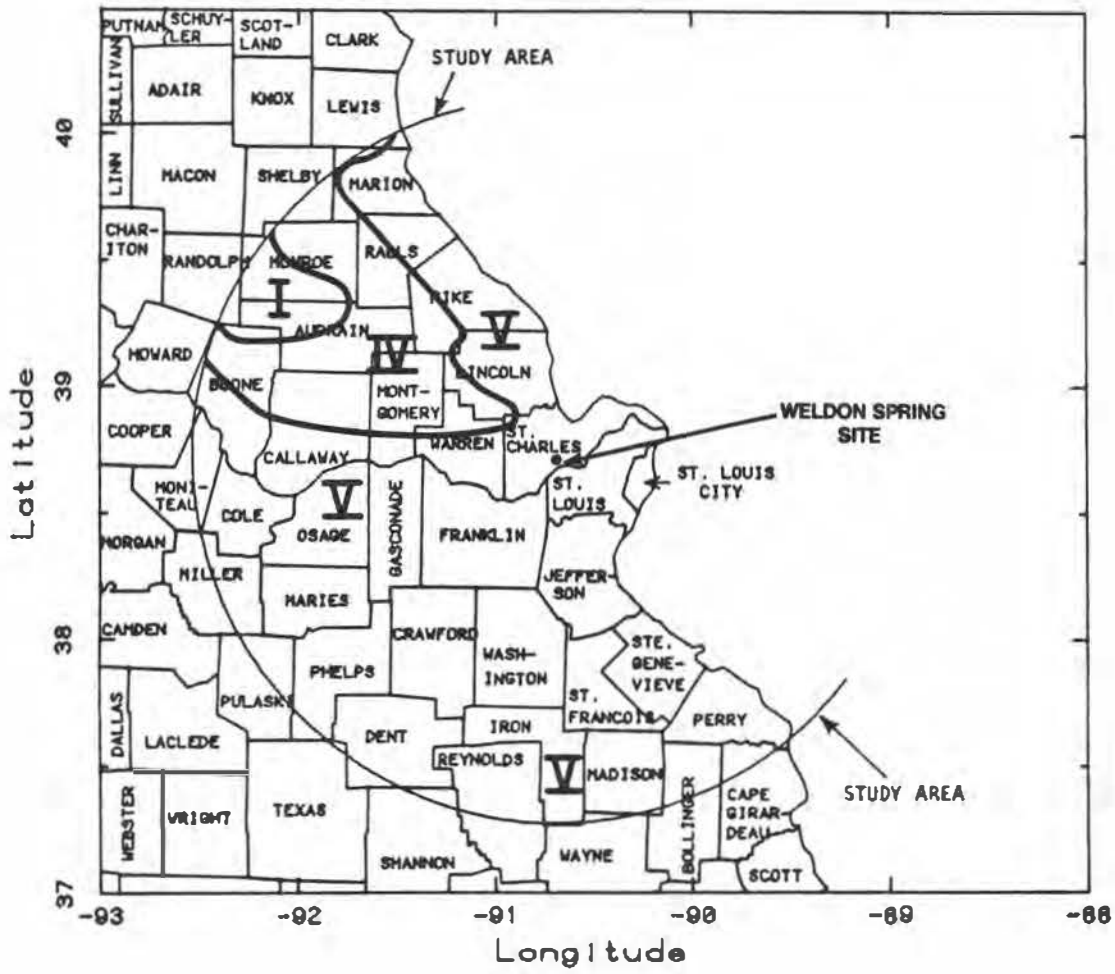
*The activity concentration of natural uranium (in pCi/g) can be calculated from the mass concentration (in ppm) by multiplying the mass concentration by 0.724.



LEGEND

Limitations Apply to Exploration and Operation

REGION I
Slight to moderate limitations
Glacial drift - 12 to 31 m (40 to 100 ft)
Shale bedrock

REGION II
Slight to moderate limitations
Residual clayey soil - some thickness >12 m (40 ft)
Shale bedrock

REGION III
Moderate limitations
Glacial drift - >31 m (100 ft)
Widespread deposits of water-bearing sand and gravel
Shale bedrock

REGION IV
Moderate limitations
Glacial drift - varying thickness
Limestone bedrock

REGION V
Severe limitations
Loess, alluvium, residual soils - varying thickness
Limestone, dolomite bedrock
Major recharge region

Figure 3.16. Geologic Limitations Map for Hazardous Waste Isolation in Missouri as Applicable to the "Nearby Site". Regions II and III do not exist in the study area for the "Nearby Site". Source: Modified from Stohr et al. (1981).

Flooding in the study area is most common during March through August. Heavy spring rains cause most of the floods, but some of the greatest floods on record have occurred in the summer as a result of intense, local summer thunderstorms. Flood problems may become more severe on tributary streams in the area as industrial and domestic development increases on the floodplains, causing an increase of impervious areas. A summary of maximum recorded floods and 50-year floods for the three major rivers in the study area are presented in Table 3.10.

Table 3.10. Summary of Maximum Recorded Floods and 50-Year Floods for Three Major Rivers

	Maximum Flood				
Location	Drainage Area (km^2)	Date of Maximum Discharge	Discharge (m^3/s)	Maximum Gage Height (m MSL)	50-Year Flood (m^3/s)
Mississippi River at Alton	439,000	Apr 29, 1973	15,200	131.7	14,000
Missouri River at Hermann	1,350,000	Jun 1844	25,300	157.6	20,400
Meramec River near Eureka	9,700	Aug 22, 1915	5,000	136.1	2,900

Conversion Factors: To convert square kilometers (km^2) to square miles (mi^2), multiply by 0.3861; to convert cubic meters (m^3) to gallons (gal), multiply by 264.2; to convert meters (m) to feet (ft), multiply by 3.281.

Source: Miller et al. (1974).

Mississippi River water is moderately mineralized and is a calcium bicarbonate type that contains significant amounts of magnesium and sulfate in the dissolved solids. The water is hard and some treatment such as softening is desirable for municipal and some industrial uses. Water in the Missouri River is moderately mineralized. The predominant chemical constituents are calcium, magnesium, sodium, bicarbonate, and sulfate. Turbidity is relatively high, and the water is hard and must be treated for most uses. Water from the Meramec River is a calcium carbonate type. Calcium and magnesium are the predominant constituents dissolved in the water. The water is hard and turbidity is normally low (Vineyard and Feder 1982).

3.3.2.2 Groundwater

The primary source of groundwater recharge in the study area is precipitation. The amount of recharge from precipitation to bedrock depends upon the general configuration and physical character of the land surface, the amount and type of vegetation, the distribution and quantity of precipitation, and the composition and moisture content of the soil and underlying rocks. In areas where bedrock is exposed at the surface, recharge to the groundwater reservoir is minimal and almost all of the precipitation leaves the area directly as runoff. Shallow bedrock aquifers in hydraulic connection with rivers also receive recharge from natural infiltration of the rivers primarily during sustained high river stage and overbank flooding. Alluvial aquifers in the area, being located along the river valleys, are recharged by infiltration of stream water, by direct precipitation, and by underflow from the underlying and adjacent bedrock.

Locations of measured springs in the study area are given in the report of Vineyard and Feder (1982). Many springs contribute to the high sustained base flow of the Meramec River and many of its tributaries. Springs in the Meramec River basin yield moderately mineralized water. Calcium, magnesium, and bicarbonate are the predominant constituents dissolved in the water. The dissolved-solids content of spring water throughout the basin ranges from about 120 to 340 mg/L. Hardness ranges from about 96 to 340 mg/L (Vineyard and Feder 1982). The springs in northern Missouri are generally very small because of the limited storage in limestone, dolomite, sandstone, and shale rocks. Most of the saline springs are located along the freshwater-saltwater interface in some areas along the Mississippi River. Western counties (Phelps, Maries, Pulaski, Casconade, and Osage) in the study area have one of the largest concentrations of big springs in the state. Spring water in these counties is a calcium-magnesium bicarbonate type. The waters are moderately mineralized with a dissolved-solids content ranging from about 130 to 310 mg/L (Vineyard and Feder 1982). Springs are few in the southern part of study area, including St. Francois, Iron, Madison, and Reynolds counties. Water from most springs in this area is a moderately mineralized calcium-magnesium bicarbonate type. The dissolved-solids content ranges from about 55 to 280 mg/L and hardness ranges from about 49 to 320 mg/L (Vineyard and Feder 1982). Several springs in Crawford, Iron, and Phelps counties, the southwestern part of the study area, are used for fish hatcheries and rearing ponds (Vineyard and Feder 1982).

3.3.3 Climate and Meteorology

The climate of the area is similar to the climate of the Weldon Spring site (see Section 3.1.3). Climatological data from Columbia, Jefferson City,

and the Callaway Nuclear Power Plant (located in Callaway County, west of the Weldon Spring site--U.S. Nucl. Reg. Comm 1975) were used to describe the meteorological characteristics at the "Nearby Site".

The climate can be described as continental and is characterized by rapid changes in temperature and marked extremes, resulting in hot summers and moderately cold winters. The study area lies near the principal track of winter and spring storms that move northeast and east through the region. Mean monthly temperatures range from about -1.1°C (30°F) in January to about 21°C (70°F) in July. Record maximum and minimum temperatures of 45°C (113°F) and -32°C (-26°F) were reported at Columbia, Missouri.

The annual average precipitation is about 94 cm (37 in.). The maximum mean monthly precipitation of about 12 cm (4.7 in.) occurs in May, and the minimum mean monthly precipitation of about 4.3 cm (1.7 in.) occurs in January. The maximum 24-hour rainfall at Columbia was about 17 cm (6.6 in.) in September 1918. Annual snowfall averages about 48 cm (19 in.). At Columbia, the maximum monthly snowfall in the last 20 years was 64 cm (25 in., March 1960), and the maximum 24-hour snowfall was 33 cm (13 in., March 1937).

Wind data from the 10 m (30 ft) level at the Callaway site for the period May 4, 1973, through May 4, 1974, indicates a prevailing wind direction from the south (12.4%), with winds from the southeast clockwise through southwest totaling about 49%. Ten years (1960-1969) of wind data from Columbia indicate prevailing winds from the south occur about 13% of the time. Mean wind speeds at Columbia and at the Callaway Nuclear Plant are 15 km/h (9 mph) and 13 km/h (8 mph), respectively.

Severe weather is not uncommon. Thunderstorms can be expected to occur on about 55 days per year, being most frequent during May through August. In the period 1955 through 1967, 13 tornadoes were reported in the 1° latitude-longitude square containing the Callaway Nuclear Power Plant, with a mean annual frequency of 1.0. The computed recurrence interval for a tornado at the plant site is 1,300 years.

From 1936 to 1970, about 11 atmospheric stagnation cases, totaling about 47 days, were reported in the Callaway site area. The maximum monthly frequency is in August.

3.3.4 Ecology

The "Nearby Site" is located within the Prairie Parkland province (Bailey 1978; Galvin 1979), which is comprised of the Oak-Hickory Forest (northern) subsection and the Bluestem Prairie, Oak-Hickory Forest (northern) subsection. The former predominates within the Mississippi and Missouri river

valleys whereas the latter is more prevalent away from the rivers (Galvin 1979). The biotic composition at the "Nearby Site" would be similar to that occurring within the Weldon Spring vicinity (see Section 3.1.4).

A number of state and federally listed rare and endangered species occur within the Missouri counties located within a 160-km (100-mi) radius of the Weldon Spring site. These species are listed in Appendix G, Table G.2.

3.3.5 Land Use, Visual and Cultural Resources

Land use within the study area of the "Nearby Site" is a mosaic of urban, industrial, and residential areas mixed with large areas dedicated to farms. In general, residential and industrial properties would be the least suitable land types for a waste-disposal site. Rural agricultural sites could be more suitable from a demographic perspective because of lower population densities. Zoning could be a limiting factor with regard to locating a waste-disposal site in these less-developed areas.

The visual resources of the "Nearby Site" are very diverse due to the complex topography and land use. Site-specific information would be required in order to identify and characterize the major viewsheds and points of scenic interest.

The study area of the "Nearby Site" has a rich and complex cultural history that extends from Paleo-Indian through to the Historic Period. In the nearby American Bottoms, large agricultural populations associated with the Mississippian tradition were established. On the alluvial floodplains of this area, there is the largest concentration of platform mounds in the United States (Griffin 1964; Willey 1966). Numerous historic sites and structures remain in the area and are listed in the National Register.

Surveys to identify cultural resource sites have not been systematically made throughout the study area. Therefore, it is likely that a surface and subsurface cultural resource survey would be required to identify and evaluate cultural resources at a specific location.

3.3.6 Population and Socioeconomics

The important demographic criteria for locating a waste disposal site in a given locale are low population density and no or slow population growth (or population loss). Counties with these characteristics generally tend to be agricultural, i.e., have a high proportion of total land area in farms. With the exception of the St. Louis metropolitan area, counties in the study area range in population from about 10,000 to 100,000 and in population density

from 8 to 15 persons/km^2 (20 to 39 persons/mi^2) up to more than 40/km^2 (100/mi^2). The percentage of people living in urban areas ranges from 0 to 69% in these same counties. Some counties in the region have experienced rapid population growth from 1970 to 1980 whereas others have experienced only modest growth. None lost population from 1970 to 1980.

3.3.7 Existing Radiological Environment

It is assumed that the existing radiological environment in the study area is similar to the average radiological conditions in the state of Missouri. Based on surface soil samples taken from various locations in Missouri, the concentrations of radium-226, thorium-232, and uranium-238 range from 0.31 to 1.4 pCi/g, 0.32 to 1.3 pCi/g, and 0.33 to 1.7 pCi/g, respectively. Corresponding average concentrations are 1.1 pCi/g, 1.0 pCi/g, and 1.1 pCi/g, respectively. Measured external exposure rates range from 4.6 to 10.0 μR/h (Myrick et al. 1981).

3.4 URANIUM PROCESSING SITE

Under Alternative 3c, the raffinate sludge will be transported to an operating uranium processing plant in the Four Corners area (New Mexico, Utah, Colorado, Arizona) of the southwestern United States for reprocessing to recover the uranium remaining in the sludge. A more detailed description of this area can be found in a report of the U.S. Nuclear Regulatory Commission (1980).

3.4.1 Topography, Geology, Soils, Seismology, and Mineral Resources

The Four Corners area is underlain by gently southward-dipping sedimentary strata ranging in age from Precambrian to Cretaceous. The topographic relief of the area is strongly influenced by the weathering patterns of these sedimentary rocks.

Soils in the area are typically classified as entisols, which are young, poorly formed soils. In the central portion of the region, fine sandy loam entisols are underlain by sandy alluvium, and a prominent lime zone occurs below 100 cm (40 in.). Soil depths are shallow (10-50 cm [4-20 in.]) even on gentle to flat slopes (U.S. Nucl. Reg. Comm. 1980).

The area is considered to be tectonically stable, although scattered earthquakes of Modified Mercalli intensities greater than VI have occurred (U.S. Nucl. Reg. Comm. 1980).

3.4.2 Hydrology, Water Use, and Water Quality

The principal surface drainage features in the Four Corners area are the Upper Colorado River and its tributaries. The Upper Colorado River basin is drained by the Colorado River above Lee Ferry, Arizona. The average annual precipitation ranges from less than 15 cm (6 in.) in the arid parts of the basin to more than 150 cm (60 in.) in parts of the Wind River Range and San Juan Mountains. Calculation of an annual water budget for the Upper Colorado River basin, using a basin-wide average precipitation of 40 cm (16 in.), results in a total estimated average precipitation of 1×10^{11} m^3 (3.5×10^{12} ft^3). Of this amount, the greatest water loss from the basin is by evapotranspiration [8×10^{10} m^3 (2.8×10^{12} ft^3)]. Six major rivers flow into the Colorado River in the upper basin from north to south: the Green, Yampa, White, Gunnison, Dolores, and San Juan.

Because water resources in the semiarid region are scarce, one of the critical surface water management issues affecting the Upper Colorado River basin is the apportionment, by interstate compacts, of streamflow from the basin among the states within the basin. The water resources are used for domestic and municipal purposes, irrigated agriculture, industrial demands, energy-related developments, preservation of fish and wildlife, watering of livestock, and maintenance of recreational and scenic values. Eventually, the availability of water may limit regional growth, including accelerated mineral-resource development, increased recreational activities, and the expansion of industrial developments (Yang and Vocke 1981).

Historically, the Colorado River has carried a large load of dissolved minerals. Natural forces and many human activities contribute to salinity in the river. This increase in salinity is the result of the combined effects of irrigation, evapotranspiration, and subsurface inflow (Yang and Vocke 1981). Detrimental effects of rising salinity levels in the Lower Colorado basin have resulted in a treaty with Mexico and in the formation of the Colorado River Salinity Control Forum. A more detailed description of surface water hydrology can be found in a water-availability study by Yang and Vocke (1981).

Groundwater resources in the study area are available in the consolidated strata and unconsolidated deposits. The best aquifer is the Paleozoic carbonates, followed by the Cretaceous and Jurassic sandstones and the Triassic siltstones. The Cretaceous shales and Precambrian rocks have low permeability. The surficial unconsolidated deposits are permeable and, if saturated, yield large amounts of water. The quality of water in these deposits is variable and generally decreases with increasing depth. Permeable strata have better water quality than strata of low permeability. Groundwater uses may include domestic, stock watering, municipal, industrial, and

irrigational. Geohydrological and water-use data will depend upon site-specific conditions for the uranium processing plant.

3.4.3 Climate and Meteorology

As is typical of the southwestern United States, the weather in the Four Corners region is dominated by the influences of elevation and of high and low pressure systems that pass through the area during the year. The area is semiarid, with mild summers and cold winters.

Precipitation is comparatively low, but there are relatively large variations in the seasonal and monthly totals from year to year. Precipitation during late spring and summer occurs largely as thunderstorms. These thunderstorms occasionally lead to tornadoes. Dust devils occur frequently and occasionally cause slight damage to objects in their path. It is common to have winter snow storms with low temperatures and high winds. Dispersion conditions depend upon the topography of the area, such as proximity to plateaus, hills, and general elevation changes.

3.4.4 Ecology

The uranium processing site might be located within one of several vegetative communities due to the diversity of habitat types occurring within the southwestern United States. Following is a general description largely abstracted from a generic environmental impact statement on uranium milling (U.S. Nucl. Reg. Comm. 1980).

Plant communities in the region include ponderosa pine and Douglas fir in the northern mountain area; pinyon-juniper in the northern foothills, along the rim of the Wide Plateau in the south and in dry washes leading down from the rim; and desert shrub in the Wide Plateau (Galvin 1979; U.S. Nucl. Reg. Comm. 1980). The uranium processing site would probably be located in the central portion of the region where the vegetation is predominantly shortgrass prairie subject to heavy grazing. Blue gramma grass and buffalo grass would be the major plant species, with sagebrush and rabbitbrush also occurring.

Common mammals of the region include ground squirrel, jackrabbit, least chipmunk, deer mouse, kangaroo rat, badger, and coyote. Game species include pronghorn antelope, mule deer, desert cottontail, and blue grouse. It is anticipated that aquatic habitat near the uranium processing site would be absent or limited to ephemeral tributary streams. In the latter case, biota would primarily be limited to plankton, invertebrates, and amphibians. The endangered and threatened species cannot be justifiably listed due to the extensive area included and the uncertainty regarding the specific site location.

3.4.5 Land Use, Visual and Cultural Resources

Lands associated with an established uranium processing plant would already have been dedicated to an industrial use. Land use and visual resources near the plant site would be site-dependent. Cultural resource sites are well known in the Four Corners area and include a wide variety of prehistoric, ethnohistoric, and historic cultures and chronologies. The cultural resource sites that are known in the vicinity of the uranium processing plant would be determined once a specific site was selected.

3.4.6 Population and Socioeconomics

Arizona, Colorado, New Mexico, and Utah have significant proportions of their land owned by the federal government (Table 3.11). Population density is low because of the large land areas and relatively small populations. The population per square kilometer for each state varies from 4.1 (New Mexico) to 10.7 (Colorado); these densities have increased with population growth since 1970. According to the U.S. Bureau of the Census, the population in these states (Table 3.11) is classified as urban where there are 2,500 or more people. In New Mexico, 72.1% of the population resides in urban areas. In Arizona, Colorado, and Utah, at least 80% of the population lives in urban areas. The percent total land area devoted to agriculture ranges from a low of 20% in Utah to a high of 62% in New Mexico.

Table 3.11. Selected Demographic and State Characteristics for Potential Uranium Processing Sites, Alternative 3c

State	1980 Population	Land Area (km^2)	Population (no./km^2)	Population Growth Rate, 1970-1980	Percent Urban	Percent Land Owned by Federal Government
Arizona	2,718,000	295,260	9.2	53.1	83.9	44.2
Colorado	2,890,000	269,596	10.7	30.8	80.6	36.4
New Mexico	1,303,000	314,926	4.1	28.1	72.1	33.3
Utah	1,461,000	219,888	6.6	38.0	84.4	63.8

Conversion Factor: To convert square kilometers (km^2) to square miles (mi^2), multiply by 0.3861.

Source: Columns 2 through 6, U.S. Bureau of the Census (1982); Column 7, U.S. Bureau of the Census (1986).

3.4.7 Existing Radiological Environment

Surface soil concentrations of radium-226, uranium-238, and thorium-232 have been measured at locations near Grants, New Mexico. The average surface soil concentrations are 0.97, 0.97, and 0.73 pCi/g, respectively; the average exposure rate is 11 μR/h. These values are all within one standard deviation of the national averages (Myrick et al. 1981).

The radon gas concentration at a site in New Mexico averaged over a 6-year period was 0.24 pCi/L. This is believed to be a conservative average outdoor value for western states (U.S. Nucl. Reg. Comm. 1980--Vol. II).

3.5 TRANSPORTATION ROUTES

The preferred transportation routes for alternatives involving off-site transportation of the Weldon Spring wastes (Alternatives 3a, 3b, and 3c) are described in this section. Route selection criteria, alternate routes, and alternate transportation modes are discussed in detail in Appendix F. The primary criterion for selection of the preferred route is limiting the population dose; thus, the preferred route is the shortest route that has the fewest people living near it. For purposes of analysis in this EIS, it is assumed that the wastes will be transported by rail to the Hanford site or to a uranium processing site (Alternatives 3a and 3c) and by truck to the "Nearby Site" (Alternative 3b). Rail would likely be most economical for the longer distances involved in Alternatives 3a and 3c, whereas truck would likely be most economical for the shorter distance in Alternative 3b.

3.5.1 Local Rail Access

An abandoned 6-km (4-mi) rail spur connects the Weldon Spring site to the main track of the MKT Railroad. This spur passes within 91 m (300 ft) of Pits 1 and 2, and a secondary spur branches into the quarry area. Most of the right-of-way is intact, but the spur would require reconstruction (Bechtel Natl. 1984a). In addition, turnouts would have to be constructed for loading, decontamination, and train make-up.

The MKT Railroad would pick up a completed train with its locomotive and caboose. For transport to the Hanford site (Alternative 3a), the MKT Railroad would take the train to an interline location near St. Louis and transfer the train to the Burlington Northern Railroad for the remainder of the trip to the Hanford site. For transport to the Four Corners area (Alternative 3c), a likely rail routing would be the MKT Railroad to southeast Kansas where the train would be transferred to the Santa Fe Railroad.

The MKT Railroad filed a petition with the Interstate Commerce Commission on August 21, 1986, requesting permission to abandon the 320-km (200-mi) rail line from Machens to Sedalia, Missouri. If MKT is granted this request, the availability of a local rail line to either the Burlington Northern Railroad (for Alternative 3a) or the Santa Fe Railroad (for Alternative 3c) could be comprised. However, DOE could utilize the abandoned rail line after determining its acceptability and performing any required upgrading and maintenance. If Alternative 3 is selected, DOE will reevaluate the various transportation alternatives to ensure that the most expedient and safe means are used.

3.5.2 Local Truck Access

Current access roads to the raffinate pits area and the quarry from Missouri State Route 94 are not adequate for heavy truck traffic. These roads and their intersections with State Route 94 will have to be upgraded for trucks hauling fill material and wastes from the vicinity properties and for trucks hauling wastes from the Weldon Spring site if they are to be shipped to another location by truck. The location of State Route 94 relative to the raffinate pits, chemical plant, and quarry is shown in Figure 3.7.

3.5.3 Route to Hanford

Rail access is available at both the Weldon Spring site and the Hanford site for Alternative 3a. The preferred rail route is the MKT Railroad to St. Louis and the Burlington Northern Railroad from St. Louis to Hanford. The route is 3,500 km (2,200 mi) long and passes through Missouri, Iowa, Nebraska, Wyoming, Montana, Idaho, and Washington (see Appendix F, Section F.2). The average freight volume along the route is 18 million tons per year, and the average population density of counties along the route is 10 persons/km^2 (26 persons/mi^2).

3.5.4 Route to "Nearby Site"

Because of the short distance, truck transport would be most economical for waste transport to the "Nearby Site" (Alternative 3b). Local access to the interstate system is available via State Route 94 to U.S. 40/61 and then on U.S. 40/61 to Interstate 70, a total distance of 18 km (11 mi). The route is assumed to pass through fairly rural counties with a population density of 34 persons/km^2 (88 persons/mi^2).

3.5.5 Route to Uranium Processing Site

It is assumed that the uranium processing site for Alternative 3c is in the vicinity of Grants, New Mexico, and that rail will be used for this

alternative. The preferred route is the MKT Railroad to Chanute, Kansas, and the Santa Fe Railroad from there to Grants, New Mexico. This route is 1,900 km (1,200 mi) long and passes through Missouri, Kansas, Oklahoma, Texas, and New Mexico (see Appendix F, Section F.2). The average freight volume along the route is 24 million tons per year, and the average population density of counties along the route is 9 persons/km^2 (24 persons/mi^2).

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*In this reference list, (1) the term "personal communication" is used to indicate either a telephone conversation or a face-to-face conversation and (2) all letters and memos cited are on file in the Energy and Environmental Systems Division, Argonne National Laboratory; copies are available upon request from: J.M. Peterson, EES, Bldg. 362, Argonne National Laboratory, Argonne, Illinois 60439.

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4. ENVIRONMENTAL CONSEQUENCES

The potential impacts of all alternatives are described in terms of the following time periods:

- Action Period (approximately 10 years): The period during which physical actions such as excavation, transportation, and stabilization will take place.

- Long-Term Management: The time following the action period during which the wastes will continue to be managed. Human access to disposal cell areas will be limited, and the federal government will continue to own these areas and use them solely for waste-management purposes. Containment structures will be maintained, any releases to the environment will be monitored, and corrective remedial actions will be taken, as necessary. In this EIS, the cumulative impacts for each alternative over 1,000 years are assessed. Although the federal government intends to manage the wastes, impacts that might occur if there was loss of institutional control are also discussed (see Section 4.7).

The 1,000-year time frame is selected to be consistent with the time frames identified in regulations of the U.S. Environmental Protection Agency (EPA) for management of inactive uranium mill tailings (U.S. Environ. Prot. Agency 1983b). The naturally occurring radionuclides found in uranium mill tailings (principally the uranium-238 decay series) constitute the bulk of radioactivity in the Weldon Spring wastes. EPA considers "the single most important goal of control to be effective isolation and stabilization of tailings for as long a time period as is reasonably feasible, because tailings will remain hazardous for hundreds of thousands of years." Furthermore, "the longevity of control is governed by the possibility of intrusion by man and erosion by natural forces." After considering several time periods for control, EPA required that "control measures be carried out in a manner that provides reasonable assurance they will last, to the extent reasonably achievable, up to 1,000 years, and, in any case, for a minimum of 200 years." Uncertainties increase significantly beyond 1,000 years and it would not be reasonable to require assurances of control for longer time periods. Because the 1,000-year time period has been deemed to be a reasonable basis for EPA's decisions regarding inactive uranium mill tailings, DOE considers the same time period to be a reasonable reference point for analyzing environmental impacts in this EIS to support DOE's decision on long-term management of the Weldon Spring wastes.

The environmental impacts associated with the various alternatives are summarized in Table 4.1. This summary is intended to highlight the major environmental consequences of the various alternatives. These impacts are addressed in greater detail in the rest of Chapter 4.

4.1 GEOLOGY AND HYDROLOGY

4.1.1 Site Integrity

Several natural forces could have adverse impacts on the integrity of disposal cells. The action of wind, precipitation, and flooding could result in erosion of the soil cover; drought, fire, and disease could destroy the vegetative cover; and severe natural phenomena (e.g., earthquakes or tornadoes) could compromise the integrity of the disposal cell and allow the release of contaminants to the environment.

Criteria for the disposal cell will be developed during design engineering activities. Design-basis natural phenomena (e.g., earthquake, tornado, flood) will be determined, and the disposal cell will incorporate features to preserve cell integrity. A safety analysis report will be prepared to address the safety aspects of all phases of this project, during both the action period and long-term management. Any degradation of the disposal cell resulting from the effects of natural forces will be repaired during long-term management.

4.1.2 Groundwater

The potential contamination of groundwater near the raffinate pits, quarry, and alternative sites was analyzed by means of two models. For the raffinate pits area and Hanford site, an analytical model was used to calculate contaminant concentrations in groundwater. For the quarry, a numerical model was used because of the complexity of groundwater flow conditions in the vicinity of the quarry.

4.1.2.1 Model Descriptions

Analytical Model for the Raffinate Pits Area and the Hanford Site. The transport medium near the raffinate pits area and the Hanford site may be illustrated as shown in Figure 4.1. Precipitation falling on the waste field would infiltrate through the wastes and transport contaminants down through the unsaturated (vadose) zone (D_u) into the saturated (aquifer) zone. In this analysis, the waste field was assumed to be a rectangular block with dimensions L_x, L_y, and L_z. The geometry of the contaminated layer was assumed to remain constant during vertical migration in the vadose zone. After the

Table 4.1. Summary of Environmental Impacts Associated with the Various Alternatives for Management of the Weldon Spring Wastes[a]

Alternative 1 IMPROVED CONTAINMENT IN EXISTING RAFFINATE PITS	Alternative 2a NEW CELL, PARTIALLY ABOVE GRADE	Alternative 2b NEW CELL, COMPLETELY ABOVE GRADE
GROUNDWATER[b]		
County Well Field		
Maximum average natural uranium concentration contribution 0.03 pCi/L in 800 yr; negligible increase compared to reported background concentration (<1.5 to <3.6 pCi/L).	Same as Alternative 1.	Same as Alternative 1.
Chemical concentration contributions considerably below regulatory limits.	Same as Alternative 1.	Same as Alternative 1.
Weldon Spring Site		
No radionuclide contribution to limestone aquifer in 1,000 yr; maximum natural uranium concentration contribution of 0.45 pCi/L in 3,900 yr.	No radionuclide contribution to limestone aquifer in 1,000 yr; maximum natural uranium concentration contribution of 0.37 pCi/L in 3,900 yr.	No radionuclide contribution to limestone aquifer in 1,000 yr; transport time to water table greater than Alternative 1; maximum concentration contribution less than Alternative 1.
Chemical species having distribution coefficients greater than 100 mL/g are not predicted to reach the limestone aquifer in 1,000 yr; maximum concentration contributions from those species that reach the limestone aquifer are predicted to be less than Missouri groundwater limits.	Transport times same as Alternative 1; maximum concentration contributions slightly less than Alternative 1 and less than Missouri groundwater limits.	Transport times greater than Alternative 1; maximum concentration contributions less than Alternative 1 and less than Missouri groundwater limits.

Table 4.1. Continued

Alternative 1 IMPROVED CONTAINMENT IN EXISTING RAFFINATE PITS	Alternative 2a NEW CELL, PARTIALLY ABOVE GRADE	Alternative 2b NEW CELL, COMPLETELY ABOVE GRADE
GROUNDWATER (Continued)		
Off-site Disposal Areas		
Not applicable.	Not applicable.	Not applicable.
SURFACE WATER		
During action period, water disposed of by spray irrigation will meet Missouri irrigation limits and Missouri requirements for runoff water (DOE limits will be met for radionuclide concentrations); negligible impacts over long term.	Same as Alternative 1.	Same as Alternative 1; long-term concentration of lead in runoff water expected to be no greater than lead in runoff from areas in Missouri having high lead concentrations in soils and rocks.
RADIOLOGICAL		
Action Period (10 years) - General Public		
31 person-rem (primarily from air pathway); 0.0053 health effects.	Same as Alternative 1.	Same as Alternative 1.
Action Period (10 years) - Workers		
110 person-rem (primarily from waste handling); 0.019 health effects.	120 person-rem (primarily from waste handling); 0.020 health effects.	Same as Alternative 2a.

Table 4.1. Continued

Alternative 1 IMPROVED CONTAINMENT IN EXISTING RAFFINATE PITS	Alternative 2a NEW CELL, PARTIALLY ABOVE GRADE	Alternative 2b NEW CELL, COMPLETELY ABOVE GRADE
RADIOLOGICAL (Continued)		
Long Term (Cumulative over 1,000 years) - General Public		
180 person-rem, including 120 person-rem from water pathway at county well field (compares with 230,000,000 person-rem from background radiation at Weldon Spring); 0.031 health effects (compares with 4,300,000 cancer deaths normally expected in the exposed population).	160 person-rem, including 120 person-rem from water pathway at county well field (compares with 230,000,000 person-rem from background radiation at Weldon Spring); 0.027 health effects (compares with 4,300,000 cancer deaths normally expected in the exposed population).	Same as Alternative 1.
ECOLOGICAL		
Negligible during action period; maintenance will counteract and repair damage to disposal cell by plants and animals.	Same as Alternative 1.	Same as Alternative 1.
AIR QUALITY		
National Ambient Air Quality Standards for total suspended particulates easily met using standard dust-control measures.	Same as Alternative 1.	Same as Alternative 1.

Table 4.1. Continued

Alternative 1 IMPROVED CONTAINMENT IN EXISTING RAFFINATE PITS	Alternative 2a NEW CELL, PARTIALLY ABOVE GRADE	Alternative 2b NEW CELL, COMPLETELY ABOVE GRADE
SOCIOECONOMIC		
Transportation		
0.096 deaths, 1.6 injuries; some congestion on local roads during action period.	Same as Alternative 1.	0.18 deaths, 3.1 injuries; some congestion on local roads during action period.
Land Permanently Committed to Waste Disposal		
20 ha (49 acres) for disposal area at Weldon Spring.	18 ha (45 acres) for disposal area at Weldon Spring.	23 ha (58 acres) for disposal area at Weldon Spring.

Table 4.1. Continued

Alternative 3a HANFORD SITE	Alternative 3b "NEARBY SITE"	Alternative 3c URANIUM PROCESSING SITE	Alternative 4 NO ACTION
GROUNDWATER[b]			
County Well Field			
Same as Alternative 1.	Same as Alternative 1.	Same as Alternative 1.	Maximum average natural uranium concentration contribution 0.2 pCi/L in 1,700 yr; small increase compared to reported background concentration (<1.5 to <3.6 pCi/L).
Same as Alternative 1.	Same as Alternative 1.	Same as Alternative 1.	Chemical concentration contributions greater than Alternative 1 but considerably below regulatory limits.
Weldon Spring Site			
Minimal impact.	Minimal impact.	No radionuclide contribution to limestone aquifer in 1,000 yr; transport times same as Alternative 1 but concentration contributions less than Alternative 1.	No radionuclide contribution to limestone aquifer in 1,000 yr; maximum natural uranium concentration contribution 4.3 pCi/L in 1,900 yr.
Minimal impact.	Minimal impact.	Transport times for chemical species same as Alternative 1; maximum concentration contributions much less than Alternative 1 and much less than Missouri groundwater limits.	Maximum chemical concentration contributions reached in half the time as Alternative 1, but most concentrations half of Alternative 1 or less; predicted peak concentration contributions of mercury (0.05 µg/L) and fluoride (2.6 mg/L) are equal to or greater than Missouri groundwater limits.

Table 4.1. Continued

Alternative 3a HANFORD SITE	Alternative 3b "NEARBY SITE"	Alternative 3c URANIUM PROCESSING SITE	Alternative 4 NO ACTION
GROUNDWATER[b] (Continued)			
Offsite Disposal Areas			
No radionuclide contribution to groundwater in 1,000 yr; maximum natural uranium concentration contribution 0.37 pCi/L in 20,000 yr.	No radionuclide contribution to groundwater in 1,000 yr; transport times at least as long as Alternative 2a; maximum concentration contributions no higher than Alternative 2a.	Negligible incremental contribution of radionuclides to leachate from existing uranium mill tailings pile.	Not applicable.
Most chemical species not expected to reach groundwater in 1,000 yr; maximum concentration contributions of more mobile species in groundwater at point of discharge to Columbia River expected to be less than drinking water limits.	Most chemical species not expected to reach groundwater in 1,000 yr; concentration contributions of more mobile species less than Missouri groundwater limits; transport times at least as long as Alternative 1; maximum concentration contributions no higher than Alternative 1.	Negligible incremental contribution of chemicals to leachate from existing uranium mill tailings pile.	Not applicable.
SURFACE WATER			
Same as Alternative 1 at Weldon Spring during action period and none thereafter; none at Hanford site.	Same as Alternative 1 at Weldon Spring during action period and none thereafter; negligible at "Nearby Site".	Same as Alternative 1 at Weldon Spring; negligible incremental contribution at uranium processing site.	Negligible.

Table 4.1. Continued

Alternative 3a HANFORD SITE	Alternative 3b "NEARBY SITE"	Alternative 3c URANIUM PROCESSING SITE	Alternative 4 NO ACTION
RADIOLOGICAL			
Action Period (10 years) - General Public			
250 person-rem (primarily from transportation of wastes); 0.043 health effects.	120 person-rem (primarily from transportation of wastes); 0.020 health effects.	39 person-rem (primarily from air pathway); 0.0066 health effects.	45 person-rem (primarily from air pathway); 0.0077 health effects.
Action Period (10 years) - Workers			
130 person-rem (primarily from waste handling); 0.022 health effects.	230 person-rem (about half from waste handling and half from truck transport of wastes); 0.039 health effects.	Same as Alternative 2a.	5.1 person-rem (from maintenance and monitoring); 0.00087 health effects.
Long-Term (Cumulative over 1,000 years)			
720 person-rem, including 600 person-rem from air pathway at Hanford and 120 person-rem from groundwater pathway at Weldon Spring (compares with 34,000,000 person-rem at Hanford and 230,000,000 person-rem at Weldon Spring from background radiation); 0.12 health effects (compares with 4,300,000 cancer deaths normally expected in the exposed population at Weldon Spring plus 600,000 cancer deaths normally expected in the exposed population at Hanford).	Same as Alternative 2a.	Slightly less than Alternative 2a at Weldon Spring (130 person-rem; 0.022 health effects); small incremental impact at uranium processing site.	11,000 person-rem (primarily from air pathway), including 420 person-rem from water pathway at county well field (compares with 230,000,000 person-rem from background radiation at Weldon Spring); 1.9 health effects (compares with 4,300,000 cancer deaths normally expected in the exposed population).

Table 4.1. Continued

Alternative 3a HANFORD SITE	Alternative 3b "NEARBY SITE"	Alternative 3c URANIUM PROCESSING SITE	Alternative 4 NO ACTION
ECOLOGICAL			
Similar to Alternative 1.	Similar to Alternative 1.	Similar to Alternative 1.	Similar to Alternative 1.
AIR QUALITY			
Same as Alternative 1.	Same as Alternative 1.	Same as Alternative 1.	Negligible total suspended particulates.
SOCIOECONOMIC			
Transportation			
2.5 deaths, 34 injuries; some congestion on local roads during action period.	0.51 deaths, 8.7 injuries; some congestion on local roads during action period.	0.21 deaths, 3.1 injuries; some congestion on local roads during action period.	0 deaths, 0 injuries; no congestion.
Land Permanently Committed to Waste Disposal			
120 ha (300 acres) for disposal area at Hanford.	15 ha (37 acres) for disposal area at "Nearby Site".	11 ha (28 acres) for disposal area at Weldon Spring	93 ha (230 acres) for raffinate pits area, chemical plant area, and quarry.

[a] All results rounded to two significant figures.

[b] Groundwater modeling results are reported as "concentration contributions", i.e., the incremental concentration resulting from leaching of the Weldon Spring wastes.

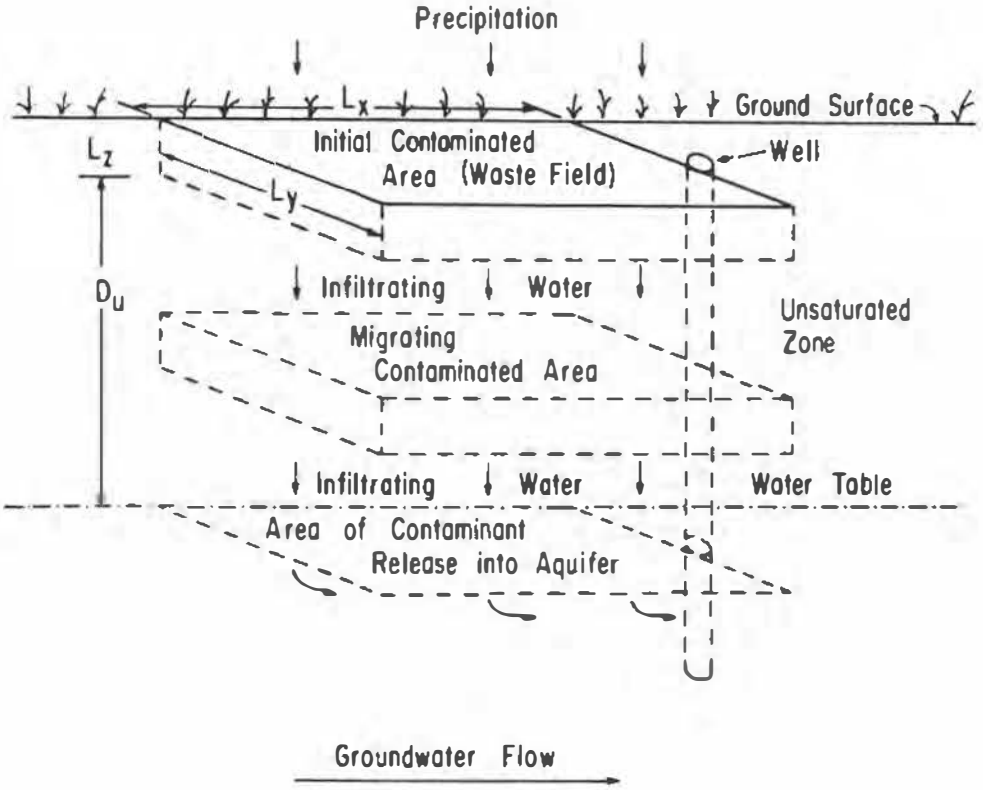

Figure 4.1. Schematic Diagram of Model for Estimating Contaminant Concentrations in Groundwater near the Raffinate Pits Area and the Hanford Site. Source: Modified from U.S. Department of Energy (1986a).

contamination reached the water table, it would spread out and be diluted in the groundwater and would generate a contaminant plume that would move downgradient. For the raffinate pits area, the vadose zone was divided into two layers to account for the blocky fractures that occur at the lower layer of the clay overburden (Bechtel Natl. 1984b) (see Section 3.1.1). The computer code used to simulate the migration of contaminants was a modified version of a three-dimensional solute transport model, AT123D, developed at Oak Ridge National Laboratory (Yeh 1981). AT123D solves a generalized transport equation analytically and is applicable only under saturated aquifer conditions. Transport and depletion mechanisms included in the model are advection, dispersion, sorption, and radioactive decay.

The code was modified so that it could handle saturated-unsaturated conditions by using a contaminant travel time method described by Gilbert et al. (1983). The source input option of the AT123D model was also modified by incorporating a continuous waste-leaching term derived from a first-order leaching model (Baes and Sharp 1983). Other modifications of AT123D included

extending the application of the code to macropore/fracture conditions and using three sets of parameters for wastes, unsaturated zone, and saturated zone. All these modifications to the AT1230 model are discussed in detail in Appendix I, Section I.1.

Numerical Model for the Quarry Area. The hydrogeologic conditions at the quarry area and county well field are more complex than the conditions at the raffinate pits area. Therefore, a more complex groundwater flow code was used for analysis -- i.e., a pseudo-three-dimensional, finite-element code (Tracy and Carlton 1982). Detailed descriptions of the code, the boundary conditions, and the values selected for the input parameters are provided in Appendix I, Section I.2. The model included several important hydrogeologic features at the quarry area as input, such as (1) leakage between the alluvium and limestone aquifers, (2) discharge from well field pumpage, (3) interaction of the aquifer with surface streams, and (4) location-dependent transmissivity and thickness of the alluvium. The alluvium was divided into two zones: the near alluvium with low transmissivity and the river alluvium with high transmissivity. The flow model was first calibrated using the field-measured data and was then used to simulate groundwater flow and contaminant concentration patterns for both the no-action and action alternatives.

4.1.2.2 Source Term and Hydrogeological Parameter Values

The data used to estimate the quantities of contaminants available for release (i.e., the source term) at the raffinate pits, quarry, and other potential disposal sites are presented in Sections 3.1.7 and 3.1.8 and Appendices H and I. The values of various hydrogeological parameters selected to generate estimates of contaminant concentrations in groundwater are discussed in Appendix I.

4.1.2.3 Radionuclides in Groundwater

Uranium and radium are the radioactive contaminants of importance in estimating potential radiological impacts from groundwater contamination (see Appendix I, Section I.1.2). The concentrations of thorium in groundwater are expected to be negligible because thorium is strongly bound to soil -- i.e., it has a high distribution coefficient of about 60,000 mL/g (Gilbert et al. 1983). The results of the model calculations are reported here as pCi/L natural uranium and pCi/L radium-226, unless otherwise specified. For this analysis, natural uranium was considered to be essentially equivalent to total uranium; the concentration of natural uranium (in pCi/L) is equal to 2.046 times the concentration of uranium-238. Radium-228 was neglected because its relatively short half-life (5.8 years) would result in the disappearance of radium-228 by radioactive decay enroute to potable

groundwater supplies. Results are reported as "concentration contributions", i.e., the incremental concentrations resulting from the leaching of the Weldon Spring wastes. The background concentration of natural uranium in water in the river alluvium of the quarry area ranges from < 1.5 pCi/L (Well 5--U.S. Geol. Surv. 1984) to < 3.6 pCi/L (LW series wells--Layne Western 1986). Background concentrations should be added to the concentration contribution to get the total concentration.

Raffinate Pits Area. The model simulations indicate that radium and uranium would not reach the groundwater in the limestone under the raffinate pits area in 1,000 years for Alternatives 1, 2a, 2b, 3c, and 4. In Alternatives 3a and 3b, no wastes would remain on-site. Uranium is more mobile than radium, but even for the no-action Alternative 4, the travel time for uranium to reach the water table (the radionuclide travel time) is calculated to be 1,900 years (see Appendix I, Table I.3). For Alternative 2b (above-grade disposal cell), the lead sheet in the cover would reduce water infiltration into the wastes for a period of time, resulting in an even longer travel time to the water table. Concentrations in the groundwater (after the radionuclides reached the groundwater) would be smaller for Alternative 3c due to the smaller source at the raffinate pits area. Thus, the model results predict that the radionuclides would remain in the unsaturated zone on-site for a period in excess of 1,000 years whether action is taken or not.

Quarry. The time variations of radionuclide concentrations in the quarry area were calculated using currently available data. The time for emplacement of wastes into the quarry is assumed to be 1963. For the action alternatives, the quarry is assumed to be cleaned up in 1990.

The simulation was first performed to determine the values of various hydrodynamic variables. These were computed by considering the pumping at the well field and using the flow model to yield flow characteristics beneath the quarry. The predicted groundwater elevations with 11 pumping wells (i.e., Wells 1, 2, 3, and 5 through 12) (Figure 4.2) indicate that the cones of depression resulting from well pumping would overlap and the maximum drawdown would be about 7.0 m (23 ft) near Wells 9 through 11 (the locations of these wells are shown in Figure 3.9). The results also indicate that recharge to the well field would be primarily from the Missouri River and secondarily from upgradient groundwater. The total pumping rate for the 11 county wells was taken to be about 1.4 m^3/s (49 cfs) (Hovatter 1985). Of this, only about 0.076 m^3/s (2.7 cfs) was recharge from upgradient groundwater; the rest was from the Missouri River. This result is reasonable and is in agreement with other studies (Layne Western 1986; Kleeschulte et al. 1986), given the high transmissivity of the river alluvium and the availability of large amounts of water.

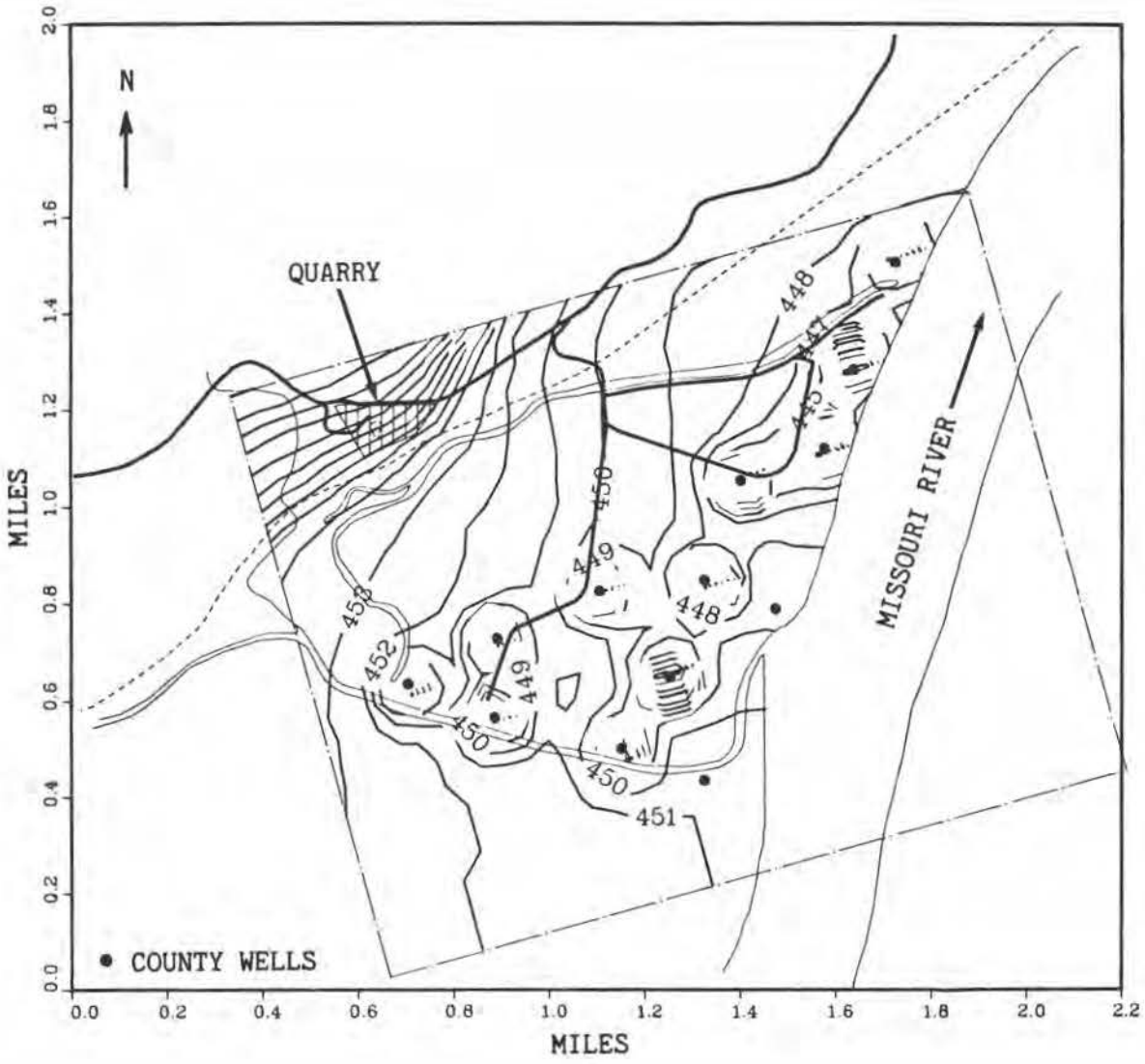

Figure 4.2. Predicted Groundwater Elevations (ft) with 11 Wells Pumping. Contour Interval = 1 ft. Conversion Factors: To convert feet to meters, multiply by 0.3048; to convert miles to kilometers, multiply by 1.609.

Detailed model calculations were not made for radium-226 because the impact on the groundwater in the near alluvium and river alluvium would be negligible over the time frame of interest. Radium is very tightly bound in the wastes (Kd = 14,000 mL/g--see Appendix I, Section I.3.1.2) and disperses very slowly once it is leached from the wastes. It would take about 26,000 years for the radium-226 to move 80 m in the limestone and reach the near alluvium. Using the parameters given in Appendix I, Section I.4.2, the maximum radium-226 concentrations currently in the groundwater at the quarry are estimated to be 2 to 4 pCi/L. Measured values were less than 2 pCi/L in

1985 (Bechtel Natl. 1985b, 1986). The amount of radium-226 in the quarry wastes is increasing slowly as radium-226 approaches radiological equilibrium with its parent thorium-230. The maximum concentration of radium-226 in the quarry wastes is predicted to occur about 9,100 years in the future. For this reason, concentrations of radium-226 in groundwater at the quarry should increase slowly to maximum values of 20 to 40 pCi/L about 9,100 years in the future for the no-action Alternative 4 (see Appendix H, Section H.1.5, for a discussion of the ingrowth of radium-226 from thorium-230). Radium-226 concentrations for the action alternatives would be significantly lower because the wastes would be removed from the quarry.

The predicted existing concentration contributions of natural uranium around the quarry are presented in Figure 4.3a. The predicted values of 400 to 1,200 pCi/L near the slough are in reasonable agreement with values measured in 1984 and 1985 of 3,800, 4,200, 62, and 2,200 pCi/L in Wells OB-6, OB-10, OB-11, and OB-14 (see Appendix I, Figure I.10, for the locations of these wells) in the alluvium between the slough and the quarry (Layne Western 1986). The value of 5,800 pCi/L in the quarry groundwater is in reasonable agreement with the measured values of 550 to 8,200 pCi/L in groundwater in boreholes in the quarry wastes and 620 to 1,500 pCi/L in the quarry pond (U.S. Geol. Surv. 1984; Bechtel Natl. 1985b; Layne Western 1986). The agreement between model predictions and measured values is reasonable given the wide variation in uranium concentrations in the wells near the slough and the possibility of additional sources of uranium on the ground surface of the near alluvium (Berkeley Geosci. Assoc. 1984). Although background concentrations are not available for the groundwater in the limestone or near alluvium, background concentrations at different locations in the river alluvium range from < 1.5 to < 3.6 pCi/L (U.S. Geol. Surv. 1984; Layne Western 1986).

For comparative purposes, the predicted concentration contributions of natural uranium in 1,000 years for the no-action and action alternatives are presented in Figures 4.3b and 4.3c (note that the contour interval is 200 pCi/L for the no-action Alternative 4 and 2 pCi/L for the action alternatives). Based on the calculated leaching times for uranium in different regions of the quarry (51 to 4,300 years--Appendix I, Section I.4.2), much of the uranium is expected to remain in the quarry. Comparison of the predicted concentration patterns shown in Figures 4.3b and 4.3c indicates that, over 1,000 years, there would be a much lower uranium concentration in the groundwater near the quarry for the action alternatives than for the no-action Alternative 4.

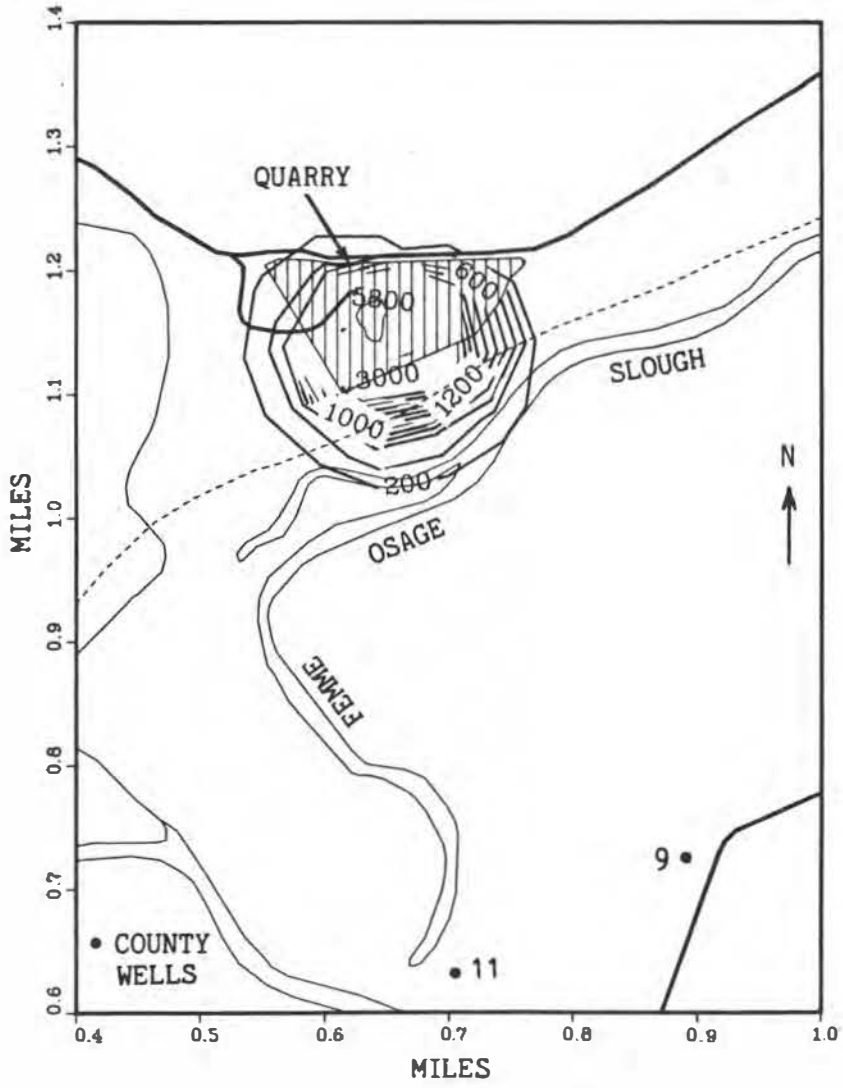

Figure 4.3a. Predicted Existing Concentration Contributions (pCi/L). (Contour interval = 200 pCi/L.)

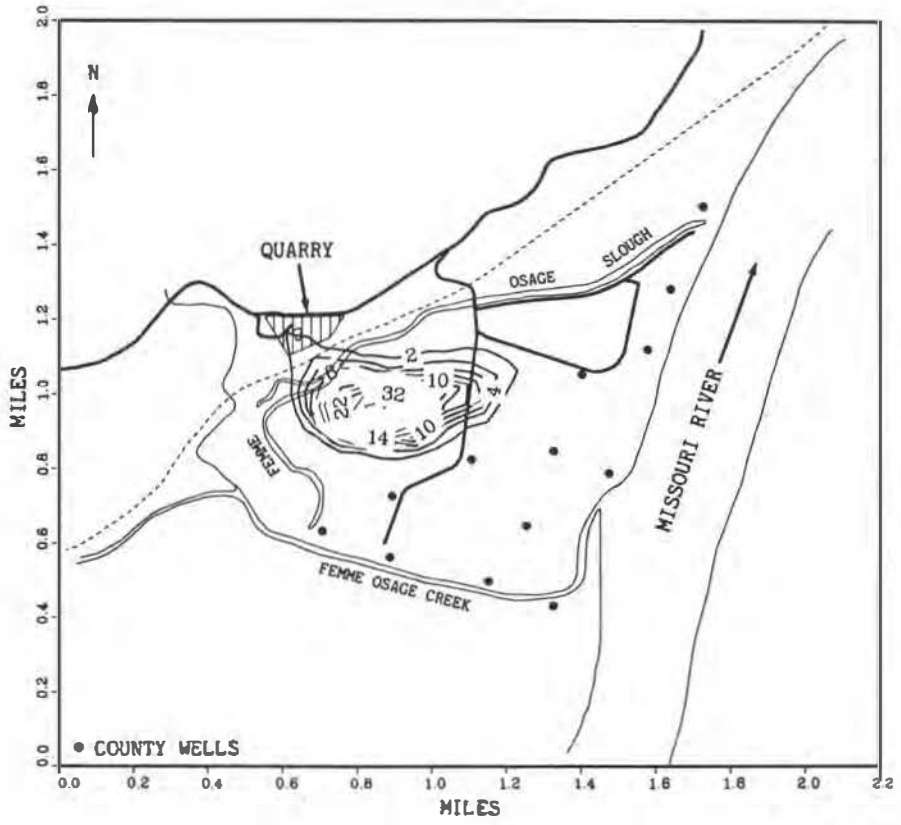

Figure 4.3b. Predicted Concentration Contributions (pCi/L) in 1,000 Years for the No-Action Alternative. (Contour interval = 200 pCi/L.)

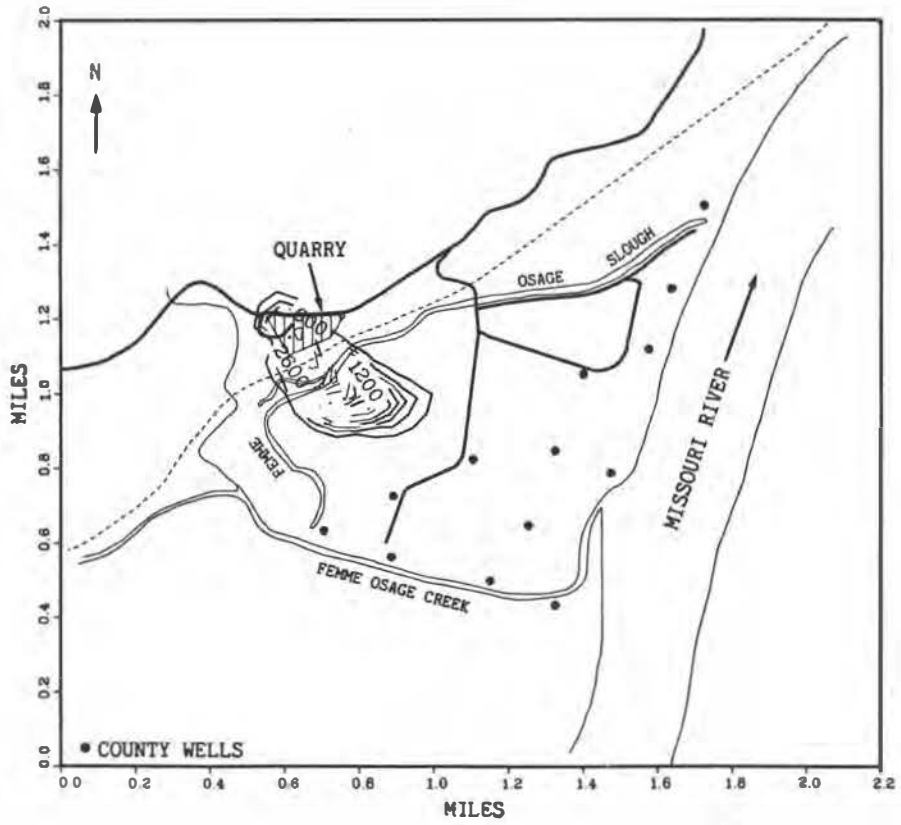

Figure 4.3c. Predicted Concentration Contributions (pCi/L) in 1,000 Years for the Action Alternatives. (Contour interval = 2 pCi/L.)

Figure 4.3. Comparison of Predicted Natural Uranium Concentration Contributions for the No-Action and Action Alternatives. Background concentrations are not included. The quarry area is shown by hatching. Conversion Factor: To convert miles (mi) to kilometers (km), multiply by 1.609.

The predicted uranium concentration patterns also indicate that the plume would be affected by well pumping and would migrate toward the well field. Wells 5 through 9 (Figure 3.9) would be affected to a small degree by the edge of the contaminant plume; the effect on the rest of the pumping wells would be negligible. This result can probably be attributed to the fact that most of the recharge to the wells is from the Missouri River. The predicted results further demonstrate that Well 8 would have the highest uranium concentration of all the wells.

The predicted time variations of the uranium concentration contributions in Well 8 are shown in Figure 4.4. Under the no-action Alternative 4, the predicted maximum concentration contribution of natural uranium in Well 8 would be about 2.0 pCi/L, occurring about 1,700 years after the time of waste emplacement. The concentrations averaged over all wells would be 0.2 pCi/L. These values represent the uranium contributions from water in the well field from leaching of the quarry wastes; they are comparable to reported background concentrations, which range from < 1.5 to < 3.6 pCi/L (U.S. Geol. Surv. 1984; Layne Western 1986), and they are below the DOE guideline of 1,100 pCi/L for natural uranium (U.S. Dept. Energy 1986b). Adding the maximum concentration contribution of 2.0 pCi/L to the upper value reported for the background concentration of uranium (3.6 pCi/L) gives a total estimated concentration of uranium in Well 8 of < 5.6 pCi/L. For the action alternatives, the maximum concentration contribution in Well 8 would be reduced by a factor of about 7 to 0.28 pCi/L and would occur about 800 years after waste emplacement; the average concentration for all wells would be 0.03 pCi/L. These values are less than background concentrations. The results showing the time variations of the average concentrations for the 11 pumping wells are presented in Figure 4.4. The radiological impacts due to the ingestion of contaminated well water are discussed in Section 4.2.1.2.

Hanford Site, "Nearby Site", and Uranium Processing Site. At the Hanford site (Alternative 3a), there would be very slow migration of radioactive contaminants through the unsaturated zone from the disposal area, despite the high permeability of the sandy soils, because of low annual precipitation. Based on the slow migration rate and the depth to the water table (Appendix I, Section I.1), radium and uranium are not expected to reach the water table in 1,000 years.

For Alternative 3b, the wastes would be taken to a "Nearby Site" that would have more favorable conditions (e.g., thicker clay, lower hydraulic conductivity, deeper groundwater table, and/or higher sorption capacity) than the Weldon Spring site. The radionuclides would not be expected to reach the groundwater table within 1,000 years at the "Nearby Site". The assumed better

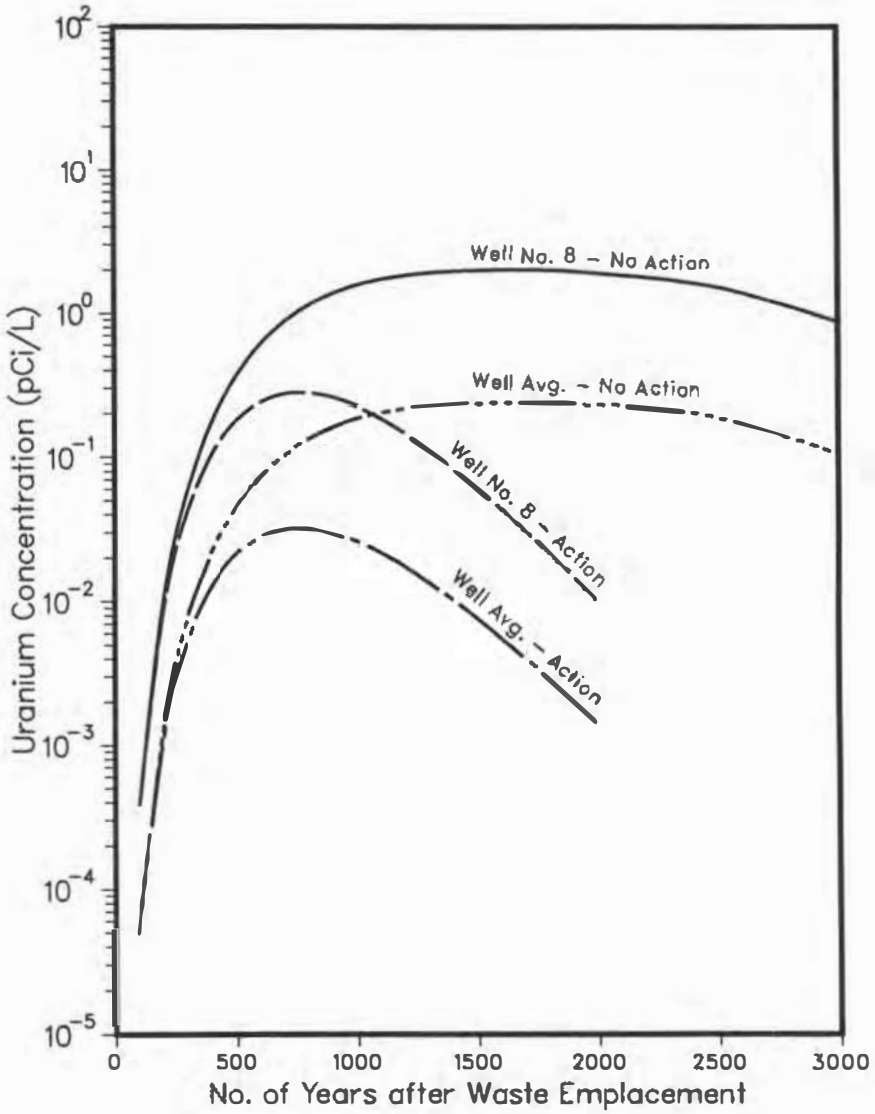

Figure 4.4. Time Variations of the Maximum and Average Concentration Contributions of Natural Uranium in Water from the County Well Field.

conditions at the "Nearby Site" would be expected to delay the migration of radionuclides into groundwater even longer than at the raffinate pits area.

For Alternative 3c, the raffinate sludge would be transported for reprocessing to an existing uranium processing site somewhere in the Four Corners area of the southwestern United States. The greatest potential impact to groundwater due to operation of the plant would result from seepage from the uranium mill tailings. Contamination of groundwater by such seepage is largely site-specific, and predicting movement and dispersion of contaminants depends on local geologic and hydrologic characteristics (U.S. Nucl. Reg.

Comm. 1980a). The impacts of any wastes generated by the reprocessing of the Weldon Spring sludge would be negligible in comparison to the overall groundwater contamination potential from all contaminated materials at the uranium processing site.

4.1.2.4 Chemicals in Groundwater

Raffinate Pits Area. For Alternatives 1, 2a, 2b, 3c, and 4, the impacts on groundwater of chemicals leaching out of the wastes in the raffinate pits area were estimated by the same model that was used to estimate radiological impacts. For the chemical analysis, distribution coefficients (Kd values) were incorporated that are appropriate for the chemical species of concern (Appendix I, Section I.3). The source term for modeling the raffinate pits area is described in Appendix I, Section I.4.1. Results are reported as "concentration contributions", i.e., the incremental concentrations resulting from leaching of the Weldon Spring wastes.

According to model calculations for Alternative 1, chemical species with Kd values greater than 100 mL/g in the underlying clay would not reach the groundwater in the Burlington-Keokuk limestone under the pits (the uppermost aquifer under the pits--see Section 3.1.2.2) within 1,000 years. Several chemical species with Kd values less than or equal to 100 mL/g in the wastes and clay are predicted to reach peak concentration contributions in the limestone aquifer that are between 10% and 70% of the Missouri groundwater limits* in 1,000 years or less. Concentration contributions are expected to remain at these levels for several kilometers downgradient from the raffinate pits, and these values would occur in less than 10 years after the chemicals reached the groundwater.

Chemical species whose peak concentration contributions are predicted to be between 10% and 70% of the appropriate regulatory limits are selenium, arsenic, copper, mercury, nitrate, and fluoride. The peak concentration contributions that would occur in the limestone aquifer at the DOE north fence line and at 4 km (2.5 mi) downgradient** are as follows: selenium, 3.3 µg/L at the DOE fence line in 34 years and 2.6 µg/L at 4 km downgradient in 39 years; arsenic, 14 µg/L at the DOE fence line in 34 years and 11 µg/L at

*Background concentrations for groundwater in the Burlington-Keokuk limestone near the raffinate pits area are not available for most of the chemical species of concern. The regulatory limits apply to the total concentration (background plus contribution from leaching of the Weldon Spring wastes).

**Distances are measured from the upgradient edge of Pits 3 and 4, which would contain most of the wastes for Alternative 1. The DOE fence line is about 600 m downgradient from the upgradient edge of Pits 3 and 4.

4 km downgradient in 39 years; copper, 3.2 μg/L at the DOE fence line in 270 years and 2.6 μg/L at 4 km downgradient 5 years later; mercury, 0.016 μg/L at the DOE fence line in 1,000 years and 0.013 μg/L at 4 km downgradient 5 years later; nitrate, 17 mg/L at the DOE fence line in 2.2 years and 13 mg/L at 4 km downgradient in 7.2 years; and fluoride, 1,400 μg/L at the DOE fence line in 34 years and 1,100 μg/L at 4 km downgradient in 39 years. The corresponding Missouri groundwater limits (Appendix H, Table H.12) are selenium, 10 μg/L; arsenic, 50 μg/L; copper, 20 μg/L; mercury, 0.05 μg/L, nitrate, 44 mg/L (as nitrate); and fluoride, 2 mg/L. Transport velocities of all chemical species in the limestone are assumed to be equal to the linear water velocity (i.e., no retardation).

For Alternative 2a, the peak concentration contributions for most chemicals in the Burlington-Keokuk limestone aquifer are predicted to occur at the same times as they would for Alternative 1. However, the contributions for Alternative 2a would generally be about 80% of those predicted for Alternative 1.

Peak concentration contributions for Alternative 3c are expected to be much lower than those predicted for Alternative 1 or 2a because neither raffinate sludge nor stabilizer is present to contribute to the source term. In Alternative 3c, the raffinate sludge will be dried and sent to a uranium processing plant for reprocessing.

For Alternative 2b, the infiltration of precipitation and subsequent leaching of the wastes would depend on the integrity of the 0.64-cm (0.25-in.) thick lead sheet in the cover. The duration of time that the lead sheet would retain its integrity is uncertain, however, because of uncertainties in the corrosion rate (including the pH and ion composition of the infiltrating water and the aeration of the drain layer), the number of local depressions that might develop in the sheet resulting in local subsurface water ponding, the engineering variables associated with construction of such a large lead sheet, and future settling of the underlying wastes and clay layer causing stresses on the lead sheet. If there were no large tears nor a high density of corrosion pits perforating the sheet, leaching would be negligible. If a sufficient number of pits corroded through the lead sheet and/or tears developed, the sheet would leak and infiltrating water would cause leaching of the wastes (see Appendix I, Section I.4.3). Thus, for Alternative 2b, it is expected that leaching of chemicals to groundwater would be delayed longer than for Alternatives 1 and 2a.

There would also be essentially no impact on groundwater during 1,000 years resulting from lead corroding from the sheet, moving off the cover, and accumulating in the sediments in the drainage ditch surrounding the waste pile. Based on estimated corrosion rates, model calculations predict that it would take several hundred years for lead concentrations to build up in the sediments in the drainage ditch and an additional 1,000 years for the lead to move through the clay and reach the underlying limestone.

Model calculations for the no-action Alternative 4 predict that peak concentration contributions in the limestone aquifer for most chemical species would be lower than those predicted for Alternative 1 by a factor of two or more because in Alternative 4 there would be no chemicals from the stabilizer contributing to the leachate.* Also, the model parameter values used for Alternative 4 for raffinate porosity, density, and volume (see Appendix I, Table I.1) are such that predicted concentrations of contaminants in the groundwater are higher than the concentrations obtained from the parameter values used for Alternative 1. Mercury and fluoride are exceptions. For mercury, predicted peak concentration contributions in the groundwater are 0.05 μg/L (equal to the state groundwater limit [Appendix H, Table H.12]) at the DOE fence line to the north in about 530 years and 0.040 μg/L at 4 km (2.5 mi) downgradient 5 years later. These concentrations are higher than the corresponding values for Alternative 1, primarily because mercury concentrations would be higher in the unstabilized wet raffinate sludge than in the stabilized raffinate sludge. For fluoride, the concentration would also be greater in the wet raffinate sludge than in the stabilized raffinate sludge; and the predicted peak groundwater concentration contribution of 2,600 μg/L at the DOE fence line in 17 years, which is above the state groundwater limit of 2,000 μg/L (Appendix H, Table H.12), is also higher than the corresponding value for Alternative 1.

The infiltration rate for Alternative 4 was assumed to be twice that for Alternative 1 (Appendix I, Table I.1). On this basis, peak concentration contributions would be reached in half the time for Alternative 4.

There are currently no data that indicate the presence of organics in the raffinate sludge. Small amounts of hexane and tributyl phosphate, which were used to process the uranium, are expected to be present in the sludge. Also, for Alternatives 1, 2a, 2b, and 3c, small amounts of PCBs and pesticides are expected to be present in the waste-disposal cell because they are

*See Appendix I, Section I.4.1, for discussion of assumed chemical composition of the raffinate sludge stabilizer. If DOE decides to implement Alternative 1, 2a, or 2b, detailed analyses will be carried out to determine what specific stabilizer should be used, taking into consideration physical and chemical properties of the stabilized material.

contaminants of the quarry wastes (Appendix H, Table H.13); also present would be any organics that might be in the stabilizer (Alternatives 1, 2a, and 2b only). DOE is gathering additional data on the chemical characteristics of the raffinate sludge to ensure that appropriate waste stabilization and confinement features are incorporated in the alternative selected.

The concentrations of PCBs and pesticides in groundwater can be estimated by assuming that the average concentrations of these organics would be the same in the stabilized raffinate sludge as they are in the quarry wastes (Appendix H, Table H.13). Using the same representative Kd values for the stabilized raffinate sludge and underlying clay as were used for the quarry wastes (see Appendix I, Section I.3.2), predicted peak concentration contributions at the DOE north fence line (downgradient) for Alternative 1 are 0.0001 μg/L lindane in 100 years, 0.002 μg/L endrin in 1,000 years, and 0.09 μg/L PCB 1254 in 1,000 years. The concentration contributions are predicted to be about the same at 2 km (1.2 mi) downgradient and are about 80% of the values at the DOE fence line at 4 km (2.5 mi) downgradient (conservatively assuming Kd = 0 mL/g in the fractured limestone). Peak concentration contributions for Alternative 2a are predicted to be about 80% of the above. For Alternative 2b, migration of these organic compounds would be delayed because of the lead sheet. For Alternative 4, the predicted values are about half the above if it is assumed that concentrations of these chemicals are the same in the raffinate sludge as in the quarry wastes.

According to Missouri state regulations (10 CSR Part 20.7), organics such as PCBs, endrin, and lindane are "not allowed in the waters of the state." However, the predicted peak concentrations for lindane and endrin are well below the EPA drinking water limits of 4 μg/L (lindane) and 0.2 μg/L (endrin) (40 CFR Part 141); no EPA drinking water limits are given for PCBs.

There would be a minimal chemical impact on groundwater quality in the raffinate pits area for Alternatives 3a and 3b resulting from continuing migration of the chemical species that have already migrated into the underlying clay prior to cleanup of the pits. This impact would be much less than that for Alternative 3c. The impact for Alternative 3c would be much less than for Alternative 1 because the raffinate sludge would be removed to a uranium processing site for reprocessing.

The model results described above are expected to be conservative, especially for Alternatives 1 and 2a. For these alternatives, no credit was taken for the possible reduced leachablilty of the stabilized raffinate sludge compared to that for the unstabilized sludge. This conservatism is reflected in the use of the ion-exchange model for the chemicals contained in the sludge and stabilizer because, in this model, the chemical species are assumed to be readily available for leaching. In reality, most chemical species would be

expected to be bound in the interior of the stabilizer particles or in the matrix of the stabilized sludge and they would have to diffuse to the surface of the particles or matrix before being available for leaching. Use of the ion-exchange model for Alternative 4 is more reasonable because the particles of raffinate sludge are exposed for leaching and are not bound up in a solid matrix. As a result, model predictions of contaminant concentrations in groundwater for Alternatives 1, and 2a are expected to overestimate the actual concentrations that would occur. The predicted peak concentration contributions for Alternatives 1 and 2a are less than the corresponding state groundwater limits, even with this conservative approach.

Use of the ion-exchange model is expected to give more reasonable results for Alternatives 3a and 3b because, like Alternative 4, the particles of sludge are not bound up in a solid matrix. (No credit was taken for the effect of packaging with regard to restricting the leaching of the dried sludge.) For the other waste components (i.e., soils, rubble, clay), it is not known whether or not the contaminants are in the interior of the waste particles or are localized on the surface.

Another factor leading to conservatism is the assumption that Kd = 0 mL/g for all chemical species in the limestone. To the extent that the Kd values were greater than zero, transport velocities and contaminant concentrations in the limestone would be lower than predicted.

Quarry Area. For all action alternatives, all wastes will be removed from the quarry. Consequently, the only potential chemical impacts that might occur would be associated with contamination left in the limestone and alluvium near the quarry. The potential exists for this contamination to move toward the county well field. For the no-action Alternative 4, leaching of contaminants out of the quarry wastes and into the surrounding groundwater is expected to continue. In order to determine the potential impact of chemical species leaching from the quarry, model calculations were made of the concentration contributions of chemical species to be expected at various times for (a) the county well field area and (b) the area near the quarry. The model is the same one used for radiological species and is described in Appendix I, Section I.2. The source terms are described in detail in Appendix I, Section I.4.2.

For the no-action Alternative 4, the concentration contributions at all times at the county well field are predicted to be considerably below both background values and regulatory limits for all regulated chemical species for which source term concentrations are available. This can be seen by comparing the peak concentration contributions in the well water for those chemical species that are expected to have the highest concentrations -- selenium, copper, zinc, arsenic, lead, and mercury -- relative to the corresponding

Missouri state groundwater limits (Appendix H, Table H.12). The highest predicted concentration contribution of selenium in any well at any time is 0.0001 μg/L in Well 8, about 400 years after wastes were initially dumped in the quarry. This is far below both the background value of < 5 μg/L (Hengerson 1985) and the groundwater limit of 10 μg/L (Appendix H, Table H.12). The maximum concentration contributions for copper (0.0004 μg/L) and zinc (0.002 μg/L) would be reached in Well 8 at 1,000 years after waste emplacement. These values are negligible compared to background values of < 10 μg/L for copper and 250 μg/L for zinc (Hengerson 1985) and groundwater limits of 20 μg/L for copper and 100 μg/L for zinc (Appendix H, Table H.12). The same comparison holds for arsenic for which a maximum concentration contribution of 0.0003 μg/L would be reached in Well 8 at 2,000 years after waste emplacement, which is considerably below the groundwater limit of 50 μg/L. About 3,000 years after waste emplacement, lead would reach a maximum concentration contribution of 0.0007 μg/L in Well 8 (the groundwater limit is 50 μg/L) and mercury would reach a maximum concentration of 0.000005 μg/L (the groundwater limit is 0.05 μg/L). The same is true for other regulated metals for which concentrations have been measured in the quarry wastes (Appendix H, Table H.13), i.e., the concentration contributions at the county well field would be far below both the background values and the groundwater limits at all times.

The same conclusions are true for organics such as pesticides and PCBs. The concentration of lindane (γ-benzene hexachloride) in the county well field is predicted to peak at 7×10^{-9} μg/L. Predicted peak concentration contributions of 5×10^{-7} μg/L for endrin and 3×10^{-5} μg/L for PCB 1254 would occur in 3,000 years in Well 8. According to Missouri state regulations (10 CSR Part 20.7), organics such as PCBs, endrin, and lindane "are not allowed in the water of the state." However, these values are all considerably below the EPA drinking water limits of 4 μg/L (lindane) and 0.2 μg/L (endrin) (40 CFR Part 141); no EPA drinking water limits are given for PCBs.

The above results of negligible maximum concentration contributions at the county well field compared to the regulatory limits were obtained for the no-action Alternative 4. This conclusion also holds for the action alternatives because, in these alternatives, the source material (quarry wastes) is assumed to be removed in 1990 (27 years after the wastes were assumed to be dumped in the quarry).

Model calculations were also carried out for the area close to the quarry in the limestone and alluvium. The parameter values used were the same as those used for the county well field calculations. The model results for all alternatives show a region of predicted groundwater concentration contributions that would be above the Missouri groundwater limits (Appendix H,

Table H.12) for several chemical species -- selenium, arsenic, copper, chromium, zinc, lead, mercury, and cyanide -- and above the EPA drinking water limit for endrin. The region extends into the river alluvium for some chemical species such as copper and cyanide. The size of the region and the time interval for which a region is predicted to be above concentration limits would be different for different chemical species because of the different Kd values and average concentrations in the quarry wastes.

Except for cyanide, the regions of predicted concentration contributions that are above the regulatory limits would be larger and would exist for a longer time for the no-action Alternative 4 than for the action alternatives. For selenium, a region of above-limit concentration contribution is predicted to extend across the slough up to 160 m (520 ft) into the river alluvium and is expected to last for more than 400 years for the no-action Alternative 4. For the action alternatives, the region is predicted to be smaller (extend up to 80 m [260 ft] into the river alluvium) and is expected to be reduced to below limits 200 years after wastes were initially dumped in the quarry. For copper, chromium, zinc, lead, mercury, and arsenic, the regions of above-limit concentrations would extend up to 250 m (820 ft) into the river alluvium and would exist for more than 1,000 years for the no-action Alternative 4. For the action alternatives, the regions of predicted above-limit concentrations are expected to be smaller and to be reduced to below limits in 200 years (for copper, zinc, and arsenic) and 400 years (for lead).

For cyanide, all alternatives would be equivalent because cyanide is mobile, with Kd = 0 (Fuller 1977; Theis and West 1986), and is predicted to be almost completely leached out of the wastes at the present time. The region of predicted concentration contribution that is above the state groundwater limit for cyanide is predicted to currently extend up to 320 m (1,000 ft) into the river alluvium (about 20 years after waste emplacement) for all alternatives. By the year 2020, cyanide concentrations are expected to be below the state groundwater limit at all locations.

Comparison of predicted and measured concentrations for chemical species in the groundwater near the quarry for the years 1980 through 1985 confirms the conservative nature of the model calculations because the predicted values are generally higher than the measured values. The ratio of predicted to measured concentrations ranges from 0.4 or more for zinc in Well TW-6 to 170 or more for copper in the quarry pond (see Appendix I, Table I.13). Comparison of the model predictions and measured values is discussed in more detail in Appendix I, Section I.5.2.

Hanford Site. The model used to analyze the impacts of chemicals leaching out of the wastes buried at the Hanford site is the same model that was used for the raffinate pits area. The model and appropriate input parameters are described in Appendix I, Sections I.1 and I.3. Only the most mobile chemical species are expected to reach the groundwater under the buried wastes within 1,000 years* at the Hanford site. These species are predicted to reach the groundwater in 260 years and then move with the groundwater at a speed of 600 m/yr. Eighteen years later, they would have have moved 11 km downgradient and would have reached the boundary of the Hanford site at the Columbia River (Figure 3.11). At this point, predicted peak concentration contributions in the groundwater before it enters the river are 2.0 mg/L for nitrate (as N) and 3.3 mg/L for sulfate. Groundwater concentrations measured in wells in the Hanford area range from 0.1 to 9.8 mg/L nitrate (as N) and from 22 to 210 mg/L sulfate (U.S. Geol. Surv. 1981). For comparison with regulatory limits,** it should be noted that use of groundwater by facilities on the Hanford site is controlled and will continue to be restricted for the forseeable future. No groundwater is used for a community drinking water system, but some groundwater from the unconfined aquifer is used as a limited drinking water source during work hours at some places on the site (U.S. Dept. Energy 1986c). The contribution of any contaminants from disposing of the Weldon Spring wastes would be small compared to contributions from other DOE facilities on the Hanford site.

The nearest community water supply intake is on the Columbia River at Richland, downstream from the Hanford site. The EPA primary drinking water limit (40 CFR Part 141) is 10 mg/L for nitrate (as N); sulfate is not regulated by these standards. The limit applies specifically to water delivered to a free-flowing outlet (e.g., faucet) of the ultimate user. Using a dilution factor of about 16,000 for transit in the Columbia River between discharge of the groundwater to the river at the Hanford site and withdrawal of the water at Richland (U.S. Dept. Energy 1986c), the concentration contribution is reduced to 0.0001 mg/L nitrate (as N). This value is considerably below the drinking water limits.

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*The calculations assume that the bins containing the dried raffinate sludge will corrode and expose the sludge to infiltrating precipitation in a time period that is short compared to 1,000 years.

**The regulations of the Washington Department of Ecology (1986) for dangerous waste are similar to those of the Resource Conservation and Recovery Act (RCRA) for hazardous waste under 40 CFR Parts 260 through 265. Based on test results, the raffinate sludge (Nemec 1986) and quarry wastes (Bechtel Natl. 1985b) do not exhibit any of the characteristics of dangerous waste under Washington state regulations or hazardous waste under RCRA.

"Nearby Site". The "Nearby Site" would have more favorable conditions (e.g., thicker clay, lower hydraulic conductivity, deeper groundwater table, and/or higher sorption capacity) than the Weldon Spring site. Thus, the impacts at the "Nearby Site" would be less than those predicted for the Weldon Spring raffinate pits area under the action Alternatives 1, 2a, and 2b.

Uranium Processing Site. Wastes would be generated at the uranium processing site from reprocessing the Weldon Spring sludge. The contribution to any local groundwater contamination from the reprocessing wastes would be a small increment to the contribution from other radioactive wastes at the site.

4.1.2.5 Mitigative Measures

The groundwater at each area will be monitored to verify that concentrations of contaminants are below applicable limits. If concentrations in excess of applicable standards are detected, mitigative measures will be taken. Such mitigative measures could include reengineering/reconstruction of the waste-containment system or further remedial actions. A long-term monitoring program for the selected alternative will be developed by DOE.

Following removal of the wastes from the quarry to a disposal cell for all action alternatives, DOE will evaluate the need for any further actions at the quarry such as groundwater restoration. DOE also intends to evaluate the need for groundwater restoration in the raffinate pits area following implementation of one of the action alternatives. If groundwater restoration is not needed at either area, a decision will be made regarding whether the quarry or portions of the raffinate pits area can be released for unrestricted use or whether appropriate restrictions must be imposed for a period of time. If groundwater restoration is needed at either or both areas, DOE will perform a groundwater restoration feasibility study and select an appropriate remedial action alternative. After implementation of the groundwater restoration alternative, DOE will determine if the quarry area or portions of the raffinate pits area can be released for unrestricted or other appropriate use.

4.1.3 Surface Waters

4.1.3.1 Action Period

During the action period, cleanup and disposal of the contaminated materials would result in slightly increased concentrations of radiological or chemical contaminants in runoff from the contaminated areas. Good construction practices, such as diversion and control of runoff, will minimize the amount of contaminated materials in runoff.

It is planned for all water from the raffinate pits and chemical plant areas to be diverted to settling ponds, treated (as needed), and then used for spray irrigation. Water removed from the quarry in order to clean up the quarry wastes will also be collected and treated prior to being used for spray irrigation. As an alternative to spray irrigation, this water could be released to the Missouri River under controlled conditions and in compliance with applicable Missouri state requirements. Water disposal and treatment options are discussed in Appendix E, Section E.2.

A 45-ha (110-acre) area south of the raffinate pits area has tentatively been identified as being appropriate for spray irrigation. It is estimated that a total of about 314,000 m^3 (83,000,000 gal) of water will be collected (Bechtel Natl. 1984a, 1985a). Concentrations of nonradioactive chemicals will be reduced below state regulatory limits for irrigation water, and concentrations of radioactive species will be reduced below limits specified by DOE: 100 pCi/L for radium-226 and radium-228, 50 pCi/L for thorium-232, 500 pCi/L for uranium-234, and 600 pCi/L for uranium-238 (U.S. Dept. Energy 1986b). DOE will obtain any necessary permits from the state and follow the requirements that must be met for irrigation water, for runoff water from the irrigation area, and for monitoring the water treatment and spray irrigation systems.

In order to estimate the potential impact of water disposal by spray irrigation, it is assumed that all irrigation water would contain radium, thorium, and uranium at the concentration limits noted above. This would result in a total deposition of 7 pCi/cm^2 radium-226, 3.5 pCi/cm^2 thorium-232, 35 pCi/cm^2 uranium-234, and 42 pCi/cm^2 uranium-238 on the irrigated land. If it is assumed that these amounts are retained in the top 10 cm of soil, with an average bulk density of 1.35 g/cm^3 (Baes and Sharp 1983), then the resulting average concentrations of 0.52 pCi/g radium-226 and 0.26 pCi/g thorium-232 contributed by irrigation would be less than the existing background concentrations for surface soils. The criteria for release of the land for unrestricted use (see DOE Guidelines for Residual Radioactivity [Appendix D]) would also be satisfied. Concentrations of 2.6 pCi/g uranium-234 and 3.1 pCi/g uranium-238 would be slightly above the background concentration of < 0.68 to 1.62 pCi/g (uranium-238 and uranium-234 have the same background concentrations) (Boerner 1986).

For nonradiological parameters, Missouri groundwater recharge standards may affect the rate of spray irrigation. For example, Missouri limits nitrogen application to grass/woodlands to a rate of 0.167 MT (total N)/ha·yr (150 lb [total N]/acre·yr) (Newtown 1986). Estimation of the application rate necessary to satisfy this criterion is difficult because water volumes and nitrate concentrations fluctuate widely (see Appendix H, Tables H.1 and H.11). A conservative estimate can be obtained by combining the total volume

of water in the raffinate pits (216,000 m^3 [57,000,000 gal]) and a weighted average nitrate concentration (1,400 mg/L) in the surface water in the four pits to give a total of 300 MT (330 tons) of nitrate, or 68 MT (75 tons) total nitrogen. To dispose of 68 MT (75 tons) total nitrogen at a rate of no more than 0.167 MT/ha-yr would take about 9 years. Alternatively, the water could be treated to remove nitrates (see Appendix E, Section E.2.2.3).

Groundwater recharge standards for other nonradiological parameters are not likely to impact water disposal by spray irrigation if the expected 314,000 m^3 (83,000,000 gal) of water is sprayed over a 9-year period. At this rate, a net precipitation infiltration of 10 cm/yr (4 in./yr) would dilute the spray-applied water by a factor of about 2.2. This dilution factor would reduce the average concentrations of all regulated parameters to values that are equal to or less than the groundwater limits (Appendix H, Table H.12).

The specific location of the area to be used for spray irrigation has not yet been chosen. If the chosen area contains regions of radioactive contamination above DOE Guidelines for Residual Radioactivity (Appendix D), e.g., in the Army Reserve Property (Figure 3.7), then the contaminated areas will be excavated and cleaned up before spray irrigation begins. The mobilization in the soil of residual radioactivity and chemical contamination in the irrigated area is expected to be larger than in nearby nonirrigated areas. The increase should be less than a factor of two because the amount of water contributed annually by spray irrigation (7 cm [2.8 in]) would be less than the annual amount of precipitation that infiltrates the soil (10 cm [4 in.]).

The activities carried out during the action period are expected to result in a negligible radiological or chemical impact on surface waters at the Hanford site, the "Nearby Site", or the uranium processing site. A small amount of spillage is expected from unloading and placing the less-contaminated, bulk-loaded wastes into the disposal cell at Hanford or the "Nearby Site." The more highly contaminated sludge will be placed in bins and either buried directly (at Hanford or the "Nearby Site") or removed from the bins at the uranium processing facility. Good construction and operating practices should minimize the opportunity for transport of contaminated materials off-site via runoff water.

4.1.3.2 Long Term

Some surface waters -- such as Lakes 34, 35, and 36 in the Busch Wildlife Area; Burgermeister Spring; and Femme Osage Slough (see Figure 3.7 for locations) -- contain elevated levels of uranium and gross alpha activity (Appendix H, Table H.8). The concentration of uranium is below the applicable DOE guideline of 1,100 pCi/L for natural uranium (U.S. Dept. Energy 1986b); there is no applicable gross alpha limit. It is expected that the levels in

the lake and spring water will be gradually reduced for all action alternatives after completion of cleanup of the Ash Pond and Frog Pond areas around the chemical plant and after removal of contaminated soils from the vicinity properties. Uranium concentrations in Femme Osage Slough are expected to decrease after cleanup of the quarry and the contaminated soils in the alluvium area between the slough and quarry. Under the no-action Alternative 4, uranium and gross alpha concentrations in surface waters are expected to stay approximately the same as they are at present. Over thousands of years, the concentrations will gradually decrease as a result of leaching and erosion.

For Alternatives 1, 2a, 3a, 3b, and 3c, no long-term radiological or chemical impacts on the surface waters at Weldon Spring or the alternative sites are expected because the disposal cells will be maintained to prevent the escape of contaminants to surface waters. For the no-action Alternative 4, inspection and maintenance of walls and dikes at the raffinate pits area would prevent contaminated runoff from reaching nearby surface waters. However, runoff from other contaminated areas could be carried off-site.

In Alternative 2b, there will be a lead sheet in the disposal cell cover. Almost all precipitation that falls on the cover and penetrates the top 0.46-m (18-in.) layer of topsoil will percolate down to the lead sheet. Percolation water running off the lead sheet will collect in the drainage ditch around the disposal cell. The amount of water penetrating the cover is estimated to be about 13 cm (5 in.) per year. This is based on an annual precipitation rate of 94 cm (37 in.) (Table I.1) and the assumption that 14% of the precipitation that falls on the cover will penetrate the topsoil, sand, and riprap layers (Metry et al., undated).

Lead is resistant to corrosion, as exemplified by its use on roofs and as a sheathing for underground cables. In studies where lead pipe or sheeting was buried in soil for several years, reported corrosion rates range from < 0.0025 mm/yr (0.0001 in./yr) to 0.01 mm/yr (0.0004 in./yr), with the removal rate depending on the type of soil in which the lead was buried and on other factors (Amistadi 1985). At the lower corrosion rate, the total amount of lead removed by corrosion in 1,000 years would be 2.5 mm (0.1 in.) or 39% of the lead sheet. This amounts to a removal of 3.6 MT/yr (4.0 tons/yr) of lead from the sheet by corrosion, or 3,600 MT (4,000 tons) of lead in 1,000 years. At the higher corrosion rate, lead would be removed at a rate of 14 MT/yr (16 tons/yr), and the sheet would be completely corroded away in 640 years. It is expected that the lower corrosion rate is more applicable to the conditions at the Weldon Spring site under Alternative 2b. Prior to total corrosion of the lead sheet, pit corrosion would result in the development of holes, thus allowing infiltrating water to seep into the underlying wastes (see Appendix I, Section I.4.3).

Based on data from the Missouri Lead Study (Wixson 1977), the following scenario seems reasonable for the fate of lead removed by corrosion in the first years. Following site closure, very little lead is expected to appear in the drainage ditch because most of the lead corroded from the sheet would be deposited or bound on soil and sand particles in the cover. However, as time passed, lead would be transported off the cover and would appear in sediments and water in the drainage ditch surrounding the containment area. The average transport time is estimated to be about 23 years after site closure (see Appendix I, Section I.4.3). Concentrations of lead in the drainage ditch sediments could build up over the years to 200 to 40,000 ppm (Wixson 1977). Concentrations of lead in the percolation water running off the lead sheet are expected to be in the range of 0.006 to 0.09 mg/L if the runoff pH is between 7 and 8. In more acidic water (pH 5 to 6), lead concentrations could reach 6.5 mg/L. Peak concentrations of 10 mg/L or more of lead have been measured in runoff from areas having high natural lead concentrations. However, these high values last for only a few hours (Wixson 1977). Concentrations of lead in runoff from the cover over the Weldon Spring wastes are expected to be no greater than lead in runoff from areas in Missouri having high lead concentrations in soils and rock. The transport of lead off the cover by other means, such as erosion of the cover with the lead bound to the eroding soil particles, is not expected to be significant because the cover will be maintained to prevent erosion.

Potential off-site transport of lead corroded from the lead sheet in Alternative 2b can be mitigated by careful monitoring of lead in the sediments and water in the drainage ditch around the disposal cell and in the overflow from that ditch. If lead concentrations built up to unacceptable levels, corrective actions would be taken, as necessary.

4.1.4 Geologic Resources

For all alternatives, few adverse impacts on the local geological resources are expected. The various action alternatives would require the consumptive use of quarried rock, sand, gravel, and clay. Adequate supplies of these materials, as required for each alternative design, should be readily available at all alternative sites. With proper siting of the "Nearby Site", the availability of mineral resources should not be adversely affected. Consumptive use of petroleum products (e.g., diesel fuel and gasoline) are not expected to strain local supplies.

4.2 RADIOLOGICAL IMPACTS

Evaluation of radiological impacts involves calculating radiation doses to workers and the general public that might be expected to occur during the

remedial action period (including cleanup, transportation, and disposal of the Weldon Spring wastes) and during long-term management. The people considered in these calculations are the workers who would perform the remedial action, persons living near the disposal sites [within 80 km (50 mi)], and persons in proximity to the wastes as they are being transported to an alternative disposal site. Also evaluated is the dose to a hypothetical nearby individual.

DOE regulations require that the effective dose equivalent to any member of the general public as a result of DOE activities be limited to 500 mrem/yr for exposure lasting less than 5 years and 100 mrem/yr for periods of exposure in excess of 5 years. In addition, to be consistent with the limits given in 40 CFR Part 61, the dose from the air pathway only must be limited to 25 mrem/yr to the whole body and 75 mrem/yr to any organ (Vaughan 1985). Implementation of one of the proposed action alternatives will be done in a manner to ensure that doses to the general public are below these limits and are as low as reasonably achievable (ALARA). An environmental monitoring program appropriate to the selected alternative will be established to verify that these requirements are met.

Work practices and procedures will be developed to ensure that occupational doses are controlled and are below the allowable limits specified in DOE Order 5480.1A (i.e., 5 rem/yr to the whole body and not more than 3 rem/yr during each calendar quarter). The effects of both internal and external doses are included in this limit. Extensive efforts will be made by DOE to reduce worker exposure to levels that are ALARA under these limits, including detailed planning of all work activities involving potential radiation exposure to reduce exposure time, provision of adequate shielding, and protection against radionuclide uptake. Procedures may include specified time limits and specified types of protective clothing and equipment. The work will be carried out under written procedures that have been approved by health physicists and, depending on the radiation and contamination potential, the work may be continuously monitored by health physicists.

The principal environmental pathways through which people may be exposed to radiation from the wastes are (1) direct exposure from the wastes; (2) inhalation of radionuclides into the lungs, followed by redistribution to other organs of the body; and (3) ingestion of radionuclides through drinking water and foodstuffs. These principal pathways are diagrammed in Figure 4.5.

The principal mechanisms by which the radioactivity in these wastes could result in exposure to the general public is through atmospheric or hydrologic transport. Atmospheric transport of released radioactivity would be the principal mechanism of exposure during the action period. Over the long term, hydrologic transport would become a proportionately greater contributor to exposure. All pathways that could contribute to the dose incurred by the

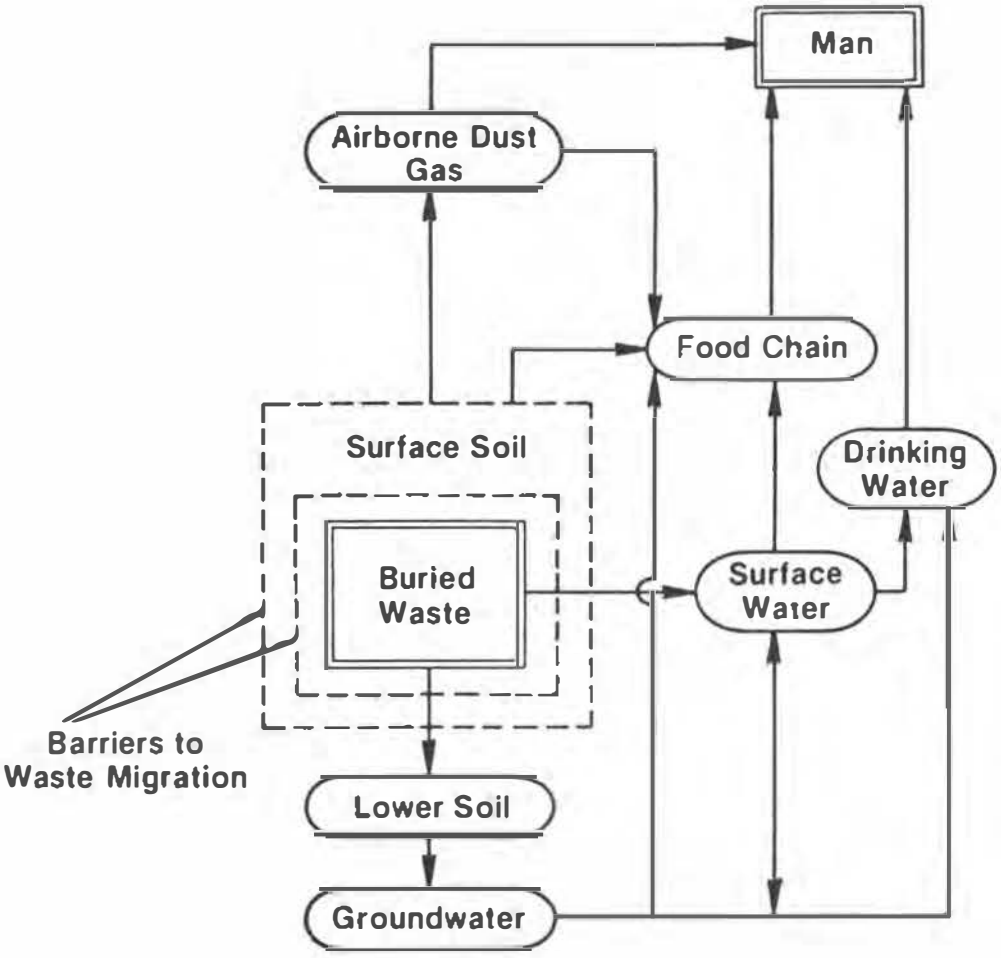

Figure 4.5. Principal Environmental Pathways of Radioactive Materials from Buried Wastes to Man.

general public are included in this assessment; however, the cumulative dose from all pathways would be a very small fraction of the background dose.

Radiation doses via the atmospheric pathway to the maximally exposed individual (a hypothetical individual in the vicinity of the wastes) and the cumulative dose to the general public were evaluated using the Uranium Dispersion and Dosimetry (UDAD) computer code (Momeni et al. 1979) and THODAD, a modified version of UDAD, for the thorium-232 series. The codes provide estimates of the potential radiation dose in the vicinity of the wastes. The input data required by UDAD and THODAD include population distribution data, meteorological data, agricultural productivity data, and radionuclide release data. The required radionuclide release data include radon gas emission rates from the waste surfaces, radionuclide release rates for particulate emissions, and the time periods over which the releases occur.

Radiation dose commitments from release through the groundwater pathway were calculated using the model and parameters described in Section 4.1.2.

Meteorological data collected at St. Louis-Lambert International Airport from 1970 through 1974 were used as input to the UDAD and THODAD codes for the Weldon Spring site and "Nearby Site". Meteorological data for the Hanford site were obtained from data collected at the Hanford Meteorological Station (Stone et al. 1983). The codes calculate air concentrations of radionuclides using a modification of the Gaussian plume model developed by Pasquill and Gifford (Gifford 1976; Eimutis and Konicek 1972). The concentration of each radionuclide was averaged over the crosswind direction to give the estimated ground-level concentration downwind of the source of emission. Concentrations of radionuclides suspended in the air and deposited on the ground were calculated at distances up to 80 km (50 mi) from the point of release.

Estimated exposures to radionuclides that originate in the releases from the Weldon Spring site, alternative disposal sites, and during transportation were converted to radiation doses to individuals. Radioactive materials taken into the body by inhalation or ingestion will continuously irradiate the body until removed by the processes of metabolism and radioactive decay. In this assessment, the estimates for internal dose from inhalation and ingestion are obtained by integrating over the assumed average remaining lifetime of 50 years of the exposed individual. The radiation dose to the total body and to internal organs from external exposure to penetrating radiation are approximately equal, but the dose from internal exposure may vary considerably because some radionuclides concentrate in certain organs of the body. In this EIS, "dose" represents the committed 50-year effective dose equivalent expressed in units of rem or millirem (mrem). Committed effective dose equivalents (i.e., whole-body equivalent doses) were calculated based on the methodology and weighting factors given in Report No. 26 of the International Commission on Radiological Protection (ICRP) (Int. Comm. Radiol. Prot. 1977). The effective dose equivalent for inhalation of radon decay products was based on the methodology given in ICRP Report No. 32 (Int. Comm. Radiol. Prot. 1981).

The doses to the general public are reported in terms of the environmental dose commitment, which includes both the doses received during actual releases plus the doses associated with lingering traces of contamination in the environment. The environmental dose commitment (EDC) is defined as the sum of all doses to individuals over the entire time period that the materials persist in the environment in a state available for interaction with humans. A 100-year integrating period was used. The 100-year EDCs computed are within about 10% of those that would have otherwise been computed based on an infinite integrating period.

4.2.1 Doses to the General Public

The major radiation doses to the general public would result from radionuclide transport via atmospheric and groundwater pathways and from transportation activities. Several individuals in the vicinity of the various sites were selected for analysis as being the potentially maximally exposed individuals for the various alternatives (Table 4.2). Because agricultural activity occurs in the area of the Weldon Spring site (within 80 km [50 mi]), especially to the north, it is important to consider the food chain pathway. The conservative assumption was made that 10% of the dietary intake of the population within 80 km (50 mi) is supplied by locally grown food (Hoormand 1985).

4.2.1.1 Doses to the General Public from Atmospheric Transport

Action Period. During the action period, radiological impacts to members of the general public would occur at the Weldon Spring site and at the alternative disposal sites. Excavation, transfer, reburial, gas diffusion, and wind erosion would result in releases to the atmosphere of radon gases

Table 4.2. Nearby Individuals at the Various Sites

Site	Description	Distance (km)[a]	Direction
Weldon Spring	Worker at Highway Department Maintenance Facility[b]	0.90	NE
Hanford Site	Worker at 200-West burial area[c]	1.0	E
	Resident of Richland, Washington[d]	38	SE
"Nearby Site"	Hypothetical individual	0.90	NE
Uranium processing site	Hypothetical individual	1.0	E

[a] The origin at the Weldon Spring site is taken to be the centroid of Pit 3. All distances are rounded to two significant figures.

[b] Continuous residence is assumed.

[c] A worker is assumed to be the maximally exposed individual for the action period.

[d] A Richland resident is assumed to be the maximally exposed individual during long-term management.

Conversion Factor: To convert kilometers (km) to miles (mi), multiply by 0.6214.

(radon-220 and radon-222) and contaminated particulates. Details on the methodology utilized to estimate radioactive releases to the atmosphere are given in Appendix J.

Radon gas releases consist of two separate sources: (1) the steady or continuous releases from exposed waste surfaces and (2) the "puff" or temporary releases associated with the disturbance of these wastes. The steady radon gas diffusion out of the waste materials was estimated using a one-dimensional diffusion equation (U.S. Nucl. Reg. Comm. 1983) that predicts the radon flux (rate per unit area) out of the contaminated area. The radon gas release rate is given by the product of the flux and the exposed surface area. When the wastes are disturbed and exposed to the air (e.g., when they are excavated or unloaded at the alternative sites), radon gases that have built up in the wastes may be released in "puffs". The portion of the radon gas that does not escape from the radioactive particulates (i.e., 80% for radon-222 and 90% for radon-220) undergoes radioactive decay to solid products. For this analysis, it is assumed that 20% of the total radon-222 and 10% of the total radon-220 would migrate out of the waste particulates into the surrounding void spaces and could be released in puffs. These values have been used for analyzing radon gas releases from uranium mill tailings and are referred to as the emanating power (U.S. Nucl. Reg. Comm. 1980a). After the wastes were again covered with clay, soil, etc., gas releases from the ground surface would be markedly reduced.

Radioactive particulates would also be released during the action period. These releases would be associated with activities involving movement of contaminated materials as well as from exposed surfaces. It is assumed that control methods (such as periodic watering and minimization of exposed surfaces) would be used at all sites. The annual particulate release rate was estimated using the methodology described in a report prepared by Argonne National Laboratory (1982) together with appropriate weather and other correction factors (U.S. Environ. Prot. Agency 1977). The releases are dependent primarily on the effectiveness of control methods and weather conditions during operations.

A summary of the estimated radioactive gas and particulate releases during the action period is presented in Table 4.3. The major source of radioactive release (in terms of total curies) is radon gas. The major source of radon gas release would be the continuous release from the exposed surface of the Weldon Spring wastes. The "puff" releases from disturbance of the wastes would be of short duration and of minor importance.

Table 4.3. Estimated Radioactive Releases during the Action Period

Alternative	Site	Release Rates Above Background					
		Particulates (mCi/yr)				Gases (Ci/yr)	
		Uranium-238[a]	Thorium-230	Radium-226[b]	Thorium-232[c]	Radon-222	Radon-220
1,2a,2b	Weldon Spring site	0.41	3.7	0.15	0.054	23	58
3a	Weldon Spring site	0.41	7.2	0.21	0.077	22	58
	Hanford site	3.0	9.0	0.94	0.29	7.2	0.1
3b	Weldon Spring site	0.41	7.2	0.21	0.077	22	58
	"Nearby Site"	0.17	0.52	0.054	0.017	0.81	0.1
3c	Weldon Spring site	0.58	7.7	0.76	0.093	23	58
	Uranium processing site	0.30	6.9	0.19	0.064	0.38	0.06
4	Weldon Spring site	0.042	0.0086	0.0055	0.0047	43	120

[a] Includes thorium-234, protactinium-234m, and uranium-234 -- which are assumed to be present in the same activity concentration as uranium-238.

[b] Includes all radioactive decay products down to polonium-210, which are assumed to be present in the same activity concentration as radium-226.

[c] Includes all radioactive decay products in the thorium-232 series, which are assumed to be present in the same activity concentration as thorium-232.

The particulate releases during the action period would be lowest for Alternative 4 (no action) because the wastes are not disturbed in this alternative. The particulate releases for this alternative would be mainly from wind-eroded particulates from the contaminated vicinity properties and the chemical plant. The particulate releases during the action period would be highest for Alternative 3a (Hanford) during waste disposal activities because of the arid climate at the Hanford site.

The estimated radiation doses via atmospheric transport to the general public and the maximally exposed individual during the action period are summarized in Table 4.4. The dose from airborne releases to the general public in the vicinity of the Weldon Spring site or other alternative sites would be comparable for all action alternatives; the estimated dose ranges

Table 4.4. Estimated Cumulative Doses to the General Public from Atmospheric Releases during the Action Period[a]

Alternative	Site	Environmental Dose Commitment (person-rem)			Maximally Exposed Individual[b] (mrem)
		Radon Gases	Particulates	Total Dose	
1,2a,2b	Weldon Spring site	23	7.5	31	1.0
3a	Weldon Spring site	23	13	36	-
	Hanford site	0.59	0.60	1.2	2.3
	TOTAL	24	14	37	-
3b	Weldon Spring site	23	13	36	1.6
	"Nearby Site"	0.63	1.4	2.0	-
	TOTAL	24	14	38	-
3c	Weldon Spring site	23	14	37	1.8
	Uranium processing site	1.1	0.72	1.8	-
	TOTAL	24	15	39	-
4	Weldon Spring	44	1.4	45	0.41

[a] The 1980 census data are used for the population distribution within 80 km (50 mi) of the sites under consideration (i.e., 2.3 and 0.3 million for the Weldon Spring and Hanford areas, respectively). All values are rounded to two significant figures.

[b] For alternatives involving more than one location, the maximally exposed individual dose is given for the location having the larger dose.

from 31 to 39 person-rem over the 10-year period. The dose from the no-action Alternative 4 is estimated to be slightly higher (45 person-rem). At Hanford (Alternative 3a), the dose to the general public resulting from activities associated with the Weldon Spring wastes would be about 1 person-rem compared to a 40-person-rem dose over the same 10-year period for other waste-management activities at the Hanford site (U.S. Dept. Energy 1986a). All these doses are only a very small fraction of the dose received from background radiation.

The airborne releases associated with the off-site Alternatives 3a, 3b, and 3c would result in a higher radiological impact to the general public than the on-site disposal alternatives. The higher radiological impact would be mainly due to the additional releases resulting from drying the sludge prior to transport off-site.

The doses to the maximally exposed individuals at various sites for the entire action period range from a low of 0.41 for an individual located near the Weldon Spring site (Alternative 4) to a high of 2.3 mrem for a person working on the Hanford site (Alternative 3a) 1.0 km from the burial area for the Weldon Spring wastes. The nearby worker at the Hanford site would receive the highest dose because of his/her proximity to the burial area and because of the higher airborne releases of radioactivity at the arid Hanford site compared to the other alternative sites. For comparison, the dose to a person living near the Weldon Spring site from external background radiation is about 99 mrem/yr (Bechtel Natl. 1986) and that for an individual living near the Hanford site is about 100 mrem/yr (Price 1986).

Long-Term Management. During long-term management, it is assumed that maintenance, monitoring, and corrective remedial actions will be carried out as necessary. For all action alternatives, the earthen covers will be kept in repair at all sites, which essentially will eliminate radon-220 gas and particulate releases. However, some radon-222 will escape, and the primary source of airborne exposure during this period for all action alternatives would be inhalation of short-lived radon-222 decay products.

The calculated fluxes of radon gases are several orders of magnitude lower for the humid sites than for the arid sites due to the higher attenuation of radon gases in the moist soil. Radon-222 flux at the Hanford site would be higher than at Weldon Spring, but at both sites the flux is expected to be well below the DOE regulatory limit of 20 $pCi/m^2 \cdot s$. Radon-222 flux at the "Nearby Site" is also expected to be much below the regulatory limit because the containment design and meteorological conditions are similar to those at Weldon Spring. The flux at the uranium processing site would be site-specific but is expected to be similar to the arid Hanford site.

Calculation of doses via the atmospheric pathway was performed in the same manner for long-term management as for the action period. Meteorological patterns and population distributions were assumed to be the same as those for the action period. The estimated doses to the general population for long-term management are given in Table 4.5. Because of the arid climate at the Hanford site, Alternative 3a would have the highest radiological impact from radon gas diffusion through the cover. A population dose of about 600 person-rem at the Hanford site is estimated for the atmospheric pathway for Alternative 3a over 1,000 years. The doses to the general public would be significantly lower for the other action alternatives. For the no-action Alternative 4, a population dose of about 10,000 person-rem via the atmospheric pathway is estimated for the same time period. This dose would be primarily due to resuspension of radioactive particulates and radon gas diffusion from the contaminated materials. The doses via the atmospheric pathway to the maximally exposed individuals near the disposal areas would be less than 1 mrem/yr for all alternatives at all sites.

4.2.1.2 Doses to the General Public from Groundwater Transport

The significant sources for potential contamination of the surrounding groundwater of the raffinate pits area are radium-226 and uranium. Because of the slow migration of radium and uranium in the clay materials beneath the raffinate pits area, no impacts via groundwater are expected during either the action period or long-term management (see Section 4.1.2). No contamination at the county well field near the quarry is expected during the action period. During the long term, slightly elevated uranium levels may occur in the well field (Section 4.1.2.3), but concentrations of radionuclides (including uranium) are not expected to exceed regulatory limits.

The doses as a function of time for an individual who consumes water from the county well field at a rate of 370 L/yr (98 gal/yr) (U.S. Environ. Prot. Agency, undated) are shown in Figure 4.6 for the action and no-action alternatives. For the no-action Alternative 4, it is predicted that, within 1,000 years, the maximum uranium concentrations in well water resulting from the migration of contaminants from the quarry would increase to 2.0 pCi/L at Well 8 and to 0.2 pCi/L averaged over all wells;* for comparison, background concentrations are < 1.5 to < 3.6 pCi/L (Section 4.1.2.3). The EPA has recently proposed a guidance level of 10 pCi/L for uranium in drinking water (Cothern et al. 1983). The estimated ingestion doses for an individual are 0.2 mrem/yr (maximum from Well 8 alone) and 0.02 mrem/yr (average for the entire well field).

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*The uranium radionuclides are assumed to be in the natural activity ratio of uranium-238:uranium-235:uranium-234 of 1:0.046:1

Table 4.5. Estimated Doses to the General Public from Releases during Long-Term Management[a]

Alternative	Site	Cumulative Environmental Dose Commitment (person-rem)[b]				Maximally Exposed Individual[e] (mrem/yr)
		Air Radon Gases[c]	Air Particulates[c]	Groundwater[d]	Total Dose	
1	Weldon Spring site	60	-	120	180	0.034
2a	Weldon Spring site	39	-	120	160	0.032
2b	Weldon Spring site	62	-	120	180	0.034
3a	Weldon Spring site	-	-	120	120	0.028
	Hanford site	600	-	-	600	-
	TOTAL	600	-	120	720	-
3b	Weldon Spring site	-	-	120	120	0.028
	"Nearby Site"	42	-	-	42	-
	TOTAL	42	-	120	160	-
3c	Weldon Spring site	8.0	-	120	130	0.029
4	Weldon Spring site	10,000	270	420	11,000	0.27

[a] All numbers rounded to two significant figures.

[b] Cumulative dose over 1,000 years.

[c] The 1980 census data are used for the population distribution within 80 km (50 mi) of the sites under consideration (i.e., 2.3 and 0.3 million for the Weldon Spring and Hanford areas, respectively). All values are given to two significant figures.

[d] Based on an assumed population of 65,000 persons consuming water from the county well field.

[e] Individual doses are given for the year of maximal exposure. For alternatives involving more than one location, the maximally exposed individual dose is given for the location having the larger dose.

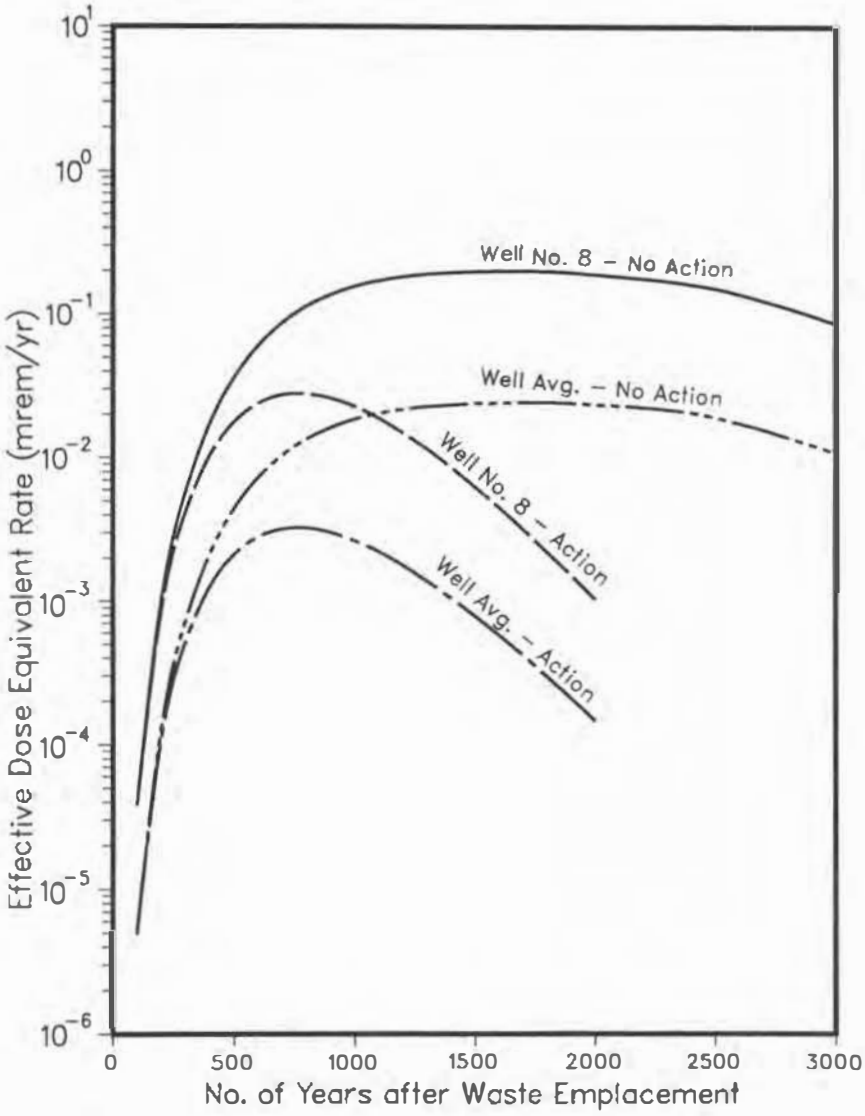

Figure 4.6. Individual Dose Due to Ingestion of Well Water from the Well Field.

The estimated cumulative (time-integrated) doses via the groundwater pathway to the general public over 1,000 years are given in Table 4.5. The dose for the no-action Alternative 4 is estimated to be 420 person-rem from consuming water from the well field near the quarry. Because the quarry wastes will be removed in all action alternatives, the time-integrated dose over the same time period for the groundwater pathway would be reduced to 120 person-rem for these alternatives. It should be noted that the water from the well field near the quarry is currently softened with lime for the public water supply (Parsons 1985); under appropriate operating conditions, a lime softening process could remove more than 80% of the uranium (Reid et al.

1985). Because the dose calculations are based on untreated well water, the actual dose due to ingesting water from the well field is expected to be lower.

The dose to the general population in the vicinity of the uranium processing site was not calculated because the incremental dose commitment from any wastes remaining from reprocessing the Weldon Spring sludge would be negligible in comparison to that from other radioactive materials remaining from other processing activities at this site.

4.2.1.3 Doses to the General Public from Transportation of the Wastes

Under Alternatives 3a, 3b, and 3c, the wastes will be transported off-site. Under Alternative 3a, all the wastes will be transported by rail to the Hanford site near Richland, Washington, 3,500 km (2,200 mi) from the Weldon Spring site. Under Alternative 3b, all the wastes will be transported by truck to a "Nearby Site", which for this analysis was assumed to be 160 km (100 mi) away. Under Alternative 3c, only the sludge from the raffinate pits and quarry will be transported by rail to a uranium processing site assumed to be located in the Four Corners area of the southwestern United States near Grants, New Mexico, about 1,900 km (1,200 mi) away. Details on the evaluation of impacts associated with transportation of these radioactive wastes are given in Appendix F.*

All sludge from the raffinate pits (62,000 MT [68,000 tons]) and quarry (1,100 MT [1,200 tons]) is assumed to be shipped in containers. The remaining wastes from the chemical plant, vicinity properties, and quarry as well as clay from the pits (840,000 MT [920,000 tons]) will be shipped unpackaged.

The general public exposed to radiation consists of people living or working along the transport route (off-link) and people traveling along the route in either the same or opposite direction (on-link). Several mechanisms could contribute to radiation exposure: (1) direct radiation, (2) dispersion of particulates, and (3) radon-220 and radon-222 gas emanation. A simplified version of the method of Chen et al. (1981) was used to obtain estimates of radiation doses from direct exposure. The gamma spectra of the uranium-238 and thorium-232 decay series radionuclides were obtained from the isotope library contained in the ORIGEN computer code (Bell 1973). A line source

*The radioactive materials in the quarry will be transported to the raffinate pits area for all action alternatives. The radiological impacts associated with this movement are included in those given in Section 4.2.1.1 (for doses to the general public) and Section 4.2.2.2 for waste-handling activities (for occupational doses).

model (Yuan and Chee 1982) was used to assess the radiation doses from the dispersion of particulates and radon gases.

The estimated doses to the general public associated with transport of the Weldon Spring wastes for the different alternatives are summarized in Table 4.6. The potential cumulative population doses during normal transport conditions would be 210 person-rem for shipping all of the wastes to the Hanford site (Alternative 3a), 82 person-rem for shipping all of the wastes to the "Nearby Site" (Alternative 3b) and 0.036 person-rem for shipping the sludge from the raffinate pits and quarry to a uranium processing site for reprocessing (Alternative 3c). The radiation doses are much larger for Alternatives 3a and 2b due to the bulk (unpackaged) shipment of all contaminated materials not having radioactivity concentrations high enough to classify them as being radioactive for transportation purposes. Bulk shipment of this material will result in the emission of radioactive particulates along the transportation route.

Transport of these radioactive materials involves the potential for accidents, with subsequent exposure of the general public to any radioactive materials that would be spilled. Based on the large number of shipments and the distances involved, several accidents could occur and spilled material could become available for release to the environment. The cumulative population doses for transportation accidents are estimated to be 4.8 person-rem for Alternative 3a, 0.98 person-rem for Alternative 3b, and 0.88 person-rem for Alternative 3c. Details of this analysis are presented in Appendix F, Section F.4.

The dose to the maximally exposed individual from transportation of the wastes was calculated for a person living 30 m (100 ft) away from the railroad track or roadway for the whole duration of the shipments. The estimated doses

Table 4.6. Cumulative Dose to the General Public during Transport of the Weldon Spring Wastes

Alternative	Destination	Mode of Transport	100-Year Environmental Dose Commitment (person-rem)	
			Normal Conditions	Accidents
3a	Hanford site	Rail	210	4.8
3b	"Nearby Site"	Truck	82	0.98
3c	Uranium processing site	Rail	0.036	0.88

are 0.58 mrem for Alternative 3a (Hanford site), 0.57 mrem for Alternative 3b ("Nearby Site"), and 0.11 mrem for Alternative 3c (uranium processing site). The doses are larger for Alternatives 3a and 3b due to particulate releases from the unpackaged wastes.

4.2.2 Doses to Workers

Implementation of any of the alternatives would result in workers being exposed to radioactive materials and thus incurring radiation doses. In this EIS, the doses to workers (occupational doses) are estimated for the following activities: (1) activities at Weldon Spring, (2) transportation activities, and (3) activities at alternative long-term management sites.

4.2.2.1 General Assumptions and Methodology for Calculating Occupational Doses

Three major sources of radioactivity could lead to occupational doses: (1) direct gamma exposure, (2) release of radioactive particulates, and (3) release of radon-220 and radon-222 gases. The first source would result in an external whole-body dose whereas the second and third sources would result in doses to internal organs via inhalation of radionuclides.

External doses to workers were calculated by multiplying the length of time workers would spend in various radiation fields by the radiation field strength and summing over the number of workers involved. Internal doses were calculated in a similar manner using the estimated concentrations of airborne radionuclides in the work environment and the average rates of inhalation. The measured on-site radiation exposure level at the Weldon Spring site ranges from 0.005 to 0.5 mrem/h (Bechtel Natl. 1985a). An average dose rate of 0.05 mrem/h was used to estimate the dose from exposure to external gamma radiation. The amount of dust in the vicinity of earth-moving activities has been estimated by NRC to be about 0.01 g/m^3 of material handled, with 10% being in the respirable particle-size range, i.e., <10 μm in diameter (U.S. Nucl. Reg. Comm. 1980b). Frequent watering is assumed to reduce particulate concentrations by a factor of two. The release rate of respirable dust is therefore estimated to be about 0.0005 g/m^3.

The average concentration of radon-222 decay products that workers handling the wastes would be exposed to is estimated to be 0.001 Working Level (WL), based on past experience at uranium milling facilities (U.S. Nucl. Reg. Comm. 1980a) and on the radium-226 concentration in the Weldon Spring wastes. Doses to workers from radon-220 are estimated to be 10% of those from radon-222 based on the much shorter half-life of radon-220 (55 s) compared to radon-222 (3.8 d).

4.2.2.2 Occupational Doses during the Action Period

The occupational doses for activities at the various locations during the action period were estimated using the methodology described above and estimated work force requirements for the various sites. The occupational doses are summarized in Table 4.7.

Occupational Doses from Waste-Handling Activities. The radioactive wastes will be handled at the Weldon Spring site, to varying degrees, for all of the alternatives. In addition, the wastes will also be handled at alternative disposal or treatment sites for Alternatives 3a, 3b, and 3c. The largest component of the occupational doses at the Weldon Spring site would be associated with excavation, packaging, and loading of the wastes. Because the work load and amount of material to be handled are not appreciably different among all the action alternatives, the occupational doses for all waste-handling activities would be similar, estimated to be about 110 to 130 person-rem.

When the packaged wastes are received for disposal at the Hanford site (Alternative 3a) or the "Nearby Site" (Alternative 3b), they will be removed from the transport vehicles and placed into trenches for burial. Before placement, it is assumed that the lids of the waste packages will be removed briefly to fill the void spaces above the wastes with soil. Workers will be

Table 4.7. Occupational Doses for the Various Alternatives during the Action Period (10 Years)

Alternative	Dose (person-rem)		
	Handling Wastes	Transporting[a] Wastes	Total[b]
1: Improved containment in the existing raffinate pits	110	-	110
2a: New cell, partially above grade	120	-	120
2b: New cell, completely above grade	120	-	120
3a: Hanford site	130	2.7	130
3b: "Nearby Site"	130	98	230
3c: Uranium processing site	120	1.0	120
4: No action	5.1	-	5.1

[a] Doses to crew members transporting the wastes.

[b] Totals may not add due to roundoff to two significant figures.

exposed to radon gases and particulates from the exposed wastes during these activities. The occupational dose at the two alternative disposal sites is estimated to be 20 person-rem. This dose is lower than the doses at the Weldon Spring site because of the considerably smaller work force requirements associated with the disposal activities at these two alternative disposal sites. The occupational dose from handling the packaged materials at the uranium processing site for Alternative 3c is also estimated to be about 20 person-rem based on the similar activities involved.

For the no-action Alternative 4, the occupational dose is estimated to be 5.1 person-rem because of the minimal work involved.

Occupational Doses Due to Transportation of the Wastes. The principal pathway by which drivers transporting the wastes would be exposed to radiation would be direct external exposure to gamma rays emitted from the wastes. Doses from exposure to contaminated particulates and radon-220 and radon-222 gases would be negligible. For truck transport to the "Nearby Site" (Alternative 3b), it was assumed that the exposure rate for a driver would be the same as the exposure rate at the surface of the waste shipment and that the drivers would be exposed to radiation from the wastes for the entire transport distance. For train transport (Alternatives 3a and 3c), it was assumed that the workers would come in close proximity to the wastes 10% of the time during transport and during the remainder of the trip would be exposed to a very low level of radiation. A crew of five for train transport and a crew of two for truck transport were assumed for estimating the occupational dose. The collective occupational doses are estimated to be 2.7, 98, and 1.0 person-rem for Alternatives 3a, 3b, and 3c, respectively.

Summary of Occupational Doses. The estimated cumulative doses to workers during the action period range from 5.1 person-rem (Alternative 4, no action) to 230 person-rem (Alternative 3b, removal of all wastes to a "Nearby Site"). The highest dose would be incurred for Alternative 3b because of the higher worker dose from truck transportation to the "Nearby Site". The lowest dose would be incurred for the no-action Alternative 4 because of the minimal amount of activities associated with this alternative.

4.2.2.3 Occupational Doses During Long-Term Management

During long-term management, doses to workers at the Weldon Spring site and the alternative sites are expected to be negligible for all action alternatives because there would be only maintenance and monitoring activities and few workers would be involved. Workers would periodically be on-site to collect air and water samples, to inspect the condition of the containment system, to maintain the fences and mow the grass, to patrol the site for security purposes, and to perform other routine maintenance and monitoring

activities. Occasionally, workers might be on-site to take remedial actions such as repairing any failures in the containment system. During long-term management, workers would not be exposed directly to the radioactive wastes. The dose from all pathways from the buried wastes is expected to be negligible, < 0.1 person-rem/yr for the workers over the long term.

Occupational doses for the no-action Alternative 4 would be higher during long-term management because the waste materials would not be as well contained. It would therefore be necessary for workers to perform more repairs more frequently than for the action alternatives. The occupational doses for these activities would not be expected to be more than a few person-rem/yr unless major repairs were needed. Major repairs would result in occupational doses comparable to those for implementation of the action alternatives (i.e., about 100 person-rem).

4.2.3 Cumulative Radiological Impacts

4.2.3.1 General Public

The estimated cumulative (time-integrated) doses for all pathways to the general public in the vicinity of the Weldon Spring site and other alternative disposal sites are summarized in Table 4.8. The action period doses include doses at the Weldon Spring site, at the alternative disposal sites, and along the transportation routes. During the action period, estimated doses to the general public range from a low of 31 person-rem for the on-site disposal alternatives to a high of 250 person-rem for Alternative 3a (Hanford). The dose for Alternative 3a is the largest among all the alternatives because Alternative 3a involves the longest transportation route and exposure during transportation is a large contributor to the population dose for this alternative.

During long-term management, population doses would be incurred in the vicinity of the sites containing radioactive materials due to the gradual migration of radioactivity via groundwater, radon gas diffusion, or wind erosion. All the action alternatives would result in a reduction in radiological impacts over the long term largely because the radioactive wastes would be removed from the quarry and all the wastes would have better containment at the disposal site selected. The estimated dose to the general public associated with the groundwater pathway would increase gradually over time and would reach a peak in about 800 years for the action alternatives and in about 1,700 years for the no-action Alternative 4. The estimated total cumulative doses via all pathways to the general public over 1,000 years range from a low of 130 for Alternative 3c (uranium processing site) to a high of

Table 4.8. Summary of Cumulative Doses to the General Public from All Pathways

Alternative	Time (yr)	Cumulative Dose[a] (person-rem) Air[b]	Water	Transportation	Total
1: Improved containment in the existing raffinate pits	10	31	-	-	31
	1,000	60	120	-	180
2a: New cell, partially above grade	10	31	-	-	31
	1,000	39	120	-	160
2b: New cell, above grade	10	31	-	-	31
	1,000	62	120	-	180
3a: Hanford site	10	37	-	210	250
	1,000	600	120	-	720
3b: "Nearby Site"	10	38	-	82	120
	1,000	42	120	-	160
3c: Uranium processing site	10	39	-	0.036	39
	1,000	8.0	120	-	130
4: No action	10	45	-	-	45
	1,000	10,000	420	-	11,000

[a] Environmental dose commitment integrated over the 10-year action period or long-term management period. All values are rounded to two significant figures; totals may not add due to rounding.

[b] Includes radon gases and particulates (see Table 4.5).

11,000 for Alternative 4 (no action). For comparison, the population near the Weldon Spring site would receive a dose of 230,000,000 person-rem from background sources of radiation over the same time period based on a dose rate of 99 mrem/yr (Bechtel Natl. 1986) and an exposed population of 2.3 million people.

The doses to the maximally exposed individual for all alternatives are given in Table 4.9. During the entire action period, the highest maximum individual dose is estimated to be 2.3 mrem for Alternative 3a. During long-term management, the maximum individual dose is estimated to be less than 1 mrem/yr for all alternatives. For comparison, a person living near the Weldon Spring site receives a dose of about 99 mrem/yr from background external radiation (Bechtel Natl. 1986).

Table 4.9. Doses to Maximally Exposed Nearby Individuals

Alternative	Time (yr)	Dose [a] (mrem)
1: Improved containment in the existing raffinate pits	10	1.0
	1,000	0.034/yr
2a: New cell, partially above grade	10	1.0
	1,000	0.032/yr
2b: New cell, completely above grade	10	1.0
	1,000	0.034/yr
3a: Hanford site	10	2.3
	1,000	0.028/yr
3b: "Nearby Site"	10	1.6
	1,000	0.028/yr
3c: Uranium processing site	10	1.4
	1,000	0.029/yr
4: No action	10	0.41
	1,000	0.27/yr

[a] During the action period, the doses are integrated over the entire 10-year period. During long-term management, the annual doses in the year of maximum exposure are given. All values are rounded to two significant figures.

4.2.3.2 Workers

The estimated cumulative doses to the work force for each alternative during the action period (10 years) are given in Table 4.7. These doses would be incurred during implementation of the various alternatives and include activities at the Weldon Spring site, along transportation routes, and at alternative long-term management sites. The highest dose to workers is estimated to be 230 person-rem for Alternative 3b ("Nearby Site"), primarily because of the extra worker dose during truck transport. There would be no significant doses to workers during long-term management at all sites.

4.2.4 Estimation of Health Effects

4.2.4.1 Risk Estimator

The significance of the estimated doses for the various alternatives may be placed in perspective by estimating their effects on human health. The

potential health effects from exposure to low levels of radiation may include a small (immeasurable) increase in the occurrence of cancer, depending on what particular organ is irradiated, and possible genetic effects that may occur in future generations. For the Weldon Spring site, implementation of any of the alternatives is not expected to result in exposures to radiation that are large enough to produce either immediately observable health effects in any individual or long-term effects that could be identified by statistical methods in the exposed population.

The probability that a unit dose of radiation will produce a deleterious effect in an individual (or effects in a population) is commonly referred to as a risk factor or risk estimator. Estimates of risk factors from exposure to radiation, for both individuals and populations, are available in a number of publications. The National Academy of Sciences issued a report in 1960 from the Advisory Committee on Biological Effects of Atomic Radiation called the BEAR Report (Natl. Acad. Sci. 1960). In 1972, the National Academy of Sciences Advisory Committee on the Biological Effects of Ionizing Radiation issued a report commonly referred to as "BEIR I" (Natl. Acad. Sci. 1972). The committee was subsequently asked to review the risk estimators, and the results of that review were issued in the 1980 report referred to as "BEIR III" (Natl. Acad. Sci. 1980). Other groups of experts also have published risk estimators for radiation exposure; for example, the International Commission on Radiological Protection (ICRP) and the National Council on Radiation Protection and Measurements (NCRP) have both studied radiation effects. The ICRP issued Publication 26 in 1977 (Int. Comm. Radiol. Prot. 1977). In the same year, a United Nations study group (U.N. Scientific Committee on the Effects of Atomic Radiation, or UNSCEAR) published an extensive report that included estimates of risks of cancer from ionizing radiation (U.N. Sci. Comm. Effects Atom. Radiat. 1977).

Health effects in this EIS specifically refer to the induction of fatal cancers and genetic defects. There have been no direct measurements of increased cancer for low-level radiation exposures; data exist only for higher-level exposures. Risks at lower doses have been estimated by assuming that the same dose/health-effects relationship applies to low doses as to high doses and then extrapolating from data taken at higher dose levels. The cancer risk estimates developed by various professional organizations are presented in Table 4.10. The values represent the estimated range of additional cancer mortality (above the normal mortality from such cancers) per 0.5 rem of radiation to a population of one million people continuously exposed to external gamma radiation. Comparison of the values in Table 4.10 indicates that there is general agreement among the estimates developed by the three studies. In application of these estimates, care must be taken to

Table 4.10. Estimated Cancer Mortality from Exposure to Low-Level Radiation

Source	Number of Additional[a] Fatal Cancers Estimated to Result from Continuous Exposure of One Million People to 0.5 rem of External Gamma Radiation[b]
National Academy of Sciences (1980)--BEIR III	33-85
International Commission on Radiological Protection (1977)	50
United Nations Scientific Committee on the Effects of Atomic Radiation (1977)	38-88

[a] Additional means above the normal cancer mortality.

[b] The number of additional cancers can be multiplied by two to estimate the total incidence of cancer.

Source: U.S. Department of Energy (1983).

identify the limitations of each study. The UNSCEAR and ICRP approach provides average lifetime risk factors. The UNSCEAR report discusses absolute risk, or the number of expected cancer cases that will result from exposure of a given population. The BEIR III report considers both absolute and relative risk. Relative risk is the ratio of incidence of cancer in an exposed population to the incidence in a control population.

The American Cancer Society (1978) indicates that about half of all cancer cases are fatal. Thus, the numbers in Table 4.10 can be multiplied by two to estimate total incidence (including morbidity and mortality). These cancers would be in addition to those normally expected in a population. According to the American Cancer Society, the individual risk of getting cancer for any member in the population is 1 chance in 4. The cure rate for these cancers ranges from 90% to 5%, depending on the type of cancer.

The values given in Table 4.10 are considered by many radiation protection specialists to be the best estimates that can be provided. Press releases have publicized radiation-effects studies in which the authors have indicated that the risk is much higher. For example, epidemiological studies on workers from the Portsmouth Naval Shipyards and the Hanford site indicate that a greater risk may exist from radiation exposure than has been reported in the past. The Committee on the Biological Effects of Ionizing Radiation

(Natl. Acad. Sci. 1980) reviewed these studies and took the results into consideration in making its recommendations in the BEIR III report.

Genetic effects can be estimated as a result of exposure of the parents to a given radiation level. The risk estimator for serious genetic effects, as expressed in the first two generations, is about 4×10^{-5}/person-rem (Int. Comm. Radiol. Prot. 1977). The total risk of genetic damage that may be expressed in all subsequent generations is about twice that which is expressed in the first two generations only.

The risk estimator selected for this EIS was based on consideration of the risk estimators from the various publications discussed above. Thus, to estimate health risks to both the general public and workers for the various alternatives evaluated in this EIS, a risk estimator of 1.7×10^{-4}/person-rem (Int. Comm. Radiol. Prot. 1977) was used.

4.2.4.2 Risk of Adverse Health Effects for Each Alternative

General Public. The major component of the health effects associated with the radioactive releases under all alternatives would be the occurrence of cancer deaths; more than 90% of the estimated health effects would be fatal cancers. Estimates of increased risk of adverse health effects to the general public as a result of doses incurred during the action period and long-term management are shown in Table 4.11. During the action period, the estimated number of adverse health effects ranges from a low of 0.0053 (for Alternatives 1, 2a and 2b) to a high of 0.043 (Alternative 3a). All of these alternatives have a very low probability of producing an adverse health effect during the action period.

During long-term management, Alternative 4 (no action) would have the highest risk of increased adverse health effects. Over a period of 1,000 years, the no-action Alternative 4 would be expected to result in about two adverse health effects. For comparison, 4,300,000 cancer deaths would normally be expected in the same exposed population over the same period of time (assuming that the yearly death rate is 8.8/1,000 and that 21% of all deaths are caused by cancer--U.S. Depart. Health Human Serv. 1985). All of the other alternatives would be expected to produce less than one adverse health effect.

Workers. Estimates of increased risk of adverse health effects to workers as a result of doses incurred during the action period are given in Table 4.12. The estimated number of health effects is lowest for Alternative 4 because of the minimal amount of work involved. The estimates of adverse health effects for all action alternatives are similar, ranging from a

Table 4.11. Cumulative Health Effects to the General Public

Alternative	Time (yr)	Cumulative Health Effects[a]
1: Improved containment in the existing raffinate pits	10	0.0053
	1,000	0.031
2a: New cell, partially above grade	10	0.0053
	1,000	0.027
2b: New cell, completely above grade	10	0.0053
	1,000	0.031
3a: Hanford site	10	0.043
	1,000	0.12
3b: "Nearby Site"	10	0.020
	1,000	0.027
3c: Uranium processing site	10	0.0066
	1,000	0.022
4: No action	10	0.0077
	1,000	1.9

[a] Health effects integrated over the 10-year action period or 1,000-year long-term management period.

Table 4.12. Estimated Health Effects to Workers during the Entire Action Period (10 years)

Alternative	Total Health Effects
1: Improved containment in the existing raffinate pits	0.019
2a: New cell, partially above grade	0.020
2b: New cell, completely above grade	0.020
3a: Hanford site	0.022
3b: "Nearby Site"	0.039
3c: Uranium processing site	0.020
4: No action	0.00087

low of 0.019 for Alternative 1 to a high of 0.039 for Alternative 3b. Alternative 3b has the highest number of estimated adverse health effects for workers as a result of the occupational dose incurred during truck transportation of the wastes to the "Nearby Site". For all alternatives, the expected number of health effects is significantly less than one. It is very unlikely that there would be any adverse health effects in the entire work force as a result of activities associated with this project.

4.2.5 Mitigation of Radiological Impacts

The radiological impacts are estimated based on several assumptions regarding engineering design, construction practices, health-physics safety procedures, compliance with existing regulations, and the degree of physical and institutional control for each time period. Following are additional measures that could be taken to mitigate the predicted radiological impacts.

4.2.5.1 Action Period

During the action period, radiation doses to the general public near the Weldon Spring site and the alternative sites would result primarily from releases of radon-220 and radon-222 gases and particulates from the Weldon Spring wastes. These releases could be reduced by (1) minimizing the amount of waste surfaces exposed at any given time, e.g., limiting the excavation area, (2) minimizing the amount of time the wastes are exposed, e.g., use of temporary coverings and construction of new permanent covers at alternative sites as quickly as possible, (3) minimizing disturbance of exposed surfaces, e.g., limiting equipment speeds, (4) wetting exposed surfaces to minimize dust and diffusion of radioactive gases, and (5) ceasing operations during adverse weather conditions.

Impacts to workers would be minimized by implementing an effective health-physics program in accordance with DOE's ALARA philosophy. Specific measures to reduce occupational doses include (1) minimizing the time spent in proximity to the wastes, (2) encouraging attention to personal hygiene, (3) using protective clothing and respirators, (4) using extra shielding, as necessary, between the workers and the wastes, and (5) employing simulation training to improve work skills that can shorten the time needed to accomplish the job, i.e., minimizing the time to handle wastes.

4.2.5.2 Long-Term Management

During the long term, corrective actions will be taken, as necessary, to maintain the containment system and to return any dispersed contaminated materials to the disposal site.

4.3 ECOLOGY

4.3.1 Action Period

4.3.1.1 Terrestrial

Raffinate Pits and Chemical Plant Areas. All action alternatives would result in short-term loss of primarily old-field/pasture-like habitat at the raffinate pits and chemical plant areas. A minor amount of upland forest habitat might also be eliminated. Areas that would be affected include the 11-ha (26-acre) area occupied by the existing raffinate pits and, under Alternatives 2a and 2b, an additional 18-ha (45-acre) and 23-ha (58-acre) area of land, respectively, for the new disposal cell. The total affected area for Alternative 1 would be 20 ha (49 acres), including the additional area required for the cover and buffer zone (see Section 4.5.2). A smaller amount of land would be affected for Alternative 3c. Additionally, the 45-ha (110-acre) spray irrigation area and its associated buffer zone would experience short-term habitat modification and/or destruction during a wastewater disposal period of 10 years. The area tentatively designated for spray irrigation contains mostly old-field/pasture-like habitat. A portion of upland forest adjacent to Pit 4 might also be affected by construction activities, especially under Alternatives 1 and 3c.

Destruction of the habitat at the raffinate pits and chemical plant areas would displace birds into unaffected areas where, depending on existing carrying capacity, they might be subject to greater intra- or interspecific competition for nesting habitat and food resources. The raffinate pits and chemical plant areas are not optimal wildlife habitats because of site maintenance actions, low vegetative diversity, and the existence of buildings. As a result, the number of any species inhabiting the area is not large. Therefore, the old fields, cultivated fields, and pasture habitats in St. Charles County should support most of the displaced individuals. The ponds and lakes in the vicinity, including those in the Busch Wildlife Area and on the U.S. Army Reserve Property, would support the displaced waterfowl. The upland forest in these areas would also support displaced fauna if construction destroyed forest habitat near the pits.

Small mammals and herpetofauna would be destroyed or displaced by construction activities at the raffinate pits and chemical plant areas. Larger mammals would be less affected because of their mobility.

Fugitive dust might disrupt gaseous exchange, impede photosynthesis, and render vegetation less palatable. The greatest particulate levels would occur in the construction area. Therefore, impacts would be minimal because most animals would avoid the construction area. Exhaust emissions -- e.g., CO,

NOx, HC, SOx, and organic acids -- are not likely to be of a magnitude considered toxic to biota (Argonne Natl. Lab. 1982).

Fauna inhabiting areas within auditory or visual range of construction might be disturbed, resulting in their leaving or avoiding such areas and impacting habitats not disturbed by waste-removal activities. Animals remaining near the construction area might have their feeding or reproductive activities affected. Trees and other barriers would dampen noise, restricting annoyance to the construction area. The consequences of noise (or visual) distractions to fauna are unpredictable (Argonne Natl. Lab. 1982) but are expected to be small.

Although spray irrigation will not involve standing water (Bechtel Natl. 1984a), it would create conditions adverse to many species. The spray area would be unsuitable for species that breed in old-field/pasture-like habitat. Small burrowing mammals would avoid the area because burrows would be saturated and/or collapse.

Because of the complexities of the food web, assessing the effects of trace elements on ecosystems is virtually impossible (Dvorak et al. 1978), but the effects are expected to be small because (1) prior to spray irrigation, water from the quarry and raffinate pits will be treated, as necessary, to reduce contaminant concentrations and (2) the size of the spray area will ensure a low trace-element concentration in the soil. Furthermore, the concentrations of trace elements in the water of the raffinate pits (Bechtel Natl. 1984a--Table D-1), even before treatment, are below drinking water limits for livestock (Dvorak et al. 1978--Table 30). They are also lower than concentrations used in studies showing that heavy metals do not accumulate to hazardous levels in herbivorous animals inhabiting irrigated fields (Anthony and Kozlowski 1982; Anderson et al. 1982). Some elements of concern, such as arsenic, accumulate in roots; however, only low concentrations are translocated to edible plant parts (Liebig 1966; Berry and Wallace 1974). Therefore, trace elements from spray irrigation are expected to have little impact on biota.

Increased terrestrial habitat would become available to fauna after completion of remedial actions and restoration to grassland of the areas now occupied by the raffinate pits and chemical plant. The potential presence of woodchuck and other burrowing species might be of concern relative to cover maintenance (see Appendix K, Section K.1).

Quarry. All action alternatives would result in the long-term loss of a small area of bottomland forest, interspersed with old-field habitat, at the quarry area. A minor amount of slope forest might also be eliminated. This habitat loss would be associated with actions in the yellow, red, and high

bench areas of the quarry (see Appendix H, Figure H.4). This action would result in the loss of about 1 ha (2.5 acres) of forested habitat within the 3.6-ha (9-acre) quarry area.

Clearing of forest would reduce wildlife habitat and cause adjustments in population distributions similar to that described for the raffinate pits and chemical plant areas. Due to the small area affected, the forested land surrounding and within the quarry area could support displaced individuals. Once the construction area was covered and revegetated with grass, wildlife species would make use of the quarry area. The creation of forest edge should enhance habitat for cardinal, field sparrow, American goldfinch, indigo bunting, common crow, white-footed mouse, least shrew, eastern cottontail rabbit, woodchuck, black rat snake, corn snake, and American toad, although it would be detrimental to species that are more restricted to forest habitat. These would include species such as short-tailed shrew and gray squirrel that would probably be displaced from the disturbed areas until forest habitat was reestablished. White-tailed deer, opossum, skunk, and raccoon would initially leave the construction vicinity, but would probably acclimatize to the construction activities. Although a localized displacement of wildlife is expected, the overall effect on wildlife populations within the Missouri River floodplain area surrounding the quarry is anticipated to be minimal. Several of the small mammal species -- including the white-footed mouse, least shrew, eastern cottontail rabbit, and woodchuck -- might inhabit the construction area after it was covered and revegetated.

Reptiles and amphibians within the construction area would be displaced or destroyed. Most species that occur in the bottomland forest also occur in other habitats, e.g., old fields and upland forests. Therefore, the habitat to be disturbed by waste-removal operations represents only a very small proportion of habitat available to herpetofauna. Some reptiles -- such as black rat snake, black racer, and corn snake -- might increase in the area after the site was covered and revegetated. This would be particularly likely to occur if the white-footed mouse and other small mammals increased in the area. The American toad and Fowler's toad might also increase following revegetation.

Fugitive dust and noise impacts would be similar to those discussed for the raffinate pits area but of a lower magnitude based on the smaller size of the quarry area.

Vicinity Properties. All action alternatives would result in short-term loss of habitat at the contaminated vicinity properties because of waste-removal activities and the construction of access roads. Biotic impacts associated with waste removal from the vicinity properties would be similar to those previously discussed for the raffinate pits and quarry. The amount of

habitat to be disrupted is not anticipated to be extensive nor unique to the area. Therefore, impacts would be localized and are expected to be reversible. However, some of the cleanup of the vicinity properties would involve construction of access roads and movement of heavy equipment into the August A. Busch and Weldon Spring wildlife areas. Prior to removing wastes from these areas, DOE will develop mitigation plans to protect and restore the areas in order to minimize the extent and duration of impacts. Special attention will be given to sensitive areas such as wetlands, nature trails, and steep slopes.

Other Sites. Under Alternatives 3a (Hanford) and 3c (uranium processing site), a small amount of additional habitat -- old-field, upland forest, and bottomland forest -- would be destroyed by construction to upgrade the 6.4 km (4 mi) rail spur at the Weldon Spring site. Impacts associated with the elimination of these habitats would result in only minimal additional effects to the biota of the area.

Disposal of the Weldon Spring wastes at the Hanford site would result in destruction of up to 120 ha (300 acres) of the sagebrush/cheatgrass or Sandberg's bluegrass habitat already designated for use as a waste management area. A 0.8-km (0.5-mi) rail spur might be needed to transport the wastes into the disposal area at Hanford, and its construction would destroy a small amount of habitat. Large tracts of sagebrush/cheatgrass plant communities occur in the vicinity of the Hanford site, so loss associated with construction is considered insignificant. Impacts to wildlife would be similar to those already discussed, e.g., wildlife destruction or displacement. As with the Weldon Spring raffinate pits area, the Hanford disposal area would be reoccupied after cell closure by various biota, including burrowing animals that could be of concern relative to cover integrity (see Appendix K, Section K.1).

Under Alternative 3b ("Nearby Site"), the amount of habitat to be disturbed would be about 15 ha (37 acres). If the site was mostly old-field/pasture-like habitat, the impacts associated with construction would be similar to those previously discussed for the raffinate pits and chemical plant areas. If the site was primarily upland forest habitat, impacts would be superficially similar to those discussed for the quarry area but would involve a larger land area. Also, the species composition for the upland forest is somewhat different than that for the bottomland forest associated with the quarry area. The overall impact at the "Nearby Site" would be a local reduction in the population of species indigenous to the habitats destroyed.

Under Alternative 3c, ecological impacts associated with reprocessing the Weldon Spring wastes at the uranium processing site would be minimal because

habitats would already have been destroyed or extensively disturbed from other on-site activities. Impacts at the Weldon Spring site for both construction and management of the remaining wastes would be similar to those previously discussed.

No-Action Alternative. Under Alternative 4 (no action), ecological conditions at the various sites would continue in their current state because maintenance practices currently in effect would be continued. The potential for radioactive and chemical contamination in the area of the Weldon Spring site would also continue. This could occur through mechanisms such as biotic transport, seepage, and erosion. At waste-disposal sites, plant and animal intrusions into waste layers have been shown to mobilize toxic chemicals and radionuclides (Arthur and Markham 1983; Caldwell et al. 1983; Hakonson et al. 1983). Such mobilizations can contribute to doses received by man (Hakonson et al. 1983). If this can occur at sites where wastes have been contained to some extent, then a greater potential for waste mobilization would continue to occur at the Weldon Spring site under the no-action Alternative 4.

The potential for pit washout from dike failure would also exist, especially for Pit 4. Immediate (short-term) effects of dike failure would include destruction of herbaceous vegetation and possibly small animals. Continuing (long-term) effects would include additions of potentially toxic levels of trace elements and radionuclides to the soil in the pathway of the washout, with eventual adverse effects to vegetation and herbivores (Dvorak et al. 1978). Thus, pit washout could add to the amount of contamination in the vicinity properties. Such contamination would be expected to be of short duration until the dike could be repaired and the washout cleaned up.

4.3.1.2 Aquatic

The aquatic habitats existing within the quarry pond and raffinate pits would be eliminated under any action alternative. Elimination of these habitats would destroy most aquatic biota that inhabit them. Some semiaquatic species, mainly adult herpetofauna, would probably be able to relocate to other aquatic habitats in the area. This similarly applies to waterfowl and mammals (e.g., muskrats) that might make use of the ponds. Elimination of these water bodies would eliminate a source of contamination to the vicinity biota from activities such as waterfowl and wildlife feeding and drinking at the ponds and waste transport from the ponds by emerging aquatic insects.

An early successional pond-like habitat would develop in the water-holding ponds that comprise a portion of the spray irrigation system. Species with rapid colonization potential and/or life cycles would inhabit the ponds -- primarily plankton, invertebrates, and herpetofauna. The continual cycling

of water through the ponds and the closeness to humans usually keep such ponds from benefiting large numbers of waterfowl or wildlife (Dvorak et al. 1978).

The potential impacts to vicinity streams would primarily include increased erosion and subsequent siltation, disturbance or destruction of small tributary streams, and water chemistry changes and/or altered flows in streams receiving site dewatering and rainfall runoff. Impacts to aquatic biota associated with suspended solids and siltation include loss of food resources, habitat destruction, scouring of algae and benthic invertebrates, reduced primary production, interference with respiratory organs, altered behavior, and changes in species composition (Aitken 1936; Ellis 1936; Eur. Inland Fish. Advis. Comm. 1965). However, adverse impacts associated with suspended solids and siltation, even under conditions of direct instream disturbance, are temporary; and biota usually recolonize affected areas within a year after disturbance has ceased (Barton 1977; Reed 1977; Tsui and McCart 1981). Therefore, impacts to affected streams are expected to be minor, temporary, and reversible.

Chemical contamination of receiving streams related to construction should be negligible. Concentrations in runoff from the spray-irrigated field into receiving streams would be within state limits required for spray irrigation. Receiving streams would be further protected through retardation of trace elements by soils. Additional absorption/desorption of trace elements would occur in the irrigation field.

Few aquatic systems occur at the Hanford site and, therefore, waste disposal under Alternative 3a would not be expected to affect aquatic biota at Hanford. Under Alternative 3b, the "Nearby Site" would probably be selected in an area that is removed from standing or flowing water bodies; thus, no anticipated aquatic impacts would be associated with this alternative. Impacts to aquatic habitats at the uranium processing site (Alternative 3c) would be site-specific. Under the no-action Alternative 4, the chances for biological uptake and transport of contaminants would continue to exist. Additionally, the quarry and raffinate pits areas are undoubtedly breeding grounds for mosquitoes, which are vectors of encephalitis in the St. Louis area.

4.3.1.3 Rare and Endangered Species

Several of the federally and/or state listed species occurring in St. Charles County (Appendix G, Table G.1) could frequent the Weldon Spring site. Under most situations, such species usually avoid areas of disturbed habitat in favor of habitat that is not being affected by man's activities. DOE will consult with the U.S. Fish and Wildlife Service prior to taking any actions at the Weldon Spring site to determine appropriate mitigative measures

to minimize impacts to threatened or endangered species. There are no endangered or threatened plant species in the 200-West Area of the Hanford site. Federally listed bird species, and bird species that are rare but lack a designated status, are only migrants or nest in low numbers at the Hanford site.

A number of federally or state listed species occur in the area considered for the "Nearby Site" (Appendix G, Table G.2). Before implementing Alternative 3b, DOE would consult with the U.S. Fish and Wildlife Service to determine if listed species are present on the site and, if so, what measures should be taken to avoid impacting those species. If, in cooperation with the U.S. Fish and Wildlife Service, it was determined that impacts to such species could not be mitigated or avoided, DOE would probably not be able to implement Alternative 3b at that site. For the uranium processing site (Alternative 3c), the presence of endangered or threatened species and any mitigative measures would have to be determined on a site-specific basis.

4.3.1.4 Potential Wetland and Floodplain Impacts

Remedial actions on vicinity properties could impact two areas that are located within the Missouri River floodplain, i.e., the lower portion of the drainage ditch that flows from the raffinate pits area to the river and a small area of land located between the quarry and Femme Osage Slough (Figure 1.4). Wetlands may also exist on various vicinity properties. A floodplain/wetlands assessment will be prepared and included in the Final EIS. This assessment will describe the project, discuss the effects of the project on the floodplain and wetlands, and discuss alternatives including mitigative measures. The assessment would be in accordance with the requirements given in Executive Order 11988 (Floodplain Management) and Executive Order 11990 (Protection of Wetlands). It is DOE's policy to avoid adverse impacts on wetlands and floodplains to the extent possible and to minimize any unavoidable adverse impacts (10 CFR Part 1022).

Remedial actions could require clearing of vegetation, construction of access roads, and use of heavy machinery. The potential effects on wetlands resulting from these activities include disruption of drainage patterns, increased erosion and sedimentation, habitat destruction, release of contaminants to the environment, and displacement or destruction of biota. The relative effects would depend upon the type of wetland, the time of year (season) the remedial actions were carried out, and the particular excavation methods and mitigative measures employed. Effects could be minimized by conducting remedial actions during dry periods. The amount of wetlands that would be potentially impacted is small relative to the total amount of wetlands occurring in the Weldon Spring area. Thus, overall impacts to

wetland habitat would not be of sufficient magnitude to cause localized extinctions of any wetland species. The wetland areas are a source of contamination that could adversely impact man and wildlife, so decontamination of these areas is appropriate. Stipulations that would be included in required permits, as well as applicable regulations relating to work in wetlands, would require that the affected areas be restored to preconstruction conditions or that compensatory wetland habitat be created in another designated area.

Concerns about floodplain effects primarily relate to displacement of floodplain storage volume. Remedial actions will be scheduled during dry periods when the potential for flooding is low. In addition, appropriate mitigative measures will be taken to minimize any effects that might occur from localized flooding conditions. No significant modification of flood storage volumes would be associated with the remedial actions because the affected vicinity properties would be restored to preconstruction conditions. Thus, no significant impoundment, destruction, or other modification of flood waters would result. Other impacts to floodplain habitat and biota would be similar to those discussed for the raffinate pits and quarry areas in Section 4.3.1.1.

4.3.2 Long-Term Management

4.3.2.1 Terrestrial

Institutional care in the long term would include maintenance of the disposal area in an early successional stage (e.g., old field) or in a cultivated lawn-like condition at the eastern sites (Weldon Spring site or "Nearby Site"). Early successional plant species could include grasses and herbs (broomsedge, milkweed, aster, goldenrod, panic-grass, Kentucky bluegrass, and fescue) and low-growing trees and shrubs such as sumac, persimmon, wild cherry, hawthorn, boxelder, and sassafras (Mo. Bot. Garden 1975; Galvin 1979). Site maintenance would limit wildlife diversity and use of the area. Fauna that could inhabit the site include deer mouse, white-footed mouse, eastern cottontail rabbit, woodchuck, least shrew, field sparrow, American goldfinch, common grackle, starling, robin, eastern meadowlark, American toad, eastern garter snake, and black rat snake (Burt and Grossenheider 1964; Conant 1975; Robbins et al. 1983).

At the Hanford site, a shrub-steppe habitat -- consisting of grasses, forbs, and shrubs adapted to arid conditions -- would develop on the covers over the waste trenches. Fauna that could inhabit the site include the Great Basin pocket mouse, northern pocket gopher, mountain cottontail rabbit, western meadowlark, and sage sparrow. At the uranium processing site, local

native plants and animals would colonize the cover on the tailings pile (see Section 3.4.4).

4.3.2.2 Aquatic

No significant impacts to aquatic habitats related to containment of the Weldon Spring wastes during the long term would be expected. Maintenance activities would be confined to the immediate disposal areas, away from water bodies. Maintenance of the cell covers would also control erosion, excessive runoff, and other sources that could impact aquatic systems.

4.3.3 Biotic Effects on Disposal Cells

The integrity of disposal cells can be affected by such biotic actions as burrowing and grazing by animals (mostly mammals and insects), penetration by roots, and soil aggregation by microorganisms. These actions, their effects, and possible mitigative measures are discussed in detail in Appendix K, Section K.1.

4.4 AIR QUALITY

The most significant releases affecting air quality will occur during the 10-year action period and will be associated with transport of the wastes and placement of the earthen cover over the wastes. All operations will be carried out and controlled, as necessary, so that National Ambient Air Quality Standards (NAAQS) for total suspended particulates (TSP) are maintained at the site boundary. The maximum 24-hour standard for TSP (including ambient contributions) is 150 $\mu g/m^3$ for environmental preservation and 260 $\mu g/m^3$ for human health protection. The annual TSP standard is 75 $\mu g/m^3$. It is expected that the maximum 24-hour standard of 150 $\mu g/m^3$ at the site boundary will be the limiting standard.

The U.S. Environmental Protection Agency is considering changing to a "PM10 standard" in which only particulates smaller than 10 μm are addressed. Under this new standard, particulates larger than 10 μm would not be considered to be a hazard because they are assumed not to enter the respiratory tract. Because the PM10 standard has not yet become law, this analysis is based on current TSP standards.

Exact details of materials handling and time scheduling for the various alternatives are not available; therefore, a "worst-case day" analysis was made for each alternative. The use of a bounding analysis to show that NAAQS are satisfied is permitted by the U.S. Environmental Protection Agency as an alternative to use of the EPA-approved Industrial Source Complex model (Wilson 1986). This is a more conservative approach than was used to assess

radiological impacts of particulate emissions and is used to demonstrate compliance with existing requirements. A worst-case day is defined as a day in which the greatest fugitive dust is released, coupled with worst-case meteorology. The worst-case day is assumed to have a period of F stability (minimum atmospheric mixing), 2.5 m/s (8.2 ft/s) wind speed, and 6 hours persistency of the wind in the direction of interest. Under these conditions, very little mixing will occur, and the highest concentrations will be predicted. Considering that construction activities will take place during daytime hours, these F stability conditions are unlikely to occur.

4.4.1 Alternative 1: Improved Containment in the Existing Raffinate Pits

For Alternative 1, the worst-case day is expected to occur during the period in which the clay or topsoil is being placed on the disposal cell cover. Activities inside the raffinate pits and quarry are expected to release very little fugitive dust because the excavation activities will be below the surrounding terrain. Excavation of rubble is unlikely to release any significant fugitive dust. Although the vicinity properties are closer to points of public access, the quantity of materials to be removed from the vicinity properties is much less.

The types of equipment likely to be used for placing clay and topsoil on the cover are haul trucks, backhoes, and bulldozers. Of these, the bulldozer causes the greatest fugitive dust. The emission factors for haul trucks, backhoes, and bulldozers were obtained from the U.S. Environmental Protection Agency (1983a). For clay, the silt content was assumed to be 40% and the moisture content 13%. For topsoil, the silt and moisture contents were assumed to be 65% and 25%, respectively (Kalkwarf et al. 1984).

Under Alternative 1, the amount of material to be moved for construction of the disposal cell cover is 240,000 m^3 (310,000 yd^3) of clay and 69,000 m^3 (90,000 yd^3) of topsoil (Appendix E, Table E.1). The capacities of the backhoe, bulldozer, and haul truck were assumed to be 2.7 m^3 (3.5 yd^3), 8.4 m^3 (11 yd^3), and 12 m^3 (16 yd^3), respectively. The drop height for the backhoe was assumed to be 4 m. Based on an 8-hour day and six months (120 days) per year of activities for 5-3/4 years, a daily material moving rate was determined from which fugitive dust emissions were calculated. Five backhoes, two bulldozers, and 57 one-way trips by the haul trucks would be required on a daily basis. It was assumed that for each backhoe operation, 10 minutes of dozer time would be required to smooth out the load dumped. It was also assumed that water spraying would reduce dozer fugitive dust by a factor of two. The calculated emission rates are backhoe, 3.8 kg/d (8.4 lb/d); bulldozer, 16 kg/d (35 lb/d); and haul trucks, 7.6 kg/d (17 lb/d) (Table 4.13).

Table 4.13. Summary of Fugitive Dust Emissions and Meteorological Conditions for Worst-Case Day for the Action Alternatives[a,b,c]

Alternative	Activity	Fugitive Dust Emissions (kg/d)		
		Backhoe	Bulldozer	Haul Trucks
1	Placement of clay or soil on cover	3.8	16	7.6
2a	Placement of clay or soil on cover	—— Similar to Alternative 1 ——		
2b	Placement of clay or soil on cover	4.4	18	8.6
3a	Loading of wastes onto trains at Weldon Spring	2.7	-	-
	Placement of cover over trenches at Hanford	—— Less than Alternative 1 ——		
3b	Loading wastes onto trucks at Weldon Spring	—— Similar to Alternative 3a ——		
	Placement of clay or soil on cover at "Nearby Site"	—— Similar to Alternative 1 ——		
3c	Placement of clay or soil on cover at Weldon Spring	—— Similar to Alternative 1 ——		

[a] Meteorological conditions assumed for worst-case day for all alternatives are: Class F stability, 2.5 m/s wind speed, 6 hours wind persistency.

[b] The point of nearest public access at the Weldon Spring site is 800 m southeast of Pit 4. All calculations for the Weldon Spring site are made for that nearest point.

[c] All values are rounded to two significant figures. Daily fugitive emissions are larger for some alternatives due to the larger volume of material to be moved.

Conversion Factor: To convert kilograms (kg) to pounds (lb), multiply by 2.205.

The emissions from the backhoe, bulldozer, and haul trucks were assumed to occur from an area 50 m × 50 m (55 yd × 55 yd). The area would be located at a different portion of the disposal cell cover each day. This assumption is conservative because the haul truck emissions would be from a line source ending at the disposal cell. The plume centerline was assumed to be fixed for the 6 hours of persistent winds assumed for this analysis. In this way, maximum concentrations would be predicted at the site boundary 800 m (875 yd) southeast of the disposal area (closest point of public access). This is a conservative assumption because the wind direction is never fixed in a particular direction for any period of time but shifts direction constantly. Sector-averaging was not used although, if applied, would reduce predicted concentrations further. Also, no deposition of particulates was assumed during transport. This adds conservatism to the predictions because the larger-sized particulates within the TSP size range of 0-30 μm are likely to fall out during transport from a ground-level source.

The airborne TSP concentration at the Weldon Spring site boundary were predicted on the basis of Turner (1969). The maximum 24-hour average airborne TSP concentration is estimated to be 55 $\mu g/m^3$. Using an ambient concentration of 40 $\mu g/m^3$ (see Section 3.1.3), the total estimated concentration is 95 $\mu g/m^3$ (Table 4.14). The predicted concentration at the site boundary is below the

Table 4.14. Comparison of Estimated Off-site Total Suspended Particulate Concentrations

Alternative	Site	Maximum Estimated Off-site Concentration ($\mu g/m^3$, 24-hour avg. TSP)
1	Weldon Spring site	95
2a	Weldon Spring site	95
2b	Weldon Spring site	100
3a	Weldon Spring site Hanford site	47 Background[a]
3b	Weldon Spring site "Nearby Site"	47 95
3c	Weldon Spring site Uranium processing site	95 Background
4	Weldon Spring site	Background

[a] The TSP concentration contributions from activities associated with the Weldon Spring wastes will be much less than existing concentrations in these areas.

limit of 150 $\mu g/m^3$ for this conservative evaluation. Concentrations should be significantly lower than 95 $\mu g/m^3$ on most days.

4.4.2 Alternative 2a: New Cell, Partially Above Grade

Under Alternative 2a, 230,000 m^3 (300,000 yd^3) of clay and 64,000 m^3 (83,000 yd^3) of topsoil would be moved for construction of the disposal cell cover over an assumed 5-1/2 year period (Appendix E, Table E.2). Because the volumes of clay and soil to be moved would be somewhat less and the time period would also be a little bit less, the amount of material moved on a daily basis would be similar to that for Alternative 1. The daily emission rate would therefore be similar, and TSP concentrations at the site boundary would be expected to be similar to those for Alternative 1.

4.4.3 Alternative 2b: New Cell, Completely Above-Grade

Under Alternative 2b, 270,000 m^3 (350,000 yd^3) of clay and 85,000 m^3 (110,000 yd^3) of topsoil would be moved for construction of the disposal cell cover (Appendix E, Table E.3). The estimated emissions are backhoe, 4.4 kg/d (9.7 lb/d); bulldozer, 18 kg/d (40 lb/d) (which includes 50% reduction due to water spraying); and haul trucks, 8.6 kg/d (19 lb/d). The maximum 24-hour average airborne TSP concentration is estimated to be 63 $\mu g/m^3$. Adding an ambient level of 40 $\mu g/m^3$, the total is about 100 $\mu g/m^3$ (to two significant figures) -- which is also less than the 150 $\mu g/m^3$ limit

4.4.4 Alternative 3a: Hanford Site

Alternative 3a would involve fugitive dust emissions at both the Weldon Spring site and the Hanford site. At the Weldon Spring site, in addition to the emission sources associated with cleanup and backfill, construction of a 6.4-km (4-mi) rail spur and loading of the bulk (unpackaged) wastes onto trains are emission sources. Estimates were made for loading the bulk wastes because of the large volume, 430,000 m^3 (560,000 yd^3). Loading onto the railcars was assumed to be accomplished by a backhoe of about 2.7 m^3 (3.5 yd^3) capacity. Loading and transport were assumed to take 3 years (the actual time would be longer--see Appendix F). The fugitive release rate to the air is estimated to be 2.7 kg/d (6.0 lb/d). This leads to a predicted increase in airborne TSP levels of 6.6 $\mu g/m^3$. Adding in the ambient value of 40 $\mu g/m^3$, the total is 47 $\mu g/m^3$, which is much less than the 150 $\mu g/m^3$ limit.

At the Hanford site, it is likely that emissions would occur simultaneously from excavation, waste emplacement, and backfill activities at adjacent trenches. Much lower off-site impacts are expected primarily because the distance from the disposal area for the Weldon Spring wastes to the site boundary is much greater (several kilometers) than the distance from the

disposal cell to boundary at the Weldon Spring site (800 m). No discernible increment above background is expected at the site boundary.

4.4.5 Alternative 3b: "Nearby Site"

For Alternative 3b, the air quality impacts at Weldon Spring would be about the same as those for Alternative 3a, except that no construction of rail access would be required because the wastes are assumed to be transported by truck. The impacts for cover placement at the "Nearby Site" would be similar to those for Alternative 2a if the distance to the property line at the "Nearby Site" was similar to that at the Weldon Spring site (800 m [2,600 ft]).

4.4.6 Alternative 3c: Uranium Processing Site

For Alternative 3c, impacts from placing the cover over the portion of wastes remaining at the Weldon Spring site should be similar to the impacts for Alternative 1. At the uranium processing site, any particulates released during reprocessing of the Weldon Spring sludge would be insignificant compared to the particulates generated during other activities at the site.

4.4.7 Alternative 4: No Action

The no-action Alternative 4 is not expected to result in any significant fugitive dust emissions. No haul trucks, bulldozers, or backhoes will be used. The quarry wastes are below the surrounding terrain and therefore less subject to wind erosion. The vicinity properties are vegetated. The raffinate pits are wet most of the time, and when the surface dries out for a few weeks in the summer, a crust forms that is resistant to wind erosion.

4.4.8 Summary and Comparison

With standard dust-control measures such as water spraying, the National Ambient Air Quality Standards for total suspended particulates should be easily met. This would be true even under worst-case meteorological conditions, at all sites for all alternatives.

4.5 SOCIOECONOMICS

4.5.1 Land Use

Land-use impacts are assessed in terms of impacts on and near the disposal site as well as impacts at the properties to be decontaminated and restored. For all action alternatives, the disposal site would be permanently committed to waste disposal. Maintaining any of the waste-disposal sites

requires that surface and subsurface activities be restricted if they could affect the integrity of the containment. In addition, some restrictions may be needed on adjacent properties to limit activities that could affect the integrity of the disposal site, such as blasting.

The area committed to long-term waste disposal was assumed to be the area occupied by the cover of the disposal cell (or trenches and inter-trench area) plus 30% of that for a buffer zone. The areas committed would be 20 ha (49 acres) for Alternative 1, 18 ha (45 acres) for Alternative 2a, and 23 ha (58 acres) for Alternative 2b. Alternative 3a (Hanford) would have the largest land commitment, 120 ha (300 acres), because of the use of several shallow trenches rather than one large disposal cell (Section 2.1.4). The estimated land area committed to waste disposal is 15 ha (37 acres) for Alternative 3b at the "Nearby Site" and 11 ha (28 acres) for Alternative 3c at the Weldon Spring site. Commitment of land at the uranium processing site from disposal of wastes resulting from reprocessing the Weldon Spring sludge would be a small fraction of the land already committed to disposal of uranium mill tailings.

For all action alternatives, a 45-ha (110-acre) area would be used for disposal of treated wastewater by spray irrigation (see Section 4.1.3.1). After the action period, this area would be monitored, cleaned up as necessary, and restored for appropriate use.

Under the no-action Alternative 4, the land area remaining in restricted status -- including the quarry area, raffinate pits area, and chemical plant area -- would continue to be 93 ha (230 acres). Certain restrictions on use of the vicinity properties would also continue. For all action alternatives, the vicinity properties, quarry, and portions of the chemical plant and raffinate pits areas would be decontaminated and released for appropriate use.

Remedial actions would result in limited off-site land-use impacts. These impacts include the purchase, extraction, and transportation of fill materials. The purchase of fill would be perceived as a benefit to local business whereas the development of borrow areas might be perceived by some environmental groups as a cost. Other temporary impacts associated with these actions -- such as noise, dust, and increased traffic -- are not likely to cause significant land-use impacts. An exception could be a temporary decline in use of the wildlife areas in close proximity to the site during the action period.

Impacts to land uses surrounding the disposal area at the Hanford site are not anticipated because of continued federal control and remoteness of the disposal area. Land-use changes in the vicinity of the "Nearby Site" -- such as increased development, upgrading of local roads and services, new county

land-use goals, and reduced value of adjacent properties -- would be likely to occur. Depending on future land-use demands in the region, competition could occur over the use and development of properties in the nearby area.

4.5.2 Transportation

All of the action alternatives would cause some disruption of local traffic. At the Weldon Spring site, traffic congestion could occur for short periods of time as a result of construction workers traveling to and from the site, movement of construction vehicles and waste transport vehicles, delays on State Route 94 at railroad crossings, cleanup along roadways, and upgrading of the rail spur. Minor disruptions could also occur on transportation routes between the Weldon Spring site and the alternative disposal sites. Transportation requirements for each alternative are given in Appendix F.

Impacts would be greatest on State Route 94 where increased traffic would result in increased travel times. Increased traffic could interfere with school buses in the morning and afternoon and could result in more road noise at Francis Howell High School. To mitigate this impact, access to the Weldon Spring site could be routed via County Route "D" and a gravel road so that truck traffic would pass behind the school instead of in front. The additional traffic and loads would also result in faster road deterioration and higher maintenance costs. DOE will consult with local authorities and take any necessary actions to alleviate traffic congestion and road deterioration.

Alternative 1 would involve about 15 round-trip truck trips per day to haul quarry wastes to the raffinate pits area plus 68 round-trip truck trips per day to bring in fill materials and materials for the disposal cell. These two activities would sometimes overlap, producing a total of 83 round-trips per day. Alternative 2a would involve a total of 82 round-trips per day. Alternative 2b would involve 145 round trips per day because of the greater requirements for materials for the disposal cell.

Under Alternative 3a, 70 round-trips per day by truck would be required to haul fill materials to the Weldon Spring site and decontaminated areas for restoration. Rail transportation of the wastes to Hanford could cause traffic disruption on State Route 94 at railroad crossings. Train traffic would occur infrequently, averaging once every 2.5 days (one round-trip every 20 days for each of the four 35-car trains, for 6 months per year and a shipping schedule of 10 years--see Appendix F, Table F.2).

Under Alternative 3b ("Nearby Site"), an estimated 35 truck shipments per day would be required to the "Nearby Site" during the anticipated 6-month construction period each year. Fill materials hauled to the decontaminated areas would require 70 round-trip truck shipments per day (Appendix F,

Table F.6). During some years, these two shipping activities could coincide.

Under Alternative 3c, shipment of the sludge by rail to a uranium processing site in the southwestern United States would result in less disruption at State Route 94 rail crossings than for Alternative 3a because of the fewer number of shipments and the shorter period of time, once every 7.5 days (using 35 car trains, 6 months per year, and a 4-year shipping schedule--see Appendix F, Table F.2). In addition to the train shipments, 58 round-trips per day by truck are projected for hauling in materials for backfill and construction of the disposal cell at the Weldon Spring site (Appendix F, Table F.6).

Transportation of wastes and construction materials (backfill, clay, etc.) would increase the risk of human death and injury because of the potential for transportation accidents. The estimated transportation-related deaths and injuries associated with implementation of each alternative are given in Appendix F, Table F.13. Implementation of Alternative 3a would result in the highest number of transportation-related deaths (2.5) because all of the wastes would be moved over a long distance to the Hanford site. These risk estimates are based on average U.S. rail- and truck-accident fatality rates. Extra precautions taken because of the radioactive materials, however, could lead to accident rates that are lower than the average rates.

4.5.3 Project Work Force

The labor requirements for the various alternatives for the 10-year action period range from a low of 50 person-years for the no-action Alternative 4 to a high of approximately 1,000 person-years for the most labor-intensive Alternative 3c (uranium processing site). These requirements are expected to cause no significant problems at the Weldon Spring site, which is located within the St. Louis metropolitan area. There would be less local impact for Alternatives 3a, 3b, or 3c because the requirements would be spread out over a larger geographic area.

4.5.4 Population and Economy

The population in the Weldon Spring area might increase slightly during the action period as a result of construction activities; concomitant increases would be expected in economic activities. No significant adverse socioeconomic impacts are anticipated. Some temporary disruption to users of the wildlife areas could occur during cleanup of vicinity properties under all action alternatives. For all of the alternatives, positive effects on the economy would be realized from the expenditure of funds for labor and materials at the site. Benefits would also be gained if compensation or

payments in-lieu-of taxes were made by the federal government to the local political entities hosting the waste disposal site. There are currently no plans for such payment. Local and state governments cannot levy property taxes on federally owned installations.

The impacts for Alternatives 1, 2a, and 2b would be similar. However, Alternative 2b would have greater work force requirements and potentially greater positive economic benefits resulting from increased material purchases and increased income.

Alternative 3a (Hanford) will require less labor at Weldon Spring than Alternative 1, 2a, or 2b. Socioeconomic impacts are expected to be insignificant. At the Hanford site, the incremental activities associated with disposal of the Weldon Spring wastes are expected to be insignificant considering all the other nuclear-related activities at the Hanford site.

Impacts at Weldon Spring for Alternative 3b ("Nearby Site") are not expected to differ significantly from Alternative 3a. Socioeconomic impacts at the "Nearby Site" will depend upon the specific location of the site. Sites nearer population centers would rely on these centers for labor, thus minimizing impacts to population and public services in smaller communities near the site. Location of the site away from population centers would likely mean in-migration of labor and some related increases in demand for public services. The economic benefits of increased employment would be positive in both situations.

For Alternative 3c (uranium processing site), socioeconomic impacts at the Weldon Spring site would be similar to Alternative 1. At the uranium processing site, the reprocessing of the Weldon Spring sludge would be a small increment to the ongoing processing activities at the uranium mill and, as such, would be a small economic benefit to the community and would not have significant impacts on population or public services.

4.5.5 Visual and Cultural Resources

Under the no-action Alternative 4, the visual resources would remain unchanged. However, changes that affect the visual quality of the environment would be associated with all of the action alternatives. Impacts would be more severe during the action period than during long-term management. The magnitude of the impacts would depend on the cultural-historical importance of the land involved, the magnitude of change observed, the permanency of the change, and the number of people expected to observe the change.

Many of the vicinity properties are located in or near the wildlife/ recreational areas where the visual context is important to the public.

Earth-moving activities, traffic, noise, and dust would detract from the recreational experience of visitors. These impacts, associated with all action alternatives, would be temporary. Similar impacts would be experienced by visitors to the properties that surround the restricted disposal areas at the Weldon Spring site or Hanford site. DOE will develop plans for restoration of disturbed areas in consultation with the property owners.

Visual impacts would also occur under all action alternatives at borrow locations where materials would be extracted for use in construction of the disposal cell or for backfilling decontaminated areas. The severity of the visual impact would depend on current setting (old versus new borrow area), amount of public access, use of surrounding areas, and perceived scenic value prior to the extraction activities.

During the long term, visual impacts are expected to be low to moderate for the action alternatives assuming that access to the disposal site will be restricted and that maintenance activities will be small-scale. Landscaping will be used, as appropriate, to reduce the visual contrast between the natural background setting and the waste-containment areas.

No specific analysis of impacts on visual resources can be made for the "Nearby Site" because its exact location is unknown. For the uranium processing site, the addition of the Weldon Spring sludge to the ongoing processing activities at the uranium mill would not affect the visual resources.

Cultural resource sites, items, and structures have been reported for the areas surrounding all the alternative waste-disposal sites. Some cultural resource sites may have importance for American Indian peoples. However, most of the areas that could be adversely affected have not been systematically surveyed for surface and subsurface cultural resources. Consequently, unknown historic and prehistoric sites and structures could exist. These sites could be adversely affected by cleanup actions, including some sites potentially eligible for the National Register. The magnitude of the impacts would be related to the number, size, distribution, and importance of the cultural resources affected by a particular alternative. Adverse impacts would occur during the action period as a result of disturbance of ground surfaces and architectural structures. Other indirect impacts could occur such as off-site erosion, construction and land-use changes leading to more off-site disturbances, and aesthetic effects on sensitive areas eligible for the National Register.

The state of Missouri has determined that there is no need to conduct archeological investigations on the properties directly involved with implementation of this project in the Weldon Spring area because these areas have either been previously disturbed by construction-related activities or

exhibit a low potential for archeological resources. However, archeological investigations may be required for areas that are currently uncontaminated or have not been subject to substantial previous disturbance (Weichman 1986); DOE will consult with the state of Missouri prior to conducting ground-disturbing activities in such areas. The need for archeological investigations at the "Nearby Site" or the uranium processing site would have to be determined on a site-specific basis. DOE is currently reviewing the need for a cultural resource investigation at the Hanford site.

4.5.6 Mitigative Measures for Socioeconomic Impacts

Under all action alternatives, truck and train traffic will be coordinated to avoid conflict with traffic to or from Francis Howell High School. It is anticipated that local workers will be used as much as possible, and project materials will likely be purchased locally if available. Landscaping will be used, as appropriate, to reduce the visual contrast between the natural background setting and the waste-containment areas.

Under all alternatives, DOE will continue to work with the Missouri Department of Natural Resources, the Missouri Conservation Commission, St. Charles County, area municipalities, and local interest groups to ensure that public health and safety will be protected. A multi-agency monitoring group could be formed to accomplish this purpose.

4.6 INSTITUTIONAL ISSUES

Some institutional issues that relate to radioactive waste disposal were brought out in a Senate hearing entitled "The Socioeconomic Effects of Nuclear Waste Storage Sites on Rural Areas and Small Communities" (Senate Subcomm. Rural Dev. 1980). Specifically addressed were equitable distribution of effects, time period of institutional control and long-term institutional management, insurance or escrow arrangements to protect local communities and residents, and incentives for communities to host waste-disposal sites.

The equitable distribution of effects refers to the idea that those enjoying the benefits of certain actions should also share the costs generated by those actions. Past economic activity at the Weldon Spring site was an asset to the area. Area residents will now share in the risks (perceived and real) if the wastes remain at the site. On the other hand, area residents might be incurring a disproportionate share of the risks because benefits accrued to a larger societal group (e.g., national defense) were also derived from previous operations at the site. However, the larger societal group would pay the monetary costs for the federal cleanup. It would be difficult, however, to determine a more equitable location for the wastes currently located at the Weldon Spring site. Locating the wastes at a new site (as in

Alternative 3b) would likely be viewed as inequitable by area residents at the new disposal site.

All action alternatives would result in cleanup of the environment at the Weldon Spring site, an incentive for the local area. Alternatives 3a, 3b, and 3c would also impact other communities. Alternative 3b could result in management of wastes near a community without an existing radioactive waste-disposal or waste-processing site. Incentives would provide a means to balance costs and benefits. Incentives could take the form of payments in-lieu-of taxes to area governments, federal funding for local installations and services, or siting of a federal non-waste project in the vicinity. A large project would provide employment, economic development, and tax benefits to at least partially outweigh the negative aspects (both real and perceived) of hosting a radioactive waste-disposal site. If DOE decides to implement Alternative 3b ("Nearby Site"), not preferred at this time, DOE will work closely with the affected local community.

Another institutional issue is the irreversibility of actions and foreclosure of future options. In technology assessment, options that can be reversed and foreclose the fewest future options are more desirable (Freeman 1974). All action alternatives are reversible (although at great expense), whereas Alternative 4 (no action) might lead to further spread of contamination that, in some cases might be irreversible. All of the alternatives foreclose unrestricted use of contaminated areas or disposal areas.

Environmental regulation is also an institutional issue. DOE has primary responsibility for managing health, safety, and environmental protection programs at DOE-owned, contractor-operated installations. DOE has issued operating "Orders", pursuant to the Atomic Energy Act, that contain environmental guidelines for its various operations. The major orders that could potentially apply to the Weldon Spring alternatives are presented in Appendix C, Section C.2.

DOE installations are also subject to several other federal environmental statutes. Executive Order 12088 requires federal agencies to comply with standards established by federal environmental laws including the Toxic Substances Control Act; Clean Water Act; Safe Drinking Water Act; Clean Air Act; Noise Control Act; Solid Waste Disposal Act, as amended by the Resource Conservation and Recovery Act; and Comprehensive Environmental Response, Compensation and Liability Act. Where a state has been delegated administrative authority by the federal government, or where sovereign immunity has been waived, state requirements are also applicable. The regulatory programs and enforcement authorities for environmental statutes are generally shared by the states, the U.S. Environmental Protection Agency, the U.S. Fish and Wildlife Service (under the Department of the Interior), the

U.S. Army Corps of Engineers, and other agencies. Environmental laws that may impact a DOE decision regarding the Weldon Spring wastes are presented in Appendix C, Section C.1.

The role of state governments and other federal agencies with respect to the Weldon Spring project will be determined once an alternative has been selected.

4.7 IMPACTS OF POTENTIAL LOSS OF INSTITUTIONAL CONTROL

After cleanup, DOE intends to continue monitoring the disposal site and to take any further remedial actions that may be necessary (as indicated by the monitoring results). However, if institutional controls should cease, the following impacts could occur.

4.7.1 Radiological Impacts for Nearby Individuals and Resident-Intruders

If controls ceased, the containment system would gradually deteriorate under natural physical and biological forces (Appendix K). As the cover over the wastes deteriorated, radon gas emissions and contamination of surface water and groundwater would increase and radioactive particulates would be released to the air. Exposure of nearby persons to radiation would increase as the releases from the site increased. The rate at which these impacts would occur and the amount of releases to the environment would depend on site-specific meteorological and land-use conditions.

It is also possible that humans could inadvertently intrude into the disposal areas. Such intrusion could result in temporary exposure of the wastes or more extensive exposure associated with activities such as farming, drinking water from contaminated wells, or living in a building constructed on the site. The risk of adverse health effects for the intruder would depend on the extent of exposure from the various environmental pathways.

DOE intends to maintain controls over the disposal site to ensure that these impacts do not occur.

4.7.2 Disposal Cell Integrity

The integrity of the disposal cell could be affected by a number of natural forces acting on the cell. Such forces include erosion, flooding, subsidence, earthquakes and tornadoes. Criteria for the disposal cell will be developed during design engineering activities. Design-basis earthquakes and tornadoes will be determined, and the disposal cell will incorporate features to preserve site integrity.

The impacts of severe natural phenomena would be much greater for the no-action Alternative 4 than for any of the action alternatives because the wastes would be more susceptible to being dispersed into the environment. A tornado impacting the chemical plant in its current status could cause severe damage; such a tornado could also result in off-site transport of contaminants from the raffinate pits if a tornado struck the currently open pits. An earthquake could cause failure of the existing dikes surrounding Pits 3 and 4. These effects would be much lower after implementation of one of the action alternatives because all of the wastes would have been placed into a disposal cell and the chemical plant would have been decontaminated and demolished.

4.7.3 Hydrological

4.7.3.1 Raffinate Pits Area and Hanford Site

Infiltration into the wastes would increase as the cover over the wastes deteriorated. The estimated infiltration rate calculated for the no-action Alternative 4 at Weldon Spring represents the limiting case for both the raffinate pits area and the Hanford site for all applicable alternatives. Under the no-action alternative, maximum concentrations of uranium-238 and radium-226 in a hypothetical on-site well located at the edge of the waste field are predicted to be in the range of background concentrations and considerably below the DOE limits for uncontrolled areas (100 pCi/L for radium-226 and 1,100 pCi/L for natural uranium) (see Appendix I, Section I.1.4).

With respect to chemicals in the hypothetical on-site well, the concentrations will depend on the infiltration rate, the nature of the wastes (i.e., stabilized wastes are expected to have higher concentrations of some chemical species because of the use of a stabilizer), the presence or absence of lead in the cover, and the amount of time that has passed since waste emplacement. For example, if institutional controls ceased at 1,000 years, the more mobile chemical species such as nitrate and sulfate are expected to have been leached out of the wastes and would no longer be contributing to concentrations in the hypothetical on-site well at the Weldon Spring site.

4.7.3.2 Quarry Area

Concentration contributions of uranium at the county well field are expected to be near or below background levels for all alternatives. Peak concentration contributions of chemicals at the county well field at various times in the future are predicted to be below current regulatory limits, even for the no-action Alternative 4. Additional discussion of potential groundwater contamination in the quarry area is presented in Section 4.1.2.

If an intruder drilled a well in either the near alluvium or limestone near the quarry, concentrations of some contaminants could exceed regulatory limits. Because of the much lower potential yields associated with the near alluvium and limestone near the quarry, they are less desirable as groundwater resources than the river alluvium.

4.7.3.3 Surface Waters

For the action alternatives, surface water contamination is expected to be insignificant as long as the wastes remain covered. As the cover erodes, any exposed wastes could be eroded and carried into surface waters by runoff water. The concentrations of radioactive and chemical species in surface waters would vary with the extent of exposed wastes and the nature of precipitation events.

4.7.4 Ecological

Human use of the disposal cell area would determine long-term biotic development. Secondary succession of naturally occurring plants could occur. At the raffinate pits area and probably the "Nearby Site", an upland hardwood forest would likely develop; at the quarry, bottomland forest would develop. At Hanford, natural succession would lead to development of a sagebrush/cheatgrass community. If humans used the disposal sites for ranching, farming, housing, or industrial development, biotic assemblages unique to each type of development would become established. Biotic effects on the long-term integrity of containment systems are discussed in more detail in Appendix K, Section K.1.

4.8 SUMMARY OF MAJOR ADVERSE IMPACTS AND IRREVERSIBLE AND IRRETRIEVABLE COMMITMENT OF RESOURCES

Implementation of any of the alternatives would result in permanent commitment of land for management of the Weldon Spring wastes. The amount of land required for management of these wastes ranges from 11 ha (28 acres) for Alternative 3c (neglecting the land commitment at the uranium processing site for storage of the mill tailings) to 120 ha (300 acres) for Alternative 3a (Hanford site). Alternative 3b ("Nearby Site") would require 15 ha (37 acres) for waste-management activities. About 20 ha (50 acres) would be required for the other action alternatives, i.e., Alternatives 1, 2a, and 2b. Perpetual care of these waste management areas by the federal government (or its successor) will be required because these materials will remain hazardous for thousands of years.

Implementation of any of the action alternatives would unavoidably expose workers and the general public to radiation in addition to the amount they

would normally receive from background radiation. The estimated occupational doses for the action alternatives range from a low of 110 person-rem for Alternative 1 to a high of 230 person-rem for Alternative 3b ("Nearby Site"); the occupational dose would be largest for Alternative 3b due to transport of the wastes by truck to the disposal site. The estimated doses to the general public range from a low of 31 person-rem for Alternative 1, 2a, or 2b (i.e., management of the wastes at the raffinate pits area) to a high of 250 person-rem for Alternative 3a (Hanford site) during the action period. The dose to the general public for the action alternatives during long-term management (1,000 years) ranges from a low of 160 person-rem for Alternative 2a to a high of 720 person-rem for Alternative 3a. For comparison, the no-action Alternative 4 is predicted to result in a cumulative population dose of 11,000 person-rem during long-term management. No adverse health effects are expected for the radioactive doses incurred as a result of implementing any of the action alternatives.

Transportation of the wastes to alternative sites for long-term management would unavoidably expose workers and the general public to the risk of injuries and death associated with transportation accidents. These risks range from 1.7 injuries and 0.13 deaths for transport of the sludge to the uranium processing site (Alternative 3c) to 33 injuries and 2.4 deaths for transport of all the wastes to the Hanford site (Alternative 3a). Transport of the wastes from the quarry to the raffinate pits area is estimated to result in 0.056 injuries and 0.0033 deaths. Transport of fill materials to the waste-management sites for construction of the disposal cell is estimated to result in 1.4 to 3.0 injuries and 0.081 to 0.18 deaths (for Alternatives 3c and 2b, respectively).

Consumptive use of geologic resources (e.g., quarried rock, sand, gravel, and clay) would be required for implementing any of the action alternatives. Adequate supplies of these materials are expected to be readily available at all of the waste-management sites. Consumptive use of petroleum products (e.g., diesel fuel and gasoline) would also occur and would be largest for the alternatives involving off-site transport of the wastes for long-term management. However, adequate supplies should be available without affecting local requirements for these products.

Near the quarry, the contaminated zone is expected to be larger for a longer period of time for the no-action Alternative 4 than for the action alternatives. However, the nature of the limestone and alluvium near the quarry make it less desirable as a groundwater resource than the river alluvium where the county well field is located because the yield from a well in the quarry area or near alluvium would be much lower.

For all action alternatives, if controls cease, there would be the eventual unavoidable dispersion of radioactive and chemical contaminants into the environment. Prediction of how and when this would occur, and the resulting environmental impacts, are beyond current predictive capabilities. If it is assumed that all controls cease in 1,000 years, the predicted loss of cover over the materials ranges from several hundred years to one million years, depending on the use of the land surface. DOE intends to maintain controls over the disposal site to ensure that these impacts do not occur.

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*In this reference list, (1) the term "personal communication" is used to indicate either a telephone conversation or a face-to-face conversation and (2) all letters and memos cited are on file in the Energy and Environmental Systems Division, Argonne National Laboratory; copies are available upon request from: J.M. Peterson, EES, Bldg. 362, Argonne National Laboratory, Argonne, Illinois 60439.

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This Environmental Impact Statement has been prepared by the U.S. Department of Energy (DOE) with contractual assistance from Argonne National Laboratory (ANL), Energy and Environmental Systems Division. The Assistant Secretary for Environment, Health and Safety has approved it for publication. Kenneth Lawver, Oak Ridge Operations, was the technical monitor for DOE. John M. Peterson was the project leader for ANL.

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APPENDIX B. SCOPING

8.1 SCOPING PROCESS

As part of its Surplus Facilities Management Program (SFMP), the U.S. Department of Energy (DOE) issued a Notice of Intent (NOI) in the Federal Register on March 2, 1984 (U.S. Dept. Energy 1984), to prepare an Environmental Impact Statement (EIS) to assess the environmental impacts of alternatives for the long-term management of existing radioactive materials at the Weldon Spring raffinate pits, vicinity properties, and quarry. In accordance with regulations of the Council on Environmental Quality (CEQ) and DOE guidelines for implementing the National Environmental Policy Act (NEPA), the Department conducted a scoping process to determine the alternatives to be analyzed in the EIS, the significant issues to be analyzed in depth, and the issues to be eliminated from further detailed study. The results of the scoping process are given in the DOE Implementation Plan for this EIS (U.S. Dept. Energy 1985) and are summarized in this appendix.

Since issuance of the NOI and completion of the public scoping process, the U.S. Department of the Army has transferred custody of and accountability for the chemical plant to DOE. As a result, management of contaminated materials resulting from decontamination and decommissioning (D&D) of the chemical plant are included in the alternatives evaluated in this EIS. However, many of the issues associated with D&D of the chemical plant are not yet ready for a decision. DOE therefore plans to prepare a later NEPA document specific to D&D of the chemical plant.

Public input to the scoping process included:

- Presentations made at a public meeting held in the Francis Howell High School gymnasium, St. Charles, Missouri, on March 20, 1984, and
- Letters received by DOE regarding the scope of the EIS.

A list of the persons and organizations who provided input during the public scoping process is given in Table 8.1. Considerable input was received from private citizens; organized citizen action groups (particularly the St. Charles Countians Against Hazardous Waste); local, state, and national political representatives; and state government agencies.

Technical input to the scoping process included:

- Preliminary engineering evaluations by Bechtel National, Inc. (BNI), of several alternatives for disposition of the radioactive wastes at the Weldon Spring site;

Table B.1. Participants in the Scoping Process

Oral Comments, Weldon Spring Public Scoping Meeting, March 20, 1984.

Lt. Gov. Kenneth Rothman, State of Missouri
Joseph R. Ortwerth, State Representative, 18th District, Missouri House of Representatives
Richard Roehl, State Representative, 21st District, Missouri House of Representatives
Fred Lafser, Director, Missouri Department of Natural Resources
Gary Elmestad, Staff Assistant to Congressman Robert A. Young, 2nd District
Senator Fred Dyer, Second Senatorial District, Missouri State Senate
John R. Crellin, Missouri Department of Health
Leann Stevens, St. Charles Countians Against Hazardous Waste
Mary A. Halliday, Resident of Defiance, Missouri
Judge Richard M. Green, Presiding Judge, St. Charles County Administrative Court
Judge Peggy Coppage, St. Charles County Administrative Court
Judge Thomas Glosier, St. Charles County Administrative Court
Meredith Bollmeier, Chairperson, St. Charles Countians Against Hazardous Waste
James Whitley, Missouri Department of Conservation
Dan Bolef, Professor of Physics, Washington University in St. Louis
Wallace Howe, Division of Geology and Land Survey, Missouri Department of Natural Resources
Pamela Armstrong, President, League of Women Voters of the St. Charles Area
Bobbie Judge, Vice-President, St. Charles Countians Against Hazardous Waste
Robert M. Wester, President, R.W. Wester & Associates
Thomas J. Aley, Director, Ozark Underground Laboratories
Dr. William T. Rebore, Superintendent, Francis Howell School District
Dominick Ferranto, Jr., Resident of St. Peters, Missouri
Kenneth F. Gronewald, Resident of St. Peters, Missouri
Sandy Tabaka, Resident of St. Charles County, Missouri
Sharon Rogers, Missourians Against Hazardous Waste, Warren County
Kay Drey, Resident of University City, Missouri
Bernard Iffrig, St. Peters Old Town Association

Written Scoping Comments

Tom Nash, Field Supervisor, Columbia Field Office, U.S. Fish and Wildlife Service
Mary A. Halliday, Resident of Defiance, Missouri
Kay Drey, Resident of University City, Missouri
Charles Hajninian, Chief, Environmental Review Branch, Region VII, U.S. Environmental Protection Agency
Ann Hood, St. Charles Countians Against Hazardous Waste
Aimee Judge, Resident of St. Charles, Missouri
Richard C. Rice, Director, Missouri Emergency Management Agency
Al and Linda Hoenig, Residents of St. Peters, Missouri
Bernard J. Iffrig, Resident of St. Peters, Missouri
Robert A. Young, Congressman, 2nd District of Missouri, U.S. House of Representatives
Fred Dyer, Senator, Missouri State Senate
Richard O. Olson, Jr., M.D., St. Charles Clinic, Inc.
Dr. William T. Rebore, Superintendent, Francis Howell School District
Michael V. Garvey, D.D.S., M.S., St. Charles, Missouri

- Meetings and correspondence between ANL and the DOE operations office at Oak Ridge, Tennessee, regarding location and conceptual designs for long-term management of the radioactive wastes;

- Meetings, correspondence, and review of alternatives and issues by SFMP program managers at DOE Headquarters and DOE operations offices at Oak Ridge, Tennessee, and Richland, Washington;

- Preliminary evaluation by ANL and -- in consultation with Missouri state agencies, local government representatives, and members of the St. Charles Countians Against Hazardous Waste -- development of a conceptual design for an additional alternative of a new, above-grade disposal cell at Weldon Spring;

- Meetings with Missouri state agencies (e.g., Department of Natural Resources) and elected officials; and

- Meetings with EPA Region VII.

8.2 ISSUES OF MAJOR CONCERN

During the scoping meeting and in written comments, commenters most often raised the following issues for consideration in the EIS:

- Health risks to members of the general public,

- Chemical contamination (particularly the quarry and chemical plant) and associated potential health risks,

- Removal of the radioactive wastes from Weldon Spring,

- Seepage from the raffinate pits or quarry and potential contamination of drinking water,

- Above-grade containment of wastes,

- Inclusion of the chemical plant in the proposed actions,

- Relocation of wastes from the raffinate pits to another area on the DOE or Army properties,

- Separation/recovery of radioactive substances,

- Effects on residential and industrial development in the Weldon Spring area, and

- Regional seismic conditions.

Other issues receiving emphasis in this EIS are chemical contamination (in addition to radiological) of groundwater and the impact of potential contamination of the St. Charles County well field on development in the county. These issues were included because the Weldon Spring quarry has been nominated to the National Priority List under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and because there is evidence of hazardous wastes at the chemical plant. Meetings and correspondence with officials from the U.S. Environmental Protection Agency (EPA) Region VII have already occurred regarding these issues.

8.3 ALTERNATIVES

The three basic alternatives given in the NOI are those assessed in this EIS. However, these alternatives have been renumbered and reorganized to improve presentation. Alternative 1 in the NOI is the no-action alternative, which is Alternative 4 in the EIS. Alternative 2 in the NOI has been divided into two alternatives in the EIS: Alternative 1, long-term management of the wastes in the existing raffinate pits with improved containment; and Alternative 2, long-term management of the wastes in the raffinate pits area in a new disposal cell. Alternative 2 has been divided into two subalternatives: Alternative 2a, partially above-grade disposal cell with a leachate monitoring system; and Alternative 2b, completely above-grade disposal cell with a layer of lead in the cover and a leachate monitoring system. Alternative 2b was added as a result of input received during the scoping process.

Alternative 3 in the NOI involves long-term management of some (or all) of the wastes at alternative sites. This alternative has been divided into three subalternatives in the EIS: Alternative 3a, transport to and management of all wastes at the Hanford site near Richland, Washington; Alternative 3b, transport to and management of all wastes at a "Nearby Site" in Missouri within 160 km (100 mi) of the Weldon Spring site; and Alternative 3c, transport of the sludge from the raffinate pits and quarry to an existing uranium processing site for reprocessing and management of the remaining lower-activity wastes at the Weldon Spring site. Alternative 3c was added as a result of input received during the scoping process.

Many other subalternatives could have been developed from combinations of the various options for long-term management sites, disposal cell designs,

transportation modes, and so forth. Permutations of all options would have created an unwieldy set of alternatives for analysis and comparison. Therefore, combinations of options covering the range of alternatives to be considered by the DOE decision-maker were selected for analysis. Discussions of the options considered are presented in Appendix E (engineering options) and Appendix F (transportation).

8.4 ISSUES BEYOND THE SCOPE OF THE EIS

DOE has determined that the following issues are beyond the scope of the EIS.

1. Comparison of various regulations -- As provided under the Atomic Energy Act (as amended), DOE is exempt from regulation by states and other federal agencies with respect to radiological aspects of DOE operations. DOE develops its own regulations ("Orders") for general application to DOE operations. Development of such orders is not part of the currently proposed action and is therefore beyond the scope of this EIS. A summary of potentially applicable orders is given in Appendix C of this EIS.

2. Psychological impacts -- As a result of a U.S. Supreme Court case involving the proposed restart of one of the Three Mile Island reactors (Metropolitan Edison Company v. People Against Nuclear Energy [PANE] 103 S. Ct. 1556 [1983]), DOE has decided that analysis of indirect psychological impacts is beyond the scope of this EIS.

3. Impacts of past operations at the site -- The impacts of the various alternatives on the existing environment will be assessed in this EIS. In the above-mentioned Supreme Court decision, it was stated that "NEPA is not directed at the effects of past accidents and does not create a remedial scheme for past federal actions." Therefore, a detailed analysis of past operations, beyond that necessary to characterize the existing environment, is considered to be beyond the scope of this EIS. DOE has sponsored a study of historical radiation doses at Weldon Spring as a separate project. The results have been published in a report by Meshkov et al. (1986).

4. Monitoring of health of students and staff at Francis Howell High School -- No monitoring studies will be performed as part of this EIS. However, health impacts on students and staff are considered.

5. Detailed assessment of the chemical plant cleanup -- The impacts of specific alternatives for D&D of the chemical plant are not specifically assessed in this EIS. These alternatives will be covered in a later NEPA document tiered to this EIS (see Section 1.3 of this EIS).

6. Disposal of Weldon Spring wastes at Callaway Nuclear Power Plant -- The Callaway Nuclear Power Plant is not assessed as a long-term management site in this EIS; however, the impacts of disposal of the wastes at a "Nearby Site" within 160 km (100 mi) of the Weldon Spring site are assessed. If management of the wastes at a "Nearby Site" is chosen, this EIS will serve as the programmatic NEPA document on which assessment of impacts at a specific management site will be tiered.

7. Other management sites -- Only DOE-owned sites are currently available for management of wastes from DOE operations. Long-term management of the Weldon Spring wastes at non-DOE-owned sites is not an alternative reasonably available to the DOE decision-maker. Use of a non-DOE-owned site would involve a multistate compact or NRC licensing. No such site has been chosen or licensed, so the alternative could not be specifically assessed in the EIS. The Hanford site is the only existing DOE-owned site considered in this EIS because it is representative of alternative DOE sites. The Nevada Test Site is not considered because the conditions there are similar to Hanford and disposal at the Nevada Test Site would have similar transportation requirements but lower impacts because of the shorter distance.

8. Other radioactively contaminated sites in the St. Louis area -- EPA Region VII proposed that DOE also consider the possibility of cumulative disposal at the Weldon Spring site of the radioactive wastes currently stored at various locations in Missouri (Figure B.1). These wastes and their locations are as follows:

   - St. Louis Airport Site (SLAPS) and SLAPS Ditches -- previously used for storing ore residues, scrap, and equipment from uranium-processing operations (Newtown and Coxon 1985).

   - Hazelwood Site (9200 Latty Avenue) -- previously used for storing ore residues and wastes from uranium-processing operations (Newtown and Coxon 1985).

- Uranium Processing Facilities, St. Louis -- plant buildings and site previously used to process uranium ore or concentrates to produce uranium dioxide, uranium trioxide, uranium tetrafluoride, and uranium metal; also previously used for other activities with uranium metal and for extraction and concentration of thorium-230 from pitchblende raffinates (Newtown and Coxon 1985).

- West Lake Landfill -- previously used for disposing of soil from the Hazelwood site (Booth et al. 1982).

The SLAPS, Hazelwood, and Uranium Processing Facilities sites are already included in the DOE Formerly Utilized Sites Remedial Action Program; the West Lake Landfill is under NRC cognizance. DOE was directed by Congress in the Conference Report that accompanied U.S. Public Law 98-360 to take the

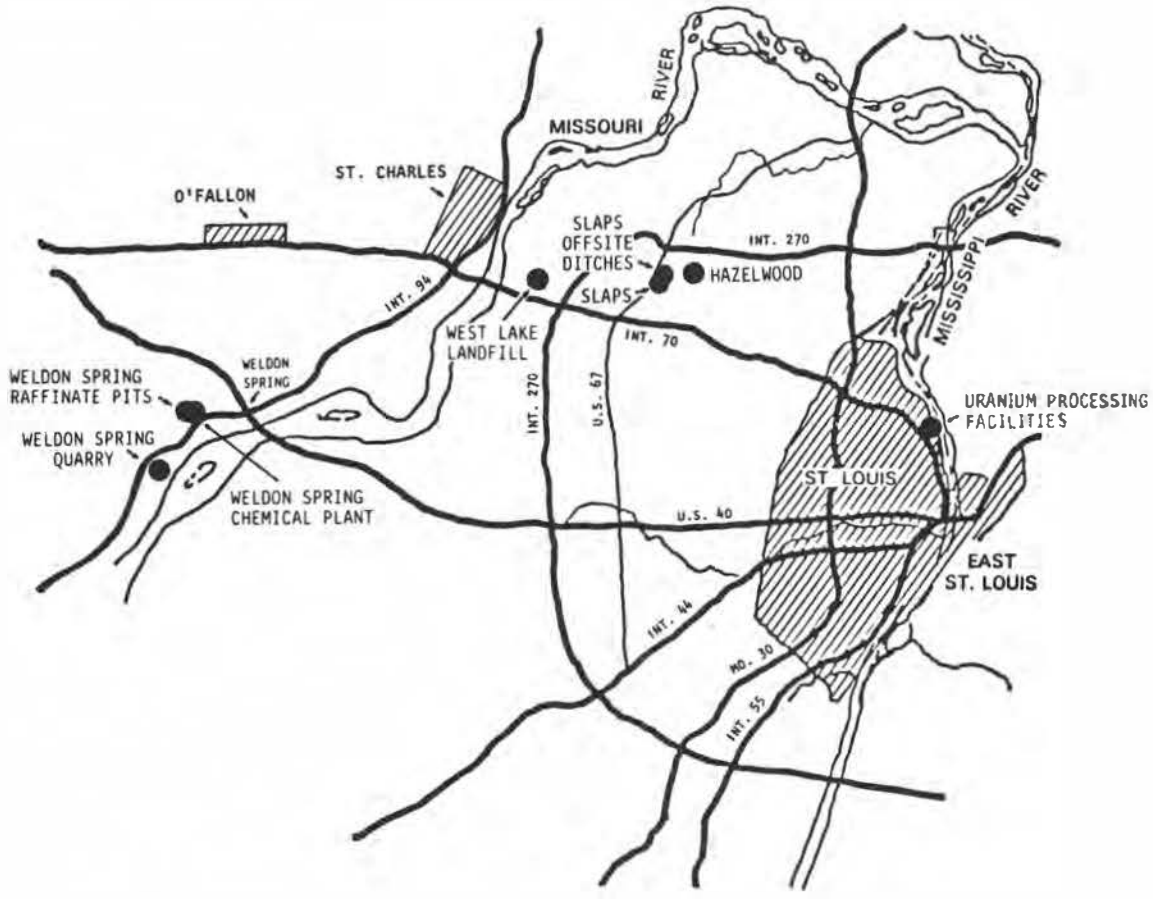

Figure 8.1. Location of Radioactively Contaminated Sites in the St. Louis Area. Source: Modified from Bechtel National (1984).

necessary steps to consolidate and dispose of the waste materials from the Hazelwood and SLAPS vicinity properties. The report directed that the materials be disposed of locally by reacquiring, stabilizing, and using SLAPS in a manner acceptable to the city of St. Louis. Plans for disposal of wastes from the Uranium Processing Facilities in St. Louis have not yet been formulated. The proposal for collecting all these wastes at the Weldon Spring site has been strongly opposed by local citizens; local, state, and national officials; and groups from the St. Louis area. DOE therefore decided not to address, in this EIS, the cumulative impacts from disposal of the wastes from other sites in the St. Louis area at the Weldon Spring site.

8.5 REFERENCES

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Booth, L.F., et al. 1982. Radiological Survey of the West Lake Landfill, St. Louis County, Missouri. NUREG/CR-2722. Prepared by Radiation Management Corporation, Northbrook, Ill., for U.S. Nuclear Regulatory Commission, Office of Nuclear Material Safety and Safeguards, Division of Fuel Cycle and Material Safety. May 1982.

Newtown, G.A., Jr., and G.D. Coxon. 1985. Department of Energy Remedial Action Activities in Missouri. Presented at the Missouri Waste Control Coalition 13th Annual Waste Management Conference, July 15-16, 1985, Columbia, Missouri. 23 pp.

Meshkov, N., et al. 1986. Weldon Spring Historical Dose Estimate. ANL/EES-TM-308. Prepared by Argonne National Laboratory, Argonne, Ill., for U.S. Department of Energy, Oak Ridge Operations Office, Oak Ridge, Tenn. July 1986.

U.S. Department of Energy. 1984. Compliance with the National Environmental Policy Act; Intent to Prepare an Environmental Impact Statement and Conduct a Public Scoping Meeting on Long-Term Management of Existing Radioactive Materials in the Vicinity of Weldon Spring, Missouri. Fed. Regist. 49(43):7851-7854 (March 2, 1984).

U.S. Department of Energy. 1985. Implementation Plan for the Environmental Impact Statement. Long-Term Management of Existing Radioactive Materials in the Vicinity of Weldon Spring, Missouri. Oak Ridge Operations Office, Oak Ridge, Tenn. Final, May 1985.

APPENDIX C. MAJOR LAWS AND REGULATIONS THAT MAY BE POTENTIALLY APPLICABLE TO THE VARIOUS ALTERNATIVES

All remedial actions undertaken at the Weldon Spring site will be done in accordance with all applicable or relevant and appropriate federal laws, regulations, and executive orders -- including the following.

C.1 FEDERAL LAWS AND EXECUTIVE ORDERS

C.1.1 Federal Laws

Archeological and Historic Preservation Act of 1974
Archeological Resources Protection Act of 1979
Atomic Energy Act of 1954, as amended
Clean Air Act of 1963, as amended
Clean Water Act, as amended (also referred to as Federal Water Pollution Control Act of 1972, as amended)
Comprehensive Environmental Response, Compensation, and Liability Act of 1980, as amended
Department of Energy Organization Act of 1977
Endangered Species Act of 1973, as amended
Fish and Wildlife Coordination Act of 1934, as amended
Hazardous Materials Transportation Act of 1974, as amended
National Environmental Policy Act of 1969, as amended
National Historic Preservation Act of 1966, as amended
Noise Control Act of 1972
Noise Pollution and Abatement Act of 1970
Occupational Safety and Health Act of 1970
Safe Drinking Water Act of 1974
Soil and Water Resources Conservation Act of 1977
Solid Waste Disposal Act, as amended by the Resource Conservation and Recovery Act of 1976, as amended in 1984 (RCRA)
Superfund Amendments and Reauthorization Act of 1986
Toxic Substances Control Act of 1976
Uranium Mill Tailings Radiation Control Act of 1978 (UMTRCA)

Based on waste characterization data and information for the Weldon Spring site, management and disposal of the wastes will be as follows: (1) those substances that are radiologically contaminated only, with no associated chemical contamination hazard, will be managed and disposed of in accordance with UMTRCA; (2) those substances that are chemically contaminated only, with no associated radiological contamination hazard, will be managed and disposed of in accordance with RCRA; and (3) those substances that are both chemically and radiologically contaminated will be managed and disposed of in accordance with the best technical approach available considering RCRA and UMTRCA to ensure maximum protection of public health, welfare, and the environment.

C.1.2 Executive Orders

Executive Order 11490, Assigning Emergency Preparedness Functions to Federal Departments and Agencies
Executive Order 11514, Protection and Enhancement of Environmental Quality
Executive Order 11738, Providing for Administration of the Clean Air Act and the Federal Water Pollution Control Act with Respect to Federal Contracts, Grants, or Loans
Executive Order 11807, Occupational Safety and Health Programs for Federal Employees
Executive Order 11988, Floodplain Management
Executive Order 11990, Protection of Wetlands
Executive Order 11991, Relating to the Protection and Enhancement of Environmental Quality
Executive Order 12088, Federal Compliance with Pollution Control Standards
Executive Order 12146, Management of Federal Legal Resources

C.2 DEPARTMENT OF ENERGY ORDERS

Order 1540.1	Materials Transportation and Traffic Management
Order 4320.1A	Site Development and Facility Utilization Planning
Order 5440.1C	Implementation of the National Environmental Policy Act
Order 5480.1A	Environmental Protection, Safety, and Health Protection Program for DOE Operations*
Order 5480.28	Hazardous and Radioactive Mixed Waste Management
Order 5480.4	Environmental Protection, Safety, and Health Protection Standards
Order 5480.14	Comprehensive Environmental Response, Compensation, and Liability Act Program
Order 5481.18	Safety Analysis Review System
Order 5482.18	Environmental Protection, Safety, and Health Protection Appraisal Program
Order 5483.1A	Occupational Safety and Health Program for Government-Owned Contractor-Operated Facilities
Order 5484.1	Environmental Protection, Safety, and Health Protection Information Reporting Requirements
Order 5000.3	Unusual Occurrence Reporting System
Order 5500.2	Emergency Planning, Preparedness, and Response for Operations
Order 5700.4A	Project Management System
Order 5700.68	Quality Assurance
Order 5820.2	Radioactive Waste Management

*Chapter XI of Order 5480.1A has been amended -- see Vaughan (1985) and U.S. Department of Energy (1986).

C.3 REFERENCES

U.S. Department of Energy. 1986. DOE-Derived Concentration Guides for Drinking Water and Breathing Air Contaminated with Radionuclides by Members of the Public. Attachment to Memorandum from R.J. Stern (Director, Office of Environmental Guidance) to Distribution, February 28, 1986.

Vaughan, W.A. 1985. Memorandum from W.A. Vaughan (U.S. Department of Energy, Assistant Secretary for Environment, Safety and Health) to Distribution, August 5, 1985. Subject: Radiation Standards for Protection of the Public in the Vicinity of DOE Facilities.

APPENDIX D. DOE GUIDELINES FOR RESIDUAL RADIOACTIVITY

U.S. DEPARTMENT OF ENERGY GUIDELINES FOR RESIDUAL RADIOACTIVITY AT FORMERLY UTILIZED SITES REMEDIAL ACTION PROGRAM AND REMOTE SURPLUS FACILITIES MANAGEMENT PROGRAM SITES

(Rev. 1, July 1985)

A. INTRODUCTION

This document presents U.S. Department of Energy (DOE) radiological protection guidelines for cleanup of residual radioactive materials and management of the resulting wastes and residues. It is applicable to sites identified by the Formerly Utilized Sites Remedial Action Program (FUSRAP) and remote sites identified by the Surplus Facilities Management Program (SFMP).* The topics covered are basic dose limits, guidelines and authorized limits for allowable levels of residual radioactivity, and requirements for control of the radioactive wastes and residues.

Protocols for identification, characterization, and designation of FUSRAP sites for remedial action; for implementation of the remedial action; and for certification of a FUSRAP site for release for unrestricted use are given in a separate document (U.S. Dept. Energy 1984). More detailed information on applications of the guidelines presented herein, including procedures for deriving site- specific guidelines for allowable levels of residual radioactivity from basic dose limits, is contained in a supplementary document -- referred to herein as the "supplement" (U.S. Dept. Energy 1985).

"Residual radioactivity" includes: (1) residual concentrations of radionuclides in soil material,** (2) concentrations of airborne radon decay products, (3) external gamma radiation level, and (4) surface contamination. A "basic dose limit" is a prescribed standard from which limits for quantities that can be monitored and controlled are derived; it is specified in terms of the effective dose equivalent as defined by the International Commission on Radiological Protection (ICRP 1977, 1978). Basic dose limits are used explicitly for deriving guidelines for residual concentrations of radionuclides in soil material, except for thorium and radium. Guidelines for

*A remote SFMP site is one that is excess to DOE programmatic needs and is located outside a major operating DOE research and development or production area.

**The term "soil material" refers to all material below grade level after remedial action is completed.

residual concentrations of thorium and radium and for the other three quantities (airborne radon decay products, external gamma radiation level, and surface contamination) are based on existing radiological protection standards (U.S. Environ. Prot. Agency 1983; U.S. Nucl. Reg. Comm. 1982). These standards are assumed to be consistent with basic dose limits within the uncertainty of derivations of levels of residual radioactivity from basic limits.

A "guideline" for residual radioactivity is a level of residual radioactivity that is acceptable if the use of the site is to be unrestricted. Guidelines for residual radioactivity presented herein are of two kinds: (1) generic, site-independent guidelines taken from existing radiation protection standards, and (2) site-specific guidelines derived from basic dose limits using site-specific models and data. Generic guideline values are presented in this document. Procedures and data for deriving site-specific guideline values are given in the supplement.

An "authorized limit" is a level of residual radioactivity that must not be exceeded if the remedial action is to be considered completed. Under normal circumstances, expected to occur at most sites, authorized limits for residual radioactivity are set equal to guideline values. Exceptional conditions for which authorized limits might differ from guideline values are specified in Sections D and F. A site may be released for unrestricted use only if the residual radioactivity does not exceed guideline values at the time remedial action is completed. Restrictions and controls on use of the site must be established and enforced if the residual radioactivity exceeds guideline values. The applicable controls and restrictions are specified in Section E.

DOE policy requires that all exposures to radiation be limited to levels that are as low as reasonably achievable (ALARA). Implementation of ALARA policy is specified as procedures to be applied after authorized limits have been set. For sites to be released for unrestricted use, the intent is to reduce residual radioactivity to levels that are as far below authorized limits as reasonable considering technical, economic, and social factors. At sites where the residual radioactivity is not reduced to levels that permit release for unrestricted use, ALARA policy is implemented by establishing controls to reduce exposure to levels that are as low as is reasonably achievable. Procedures for implementing ALARA policy are described in the supplement. ALARA policies, procedures, and actions must be documented and filed as a permanent record upon completion of remedial action at a site.

B. BASIC DOSE LIMITS

The basic limit for the annual radiation dose received by an individual member of the general public is 500 mrem/yr for a period of exposure not to exceed 5 years and an average of 100 mrem/yr over a lifetime. The committed effective dose equivalent, as defined in ICRP Publication 26 (ICRP 1977) and calculated by dosimetry models described in ICRP Publication 30 (ICRP 1978), shall be used for determining the dose.

C. GUIDELINES FOR RESIDUAL RADIOACTIVITY

C.1 Residual Radionuclides in Soil Material

Residual concentrations of radionuclides in soil material shall be specified as above-background concentrations averaged over an area of 100 m^2. If the concentration in any area is found to exceed the average by a factor greater than 3, guidelines for local concentrations shall also be applicable. These "hot spot" guidelines depend on the extent of the elevated local concentrations and are given in the supplement.

The generic guidelines for residual concentrations of Th-232, Th-230, Ra-228, and Ra-226 are:

- 5 pCi/g, averaged over the first 15 cm of soil below the surface
- 15 pCi/g, averaged over 15-cm-thick layers of soil more than 15 cm below the surface

These guidelines take into account ingrowth of Ra-226 from Th-230 and of Ra-228 from Th-232, and assume secular equilibrium. If either Th-230 and Ra-226 or Th-232 and Ra-228 are both present, not in secular equilibrium, the guidelines apply to the higher concentration. If other mixtures of radionuclides occur, the concentrations of individual radionuclides shall be reduced so that the dose for the mixtures will not exceed the basic dose limit. Explicit formulas for calculating residual concentration guidelines for mixtures are given in the supplement.

The guidelines for residual concentrations in soil material of all other radionuclides shall be derived from basic dose limits by means of an environmental pathway analysis using site-specific data. Procedures for deriving these guidelines are given in the supplement.

C.2 Airborne Radon Decay Products

Generic guidelines for concentrations of airborne radon decay products shall apply to existing occupied or habitable structures on private property that are intended for unrestricted use; structures that will be demolished or buried are excluded. The applicable generic guideline (40 CFR 192) is: In any occupied or habitable building, the objective of remedial action shall be, and reasonable effort shall be made to achieve, an annual average (or equivalent) radon decay product concentration (including background) not to exceed 0.02 WL.* In any case, the radon decay product concentration (including background) shall not exceed 0.03 WL. Remedial actions are not required in order to comply with this guideline when there is reasonable assurance that residual radioactive materials are not the cause.

C.3 External Gamma Radiation

The average level of gamma radiation inside a building or habitable structure on a site to be released for unrestricted use shall not exceed the background level by more than 20 μR/h.

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*A working level (WL) is any combination of short-lived radon decay products in one liter of air that will result in the ultimate emission of 1.3×10^5 MeV of potential alpha energy.

C.4 Surface Contamination

The following generic guidelines, adapted from standards of the U.S. Nuclear Regulatory Commission (1982), are applicable only to existing structures and equipment that will not be demolished and buried. They apply to both interior and exterior surfaces. If a building is demolished and buried, the guidelines in Section C.1 are applicable to the resulting contamination in the ground.

	Allowable Total Residual Surface Contamination (dpm/100 cm²)[a]		
Radionuclides[b]	Average[c,d]	Maximum[d,e]	Removable[d,f]
Transuranics, Ra-226, Ra-228, Th-230, Th-228, Pa-231, Ac-227, I-125, I-129	100	300	20
Th-Natural, Th-232, Sr-90, Ra-223, Ra-224, U-232, I-126, I-131, I-133	1,000	3,000	200
U-Natural, U-235, U-238, and associated decay products	5,000α	15,000α	1,000α
Beta-gamma emitters (radionuclides with decay modes other than alpha emission or spontaneous fission) except Sr-90 and others noted above	5,000β-γ	15,000β-γ	1,000β-γ

[a] As used in this table, dpm (disintegrations per minute) means the rate of emission by radioactive material as determined by correcting the counts per minute measured by an appropriate detector for background, efficiency, and geometric factors associated with the instrumentation.

[b] Where surface contamination by both alpha- and beta-gamma-emitting radionuclides exists, the limits established for alpha- and beta-gamma-emitting radionuclides should apply independently.

[c] Measurements of average contamination should not be averaged over an area of more than 1 m². For objects of less surface area, the average should be derived for each such object.

[d] The average and maximum dose rates associated with surface contamination resulting from beta-gamma emitters should not exceed 0.2 mrad/h and 1.0 mrad/h, respectively, at 1 cm.

[e] The maximum contamination level applies to an area of not more than 100 cm².

[f] The amount of removable radioactive material per 100 cm² of surface area should be determined by wiping that area with dry filter or soft absorbent paper, applying moderate pressure, and measuring the amount of radioactive material on the wipe with an appropriate instrument of known efficiency. When removable contamination on objects of surface area less than 100 cm² is determined, the activity per unit area should be based on the actual area and the entire surface should be wiped. The numbers in this column are maximum amounts.

D. AUTHORIZED LIMITS FOR RESIDUAL RADIOACTIVITY

The remedial action shall not be considered complete unless the residual radioactivity is below authorized limits. Authorized limits shall be set equal to guidelines for residual radioactivity unless: (1) exceptions specified in Section F of this document are applicable, in which case an authorized limit may be set above the guideline value for the specific location or condition to which the exception is applicable; or (2) on the basis of site-specific data not used in establishing the guidelines, it can be clearly established that limits below the guidelines are reasonable and can be achieved without appreciable increase in cost of the remedial action. Authorized limits that differ from guidelines must be justified and established on a site-specific basis, with documentation that must be filed as a permanent record upon completion of remedial action at a site. Authorized limits differing from the guidelines must be approved by the Director, Oak Ridge Technical Services Division, for FUSRAP and by the Director, Richland Surplus Facilities Management Program Office, for remote SFMP -- with concurrence by the Director of Remedial Action Projects for both programs.

E. CONTROL OF RESIDUAL RADIOACTIVITY AT FUSRAP AND REMOTE SFMP SITES

Residual radioactivity above the guidelines at FUSRAP and remote SFMP sites must be managed in accordance with applicable DOE Orders. The DOE Order 5480.1A requires compliance with applicable federal, state, and local environmental protection standards.

The operational and control requirements specified in the following DOE Orders shall apply to interim storage, interim management, and long-term management.

a. 5440.1B, Implementation of the National Environmental Policy Act

b. 5480.1A, Environmental Protection, Safety, and Health Protection Program for DOE Operations

c. 5480.2, Hazardous and Radioactive Mixed Waste Management

d. 5480.4, Environmental Protection, Safety, and Health Protection Standards

e. 5482.1A, Environmental, Safety, and Health Appraisal Program

f. 5483.1, Occupational Safety and Health Program for Government-Owned Contractor-Operated Facilities

g. 5484.1, Environmental Protection, Safety, and Health Protection Information Reporting Requirements

h. 5484.2, Unusual Occurrence Reporting System

i. 5820.2, Radioactive Waste Management

E.1 Interim Storage

a. Control and stabilization features shall be designed to ensure, to the extent reasonably achievable, an effective life of 50 years and, in any case, at least 25 years.

b. Above-background Rn-222 concentrations in the atmosphere above facility surfaces or openings shall not exceed: (1) 100 pCi/L at any given point, (2) an annual average concentration of 30 pCi/L over the facility site, and (3) an annual average concentration of 3 pCi/L at or above any location outside the facility site (DOE Order 5480.1A, Attachment XI-1).

c. Concentrations of radionuclides in the groundwater or quantities of residual radioactive materials shall not exceed existing federal, state, or local standards.

d. Access to a site shall be controlled and misuse of on-site material contaminated by residual radioactivity shall be prevented through appropriate administrative controls and physical barriers -- active and passive controls as described by the U.S. Environmental Protection Agency (1983--p. 595). These control features should be designed to ensure, to the extent reasonable, an effective life of at least 25 years. The federal government shall have title to the property.

E.2 Interim Management

a. A site may be released under interim management when the residual radioactivity exceeds guideline values if the residual radioactivity is in inaccessible locations and would be unreasonably costly to remove, provided that administrative controls are established to ensure that no member of the public shall receive a radiation dose exceeding the basic dose limit.

b. The administrative controls, as approved by DOE, shall include but not be limited to periodic monitoring, appropriate shielding, physical barriers to prevent access, and appropriate radiological safety measures during maintenance, renovation, demolition, or other activities that might disturb the residual radioactivity or cause it to migrate.

c. The owner of the site or appropriate federal, state, or local authorities shall be responsible for enforcing the administrative controls.

E.3 Long-Term Management

Uranium, Thorium, and Their Decay Products

a. Control and stabilization features shall be designed to ensure, to the extent reasonably achievable, an effective life of 1,000 years and, in any case, at least 200 years.

b. Control and stabilization features shall be designed to ensure that Rn-222 emanation to the atmosphere from the waste shall not: (1) exceed an annual average release rate of 20 $pCi/m^2/s$, and (2) increase the annual average Rn-222 concentration at or above any location outside the boundary of the contaminated area by more than 0.5 pCi/L. Field verification of emanation rates is not required.

c. Prior to placement of any potentially biodegradable contaminated wastes in a long-term management facility, such wastes shall be properly conditioned to ensure that (1) the generation and escape of biogenic gases will not cause the requirement in paragraph b of this section (E.3) to be exceeded, and (2) biodegradation within the facility will not result in premature structural failure in violation of the requirements in paragraph a of this section (E.3).

d. Groundwater shall be protected in accordance with 40 CFR 192.20(a)(2) and 192.20(a)(3), as applicable to FUSRAP and remote SFMP sites.

e. Access to a site should be controlled and misuse of on-site material contaminated by residual radioactivity should be prevented through appropriate administrative controls and physical barriers -- active and passive controls as described by the U.S. Environmental Protection Agency (1983--p. 595). These controls should be designed to be effective to the extent reasonable for at least 200 years. The federal government shall have title to the property.

Other Radionuclides

f. Long-term management of other radionuclides shall be in accordance with Chapters 2, 3, and 5 of DOE Order 5820.2, as applicable.

F. EXCEPTIONS

Exceptions to the requirement that authorized limits be set equal to the guidelines may be made on the basis of an analysis of site-specific aspects of a designated site that were not taken into account in deriving the guidelines. Exceptions require approvals as stated in Section D. Specific situations that warrant exceptions are:

a. Where remedial actions would pose a clear and present risk of injury to workers or members of the general public, notwithstanding reasonable measures to avoid or reduce risk.

b. Where remedial actions -- even after all reasonable mitigative measures have been taken -- would produce environmental harm that is clearly excessive compared to the health benefits to persons living on or near affected sites, now or in the future. A clear excess of environmental harm is harm that is long-term, manifest, and grossly disproportionate to health benefits that may reasonably be anticipated.

c. Where the cost of remedial actions for contaminated soil is unreasonably high relative to long-term benefits and where the residual radioactive materials do not pose a clear present or future risk after taking necessary control measures. The likelihood that buildings will be erected or that people will spend long periods of time at such a site should be considered in evaluating this risk. Remedial actions will generally not be necessary where only minor quantities of residual radioactive materials are involved or where residual radioactive materials occur in an inaccessible location at

which site-specific factors limit their hazard and from which they are costly or difficult to remove. Examples are residual radioactive materials under hard-surface public roads and sidewalks, around public sewer lines, or in fence-post foundations. In order to invoke this exception, a site-specific analysis must be provided to establish that it would not cause an individual to receive a radiation dose in excess of the basic dose limits stated in Section B, and a statement specifying the residual radioactivity must be included in the appropriate state and local records.

d. Where the cost of cleanup of a contaminated building is clearly unreasonably high relative to the benefits. Factors that shall be included in this judgment are the anticipated period of occupancy, the incremental radiation level that would be effected by remedial action, the residual useful lifetime of the building, the potential for future construction at the site, and the applicability of remedial actions that would be less costly than removal of the residual radioactive materials. A statement specifying the residual radioactivity must be included in the appropriate state and local records.

e. Where there is no feasible remedial action.

C. SOURCES

Limit or Guideline	Source
Basic Dose Limits	
Dosimetry Model and Dose Limits	International Commission on Radiological Protection (1977, 1978)
Generic Guidelines for Residual Radioactivity	
Residual Concentrations of Radium and Thorium in Soil Material	40 CFR 192
Airborne Radon Decay Products	40 CFR 192
External Gamma Radiation	40 CFR 192
Surface Contamination	Adapted from U.S. Nuclear Regulatory Commission (1982)
Control of Radioactive Wastes and Residues	
Interim Storage	DOE Order 5480.1A
Long-Term Management	DOE Order 5480.1A; 40 CFR 192

H. REFERENCES

International Commission on Radiological Protection. 1977. Recommendations of the International Commission on Radiological Protection (Adopted January 17, 1977). ICRP Publication 26. Pergamon Press, Oxford. [As modified by "Statement from the 1978 Stockholm Meeting of the ICRP." Annals of the ICRP, Vol. 2, No. 1, 1978.]

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APPENDIX E. ENGINEERING OPTIONS

The alternatives analyzed in this EIS are intended to represent a broad range of possible alternatives that could be carried out to properly treat the contaminated materials at the Weldon Spring site. Each alternative considered consists of several steps. In this appendix, the options available at various steps are discussed and compared.

E.1 ESTIMATED QUANTITIES OF MATERIALS

The estimated volumes of wastes (before and after remedial action) and clean materials (e.g., clay, sand, rock, and topsoil) for each action alternative are given in Tables E.1 through E.6. These waste volumes represent most of the materials that will require long-term management. Implementation of one of the action alternatives will produce small amounts of additional radiologically and chemically contaminated materials that will require appropriate treatment and management, such as contaminated clothing and wastes from water treatment for removal of certain chemical or radioactive species. The environmental impacts associated with management of these additional wastes will be small relative to the impacts for the wastes listed in Tables E.1 through E.6 and, therefore, are not specifically addressed in this EIS.

E.2 WATER DISPOSAL AND TREATMENT

E.2.1 Water Disposal

It has been estimated (Bechtel Natl. 1984a, 1985a) that 314,000 m^3 (83,000,000 gal) of contaminated water will be generated during cleanup, including 11,000 m^3 (2,900,000 gal) from the quarry pond, 216,000 m^3 (57,000,000 gal) from the raffinate pits, and 87,000 m^3 (23,000,000 gal) of washwater, rainwater, and groundwater.

E.2.1.1 Discharge to the Missouri River

One method for disposing of the standing water in the raffinate pits and quarry ponds and the washwater generated during the action period would be discharging the water to the Missouri River after necessary treatment; the discharge would be carried out under controlled conditions and in compliance with Missouri state requirements. Discharging to the river could be done by tank trucks carrying the water, by pipeline, or by direct stream flow. Use of tank trucks would probably require construction of a road from the holding ponds, where the trucks would be loaded, to the discharge point. The drainage

Table E.1. Summary of Estimated Material Quantities before and after Completion of the Actions in Alternative 1[a]

Location	Wastes (m^3), Before Action			Wastes (m^3), After Action			
	Sludge	Soil/Clay	Rubble	Sludge[b]	Soil/Clay	Rubble	Total
Pit 1	13,300	-	-	18,200	-	-	18,200
Pit 2	13,300	-	-	18,200	-	-	18,200
Pit 3	99,100	-	-	194,000	-	-	194,000
Pit 4	42,500	-	400	4,300	289,200	40,300	333,800
Quarry[c]	3,100	38,800	30,700	-	-	-	-
Chemical plant	-	229,400	9,200	-	-	-	-
Vicinity properties	-	21,000[d]	-	-	-	-	-
Total	171,000	289,000	40,000	235,000	288,000	40,000	564,000

Component	Clean Materials				
	Clay (m^3)	Sand/Rock/ Gravel (m^3)	Riprap (m^3)	Backfill (m^3)	Topsoil (m^3)
Multilayered cover[e]	240,900	45,300	129,100	-	68,600
Area restoration	-	-	-	145,900[f]	6,300[g]

[a] Adapted from Bechtel National (1984a, 1985a--Alternatives 3A and 4). NA means not applicable. A hyphen (-) means zero quantity. Component values are rounded to the nearest 100 m^3. Totals are rounded to the nearest 1,000 m^3.

[b] Values include a 37% volume increase due to addition of stabilizer (20 wt. % cement and 80 wt. % fly ash).

[c] Assumes that the total volume of quarry wastes is 72,600 m^3 and that two-thirds of the 46,000 m^3 of wastes on the quarry floor is rubble (Bechtel Natl. 1984a, 1985b; Hickey 1986).

[d] Sources: Boerner (1986); Deming (1986).

[e] Amounts are those given by Bechtel National (1984a) for its Alternative 4.

[f] Includes the raffinate pits area and the quarry area; 135,300 m^3 are needed to backfill the quarry area and 10,600 m^3 are needed for the pits area. The 10,600 m^3 can be obtained on-site as uncontaminated clay from the existing Pit 4 dike. Only 94,500 m^3 of the amount of backfill needed for the quarry must be obtained off-site because 40,800 m^3 of uncontaminated rubble is available from the chemical plant (Bechtel Natl. 1984a, 1985a).

[g] Amount consists of 2,800 m^3 needed for the raffinate pits area and 3,500 m^3 needed for the quarry area.

Table E.2. Summary of Estimated Material Quantities before and after Completion of the Actions in Alternative 2a[a]

Location	Wastes (m^3), Before Action			Wastes (m^3), After Action			
	Sludge	Soil/Clay	Rubble	Sludge[b]	Soil/Clay	Rubble	Total
Pit 1	13,300	4,500[c]	-	-	-	-	-
Pit 2	13,300	4,500[c]	-	-	-	-	-
Pit 3	99,100	32,500	-	-	-	-	-
Pit 4	42,500	56,700	400	-	-	-	-
Quarry[d]	3,100	38,800	30,700	-	-	-	-
Chemical plant	-	229,400	9,200	-	-	-	-
Vicinity properties	-	21,000[e]	-	-	-	-	-
New disposal cell	-	-	-	234,800	387,400	40,300	662,500
Totals	171,000	387,000	40,000	235,000	387,000	40,000	662,000

Component	Clean Materials							
	Clay (m^3)	Sand/Rock Gravel (m^3)	Riprap (m^3)	Backfill (m^3)	Topsoil (m^3)	Pipe (m)	Geofabric (m^2)	Sumps (m^3)
New disposal cell	62,300[f]	-	-	-	-	-	-	-
Multilayered cover	232,800	42,700	128,100	-	64,000	-	-	-
Leachate system	-	15,300[g]	-	-	-	990	230	170[g]
Area restoration	-	-	-	269,300[h]	20,200[i]	-	-	-

[a] Adapted from Bechtel National (1985a--Tables 2-6 and 2-7). A hyphen (-) means zero quantity. Componegt values are rounded to the nearest 100 m^3. Totals are rounded to the nearest 1,000 m^3.

[b] Values include a volume increase of 37% due to addition of stabilizer (20 wt. % cement and 80 wt. % fly ash).

[c] Value represents the assumed amount of contaminated clay to be removed from the bottoms and sides of the pits. Value was calculated by assuming that the amount excavated from each pit is proportional to the area of each pit (Pits 1 and 2, 0.48 ha each; Pit 3, 3.40 ha; Pit 4, 6.07 ha (Bechtel Natl. 1985a--Table 2-21) and that 89,200 m^3 is excavated from Pits 3 and 4 (Bechtel Natl. 1985a--Section 2.2.2.4).

[d] Assumes that the total volume of quarry wastes is 72,600 m^3 and that two-thirds of the 46,000 m^3 of wastes on the quarry floor is rubble (Bechtel Natl. 1984a, 1985b; Hickey 1986).

[e] Sources: Boerner (1986); Deming (1986).

[f] Needed to construct dike for new disposal cell.

[g] Excludes volume of gravel and rock needed to fill sumps. The volume of each of the nine sumps, 19 m^3, was obtained from Bechtel National (1985a--Table 3-2 and p. 38) and assumes that the total sump volume is equal to the volume of rock and gravel used in construction of the leachate system in Bechtel's Alternative 3A or 3E.

[h] Value includes backfill needed for quarry area (135,300 m^3). It excludes amount needed, if any, to restore the vicinity properties. Some of this volume may be obtained from clean material salvaged from the raffinate pits dikes.

[i] Value includes 3,500 m^3 needed for the quarry area and 16,700 m^3 needed for the raffinate pits area. Some may be available from excavation of the new disposal cell.

Table E.3. Summary of Estimated Material Quantities before and after Completion of the Actions in Alternative 2b[a]

Location	Wastes (m^3), Before Action[b]			Wastes (m^3), After Action[b]			
	Sludge	Soil/Clay	Rubble	Sludge[c]	Soil/Clay	Rubble	Total
Pit 1	13,300	4,500[d]	-	-	-	-	-
Pit 2	13,300	4,500[d]	-	-	-	-	-
Pit 3	99,100	32,500[d]	-	-	-	-	-
Pit 4	42,500	56,700[d]	400	-	-	-	-
Quarry[e]	3,100	38,800	30,700	-	-	-	-
Chemical plant	-	229,400	9,200	-	-	-	-
Vicinity properties	-	21,000[f]	-	-	-	-	-
New disposal cell	-	-	-	234,800	387,400	40,300	662,500
TOTALS	171,000	387,000	40,000	235,000	387,000	40,000	662,000

Component	Clean Materials[g]							
	Clay (m^3)	Sand/Rock/ Gravel (m^3)	Riprap (m^3)	Backfill (m^3)	Topsoil (m^3)	Perforated Clay Pipe (m)	Lead Sheet (m^2)	Sumps (m^3)
New disposal cell								
Floor	220,000	-	-	-	-	-	-	-
Leachate system	-	16,100[h]	-	-	-	16,400	-	174[i]
Drainage apron, swale	-	2,600	20,600	-	-	-	-	-
Dike	121,000	-	-	-	-	-	-	-
Multilayered cover	267,000	58,000	173,000	-	85,100	-	126,300	-
Area restoration	-	-	-	269,300[j]	20,200[k]	-	-	-

[a] A hyphen (-) means zero quantity. Component values are rounded to the nearest 100 m^3. Totals are rounded to the nearest 1,000 m^3.

[b] Values taken to be the same as for Alternative 2a, Table E.2. No before-action entries are present for soil/clay and rubble for the new disposal cell because it will be above grade (no excavation needed) in this alternative.

[c] Values include a volume increase of 37% due to addition of stabilizer (20 wt. % cement and 80 wt. % fly ash).

[d] Value represents the amount of contaminated clay to be removed from the bottoms and sides of the pits (Table E.2).

[e] Assumes that the total volume of quarry wastes is 72,600 m^3 and that two-thirds of the 46,000 m^3 of wastes on the quarry floor is rubble (Bechtel Natl. 1984a, 1985b; Hickey 1986).

[f] Sources: Boerner (1986); Deming (1986).

[g] Values are approximate and depend on design details.

[h] Excludes volume of rock/gravel needed to fill sumps.

[i] 58 sumps are needed at an assumed volume of 3 m^3 per sump. This gives a total sump volume of 174 m^3, which is about equal to the total sump volume (170 m^3) for Alternative 2a (Table E.2) (Bechtel Natl. 1985a--Table 2-3 and p. 38).

[j] Value includes backfill needed for quarry area (135,300 m^3). It excludes amount needed, if any, to restore the vicinity properties. Some of this volume may be obtained from clean material salvaged from the raffinate pits dikes.

[k] Value includes 3,500 m^3 needed for the quarry area and 16,700 m^3 needed for the raffinate pits area. Some may be available from excavation of the new disposal cell.

Table E.4. Summary of Estimated Material Quantities before and after Completion of the Actions in Alternative 3a[a]

Location	Wastes (m^3), Before Action			Wastes (m^3), After Action			
	Sludge	Soil/Clay	Rubble	Sludge[b]	Soil/Clay	Rubble	Total
Pit 1	13,300	4,500[c]	-	-	-	-	-
Pit 2	13,300	4,500[c]	-	-	-	-	-
Pit 3	99,100	32,500[c]	-	-	-	-	-
Pit 4	42,500	56,700[c]	400	-	-	-	-
Quarry[d]	3,100	38,800	30,700	-	-	-	-
Chemical plant	-	229,400	9,200	-	-	-	-
Vicinity properties	-	21,000[e]	-	-	-	-	-
Hanford site	-	-	-	51,400	387,400	40,300	479,100
Totals	171,000	387,000	40,000	51,000	387,000	40,000	479,000

Component	Clean Materials (m^3)			
	Excavated Material	Riprap	Backfill	Topsoil
Trenches (Hanford)	1,277,600[f]	NA	NA	NA
Cover (Hanford)	NA	307,000[g]	1,277,600[f]	-
Area restoration (Weldon Spring)	NA	NA	317,200[h]	22,000[i]

[a] Adapted from Bechtel National (1984a--Table 6-5). A hyphen (-) means zero quantity. NA means not applicable. Component values are given to the nearest 100 m^3. Totals are rounded to the nearest 1,000 m^3.

[b] Volume is reduced by 70% to account for water removal by heat drying.

[c] Value represents the amount of contaminated clay to be removed from the bottoms and sides of the pits (Table E.2).

[d] Assumes that the total volume of quarry wastes is 72,600 m^3 and that two-thirds of the 46,000 m^3 of wastes on the quarry floor is rubble (Bechtel Natl. 1984a, 1985b; Hickey 1986).

[e] Sources: Boerner (1986); Deming (1986).

[f] Volume of local soils excavated to make trenches. All would be reserved on the Hanford site to be used as backfill in the multilayered cover.

[g] Material brought in from off-site.

[h] Value includes 181,900 m^3 backfill needed for the raffinate pits area (Bechtel Natl. 1984a--Table 6-5) and 135,300 m^3 backfill needed for the quarry area. It excludes the amount needed, if any, to restore the vicinity properties. Some of this volume may be obtained from clean material salvaged from the raffinate pits dikes.

[i] Value includes 18,500 m^3 needed for the raffinate pits area (Bechtel Natl. 1984a--Table 6-5) and 3,500 m^3 needed for the quarry area.

Table E.5. Summary of Estimated Material Quantities before and after Completion of the Actions in Alternative 3b[a]

Location	Wastes (m^3), Before Action			Wastes (m^3), After Action			
	Sludge	Soil/Clay	Rubble	Sludge[b]	Soil/Clay	Rubble	Total
Pit 1	13,300	4,500[c]	-	-	-	-	-
Pit 2	13,300	4,500[c]	-	-	-	-	-
Pit 3	99,200	32,500[c]	-	-	-	-	-
Pit 4	42,500	56,700[c]	400	-	-	-	-
Quarry[d]	3,100	38,800	30,700	-	-	-	-
Chemical plant	-	229,400	9,200	-	-	-	-
Vicinity properties	-	21,000[e]	-	-	-	-	-
New disposal cell	-	-	-	51,400	387,400	40,300	479,100
Totals	171,000	387,000	40,000	51,000	387,000	40,000	479,000

Component	Clean Materials (m^3)				
	Clay/Soil	Sand/Rock Gravel	Riprap	Backfill	Topsoil
New disposal cell[f]	51,500	-	-	-	-
Multilayered cover[f]	192,400	35,300	105,800	-	52,900
Leachate system[f]	-	12,600	-	-	-
Area restoration (Weldon Spring)	-	-	-	317,200[g]	22,000[g]

[a] Adapted from Bechtel National (1984a--Table 6-5, 1985a--Table 2-6). A hyphen (-) means zero quantity. Component values are rounded to the nearest 100 m^3. Totals are rounded to the nearest 1,000 m^3.

[b] Volume is reduced by 70% to account for water removal by heat drying.

[c] Value represents the amount of contaminated clay to be removed from the bottoms and sides of the pits (Table E.2).

[d] Assumes that the total volume of quarry wastes is 72,600 m^3 and that two-thirds of the 46,000 m^3 of wastes on the quarry floor is rubble (Bechtel Natl. 1984a, 1985b; Hickey 1986).

[e] Sources: Boerner (1986); Deming (1986).

[f] It is assumed that the amounts of materials needed to construct the new disposal cell, the multilayered cover, and the leachate monitoring system are obtained by multiplying the corresponding material needs in Alternative 2a by the ratio of waste areas (6.61 ha/8 ha).

[g] Material brought in from off-site.

Table E.6. Summary of Material Quantities before and after Completion of the Actions in Alternative 3c[a]

Location	Wastes (m^3), Before Action			Wastes (m^3), After Action			
	Sludge	Soil/Clay	Rubble	Sludge[b]	Soil/Clay	Rubble	Total
Pit 1	13,300	4,500[c]	-	-	-	-	-
Pit 2	13,300	4,500[c]	-	-	-	-	-
Pit 3	99,100	32,500[c]	-	-	-	-	-
Pit 4	42,500	-	400	-	330,700	40,300	371,000[d]
Quarry[e]	3,100	38,800	30,700	-	-	-	-
Chemical plant	-	229,400	9,200	-	-	-	-
Vicinity properties	-	21,000[f]	-	-	-	-	-
Uranium processing site	-	-	-	51,400	-	-	51,400
Totals	171,000	331,000	40,000	51,000	331,000	40,000	422,000

Component	Clean Materials (m^3)				
	Sand/Gravel Rock	Clay	Riprap	Backfill	Topsoil
Multilayered cover[g]	26,700	142,000	76,100	-	40,400
Area restoration	-	-	-	232,300[h]	9,900[i]

[a] Adapted from Tables E.1 and E.4. A hyphen (-) means zero quantity. Component values are rounded to the nearest 100 m^3. Totals are rounded to the nearest 1,000 m^3.

[b] Volume is reduced by 70% to account for water removal by heat drying.

[c] Value represents the amount of contaminated clay to be removed from the bottoms and sides of the pits (Table E.2).

[d] The amount of wastes is greater than the pit capacity (339,800 m^3). The excess amount of wastes can be stored in the pit by either increasing the dike heights, the cover slope, or both.

[e] Assumes that the total volume of quarry wastes is 72,600 m^3 and that two-thirds of the 46,000 m^3 of wastes on the quarry floor is rubble (Bechtel Natl. 1984a, 1985b; Hickey 1986).

[f] Sources: Boerner (1986); Deming (1986).

[g] Amounts of materials are obtained by multiplying the amounts given for Alternative 1 (Table E.1) by the ratio of the area of Pit 4 (6.07 ha) to the total pit area (10.3 ha) (Bechtel Natl. 1985a--Table 2-2). It excludes the amount needed, if any, to restore the vicinity properties. Some of this volume may be obtained from clean material salvaged from the raffinate pits dikes.

[h] Value equals the sum of the amount needed for the quarry area (135,300 m^3) and the amount needed for the raffinate pits area (97,000 m^3) (Bechtel Natl. 1985a--Table 2-10).

[i] See Bechtel National (1985a--Table 2-10).

ditch from the site to the river might accommodate a discharge pipeline or direct stream flow. Because the ditch is contaminated (Appendix H, Section H.1.4), it would probably have to be cleaned before it could be used for direct stream flow.

E.2.1.2 Irrigation

Disposal of the water by spray irrigation has also been proposed. The water would be used for spray irrigation after treatment to reduce the concentrations of contaminants to regulatory limits. In this method, water from the holding ponds would be pumped to a nearby federally owned area on the U.S. Army Reserve Property where it would be sprayed over a proposed irrigation area of about 45 ha (110 acres). The size of the area is such that surface runoff and water logging of the soil would not be problems, and accumulation of contaminants in the soil as a result of irrigation would not reach levels requiring remedial action (Bechtel Natl. 1984a). Spray irrigation is addressed in greater detail in Chapter 4, Section 4.1.3.1

E.2.1.3 Evaporation

Another method of water disposal is evaporation, which can be carried out by use of an oil- or gas-fired evaporator. The pits would have to be covered to prevent additional precipitation from entering. The concentrated evaporator bottoms could be disposed of along with the raffinate sludge. To meet scheduling requirements, the evaporator would have to evaporate 22,700 L/h (6,000 gal/h) (Bechtel Natl. 1984a--Appendix D). The total amount of energy required to evaporate 314,000 m^3 (83,000,000 gal) of water would be about 200,000 MWh (7×10^{11} Btu). The total cost of this amount of energy supplied as electric energy by Union Electric Company (St. Louis, Missouri) would be about $8,000,000 (Bremer 1986).

E.2.1.4 Evaluation of Water Disposal Options

The evaporation option will not be considered further because of the large amount of energy required and the difficulties involved with repeated placement and removal of covers over the raffinate pits area during the action period.

The river discharge and irrigation options may be subject to different regulations and contaminant limits. Discharge to the river may require compliance with the Missouri drinking water limits for chemical species (Appendix H, Table H.12) (Bechtel Natl. 1983, 1984a). Also, the DOE maximum radionuclide concentration guidelines for water would have to be satisfied -- i.e., 100 pCi/L for radium-226 and radium-228, 50 pCi/L for thorium-232, 500 pCi/L for uranium-234, and 600 pCi/L for uranium-238 for uncontrolled

areas (U.S. Dept. Energy 1986b) -- and a state discharge permit would be required. The Missouri limits for irrigation water (Table H.12) and the DOE radiological guidelines for water would apply to spray irrigation. A state discharge permit may not be required. However, concentrations of chemicals in runoff cannot exceed drinking water standards without a discharge permit (Bechtel Natl. 1983).

The concentrations of various species in the raffinate pits and quarry ponds are presented in Appendix H (Tables H.3, H.4, H.11, and H.14). Comparison of these concentrations to the applicable criteria indicates that, in order to satisfy the Missouri limits for drinking water and the DOE limits for uranium and radium, water discharged to the Missouri River would first have to be treated to remove uranium, radium, nitrate, fluoride, and arsenic. This water could also be blended with other less contaminated water, thus reducing the concentrations of these chemical species to allowable levels. If necessary, there would be additional water treatment for removal of other chemicals.

For disposal by spray irrigation, the water would have to be treated to remove radium (water from all pits) and uranium (water from Pit 4 and the quarry pond). The only nonradiological species that would exceed the irrigation limits is arsenic in Pit 3 water and the quarry pond (Tables H.11 and H.14). If necessary, additional water treatment for removal of other chemicals would be carried out.

E.2.2 Water Treatment Techniques

E.2.2.1 Removal of Radium

Radium must be removed from the water in the pits and quarry in order to comply with standards, regardless of which water disposal method is used.

Removal by Precipitation. Radium can be removed from the water by addition of barium chloride and possibly some additional sulfate ion to precipitate mixed barium and radium sulfates. This procedure would require a large vessel to permit adequate time for the precipitate to coagulate and be separated from the liquid. However, precipitation might be accomplished in the pits themselves, and the barium and radium sulfate precipitates could be combined with the sludges (Bechtel Natl. 1984a--Appendix D).

Removal by Ion Exchange. An ion-exchange resin can be used to remove radium and other ions in the water. Frequent regeneration or resin replacement would be required because the other ions would load the resin. The regeneration liquids (if regeneration were done) or discarded resins would

have to be disposed of appropriately because they would be contaminated (Bechtel Natl. 1984a--Appendix D).

Use of a Complexing Resin. Radium can be removed by use of a complexing resin, such as that developed by Dow Chemical. This proprietary material removes only radium from the water, and 0.028 m^3 (1 ft^3) of resin would be sufficient to remove all the radium in the raffinate pits water (Bechtel Natl. 1984a--Appendix D). Because the resin is selective for radium only, regeneration is not necessary (Rozelle and Ma 1983).

Evaluation of Radium Removal Techniques. The precipitation method has the advantage that it is a simple and well-established technology. However, large amounts of barium chloride would be required. Also, a large vessel to carry out the precipitation is required and the process may be slow (to get the required amount of scavenging). Tests are needed to see how well the method would work if applied directly to the water in the raffinate pits.

The ion-exchange method has the disadvantage of requiring relatively large amounts of regeneration or resin replacement. A complexing resin is likely to be the method of choice because the resin would not require replacement or regeneration. The method appears to be selective even in the presence of other ions, and only a small volume of resin would have to be discarded at the end of the treatment.

E.2.2.2 Removal of Uranium

The water in Pit 4 and the quarry pond must be treated to remove uranium, regardless of which disposal option is chosen.

Removal by Precipitation. Alum or iron coagulants can be used to remove the uranium. Removal is most efficient at a pH of 10, with 10 mg/L of coagulant yielding 80% uranium removal. At pH 6, a higher coagulant dose (> 25 mg/L) is required to remove uranium (Hathaway 1983). Another method involves acidifying the water to pH 2 by addition of sulfuric acid, followed by addition of lime to bring the water to pH 10 to precipitate out the uranium (Bechtel Natl. 1984a--Appendix D). It is possible that the method using alum or iron coagulants could be carried out for the water in the raffinate pits.

Removal by Ion Exchange. Uranium dissolved in water can be removed with either an anionic or cationic exchange resin. With a cationic resin, 70% of the uranium is removed; with an anionic exchange resin, 99% of the original uranium is removed (Hathaway 1983). For purifying the raffinate pits water, the anionic exchange resin, DOWEX 21K, is preferred. The resin bed can be regenerated by eluting with salt or sodium hydroxide to yield the sodium form of the resin. The capacity of the resin for uranium removal is roughly

59 kg/m^3 (3.7 lb/ft^3), depending on the concentrations of competing contaminants and the level of regeneration. The spent regenerating liquid can be treated with peroxide to precipitate the uranium. If the resin is not regenerated, the amount of resin required would be about 21 m^3 (27 yd^3) (Bechtel Natl. 1984a--Appendix D).

Evaluation of Uranium Removal Techniques. The precipitation techniques have the advantage that other heavy metal contaminants may be removed along with the uranium. They have the disadvantage that two steps are required -- mixing in the precipitant, followed by waiting for the precipitate to settle out. Large vessels may be required and, in the case of the acid lime method, large amounts of sulfuric acid and lime would be required. Also, the acid lime method could not be carried out in the pits because the raffinate sludge would interact strongly with sulfuric acid and it would generate appreciable amounts of added contaminated waste.

The ion-exchange method has the disadvantage that regenerating solutions must be handled if regeneration is carried out. Advantages include the following: a minimum of equipment is required, operation and control of the process is simple, and more uranium can be removed than by the precipitation technique. The ion-exchange method is the preferred method (Bechtel Natl. 1984a--Appendix D).

E.2.2.3 Removal of Nonradioactive Contaminants

In order to satisfy relevant regulatory limits, removal of nonradioactive contaminants -- such as nitrate, fluoride, and arsenic -- may have to be considered for some of the water disposal options.

Nitrate Removal. Nitrates can be removed by digestion with denitrifying bacteria or by use of anaerobic-activated sludge. These methods involve mixing the bacteria or active sludge into the water, adding methanol as food, and letting the mixture stand in a large basin. After a sufficient amount of the nitrate has been removed by conversion to ammonia and water, gases are removed from the water, which then goes through a clarifier to separate the liquid from the sludge (Bechtel Natl. 1984a--Appendix D).

An advantage of these methods is that no materials are added that require further treatment for their removal. Also, preliminary experiments show that radium and other metals are efficiently removed from the water by the bacteria (Taylor 1980). Disadvantages include the fact that digestion to remove the nitrates is slow (Taylor 1980) and requires large-volume basins; continuous 24-hour, 7-day/week monitoring is needed; and methanol is relatively expensive (Bechtel Natl. 1984a--Appendix D).

Fluoride Removal. Several methods are available for removal of fluoride from water. The main process in use involves precipitation of calcium fluoride by addition of lime followed by removal of the precipitate in a clarifier. This process could not be used here because it removes fluoride only to a concentration of 10 mg/L, which is still above the state of Missouri limit for discharge to the Missouri River (Appendix H, Table H.12). (No fluoride limit exists for water disposal by irrigation.)

Other methods involve the use of activated alumina, bone char, and Fluo-Karb -- a proprietary material manufactured by the Permutit Company. All these methods produce water of the required quality to pass the state criteria for discharge to the Missouri River. Alumina is the material of choice because of its high exchange capacity (Bechtel Natl. 1984a--Appendix D).

Arsenic Removal. Arsenic can be removed by passing the water through a bed of activated alumina. Removal by this method is better than is possible using alum or lime coagulation; concentrations of 0.01 mg/L or less can be maintained. Activated carbon can also be used to remove arsenic; however, this method requires that the water be acidic (Culp and Culp 1974).

The preferred method is the use of a bed of activated alumina because it is simple and achieves the required final concentrations of 0.01 mg/L or less. Use of active carbon to remove arsenic requires the prior use of acid to acidify the water. This requires an extra process step and additional materials (acid) to remove the arsenic.

E.3 WASTE REMOVAL TECHNIQUES

Sludge, soil, and/or rubble must be removed from the quarry, pits, and vicinity properties in all action alternatives. Two possible methods are hydraulic dredging and earth-moving techniques.

E.3.1 Hydraulic Dredging

Hydraulic dredging is a method applicable to removal of sludge or sediment containing 10-30% solids; thus it is applicable to the Weldon Spring sludge, which contains about 27% solids. Various types of equipment are available that use a centrifugal pump in combination with some type of cutting or auger head. Removal rates depend on the type of equipment used and the length and diameter of the discharge pipe; they range from 38 to 190 m^3/h (50 to 250 yd^3/h) (U.S. Environ. Prot. Agency 1982). If necessary, additional water can be added to the sludge to make the consistency more appropriate for pumping. Alternatively, the contaminated water already present in the pits can be used.

E.3.2 Earth-Moving Techniques

Many methods are available for conventional earth-moving excavation -- including use of the backhoe, dragline, front-end loader, and clamshell bucket. This equipment is suitable for excavation of the solid soils, clay, and rubble waste. It can also be used for the sludge if the sludge is dewatered or partly stabilized (Bechtel Natl. 1984a--Appendix D). Typical capacities of this type of equipment range from 15 to 120 m^3/h (20 to 160 yd^3/h) (Means Co. 1984).

E.3.3 Evaluation of Removal Options

For removal of the soils, clay, and rubble, use of conventional earth-moving techniques are appropriate because the wastes are already solid. For sludge removal, the hydraulic dredging method is advantageous because it is a well-established technology that is efficient and can remove materials over a wide range of consistencies from free-flowing liquids to consolidated sludges. Disadvantages to the hydraulic dredging method include (1) the large volume of contaminated solid/water mixtures that would have to be handled and (2) the fact that large items such as drums cannot be removed (U.S. Environ. Prot. Agency 1982).

The use of conventional earth-moving methods for sludge removal has the disadvantage that prior dewatering of the sludge to obtain sufficiently solid material may be required. The main advantage is that because these methods will be used for removal of the clay, soil, and rubble that constitute the majority of the waste volume, use of the same methods for removal of the sludge may be less costly and more efficient.

E.4 SLUDGE MODIFICATION

E.4.1 Options for Dewatering the Sludge Prior to Transport

E.4.1.1 Heat Drying

Heat drying would reduce the sludge volume by 70%. The heat drying process would operate continuously 24 hours/day for each 270-day construction season and would be sized to produce 47 m^3/working day (61 yd^3/working day) or 12,700 m^3/year (16,600 yd^3/year) of dried sludge. Drying would be done by use of a heat transfer medium to transfer heat from the oil-fired heating furnace to the sludge drier. Fuel oil consumption would be about 15,000 L/d (4,000 gal/day). An additional 300 kW of electric power would be needed to operate the process. One conceptual design would require 4 years for drying and would consume 16,200 m^3 (102,000 bbl) of fuel oil (Bechtel Natl. 1984a--Appendix C).

Disadvantages of heat drying include the fact that the process is complex and costly, in terms of both equipment and energy requirements, and care must be taken to avoid release of dried sludge dust into the atmosphere. Drying the sludge also increases the rate of radon gas release.

E.4.1.2 Mechanical Dewatering

Several mechanical methods of dewatering materials are available, including centrifugation, belt filter pressing, rotary vacuum filtration, and frame pressing. These methods were not considered to be viable options here because they are designed to dewater sludge with more than 30 wt. % solids whereas the Weldon Spring sludge has only about 27 wt. % solids (Appendix H, Table H.1) (Bechtel Natl. 1984a--Appendix C). Also, these methods may require multiple stages of treatment, and this could result in higher costs.

E.4.2 Options for Modifying the Sludge Form

E.4.2.1 Matrix Techniques

Matrix techniques involve mixing the wastes with some matrix material that stabilizes or solidifies them. Materials that have been used include cement with or without additives, fly ash, bitumen, and polymers (Holcomb 1978; U.S. Environ. Prot. Agency 1982).

Cement. Portland cement, with or without additives, could be mixed with the sludge. Vermiculite and sodium silicate are often used as additives to improve the properties of the final product. Sodium silicate also greatly reduces the setting time (to as low as 10 seconds), and the final volume of the product is less than that obtained without use of sodium silicate (Holcomb 1978; Bechtel Natl. 1982--Appendix D). Advantages of methods using cement include the fact that mixing processes are simple and no wastewater is generated. Also, cement is stable and has good shielding properties. Disadvantages include the fact that the weight and volume of the product are almost twice the original values. In addition, cement is leachable and does not suppress radon gas emissions. The method is costly because of the large amounts of cement needed (Holcomb 1978; Bechtel Natl. 1982--Appendix D; U.S. Dept. Energy 1986a--Appendix C).

Fly Ash. One method of stabilizing the sludge is to mix in fly ash and cement. The resulting mixture should be easily compactible with standard earth-moving equipment and should be stable against further consolidation or subsidence. Tests carried out with surrogate sludge and samples of the Weldon Spring sludge show that addition of a mixture of 20% portland cement and 80% Type F fly ash at a rate of 1 kg (2.2 lb) for each 1.2 L (0.33 gal) of sludge is sufficient to stabilize the sludge in a satisfactory manner. The volume of

stabilized sludge is larger by 37% than that of the original sludge. One conceptual design would stabilize the sludge at a rate of about 335 m^3 (440 yd^3) of sludge per 8-hour day for 14 days/month, 9 months/year, for a total of 4 years (Bechtel Natl. 1984a--Appendices C and H). Stabilization with fly ash and cement would improve the engineering properties of the sludge. It would not significantly affect radon gas emissions. The stabilization formulas given here are conceptual; in the detailed engineering stage, DOE will take account of the chemical and physical properties of the sludge, the additives, and the stabilized product to arrive at a specific stabilization formula.

Bitumen or Asphalt. With techniques using bitumen as the matrix, the sludge would be mixed with emulsified asphalt or molten-base asphalt and the mixture heated to evaporate the water. Various methods of mixing the sludge with the asphalt and heating the mixture could be used (Holcomb 1978). A cold-mix process using a cationic asphalt emulsion and a flocculant that requires no heat to expel the water has been proposed (Bechtel Natl. 1982). Methods using bitumen have the advantages that bitumen has good coating properties, is relatively cheap and easily available, and is quite inert. Also, the leach rate of the product can be 100 to 1,000 times less than that of cement-stabilized products. Disadvantages include the fact that some of the mixing processes are complex and require strict temperature control. Also, accidental fires when the bitumen is heated must be considered. Some processes generate wastewater that also must be treated and discarded (Holcomb 1978--Appendix D; Bechtel Natl. 1982--Appendix D; U.S. Dept. Energy 1986a--Appendix C).

Resin Polymers. Another process consists of mixing wastes with a resin and a catalyst. The solidification process could take place at either ambient temperatures or elevated temperatures (about 60°C [140°F]). The resulting solid would not really solidify the wastes; instead, the long-chain molecules would form a matrix with voids that would hold particles of wastes (Holcomb 1978; Bechtel Natl. 1982--Appendix D; U.S. Dept. Energy 1986a--Appendix C).

E.4.2.2 Vitrification

Vitrification refers to methods of treatment in which wastes are partially or completely melted into a vitreous glasslike or slag material. These methods are all costly because energy requirements are considerable. The main advantage is that the resulting product is easy to contain and has a much reduced leach rate and radon gas emission rate compared with the rates for untreated materials. Some studies with soils and uranium mill tailings have shown that radon gas emissions can be reduced by a factor of up to 200 and that the leach resistance of vitrified soils is comparable to that for

Pyrex glass (Dreesen et al. 1981; Browns and Timmerman 1982). Application of these methods to the Weldon Spring sludge would probably require prior drying of the sludge.

Electric Furnace Fusion. One method of vitrifying wastes is the use of an electric furnace fusion technique. In this method, the material is resistance-heated by passing electricity through the wastes. Off-gases, which may contain radioactive and other contaminants, might have to be treated before venting to the atmosphere. The molten product is poured into molds and, after cooling, the cooled product is broken into slabs and then buried. Electric power requirements are about 270 kWh/metric ton (245 kWh/ton) of solid wastes (Bechtel Natl. 1984b--Appendix B). Fusion of the dried raffinate and quarry sludge would require about 17,000 MWh of electric energy. Even more energy must be expended to dry the sludge. The electric furnace fusion option has the advantage that it is cleaner than other methods and the resulting slag product should be much reduced in volume and easier to handle. Disadvantages include the fact that large amounts of electric energy are required.

Rotary Kiln. Another method of vitrification uses a coal-fired rotary cement kiln to sinter solid wastes. The resulting clinkers would be cooled and then buried. Pollution-control equipment to control particle, sulfur, and other emissions would be installed (Thode and Dreesen 1981). Use of the coal-fired cement kiln option has the advantage that the equipment is widely available. Also, coal is available in the required amounts and is relatively cheap compared to electric energy. The main disadvantage is that coal combustion generates large amounts of air pollutants that must be controlled.

E.4.2.3 Evaluation of Options for Sludge Form Modification

Vitrification methods have the advantage that the resulting product is reduced in volume and weight and the leach rate and radon gas emission rate are much lower than the original product. Also, the product is quite stable and easy to handle. Disadvantages include the facts that the wastes must be heat-dried prior to vitrification and, in the case of electric furnace fusion, electric energy requirements are considerable. Also, although the components needed for the rotary kiln drying method -- kiln, coal, etc. -- are widely available, the method is not yet in wide use for radioactive wastes.

The matrix isolation methods have the advantage that the technology is well developed. Also, the sludge would generally not have to be dried prior to treatment. Disadvantages include the increase in weight and volume resulting from the matrix material. Additionally, some of the processes generate additional wastewater that must be treated and released.

E.4.3 In-Situ Treatment of Sludge

E.4.3.1 Sludge Dewatering

The methods presented in this EIS for treating the raffinate and quarry sludge currently stored in the pits are based on the assumption that the quarry sludge will be placed in one of the raffinate pits prior to treatment.

Soil Preloading. The soil preloading technique involves placing a layer of soil on the wastes to compact them and squeeze out the water. An advantage of this method is that it is simple and relatively cheap. However, a major disadvantage in applying this technique to the sludge, which is fine-grained, is that compaction and settling would be quite slow and would need to continue for a very long time. Addition of the soil overburden, if it is possible at all, would have to be done very carefully to prevent displacement of the sludge as a wave of mud ahead of the soil (Bechtel Natl. 1984a--Appendix C).

Vacuum Dewatering. The vacuum dewatering technique requires the use of a vacuum system along with vertical wick drains placed in the wastes to remove water and compact them. For the Weldon Spring sludge, a cover could be placed over the sludge consisting of -- in ascending order -- sand, geotextile, fine sand, coarse sand, an impermeable liner of geomembrane, and finally a layer of soil or clay. Slotted piping would be placed in the coarse sand layer and then connected to vacuum pumps. The vertical wick drains would extend from the lower sand layer down through the raffinate sludge (Bechtel Natl. 1984a--Appendix C). When the drains, pipes, and cover are in place, the vacuum pumps would be started. The system would be operated until little or no water was being removed from the raffinate sludge. The rate of settling and water removal would depend on the permeability of the sludge, wick spacing, and effectiveness of the vacuum seal. When settling was essentially complete, the final cover would be put in place.

Electro-Osmosis. The electro-osmosis technique involves placing a line of metal-cased wells in the wastes; these wells serve as negative electrodes. A line of metal rods away from the well line serves as a line of anodes. When a current is passed between the electrodes, water moves from the region between the line of anodes and line of wells into the wells where it is removed by pumping (Bechtel Natl. 1984a--Appendix C).

Evaluation of Sludge Dewatering Options. The options discussed for in-situ dewatering of the sludge have some advantages. The main one is that, at the conclusion of the process, the dewatered sludge would be in place and need not be further treated. Also, because there would be no excavation and emplacement of the sludge, exposure to contaminants during the action period

would be reduced. However, the physical properties of the Weldon Spring sludge are such that the in-situ dewatering options do not appear feasible. Soil preloading would have to be done very carefully and, for the reasons mentioned above, it is not clear if it would work at all. The permeability of the raffinate sludge is so low that, at best, vacuum dewatering would take a long time, from 3 to 7 years. This would impose difficult scheduling requirements. Also, the pits would have to be covered during this time. Electro-osmosis does not appear to be useful because of the large energy requirements and limited water removal ability (Bechtel Natl. 1984a--Appendix C).

E.4.3.2 Vitrification

In-situ vitrification involves placing electrodes in the wastes and passing a sufficient electric current between the electrodes so that resistance heating is sufficient to fuse or vitrify the materials. The process should be carried out under a hood so that off-gases (principally water vapor) can be filtered before venting to the atmosphere. Pilot-scale field tests, carried out on soil at the Hanford site, have demonstrated the potential feasibility of this method. Electric energy requirements are considerable -- about 2.6 MWh/m^3 (2.0 MWh/yd^3) of soil vitrified. The overall cost is estimated to be about \$420/m^3 (Browns and Timmerman 1982).

Application of this method at the Weldon Spring site would probably require building a dike in Pit 4 and consolidating at least the Pit 4 sludge and quarry sludge in one section of the pit. Use of in-situ vitrification of the sludge is estimated to require about 120,000 MWh of electric energy (100,000 MWh to remove all the water contained in the sludge and 17,000 MWh to vitrify the sludge [Section E.4.2.2]; this assumes that all of the free surface water is removed before vitrification is started). Large amounts of off-gases would have to be treated before being released.

In-situ vitrification is not appropriate for use at Weldon Spring because it involves use of a new technology that has never been applied to such large amounts of material. Supplying the large amount of electric energy that is required would also be a severe problem.

E.5 DISPOSAL OPTIONS

Several possible disposal options have already been considered in the various alternatives, including partially below-grade pits or containment structures, an above-grade clay containment structure with a lead sheet in the cover, and unlined trenches (at the Hanford site). Several other disposal options could be used relative to design modification options and other disposal methods.

E.5.1 Design Modification

Modification of the design concept includes such things as changes in the sizes and shapes of the disposal cells, changes in the slopes of the walls or sides, and changes in the liner and cover materials. Examples of this are containment of the wastes in below-grade concrete bunkers or trenches at the raffinate pits area. Other cover designs include layers of concrete, asphalt, or stainless steel (Gilbert/Commonwealth 1980; Avant 1984). Other options include storage in above-ground buildings that are compartmented into many smaller cells and use of additional underground barriers such as grout curtains and slurry walls to further isolate the wastes.

E.5.2 Other Disposal Methods

Other disposal methods for the wastes can be considered. One is burial of the wastes in trenches at intermediate depths. That is, trenches are dug sufficiently deep that an engineered cover, 10 to 20 m (11 to 22 yd) thick, can be used. If the wastes are below the water table, the sides and bottoms of the trenches have to be lined with low-permeability clay (Gilbert/Commonwealth 1980).

Another method consists of drilling large-diameter holes into appropriate geological media, lining the sides of the holes if necessary, and sealing the holes after the wastes are emplaced. The holes can be from 10 to 100 m (11 to 110 yd) or more deep (Gilbert/Commonwealth 1980). Hole diameters can be 2-3 m (2-3 yd) in diameter. One design for a hole 10 m (11 yd) in depth has a gravel bottom, and the wastes are stored in fiberglass liners. Pumped grout fills the space between the liners and the wall of the hole. Another design for a 36-m-deep (39 yd) hole uses a 2-m (2-yd) steel casing pipe to isolate the wastes (Mezga 1984).

Other disposal options include placing the wastes in underground cavities, either man-made (abandoned mines, etc.) or natural (limestone solution caves), and ocean dumping. Disposal in natural cavities requires assurance that the wastes can be isolated from the groundwater. Ocean dumping at a site off the coast of New Jersey was analyzed in detail as an alternative for disposal of the Niagara Falls wastes (U.S. Dept. Energy 1986a).

Other long-term, more exotic disposal options have also been considered. Seabed disposal is a method whereby wastes are buried in sediments or rocks below the ocean floor. Techniques include trenching, drilling holes, and projectile emplacement. Another method considers the disposal of packaged wastes on or within ice sheets in Greenland or Antarctica. Emplacing wastes in the subduction zone (where an oceanic plate descends under a continental plate) by deep drilling has also been considered. The moving oceanic plate

would carry the material down into the earth. Injection by deep drilling of low-level wastes into the molten magma under the earth's crust is another possibility. In this case, known shallow magma chambers would be the repositories. Extraterrestrial disposal has also been noted -- e.g., putting the wastes either in earth orbit, in solar orbit, or into the sun, or ejecting them from the solar system. None of these exotic disposal options have been given serious consideration because they are unproven technologies and would be very expensive to implement.

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APPENDIX F. TRANSPORTATION OF THE WELDON SPRING RADIOACTIVE WASTES

Under Alternative 3a, 3b, or 3c, all or portions of the radioactive wastes from the Weldon Spring site will be shipped to other sites (Table F.1). Under Alternatives 3a and 3b, 480,000 m^3 (630,000 yd^3) of contaminated materials will be transported to another site for long-term management; the site for Alternative 3a is the Hanford site, and the site for Alternative 3b is a "Nearby Site" within a 160-km (100-mi) radius of the Weldon Spring site. Under Alternative 3c, 51,000 m^3 (67,000 yd^3) of the materials from the raffinate pits and quarry will be transported to a uranium processing site in the southwestern United States; the remainder of the wastes will be stabilized in the raffinate pits area at the Weldon Spring site.

Detailed descriptions of the Weldon Spring wastes are given in Appendix H of this EIS. For transportation purposes, the dried raffinate pits and quarry sludge is classified as low-specific-activity (LSA) radioactive waste.* Most of the remaining wastes (raffinate pits soil and clay; quarry rubble, soil, and clay; chemical plant soil, clay, and rubble; and vicinity properties soil and clay) are less contaminated and are not classified as radioactive for transportation purposes. However, it is possible that when more complete measurements are made, it will be determined that certain additional amounts of these wastes must be classified as radioactive for transportation purposes. These wastes and the wastes already classified as LSA wastes will be packaged and transported in compliance with all applicable requirements for shipment of radioactive materials given in DOE Order 1540.1 (Materials Transportation and Traffic Management) and all applicable U.S. Department of Transportation (DOT) regulations given in 49 CFR Parts 100-199.

The three potential modes of transport for the radioactive wastes are truck, rail, and barge. Barge transport has been shown to be unreasonable (Bechtel Natl. 1984) -- primarily because both river barges and ocean barges would be required, necessitating additional transfers, and much greater distances would be involved. Both rail and truck are feasible and, for purposes of defining the major alternatives in this EIS, rail transport is assumed for Alternatives 3a and 3c and truck transport for Alternative 3b. For the longer distances involved in Alternatives 3a and 3c, rail would likely

*Low-specific-activity (LSA) radioactive materials are radioactive materials that present a relatively low hazard because of their low concentration of radioactive substances. Examples of LSA radioactive materials include uranium or thorium ores and physical or chemical concentrates of these ores, unirradiated natural or depleted uranium, and unirradiated natural thorium. Regulations governing the transportation of LSA radioactive materials are given in 49 CFR Part 173.425.

Table F.I. Summary of Alternatives Involving Transport of the Weldon Spring Wastes to Other Sites[a]

Alternative, Destination	Distance (km)	Source of Materials	Volume of Materials[b,c] (m^3)		Density of Materials[c] (MT/m^3)		Weight of Materials[c,d] (MT)	
			Radioactive	Non-radioactive	Radioactive	Non-radioactive	Radioactive	Non-radioactive
3a, Hanford site	3,500	Raffinate pits	50,000[e]	98,000	1.2[f]	1.8[g]	62,000	180,000
		Quarry	930	70,000	1.2[h]	2.4[i]	1,100	170,000
		Chemical plant	0	240,000	NA	1.9[j]	0	450,000
		Vicinity properties	0	21,000	NA	1.8[g]	0	39,000
3b, "Nearby Site"	160	Raffinate pits	50,000[e]	98,000	1.2[f]	1.8[g]	62,000	180,000
		Quarry	930	70,000	1.2[h]	2.4[i]	1,100	170,000
		Chemical plant	0	240,000	NA	1.9[j]	0	450,000
		Vicinity properties	0	21,000	NA	1.8[g]	0	39,000
3c, Uranium processing site	1,900	Raffinate pits	50,000[e]	0	1.2[f]	NA	62,000	0
		Quarry	930	0	1.2[h]	NA	1,100	0

[a] All values rounded to two significant figures. NA means not applicable.

[b] Volumes obtained from data in Tables E.4, E.5, and E.6.

[c] Materials specified as radioactive or nonradioactive according to U.S. Department of Transportation regulations. Radioactive materials for transportation purposes only are those materials that have an activity concentration in excess of 2,000 pCi/g.

[d] Because of roundoff errors, weights calculated from volumes and densities and then rounded off may differ from weights calculated from rounded-off data in this table.

[e] Based on 168,000 m^3 of raffinate sludge reduced to 50,000 m^3 by heat drying (Bechtel Natl. 1984).

[f] Calculated by assuming that the dried sludge contains 15% (7,500 m^3) water by volume (Bechtel Natl. 1984) and that the sludge contains 54,500 MT solids (Appendix H, Table H.1).

[g] Density assumed to be the same as that of the soils/clay in the quarry wastes (Appendix I, Section I.4.2).

[h] Density assumed to be the same as that for the dried raffinate pits sludge.

[i] Average density of quarry wastes (Appendix I, Table I.10).

[j] Calculated by assuming that 229,000 m^3 of soils/clay and 9,200 m^3 of rubble (assumed to be 50% concrete and 50% steel) have the same densities as the corresponding components in the quarry waste as given by Bechtel National (1984--Appendix F).

Conversion Factors: To convert cubic meters (m^3) to cubic yards (yd^3), multiply by 1.308; to convert kilograms per cubic meter (kg/m^3) to pounds per cubic yard (lb/yd^3), multiply by 1.685; to convert metric tons (MT) to tons, multiply by 1.102.

be most economical; for the shorter distance in Alternative 3b, truck would likely be most economical.

The Missouri-Kansas-Texas (MKT) Railroad filed a petition with the Interstate Commerce Commission on August 21, 1986, requesting permission to abandon the 320 km (200 mi) rail line from Machens to Sedalia, Missouri. If MKT is allowed to abandon this rail line, the availability of a local rail line to either the Burlington Northern Railroad (for Alternative 3a) or the Santa Fe Railroad (for Alternative 3c) could be compromised. However, DOE could utilize the abandoned rail line after determining its acceptability and performing any required upgrading and maintenance. If Alternative 3 is selected, DOE will reevaluate the various transportation alternatives to ensure that the most expedient and safe means are used.

In this EIS, both rail and truck transport are evaluated for Alternatives 3a, 3b, and 3c to provide perspective on time and equipment requirements and on alternate transport routes. However, environmental impacts are evaluated only for the mode of transport assumed for each specific alternative. Alternate routes are analyzed only for Alternatives 3a and 3c because there is no defined termination point for Alternative 3b. The environmental impacts associated with transporting nonradioactive fill materials are also addressed for the action alternatives (i.e., all alternatives except Alternative 4).

F.1 TRANSPORTATION REGULATIONS

F.1.1 Federal

Transportation of radioactive materials is subject to the regulations and jurisdiction of many federal, state, and local authorities. Four federal agencies have jurisdiction over shipments of radioactive materials:

- The Department of Transportation (DOT) has primary responsibility for issuing regulations for the safe transportation of all hazardous materials, including radioactive materials. The DOT regulations apply to all shippers and carriers of radioactive materials except for shipments made in federal government vehicles.

- The Nuclear Regulatory Commission (NRC) issues additional regulations for certain highly radioactive materials such as spent nuclear fuel and high-level wastes. Those regulations apply to all NRC licensees. The NRC regulations also apply in certain cases to commercial carriers.

- The Interstate Commerce Commission (ICC) regulates rates, charges, and conditions of truck and rail services operating in interstate commerce. The role of the ICC in interstate commerce is diminishing as a result of gradual deregulation of the transportation industry.

- The Department of Energy (DOE) exerts operational control of the shipment activities of its government-owned, contractor-operated installations. Except for shipments made on government-owned vehicles, all DOE shipments are subject to DOT regulations. DOE's own internal directives provide that the additional safety standards imposed by NRC also apply to DOE shipments, although the administrative requirements of NRC do not apply.

Providing for adequate control of radiation is a requirement that must be met when transporting the Weldon Spring wastes. Radioactive control limits are met by providing the necessary shielding to reduce external radiation levels to within allowable limits. Because the Weldon Spring radioactive wastes will be shipped in vehicles consigned for exclusive use, the following dose limits specified in 49 CFR Part 173.441 apply:

- 1,000 mrem/h at any point on the accessible external surface of a package (closed transport vehicle),

- 200 mrem/h at any point on the accessible external surface of a package (open transport vehicle),

- 200 mrem/h at any point on the external surface of the vehicle,

- 10 mrem/h at 2 m (6.6 ft) from the sides of the vehicle, and

- 2 mrem/h in any normally occupied position in the vehicle.

In 49 CFR Part 173.403(y), radioactive material, for transportation purposes, is defined to be any material that has a specific activity greater than 0.002 μCi/g (2,000 pCi/g). The dried sludge from the raffinate pits must therefore be classified as radioactive waste because it has a radioactivity concentration in excess of 2,000 pCi/g (see Appendix H, Table H.2). These materials can be transported as LSA radioactive wastes for transportation purposes (the various types of materials classified as LSA are defined in 49 CFR Part 173.403 [n]). Most of the materials associated with decontamination of the quarry, chemical plant, and vicinity properties, as well as the clay from the raffinate pits, are estimated to have specific activities less than 0.002 μCi/g and would therefore not be classified as radioactive for

transportation purposes. The only exception to this is an estimated 930 m^3 (1,200 yd^3) of sludge from the quarry that is projected to be LSA radioactive waste. Any additional materials resulting from decontamination of the Weldon Spring site that have specific activities in excess of 0.002 μCi/g will be transported as radioactive waste.

Another major federal regulation concerning transport of the Weldon Spring wastes by truck is the gross vehicle weight limit of 36,000 kg (80,000 lb) (Public Law 97-424, Highway Improvement Act of 1982). This regulation applies to all states.

F.1.2 State and Local

Several state and local governments have issued regulations and passed statutes that impose restrictions on shipments of radioactive materials. The U.S. Congress has, by statute, given DOT preemptive regulatory authority over state and local jurisdictions in the matter of transportation of radioactive materials. The U.S. Supreme Court has upheld this preemptive authority in a case where the city of New York filed suit against DOT, challenging DOT's regulatory authority (City of New York v. U.S. Department of Transportation, Sert denied 104 S. Ct. 1403 [1984]).

Although state or local regulations regarding the transport of radioactive materials are preempted by federal law (Federal Materials Transportation Act, Section 12, Title I, of Public Law 93-633), a state or local municipality has the option of filing with the Department of Transportation for a nonpreemption determination (i.e., a waiver of preemption). A state or local requirement influencing the transport of radioactive materials will cease to be preempted by federal law if, upon application for the nonpreemption determination, the Secretary of Transportation finds that the state or local ruling (1) provides an equal or greater level of public safety than the Hazardous Materials Transportation Act, or regulations issued thereunder, and (2) does not burden commerce. Preemption determination, therefore, does offer the state or local area recourse in the case of disputes over federal preemption.

F.2 RAIL TRANSPORTATION

For rail transportation, it was assumed that all Weldon Spring radioactive wastes will be packaged to ensure compliance with regulations for transporting radioactive materials. There are a variety of packages that could be used to transport the contaminated materials, including 55-gallon drums and boxes of various sizes. For purposes of analysis in this EIS, it was assumed that large steel boxes will be used. These boxes will be similar to the Mark-III bins used for transport and disposal of low-level radioactive

wastes at Argonne National Laboratory. These bins are sturdy (made of 12-gauge welded steel), water tight (gasketed), certified for transport of LSA materials (meet DOT requirements of "strong and tight" packaging), of suitable size (1.2 m × 1.5 m × 1.8 m [4 ft × 5 ft × 6 ft]), can be filled using conventional construction equipment, and can easily be loaded and unloaded from transport vehicles using cranes. The gross weight limit for each bin is 3,600 kg (8,000 lb). The densities of the Weldon Spring wastes are such that, at the gross weight limit, bins the size of the Mark III bins will not be completely filled. Bin dimensions suitable for the Weldon Spring wastes will be 1.2 m × 1.5 m × 1.5 m (4 ft × 5 ft × 5 ft).

The bins will be transported on flatcars. A standard flatcar has a nominal capacity of 70 MT (77 tons) and bed dimensions of 16 m × 3.2 m (52 ft × 11 ft). The bed is large enough to easily hold 20 bins. However, because weight is the controlling factor, a maximum of 19 bins could be loaded.

The Weldon Spring contaminated materials that are not classified as radioactive for transportation purposes will be transported unpackaged, using gondolas. Each gondola has a capacity of 91 MT (100 tons) and will be equipped with removable fiberglass covers (Bechtel Natl. 1984). These fiberglass covers are assumed to minimize the release of materials to the atmosphere during transport. In addition, a quick-setting laminated coating can be sprayed onto these materials, further reducing airborne emissions.

F.2.1 Time and Equipment Needs

About 10,000 railcar shipments will be required to transport all the contaminated wastes from the Weldon Spring site to the Hanford site or a "Nearby Site". Transport of the dried sludge to a uranium processing facility will require about 1,000 railcar shipments. The equipment and time requirements associated with rail transport of the wastes from Weldon Spring are listed in Table F.2. Assumptions made in calculating the values in Table F.2 included: a weather-restricted construction period such that the wastes are available for transport only 6 months of the year, a one-way travel time to the Hanford site of 5 days for a dedicated train (allowing for delays in switching locomotives and in crew changes), a total one-way travel time to a "Nearby Site" of 2 days for a dedicated train (allowing for delays in switching locomotives), a total one-way travel time to a uranium processing site of 3 days for a dedicated train, and a 10% contingency reserve of extra railcars for out-of-service cars and maintenance. It was assumed that it will take 10 years to ship the wastes for Alternatives 3a and 3b (Bechtel Natl. 1984) and 4 years for Alternative 3c (Bechtel Natl. 1985).

Table F.2. Railcar Requirements for Shipment of the Weldon Spring Wastes in Alternatives 3a, 3b, and 3c[a]

	Alternative 3a Hanford Site[b]		Alternative 3b "Nearby Site"[b]		Alternative 3c Uranium Processing Site[b]	
	Train A (35 cars)	Train B (70 cars)	Train A (31 cars)	Train B (46 cars)	Train A (30 cars)	Train B (60 cars)
Railcar trips[c]						
Flatcars	1,000	1,000	1,000	1,000	1,000	1,000
Gondolas	9,200	9,200	9,200	9,200	-	-
Railcars loaded per work day[d]	12	12	12	12	8	8
Number of trains[e]	4	3	3	3	1	1
Train make-up time (total calendar days)	5	9	4	6	5	10
Train transit time[f] (total days, one way)	5	5	2	2	3	3
Train unload time (total calendar days)	5	9	4	6	4	8
Railcars required[g]						
Flatcars	15	23	10	15	30	60
Gondolas	140	210	92	140	-	-

[a] All values rounded to two significant figures or the nearest integer.

[b] Two train lengths are shown, one short and one long. Determination of which train length would be most economical depends on the actual rail rates for each train size and the capital costs or leasing costs of the railcars.

[c] Assumes 63,100 MT radioactive materials shipped in bins, 19 bins/flatcar each containing 3.3 MT, and 839,000 MT nonradioactive materials shipped in gondolas each containing 91 MT.

[d] Assumes shipping duration of 10 years for the Hanford site and the "Nearby Site", and 4 years for the uranium processing site, with materials available for transportation during 6 months per year with 20 work days per month.

[e] The numbers of cars per train and number of trains for both the Hanford site and the "Nearby Site" are designed to always have cars available for loading. For the uranium processing site, because of the low volume available per day, only one train will be used. When the train is absent, bins will be filled and stored for subsequent loading.

[f] Allows for interline delays and crew changes.

[g] Assumes 10% additional cars available for maintenance and repairs.

Although the MKT Railroad has recently filed a petition with the ICC requesting permission to abandon the 320 km (200 mi) rail line from Machens to Sedalia, Missouri, it is assumed for purposes of impact assessment that the required portion of the current MKT rail line will be available. The abandoned rail spur from the MKT main track to the raffinate pits area and a secondary spur to the quarry area will require reconstruction. Most of the right-of-way is intact. The spur will require clearing and grading; road crossings, trestles, and culverts will require reconstruction; and the track will require refurbishing. About 50% of the existing track can be used (Bechtel Natl. 1984). In addition to the rail spur, additional track near the raffinate pits area will be required for making up trains, decontaminating railcars, and storing idle cars. If Alternative 3 is selected, DOE will reevaluate the various transportation alternatives to ensure that the most expedient and safe means are used.

F.2.2 Alternate Rail Routes

For Alternative 3a (Hanford site), the procedure used for analyzing potential rail routes from the Weldon Spring site to the Hanford site (located near Richland, Washington) was to determine the most cost-effective alternatives and then select the one that has the fewest people living near it. The only originating carrier available at the Weldon Spring site is the MKT Railroad. Railroads serving Richland are the Burlington Northern and the Union Pacific System (which includes the Missouri Pacific). The MKT Railroad can transfer traffic to both the Burlington Northern and the Union Pacific in the St. Louis area. All other routings would involve more than two railroads, causing increased transfer costs, and are not considered feasible.

A comparison of the Burlington Northern and Union Pacific routes is shown in Table F.3. The Burlington Northern route, although slightly longer, travels through areas of lower population density and also has lower levels of other rail traffic; it is, therefore, the preferred route. The preferred Burlington Northern route is shown in Figure F.1; the alternate Union Pacific route is shown in Figure F.2.

For Alternative 3c (uranium processing site), there are no alternative carriers for shipments because the only carrier at the Weldon Spring site is the MKT Railroad and the only terminating carrier serving Grants, New Mexico, is the Santa Fe. Minor variations in the route are possible, but the preferred route shown in Figure F.3 is the most direct. This route has a length of 1,900 km (1,200 mi).

Table F.3. Comparison of Preferred Rail Route and Alternate Rail Route for Alternative 3a (Hanford Site)[a]

Route	Originating Carrier	Terminating Carrier	Route Length (km)	Average Population Density (no./km^2)	Average Annual Traffic Volume (MT/yr)
Preferred	MKT	BN	3,500	10	18,000,000
Alternate	MKT	MP/UP	3,300	20	30,000,000

[a] Acronyms: MKT, Missouri-Kansas-Texas; BN, Burlington Northern; MP, Missouri Pacific; UP, Union Pacific. All values rounded to two significant figures.

F.3 TRUCK TRANSPORTATION

Truck transportation is also a feasible alternative for transportation of the Weldon Spring wastes to another long-term management site. It is assumed that, as for rail transportation, all radioactive materials will be packaged in bins (see Section F.2). These bins will be transported on flatbed semitrailers, which are limited by weight restrictions to a payload of 22,000 kg (48,000 lb). Each bin has a gross weight limit of 3,600 kg (8,000 lb), and the densities of the Weldon Spring wastes are such that the weight limits for the bins will be reached. Therefore, each trailer can carry six bins, each containing 3,300 kg (7,300 lb) of contaminated material.

The Weldon Spring contaminated materials that are not classified as radioactive for transportation purposes will be transported in bulk, using 15-m^3 (20-yd^3) dump truck, semitrailer/tractor combinations. The semitrailers will have gasketed seals on the rear and sturdy weatherproof covers on top to minimize the amount of airborne particulates released during transport. In addition, a quick-setting laminated coating could be sprayed onto these materials to further reduce airborne releases.

F.3.1 Time and Equipment Requirements

About 42,000 truck trips will be required to transport all the contaminated wastes from the Weldon Spring site to the Hanford site or a "Nearby Site". To transport only dried raffinate sludge to a uranium processing site will require about 3,200 truck trips. The equipment and time requirements associated with transport of the wastes from the Weldon Spring site are listed in Table F.4. Assumptions made in calculating these values include a weather-restricted construction period such that the Weldon Spring

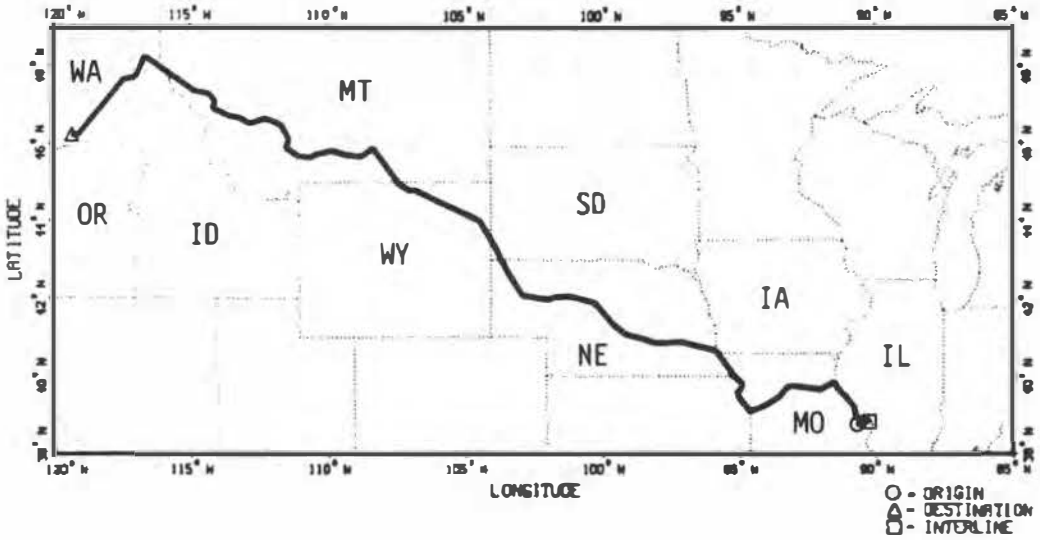

Figure F.1. Preferred Rail Route to the Hanford Site.

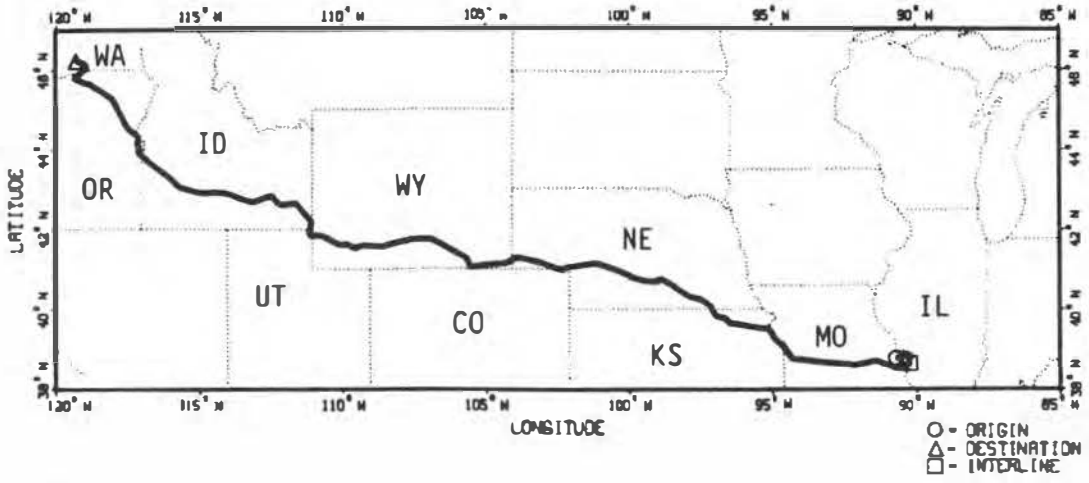

Figure F.2. Alternate Rail Route to the Hanford Site.

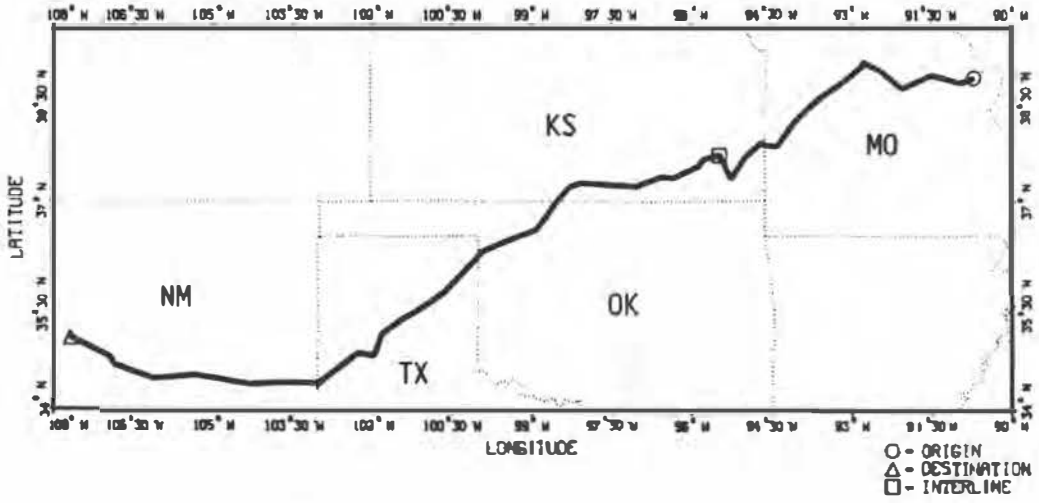

Figure F.3. Preferred Rail Route to the Uranium Processing Site.

wastes will be transported during only 6 months of the year, a one-way travel time to the Hanford site of 6 days, a one-way travel time to the "Nearby Site" of 1 day, a one-way travel to the uranium processing site of 4 days, and a 10% contingency fleet of extra trucks for out-of-service trucks and maintenance. It is assumed that it will take 10 years to complete shipment for Alternative 3a or 3b (Bechtel Natl. 1984) and 4 years for Alternative 3c (Bechtel Natl. 1985).

Currently, access to the raffinate pits area from Missouri Route 94 is not adequate for heavy truck traffic. A two-lane gravel access road approximately 300 m (1,000 ft) long from loading areas on the site to Route 94 will have to be constructed. The two-lane Army access road to Route 94 will require upgrading of 1.6 km (1 mi) of its length. Line-of-sight distances at the intersection of Route 94 and the Army access road will have to be improved, and Route 94 will have to be upgraded to U.S. Route 40/61 (3.2 km [2 mi]) (Bechtel Natl. 1984).

F.3.2 Truck Routes

The routing criteria used here for truck transport are those established by the U.S. Department of Transportation for "preferred routes" for shipment of radioactive wastes, even though the activity levels of much of the Weldon Spring wastes are low enough that preferred routes may not be legally required. Preferred routes are confined to the interstate highway system

Table F.4. Truck Requirements for Shipment of the Weldon Spring Wastes in Alternatives 3a, 3b, and 3c[a]

	Alt. 3a Hanford Site	Alt. 3b "Nearby Site"	Alt. 3c Uranium Processing Site
Truck trips[b]			
Flatbed	3,200	3,200	3,200
Dump	38,000	38,000	0
Trucks loaded per work day[c]	35	35	7
Load time (days)	1	1	1
One-way transit time (days)	6	1	4
Unload time (days)	1	1	1
Trucks required[d]			
Flatbed	27	8	48
Dump	330	93	0

[a] All values rounded to two significant figures or the nearest integer.

[b] Assumes 63,100 MT radioactive materials shipped in bins, 6 bins per flatbed trailer each containing 3.3 MT, and 839,000 MT nonradioactive materials shipped in dump trucks each containing 21.8 MT for the Hanford site, the "Nearby Site", or the uranium processing site.

[c] Assumes project duration of 10 years for the Hanford site and "Nearby Site" and 4 years for the uranium processing site, with materials available for transportation during 6 months per year, at 20 work days per month.

[d] Assumes 10% additional trucks available for maintenance and repairs.

unless otherwise specified by the states. Following the "preferred route" concept, interstate beltways or state-designated bypass routes will be used around large metropolitan areas where possible. Vehicles using preferred routes may deviate from the interstate system for emergencies, for obtaining fuel and repairs, and for travel to and from off-interstate pickup and delivery sites.

Local access to the interstate system will be via Missouri Route 94 to U.S. 40/61, a distance of 3.2 km (2 mi) and then either west on U.S. 40/61 to Interstate 70, a distance of 18 km (11 mi), or east on U.S. 40/61 to Interstate 270, a distance of 24 km (15 mi).

The route to the Hanford site (Figure F.4) will follow I-70 west to Kansas City; turn north on I-29 through Iowa into South Dakota near Sioux City, Iowa, and continue north to I-90 west of Sioux Falls, South Dakota; west on I-90 across South Dakota, Wyoming, Montana, and Idaho; and into Washington to U.S. 395 near Ritzville. The length of the route on interstate highways from U.S. 40/61 to U.S. 395 is 2,800 km (1,700 mi). After exiting I-90 on U.S. 395, the route will proceed southwest to Pasco. The route will then proceed for the last few kilometers to the Hanford site on I-182 and Washington Route 240. The route from I-90 to Richland is 120 km (80 mi) and from Richland to the disposal site is 70 km (40 mi). The total length of the route from the Weldon Spring site to the Hanford site is 3,000 km (1,900 mi).

The route to the uranium processing site (Figure F.4) will follow I-270 south to I-44 and then proceed west on I-44 across Missouri and Oklahoma to Oklahoma City. Here the route will pick up I-40 and proceed west across Oklahoma, the Texas panhandle, and New Mexico to Grants, New Mexico. The total length of the route from the Weldon Spring site to Grants is 1,800 km (1,100 mi).

F.3.3 Truck Requirements for Fill Materials

It was assumed that all fill materials will be transported to the sites by 15-m^3 (20-yd^3) dump trucks. Assuming that the empty weight of the truck is 15,000 kg (32,000 lb), the payload is 22,000 kg (48,000 lb), and the density of the fill materials is 1,800 kg/m^3 (3,000 lb/yd^3), one truck can carry a maximum weight-limited load of 12 m^3 (16 yd^3) of fill per trip. The fill requirements at the Weldon Spring site for all action alternatives, and the number of truck trips required to transport this fill are presented in Table F.5. An additional 420 truck trips will be required in Alternative 2b for delivery of lead sheet. The vehicle requirements to implement these alternatives are presented in Table F.6. The fill requirements at the Hanford site for Alternative 3a and the "Nearby Site" for Alternative 3b are given in Table F.7. Fill materials were assumed to be obtained within a 32-km (20-mi) radius of each site.

F.4 ENVIRONMENTAL IMPACTS ASSOCIATED WITH TRANSPORTATION

The environmental impacts associated with transportation activities result from both the radiological character of the wastes being transported and the nonradiological aspects of transportation such as injuries from traffic accidents and latent effects from pollutants caused by combustion of diesel fuel. The impacts described in this section include those resulting from both normal transport and accidents.

Figure F.4. Truck Routes to the Hanford Site and the Uranium Processing Site.

Table F.5. Volumes of Fill Materials Obtained from Off-site Sources for the Weldon Spring Site and Associated Transportation Requirements[a]

Alternative	Volumes of Fill Materials Obtained from Off-site Sources (m^3)[b]						No. of Truckloads[c]
	Clay	Sand/ Gravel	Riprap	Backfill	Topsoil	Total	
1	240,000	45,000	130,000	95,000	75,000	590,000	49,000
2a	300,000	58,000	130,000	-	84,000	570,000	48,000
2b	610,000[d]	77,000	190,000	135,000	110,000	1,100,000	92,000
3a	-	-	-	280,000[e]	22,000	300,000	25,000
3b	-	-	-	280,000	22,000	300,000	25,000
3c	140,000	27,000	76,000	210,000[f]	50,000	500,000	42,000

[a] All values rounded to two significant figures.

[b] Based on Tables E.1, E.2, E.3, E.4, E.5, and E.6.

[c] Assumes a weight-restricted truckload of 12 m^3 (16 yd^3).

[d] Assumes that 134,000 m^3 (175,000 yd^3) of clay is available on-site (Bechtel Natl. 1985--Table 2-6).

[e] Includes 135,000 m^3 (177,000 yd^3) required for the quarry area and 144,000 m^3 (188,000 yd^3) required for the raffinate pits area (Bechtel Natl. 1984--Table 6-5).

[f] Assumes 22,700 m^3 (29,700 yd^3) is available on-site (Bechtel Natl. 1985--Table 2-10).

F.4.1 Doses to the General Public Resulting from Normal Transport

The general public exposed to radiation consists of people living or working near the transport route (off-link) and people traveling along the route in either the same or opposite direction (on-link). Several mechanisms would contribute to radiation exposure: (1) direct radiation, (2) dispersion of particulates, and (3) radon-220 and radon-222 gas emissions. The major source of radioactivity in the Weldon Spring wastes is the raffinate sludge currently being stored in the four pits. The concentration of radioactivity in these materials in each of the four pits is given in Appendix H, Table H.2. The average concentrations of the various radionuclides in the raffinate sludge can be obtained using the data given in Table H.2 along with the solids content of the raffinate sludge in each pit given in Table H.1. The total quantity of radioactivity in these materials can be obtained by multiplying these average concentrations by the weight given in Table F.1. For assessment of transportation impacts, the quarry sludge was assumed to be

Table F.6. Truck Requirements for Hauling Fill Materials to the Weldon Spring Site[a]

Alternative[b]	No. of Truckloads[c]	Average No. of Round Trips per Day[d]	No. of Trucks Required[e]	Total Distance Traveled per Year (vehicle-km)
1	49,000	68	23	520,000
2a	48,000	67	23	510,000
2b	92,000	130	44	980,000
3a	25,000	70	24	530,000
3b	25,000	70	24	530,000
3c	42,000	58	20	450,000

[a] All values rounded to two significant figures.

[b] Assumes it will take 6 years of intermittent operations to complete hauling fill for Alternatives 1, 2a, 2b, and 3c, and 3 years of operations to complete hauling fill for Alternatives 3a and 3b.

[c] Assumes a weight-restricted truckload of 12 m^3 (16 yd^3).

[d] Assumes a 6-month (120-day) construction season, 8-hour working day, and 2.5-hour round trip loading/travel time.

[e] Includes 10% excess contingency of trucks to cover repair and maintenance.

Table F.7. Volume of Fill Materials for Alternatives 3a and 3b Obtained from Off-site Locations and Associated Transportation Requirements[a]

Site	Volume of Fill Requirements (m^3)[b]						No. of Truckloads[c]
	Clay	Sand/ Gravel	Riprap	Backfill	Topsoil	Total	
Alt. 3a							
Hanford	-	-	310,000	-	-	310,000	26,000
Alt. 3b							
"Nearby Site"	240,000	48,000	110,000	-	53,000	450,000	38,000

[a] All values rounded to two significant figures.

[b] Based on Tables E.4 and E.5.

[c] Assumes a weight-restricted truckload of 12 m^3 (16 yd^3).

contaminated with radionuclides in the same concentrations as those in the raffinate pits. It was also assumed that the radionuclide concentrations in the clay layers beneath the pits are 1% of those in the sludge currently being stored in the raffinate pits.

The parameters used to estimate doses along transportation routes are presented in the Table F.8. The average population densities for the routes to the Hanford site and the uranium processing site are those associated with the preferred routes to these sites. The average population density for the "Nearby Site" was obtained by assuming that the route will pass through fairly rural counties within a 160-km (100-mi) radius of the Weldon Spring site, based on 1980 census data for these counties (Zielen 1985).

Table F.8. Parameters Used to Estimate Radiation Doses along Transportation Routes[a]

Parameter	Alt. 3a Hanford site	Alt. 3b "Nearby Site"	Alt. 3c Uranium Processing Site
Mode of transport	Train	Truck	Train
Distance (km)	3,500	160	1,900
Population density (no./km^2)	10	34	9.4
Time per shipment (h)	110	4	60
Number of exposed persons along the route per shipment (on-link):			
In same direction	1,000	3,800	350
In opposite direction	1,100	3,800	440
Distance to nearest populated area (m)[b]	30	30	30
Distance to nearest vehicle (m):			
In same direction	10	3	10
In opposite direction	10	10	10

[a] All values given to two significant figures.

[b] The model used to estimate radiation doses assumes that there are no people within 30 m of the route. The populated zone is a strip along the roadway between 30 and 800 m, with a uniform population density (Chen et al. 1981).

Conversion Factors: To convert kilometers (km) to miles (mi), multiply by 0.621; to convert number per square kilometer (no./km^2) to number per square mile (no./mi^2), multiply by 2.59; to convert meters (m) to yards (yd), multiply by 1.094.

For train transport of radioactive materials (i.e., Alternatives 3a and 3c), the effects of both passenger and freight traffic must be considered. Freight trains traveling in the same direction as the waste shipment are not likely to be close enough to the shipment to experience any significant exposure. Therefore, only freight trains traveling in the opposite direction are considered. For shipment to the Hanford site (Alternative 3a), it is estimated that about 22 freight trains -- each with five crew members -- will pass the train carrying the wastes in the opposite direction. The only passenger service along the route is in Washington, along 250 km (160 mi) between Spokane and Pasco. There is one train daily in each direction serving that route. The average number of passengers on this train is estimated to be about 1,000. The distance for the passenger service is so short that it was assumed the waste shipment will pass only one train in each direction. Because the passenger train is expected to travel faster than the waste shipment, it was assumed that the waste shipment will wait on a side spur while the passenger train traveling in the same direction is passing.

For the trip to the uranium processing site near Grants, New Mexico (Alternative 3c), an estimated 17 freight trains -- each with five crew members -- will pass the waste transport in the opposite direction. The only passenger route is in New Mexico, where the Chicago-Los Angeles train uses a 100-km (60-mi) portion of the route between Dailies and Grants. One train travels along this route daily in each direction. The total yearly ridership on the Chicago-Los Angeles route was 260,000 persons in 1984 in both directions, or about 350 persons per train. Because not all of these people will be traveling the entire route, using this number results in a conservative estimate for the population dose. As for Alternative 3a, it was assumed that the waste shipment will pass only one passenger train in each direction.

For truck transport to a "Nearby Site" (Alternative 3b), it was assumed that the average number of passengers per car sharing the transport link is two. The one-way traffic count per hour has been estimated to be 480 cars/hour using data of the U.S. Nuclear Regulatory Commission (1977--Table 4.6). Thus, the total number of cars passing the shipment is four times the hourly count, or 1,900, because the waste shipment is expected to require 4 hours to complete; the number of exposed on-link individuals is twice that, or 3,800 in each direction.*

*The actual number of exposed on-link individuals will be larger for cars traveling in the opposite direction and smaller for cars traveling in the same direction because the truck itself is moving. These two effects will tend to offset each other.

A simplified version of the method of Chen et al. (1981) was used to obtain estimates of direct radiation exposure. It was assumed that in the dose rate expression (Chen et al. 1981--Eq. 5), the product of attenuation and buildup factor is equal to one. The actual value of this product is ≤ 1.0 for distances exceeding about 100 m. A self-shielding factor of 10 was assumed for the wastes, reducing the exposure rate on the outside of the vehicle by a factor of 10 from that which would be obtained if these materials were transparent to radiation. The gamma spectra of the uranium-238 and thorium-232 radioactive decay series were obtained from the isotope library contained in the ORIGEN computer code (Bell 1973).

The doses due to dispersion of particulates and radon gas for the unpackaged wastes were calculated in terms of 100-year environmental dose commitments using a modification of the Uranium Dispersion and Dosimetry (UDAD) code (Momeni et al. 1979; Yuan and Chee 1982). It was assumed that 0.1% of the unpackaged wastes will be dispersed as particulates during transport to the Hanford site. For transport to the "Nearby Site", a release fraction of 0.01% was assumed for the unpackaged wastes due to the shorter transport distance. These release fractions assume that high-integrity covers are utilized on the vehicles transporting the unpackaged wastes, minimizing the amount of material lost during transport. There will be minimal releases of radioactive particulates or gases from the packaged wastes; thus, packaged wastes contribute to exposure only via direct radiation.

The estimated doses to the general public resulting from transportation of the radioactive wastes are given in Table F.9 for all three modes of exposure. The population dose would be highest for shipment of the materials to the Hanford site (210 person-rem) largely because of inhalation of radioactive particulates released during shipment of the unpackaged wastes. The dose would be lower for transport of these wastes to the "Nearby Site" (82 person-rem), primarily because of the shorter transport distance. The dose for shipment of the packaged wastes to the uranium processing site would be the lowest (0.036 person-rem) because the only mode of exposure would be direct radiation.

The maximally exposed individual for Alternatives 3a and 3b was assumed to be an individual who lives 30 m from the transport route. The primary mode of exposure to this individual would be from inhalation of radioactive particulates released during shipment of the unpackaged wastes. The maximally exposed individual for Alternative 3c was assumed to be a workman on a passenger train passing in the same direction for the entire 4-year duration of shipments. The estimated doses to the maximally exposed individuals for these three alternatives are given in Table F.10. The doses are higher for Alternatives 3a and 3b because of the transport of unpackaged wastes.

Table F.9. Estimated Doses to the General Public from Transportation of the Weldon Spring Wastes[a]

Destination	Dose from Direct Radiation (person-rem)	100-Year Environmental Dose Commitment (person-rem)		
		Particulates	Radon Gas	Total[b]
Hanford site:				
Inhalation	-	170	10	-
External radiation	0.30	25	-	-
Ingestion	-	0.20	-	-
Total	0.30	200	10	210[b]
"Nearby Site":				
Inhalation	-	67	2.0	-
External radiation	2.5	9.8	-	-
Ingestion	-	0.080	-	-
Total	2.5	77	2.0	82[b]
Uranium processing site:				
External radiation	0.036	-	-	-
Total	0.036	-	-	0.036[b]

[a] All values rounded to two significant figures.

[b] Total for all three modes of exposure.

Table F.10. Estimated Doses to Hypothetical Maximally Exposed Individuals from Transportation of the Weldon Spring Wastes

Destination	Dose from Direct Radiation (mrem)	100-Year Environmental Dose Commitment (mrem)		
		Particulates	Radon Gas	Total[b]
Hanford site:				
Inhalation	-	0.46	0.00066	-
External radiation	0.026	0.0015	-	-
Ingestion	-	0.085	-	-
Total	0.026	0.55	0.00066	0.58[b]
"Nearby Site":				
Inhalation	-	0.46	0.00066	-
External radiation	0.021	0.0015	-	-
Ingestion	-	0.085	-	-
Total	0.021	0.55	0.00066	0.57[b]
Uranium processing site:				
External radiation	0.011	-	-	-
Total	0.011	-	-	0.011

[a] All values rounded to two significant figures.

[b] Total for all three modes of exposure.

F.4.2 Doses to the General Public Resulting from Accidents

It is possible that an accident could occur during transport of the wastes. The radiological consequences of such an accident are presented in Table F.11. The estimated number of accidents was obtained by multiplying the accident rate by the distance traveled and the number of trips. It was assumed that 5% of the packaged and 50% of the unpackaged materials will be spilled during an accident. The dose to the general public is directly proportional to the amount of material spilled and the population density in the area of the accident. The primary means of exposure would be inhalation of radioactive particulates. The average population densities for the various routes (i.e., 10/km^2 for the Hanford site, 34/km^2 for the "Nearby Site", and 9.4/km^2 for the uranium processing site) were used in this analysis. The estimated doses from accidents for Alternatives 3a and 3b are small fractions of those that would be incurred from normal transport conditions because of the large dose contribution from unpackaged wastes during normal transport. However, the dose from accidents for Alternative 3c would exceed that from normal transport conditions because accidents could result in the release of radioactive particulates to the environment whereas routine transport of the packaged wastes would contribute to the dose only as a result of direct radiation.

F.4.3 Doses to Workers

The principal pathway by which drivers transporting the wastes would be exposed to radiation would be direct external exposure to gamma rays from the wastes. Doses from exposure to contaminated particulates and radon gas would be negligible. To calculate the occupational dose for truck transport, it was assumed that (1) the exposure rate of a worker is the same as the exposure rate at the surface of the waste shipment, i.e., the waste shipment is assumed to be unshielded and no credit is taken for worker distance from the wastes, and (2) the workers are exposed to radiation from the wastes for the entire transport distance. For train transport, it was assumed that the workers come in close proximity to the wastes 10% of the time during transport but are exposed to a very low level of radiation during the remainder of the trip, based on their distance from the wastes (during normal transport conditions). A crew of five per train (Alternatives 3a and 3c) and two per truck (Alternative 3b) were assumed for estimating the occupational dose. Shorter-length trains (see Table F.2) were used for Alternatives 3a and 3c. The cumulative occupational dose was calculated as the product of the number of workers exposed, the average radiation dose rate from the materials being shipped, and the time of transport. The collective occupational doses are estimated to be 2.7, 98, and 1.0 person-rem for Alternatives 3a, 3b, and 3c, respectively.

Table F.11. Radiological Risk from Potential Accidents Associated with Transportation of the Weldon Spring Wastes[a]

Parameter	Alternative 3a Hanford Site		Alternative 3b "Nearby Site"		Alternative 3c Uranium Processing Site
	Packaged Waste	Unpackaged Waste	Packaged Waste	Unpackaged Waste	Packaged Waste
Accidents per vehicle-km[b]	0.93×10^{-6}	0.93×10^{-6}	1.06×10^{-6}	1.06×10^{-6}	0.93×10^{-6}
Distance (km)	3,500	3,500	160	160	1,900
Number of vehicle trips	1,000	9,200	3,200	38,000	1,000
Estimated number of vehicle accidents	3.3	30	0.54	6.4	1.8
Spilled material as fraction of total for estimated number of accidents[c]	1.6×10^{-4}	1.6×10^{-3}	8.5×10^{-6}	8.4×10^{-5}	8.9×10^{-5}
Dose to the general public (person-rem) for one accident[d]	0.49	0.11	0.61	0.10	0.49
Dose to the general public (person-rem) for estimated number of accidents	1.6	3.2	0.33	0.65	0.88

[a] A vehicle is defined as a truck or a railcar. All results are given to two significant figures.

[b] Source: U.S. Nuclear Regulatory Commission (1977).

[c] It is assumed that 5% of the packaged and 50% of the unpackaged material are spilled.

[d] 1% of the spilled material is assumed to be resuspended.

Conversion Factors: To convert accidents per vehicle-km to accidents per vehicle-mi, multiply by 1.6; to convert kilometers (km) to miles (mi), multiply by 0.621.

F.4.4 Nonradiological Impacts

The nonradiological impacts of transporting the wastes are not related to the radioactive nature of the wastes but are those impacts that would occur from the transport of any type of cargo. Such impacts result from vehicular emissions and accidents.

To compare vehicular emissions to current pollution standards, the emissions resulting from the hourly passing of one diesel-powered truck was used to calculate an average air pollution concentration (Table F.12). The results, based on Wolff (1984), indicate that the estimated concentrations of airborne pollutants are several orders of magnitude below current air quality standards. It should be noted that if truck traffic greatly exceeds one vehicle per hour, which will likely happen in the near vicinity of the Weldon Spring site, the concentration of airborne pollutants will increase in direct proportion to the truck traffic. However, the concentration of airborne pollutants is not expected to exceed air quality standards because the estimated concentration from the continual passage of one vehicle per hour, 24 hours per day (Table F.12), is much lower than these standards and efforts will be taken to minimize airborne release of pollutants (i.e., trucks will be maintained in good operating condition utilizing standard air-pollution-control devices and water sprays will be used to minimize fugitive dust generation at the Weldon Spring site).

Table F.12. Estimated Vehicular Pollutant Concentrations Associated with Transportation of the Weldon Spring Wastes

	Concentration[a] ($\mu g/m^3$)	
Pollutant	Truck	EPA Standard
Particulates	0.031	75 (annual geometric mean)
Sulfur oxides	0.012	80 (annual arithmetic mean)
Nitrogen oxides	0.031	100 (annual arithmetic mean)
Hydrocarbons	0.0078	160 (3-hour maximum)
Carbon monoxide	0.052	10,000 (8-hour maximum)

[a] Based on the continual passage of one vehicle per hour, 24 hours per day.

The nonradiological impacts of accidents during transport of these materials are more easily defined in terms of the chance for deaths and injuries. The estimated number of transportation-related injuries and deaths resulting from accidents during transport of the Weldon Spring wastes and fill materials are given in Table F.13. These estimates are based on the rate at which transportation-related injuries and deaths are projected to occur (Wolff 1984) and the total round-trip distance involved (vehicle-km). These results indicate that transportation-related injuries are likely to occur for any of the action alternatives. The likelihood for several transportation-related fatalities exists for Alternative 3a in which the radioactive materials currently being stored at the Weldon Spring site are transported to the Hanford site for long-term management. The other action alternatives are not expected to cause any transportation-related fatalities.

Table F.13. Estimated Number of Transportation-Related Deaths Associated with Transportation of the Weldon Spring Wastes[a]

Alternative/ Type of Material Transported	Total Distance Traveled (vehicle-km)	Number of Injuries[c]	Number of Deaths[d]
Alternative 1			
Contaminated (truck)[b]	1.1×10^5	0.056	0.0033
Fill (truck)	3.1×10^6	1.6	0.093
Total	3.2×10^6	1.6	0.096
Alternative 2a			
Contaminated (truck)[b]	1.1×10^5	0.056	0.0033
Fill (truck)	3.1×10^6	1.6	0.093
Total	3.2×10^6	1.6	0.096
Alternative 2b			
Contaminated (truck)[b]	1.1×10^5	0.056	0.0033
Fill (truck)	5.9×10^6	3.0	0.18
Total	6.0×10^6	3.1	0.18
Alternative 3a			
Contaminated (rail)	7.1×10^7	33	2.4
Fill (truck)	3.3×10^6	1.7	0.099
Total	7.5×10^7	34	2.5

Table F.13. Continued.

Alternative/ Type of Material Transported	Total Distance Traveled (vehicle-km)	Number of Injuries[c]	Number of Deaths[d]
Alternative 3b			
Contaminated (truck)	1.3×10^7	6.6	0.39
Fill (truck)	4.0×10^6	2.0	0.12
Total	1.7×10^7	8.7	0.51
Alternative 3c			
Contaminated (rail)	3.8×10^6	1.7	0.13
Fill (truck)	2.7×10^6	1.4	0.081
Total	6.5×10^6	3.1	0.21
Alternative 4			
None	0	0	0

[a] All values rounded to two significant figures.

[b] Associated with transportation of radioactive materials from the quarry to the raffinate pits area.

[c] Based on a truck-accident injury rate of 5.1×10^{-7}/vehicle-km and a railroad-accident injury rate of 4.6×10^{-7}/vehicle-km (Wolff 1984).

[d] Based on a truck-accident fatality rate of 3.0×10^{-8}/vehicle-km and a railroad-accident fatality rate of 3.4×10^{-8}/vehicle-km (Wolff 1984).

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APPENDIX G. RARE AND ENDANGERED SPECIES OF ST. CHARLES COUNTY, MISSOURI, AND WITHIN A 160-KILOMETER RADIUS OF WELDON SPRING, MISSOURI

Table G.1. Rare and Endangered Species of St. Charles County, Missouri

Species	Status[a]	Habitat
PLANTS		
Lycopodium lucidulum var. *lucidulum* (Shining clubmoss)	R(S)	Upland forest
Botrychium dissectum (Cut-leaved grape fern)	R(S)	Slope forest
Ophioglossum vulgatum var. *pycnostichum* (Adder's tongue)	R(S)	Slope forest
Peltandra virginica (Arrow alum)	R(S)	Bottomland forest
Ulmus americana (American elm)	E(S)	Bottomland forest
Floerkea proserpinacoides (False mermaid)	E(S)	Slope forest
Centaurium texense (Centaury)	R(S)	Slope forest
Lithospermum latifolium (Puccoon)	R(S)	Slope forest
Chelone obliqua var. *speciosa* (Rose turtlehead)	U(S)	Bottomland forest
Xanthium spinosum var. *spinosum* (Spiny cocklebur)	U(S)	Old field
Boltonia asteroides var. *decurrens*	R(S), T(F)[b]	Bottomland forest
ANIMALS		
Citheronia regalis (Regal moth)	U(S)	Slope forest
Potamilus (= *Proptera*) *capax* (Fat pocketbook pearly mussel)	E(S), E(F)	River
Lampsilis higginsi (Higgin's eye pearly mussel)	E(S), E(F)	River
Acipenser fulvescens (Lake sturgeon)	E(S)	River
Scaphirhynchus albus (Pallid sturgeon)	E(S)	River

Table G.1. Continued

Species	Status[a]	Habitat
ANIMALS (Cont.)		
Hybopsis gelida (Sturgeon chub)	R(S)	River
Hybopsis meeki (Sicklefin chub)	R(S)	River
Notropis amnis (Pallid shiner)	PE(S)	River
Cycleptus elongatus (Blue sucker)	R(S)	River
Lota lota (Burbot)	R(S)	River
Hemidactylium scutatum (Four-toed salamander)	R(S)	Bottomland forest
Rana sylvatica (Wood frog)	E(S)	Slope forest
Opheodrys vernalis (Smooth green snake)	R(S)	Old field
Sistrurus catenatus (Massasauga)	R(S)	Bottomland forest, old field
Phalacrocorax auritus (Double-crested cormorant)	R(S)	Bottomland forest
Accipiter striatus (Sharp-shinned hawk)	E(S)	Old field
Accipiter cooperii (Cooper's hawk)	E(S)	Old field
Haliaeetus leucocephalus alascensis (Northern bald eagle)	R(S), E(F)	Bottomland forest
Haliaeetus leucocephalus leucocephalus (Southern bald eagle)	PE(S), E(F)	Bottomland forest
Falco peregrinus (Peregrine falcon)	E(S), E(F)	Upland forest, slope forest, old field
Rallus elegans (King rail)	R(S)	Bottomland forest
Bartramia longicauda (Upland plover)	E(S)	Old field
Sterna albifrons (Least tern)	R(S)	Bottomland forest

Table G.1. Continued

Species	Status[a]	Habitat
ANIMALS (Cont.)		
Tyto alba (Barn owl)	R(S)	Old field
Myotis sodalis (Indiana bat)	R(S), E(F)	Upland forest, slope forest
Myotis keenii (Keen's bat)	R(S)	Upland forest, slope forest
Lutra canadensis (River otter)	E(S)	Bottomland forest

[a] R = rare, T = threatened, E = endangered, PE = probably extirpated, U = status unknown, S = state list, F = federal list.

[b] Under consideration for inclusion on the federal list.

Sources: Missouri Botanical Garden (1975); Nordstrom et al. (1977).

Table G.2. Rare and Endangered Species within the Missouri Sector of a 160-Kilometer Radius of Weldon Spring, Missouri

Species	Status[a]	Habitat
PLANTS		
Lycopodium lucidulum var. *lucidulum* (Shining clubmoss)	R(S)	Shaded crevices and ledges of sandstone bluffs
Botrychium dissectum var. *dissectum* (Cut-leaved grape fern)	R(S)	Rich wooded ravines and narrow wooded valleys
Asplenium montanum (Mountain spleenwort)	R(S)	Ravines
Dryopteris goldiana (Goldie's fern)	R(S)	North-facing sandstone bluffs bluffs
Typha glauca (Cattail)	R(S)	Marshy areas
Tridens oklahomensis	R(S)	Moist soils
Panicum calliphyllum (Panic grass)	R(S)	Oak-hickory and pine-oak woods
Carex conoidea (Field sedge)	E(S)	Upland, undisturbed prairie
Spiranthes ovalis (Ladies' tresses)	R(S)	Low or rich moist woodland
Ulmus americana (American elm)	E(S)	Woods
Sullivantia renifolia	R(S), T(F)	Moist, shaded, north-facing bluffs of limestone or St. Peter sandstone along streams
Gaylussacia baccata (Black huckleberry)	R(S)	Rocky, woody ridges above bluffs
Phlox bifida (Bifid phlox)	R(S)	Upper cherty slopes and rocky woods in shade
Veronica americana (American brooklime)	E(S)	Wet ground
Lonicera dioica var. *dioica* (Limber honeysuckle)	R(S)	Bluffs, rocky streams
Viburnum dentatum var. *deamii* (Southern arrow-wood)	R(S)	Low woods, wooded slopes
Anaphalis margaritacea (Pearly everlasting)	U(S)	Limestone bluffs
Agoseris cuspidata	R(S)	Glades, rocky prairie

Table G.2. Continued

Species	Status[a]	Habitat
ANIMALS		
Macrocotyla glandulosa (Flatworm)	NSI-L(S)	Cave stream
Leptodea leptodon (Scale shell)	E(S), T(F)	River
Acipenser fulvescens (Lake sturgeon)	E(S)	River
Alosa alabamae (Alabama shad)	R(S)	River
Hybopsis gelida (Sturgeon chub)	R(S)	River
Hybopsis meeki (Sicklefin chub)	R(S)	River
Notropis heterolepis (Blacknose shiner)	E(S)	River, stream
Fundulus kansae (Plains killifish)	R(S)	Stream, river
Opheodrys vernalis (Smooth green snake)	R(S)	Prairie, grassland
Sistrurus catenatus (Massasauga)	R(S)	Bottomland
Buteo lineatus (Red-shouldered hawk)	E(S)	Forest
Bonasa umbellus (Ruffed grouse)	R(S)	Forest
Tympanuchus cupido (Greater prairie chicken)	R(S)	Grassland
Phalacrocorax auritus (Double-crested cormorant)	E(S)	Lake, river
Circus cyaneus (Marsh hawk)	E(S)	Tall grass prairie
Accipiter striatus (Sharp-shinned hawk)	E(S)	Forest
Accipiter cooperii (Cooper's hawk)	E(S)	Forest
Rallus elegans (King rail)	R(S)	Marsh
Bartramia longicauda (Upland sandpiper)	R(S)	Grassland

Table G.2. Continued

Species	Status[a]	Habitat
ANIMALS (Contd.)		
Ammodramus henslowii (Henslow's sparrow)	R(S)	Tall grass prairie
Haliaeetus leucocephalus alascensis (Northern bald eagle)	R(S), E(F)	Bottomland forest
Pandion haliaetus (Osprey)	E(S)	Bottomland forest
Falco peregrinus (Peregrine falcon)	E(S,F)	Cliffs, bottomland forest
Myotis sodalis (Indiana bat)	E(S,F)	Caves, forest
Myotis grisescens (Gray bat)	E(S,F)	Caves
Euarctos americanus (Black bear)	R(S)	Forest
Mustela frenata (Long-tailed weasel)	R(S)	Woodland, old field, and fencerows near water

[a] E = endangered, T = threatened, R = rare, U = status unknown, NSI-L = no status indicated, but limited in occurrence and/or distribution, S = state list, F = federal list (some species so listed may be under consideration for listing).

Sources: Data largely from Nordstrom et al. (1977). Supplemental habitat data from Jones (1963).

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APPENDIX H. RADIOLOGICAL AND CHEMICAL CHARACTERIZATION OF THE EXISTING ENVIRONMENT AT WELDON SPRING

H.1 RADIOLOGICAL CHARACTERIZATION

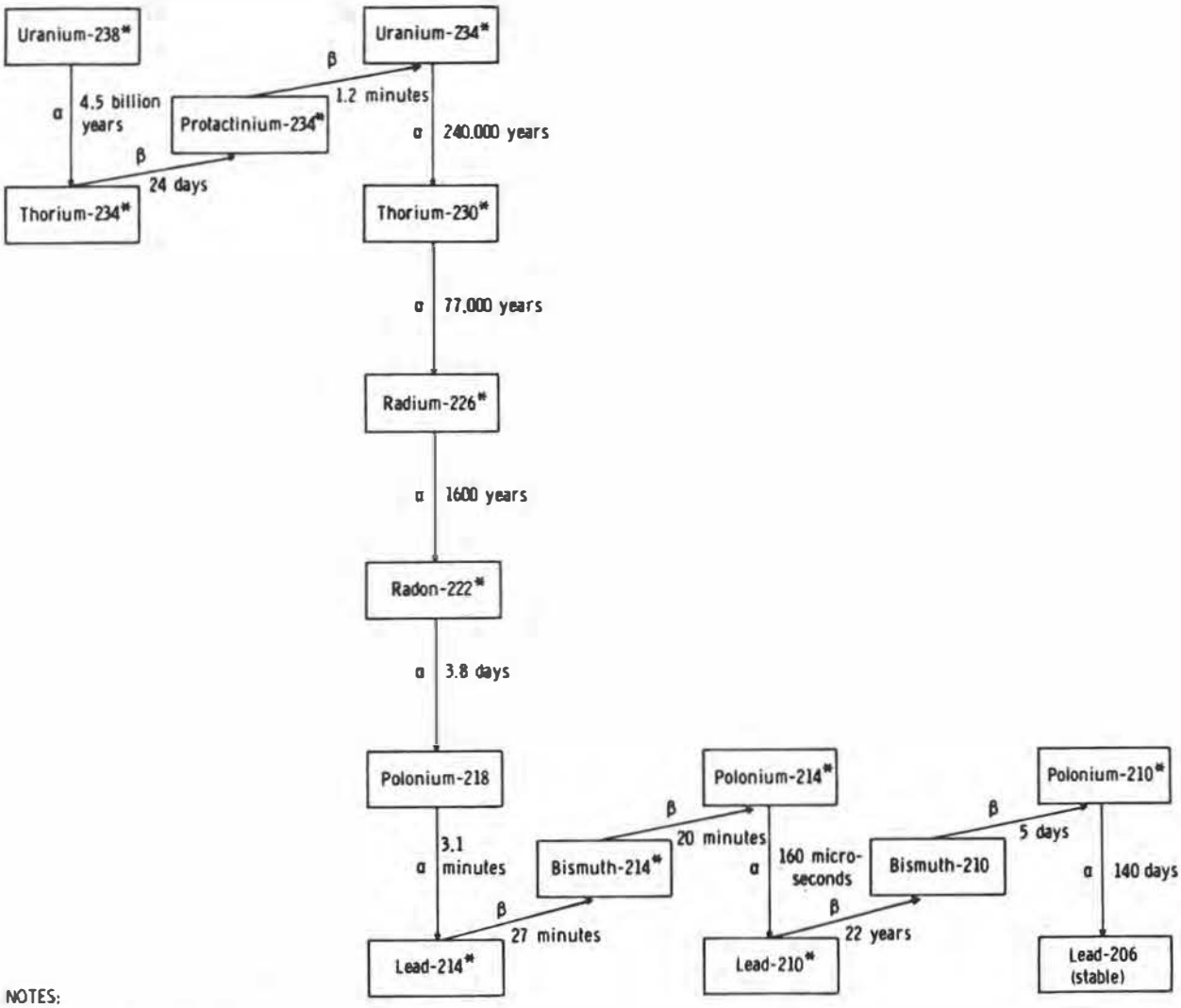
The radioactivity in the Weldon Spring wastes is due to naturally occurring radionuclides, mainly those resulting from the radioactive decay series of uranium-238 and thorium-232. These two decay series, shown in Figures H.1 and H.2, indicate the relationships among the various radionuclides in the wastes. The radioactive wastes at the Weldon Spring site resulted from the processing of uranium and thorium concentrates and are located in four areas: the raffinate pits, quarry, chemical plant, and vicinity properties.

In nature, the radionuclides in a decay series are in a state of secular equilibrium in which the activities of all radionuclides are equal. However, the processing of uranium and thorium ores alters this natural state, and deviation from secular equilibrium is expected. The rate at which secular equilibrium is reestablished depends on the half-lives of the decay products. For analysis of radiological impacts in this EIS, it was assumed that -- because of their short half-lives and the length of time since closure of the chemical plant (about 20 years) -- all radioactive decay products in the thorium-232 decay series are in secular equilibrium with thorium-232. The activities of the radionuclides in the uranium-238 decay series will change with time at a rate that depends on their original activities and half-lives and on the activities of their parent radionuclides (ingrowth effect). The concentrations of many radionuclides in the Weldon Spring wastes will not change significantly through 10,000 years. Nevertheless, the amount of radium-226 and its decay products will increase significantly over time as a result of a gradual reestablishment of secular equilibrium.

H.1.1 Raffinate Pits

Pits 1, 2, and 3 contain raffinate sludge and slag resulting from the refining of uranium ore concentrates and the recycling of scrap metal carried out at the chemical plant. Pit 4 contains similar slag and sludge as well as wastes from the processing of thorium-containing materials and drums and rubble from partial decontamination of the chemical plant (Natl. Lead Co. Ohio 1977; Bechtel Natl. 1984).

During plant operations, raffinate sludge flowed into the pits, and supernatant from the raffinate in Pits 1 through 3 flowed continuously out of the pits into the process sewer down to the Missouri River at a rate of 1,900 m^3/d (5×10^5 gpd) (Task Force 1967). After cessation of plant



NOTES:

Only the dominant decay mode is shown.
The times shown are half-lives.
The symbols α and β indicate alpha and beta decay.
An asterisk indicates that the isotope is also a gamma emitter.

Figure H.1. Uranium-238 Radioactive Decay Series.

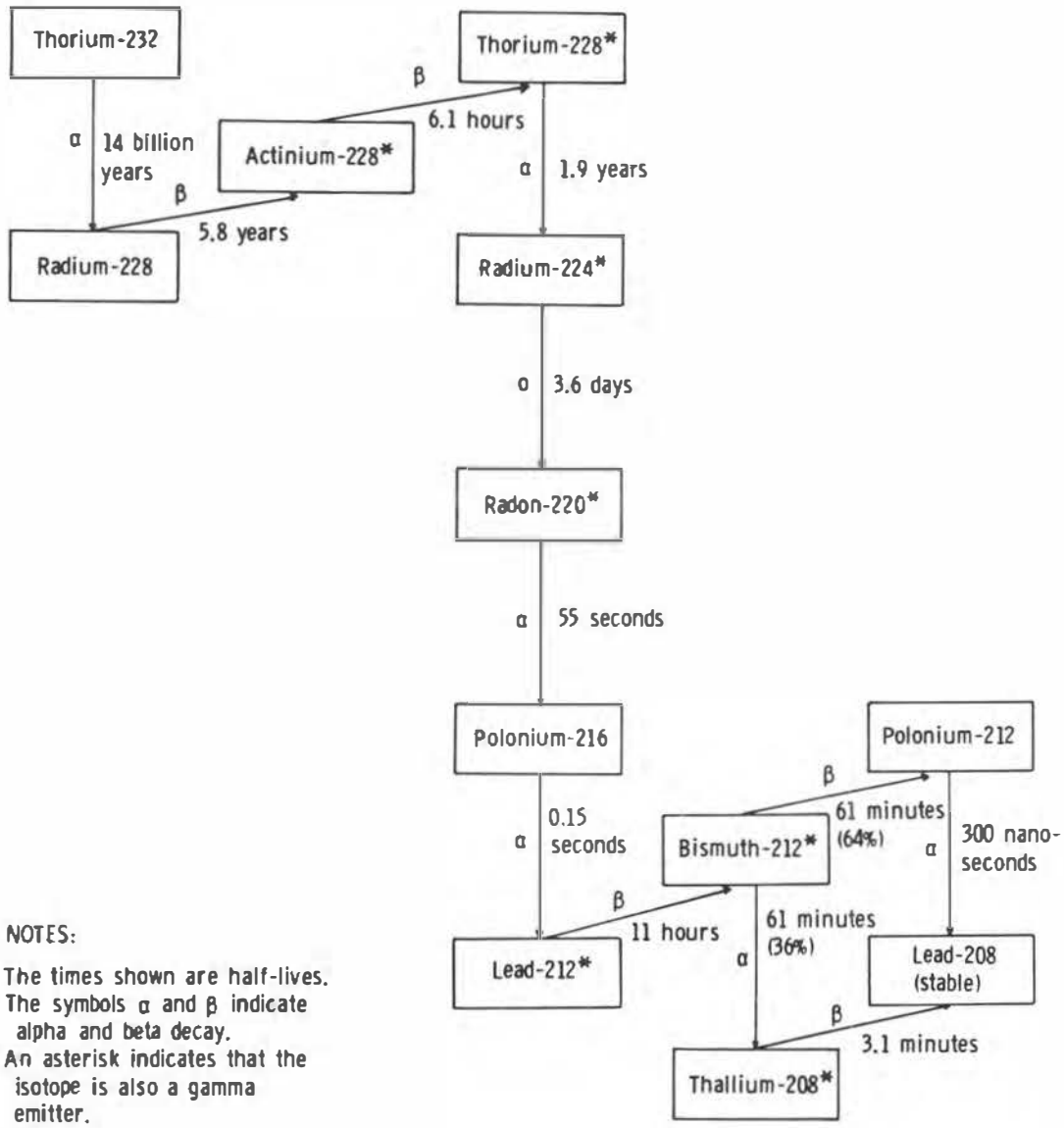


Figure H.2. Thorium-232 Radioactive Decay Series.

operations in December 1966, the flow from Pits 1 through 3 decreased to between 4.2 to 8.6 m^3/d (1,100 to 2,300 gpd) until September 1975 when the small effluent flow from the pits was stopped by plugging the pit outfall weirs (Niedermeyer 1976). At present, all drains from the pits are sealed other than the overflow from Pit 3 to Pit 4 that may occur during periods of heavy rainfall (Bechtel Natl. 1984).

The wastes in the pits are reported to be stratified and heterogeneous (Task Force 1967; Natl. Lead Co. Ohio 1977). The amount of surface water covering the wastes varies during the year. In summer, all surface water may evaporate from Pits 1 and 2 (although according to staff at the site, Pits 1 and 2 have not been dry in the past several years). Surface water is always present in Pits 3 and 4. Pit 3 is designed to overflow into Pit 4 through a pipe in the dike wall common to both pits (Natl. Lead Co. Ohio 1977; Bechtel Natl. 1984). Pit 3 contains most of the wastes. Some physical characteristics of the wastes in the pits are summarized in Table H.1.

Concentrations of various radioisotopes contained in the dried sludge are given in Table H.2. For each pit, the measurements were made on a single, blended, composite sample prepared from samples taken in 1983 at several locations in the pit (Bechtel Natl. 1984). The data indicate that most of the activity is thorium-230 and that the uranium-238 decay series is not in equilibrium. Presumably, this arises from the fact that the plant treated uranium concentrates containing 50 to 75% uranium, which were obtained from the prior processing of ores containing 0.15 to 0.5% uranium (Mallinckrodt Chem. Works 1962). Because the data presented in Table H.2 were used for analysis in this EIS, it is important to evaluate the accuracy of these data.

There are large discrepancies between the data given in Table H.2 and data gathered earlier in 1967 (Task Force 1967). In particular, the concentrations of uranium, thorium, and radium in the 1967 data are higher than the values given in Table H.2 by factors ranging up to 70. These variations may be partially explained by the stratification and heterogeneity of the sludge (Task Force 1967; Natl. Lead Co. Ohio 1977). Consequently, the 1967 data may represent sampled material with different element concentrations (and possibly different isotope distributions for each element) than the 1983 data.

The 1967 data are not believed to be representative because they imply unreasonably high inventories of 430 MT (150 Ci) uranium-238, 100 MT (11 Ci) thorium-232, and 240 Ci radium-226 in the raffinate pits. These values are larger than the reported values of 100 MT (36 Ci) uranium-238 and 65 MT (7 Ci) thorium-232 (Bechtel Natl. 1984) and 137 MT (49 Ci) uranium-238 and 64 MT (7 Ci) thorium-232 (Natl. Lead Co. Ohio 1977). Inventories calculated from the measured values given in Table H.2 are in reasonable agreement with the

Table H.1. Summary of the Physical Characteristics of Sludge in the Raffinate Pits

Pit Number	Construction Date	Pit Volume (m^3)	Percent Filled	Surface Water Volume[a] (m^3)
Pit 1	1958	14,100	94.0	Dry surface
Pit 2	1958	14,100	94.0	Dry surface
Pit 3	1959	127,500	77.8	19,000
Pit 4	1964	339,800	12.5	57,000

	Waste Characteristics			
Pit Number	Volume (m^3)	Weight Percent Solids	Wet Bulk Density (g/cm^3)	Solids Weight (MT)
Pit 1	13,300	27.6	1.191	4,370
Pit 2	13,300	29.4	1.219	4,770
Pit 3	99,200	27.3	1.206	32,660
Pit 4	42,500	25.3	1.184	12,730

[a] Volumes were determined in the summer of 1979 (Taylor et al. 1979). According to another estimate made in 1983, the water volume in the four pits is 216,000 m^3 (Bechtel Natl. 1984). Pits 1 and 2 have not been dry for the past several years.

Sources: Taylor et al. (1979); Bechtel National (1984); Rudolph (1984).

Table H.2. Radioisotope Content of Dried Sludge from the Raffinate Pits

	Concentration (pCi/g-dry)			
Isotope	Pit 1	Pit 2	Pit 3	Pit 4
Radium-226	430 ± 130	440 ± 130	460 ± 130	11 ± 3
Radium-228	850 ± 85	200 ± 20	100 ± 10	60 ± 10
Thorium-232	100 ± 20	120 ± 20	120 ± 20	120 ± 20
Thorium-230	24,000 ± 1,000	24,000 ± 1,000	14,000 ± 1,000	1,600 ± 100
Uranium-238	710 ± 70	470 ± 40	520 ± 50	620 ± 60
Uranium-234	810 ± 80	560 ± 50	570 ± 50	610 ± 60
Uranium-235	40 ± 5	30 ± 4	30 ± 4	30 ± 4

Source: Data from Bechtel National (1984--Appendix A); the ± values indicate measurement accuracy.

inventories given in reports of Bechtel National (1984) and National Lead Company of Ohio (1977). Also, the 1967 inventory of 240 Ci radium-226 is appreciably larger than the total amount of 87 Ci radium-226 assumed to be present in the 122,000 MT (134,000 tons) of uranium contained in ore concentrates received during the lifetime of the chemical plant (Section H.1.3). The 87 Ci radium-226 estimate is based on assuming that the radium-226 activity in the concentrates is 0.2% of the uranium-238 activity (Sears et al. 1975).

The accuracy of the concentration data in Table H.2 is affected by such factors as stratification and heterogeneity of the sludge and the method of sample collection and mixing (Bechtel Natl. 1984). Details of the sampling and mixing process are not available (Lampton 1984).

DOE is continuing to gather additional data characterizing the radiological properties of these wastes (see, for example, Nemec [1986]). These additional data, as well as other relevant data obtained by other sources (see, for example, Kleeschulte et al. [1986]), will be factored into detailed engineering design activities to ensure that the most appropriate design features are incorporated into the disposal cell.

Because of the large concentration of thorium-230 in the sludge relative to the radium-226 concentration (Table H.2), the concentrations of radium-226 and its decay products will increase with time. This is of particular concern when options for long-term management of the wastes are considered. The concentrations of radium-226 will increase with time up to the time of maximum concentration, 9,100 years, after which the radium-226 concentration will decay with the 77,000-year half-life of the thorium-230 parent. At 1,000 years, the radium-226 concentrations will be higher than they are currently by factors of 20 for Pits 1 and 2, 11 for Pit 3, and 50 for Pit 4. The concentration averaged over all raffinate sludge will increase by a factor of 13. At 9,100 years, the average concentration of radium-226 in the raffinate will be 12,000 pCi/g (dry wt.), which is higher than the current concentration by a factor of 34. Concentrations of radium-226 decay products such as radon-222 and lead-210 will also increase by similar factors (assuming no change in the fraction of radon gas that escapes).

Besides sludge, Pits 3 and 4 contain appreciable amounts of free water (Table H.1). The surface water from all four pits has been sampled and analyzed for radioactive species (Table H.3). The data show that concentrations of radionuclides in the surface water sampled at different times can vary widely -- e.g., in Pit 2, the surface water radium-226 concentrations vary by a factor of more than 4,000. Such variation could be caused by differing amounts of precipitation and evaporation in the period

Table H.3. Concentrations of Radioactive Materials in Water Samples from the Raffinate Pits[a]

Pit	Natural Uranium[b] Samples Dates[c]	Natural Uranium[b] Samples Number	Natural Uranium[b] Concentration (pCi/L) Range	Natural Uranium[b] Concentration (pCi/L) Average	Thorium-232[b] Samples Dates[c]	Thorium-232[b] Samples Number	Thorium-232[b] Concentration (pCi/L) Range	Thorium-232[b] Concentration (pCi/L) Average	Thorium-230 Concentration (pCi/L)
1	3/75 to 9/84	4	6 - 22	17	-	-	-	-	8 ± 1
2	3/75 to 9/84	6	16 - 230	70	-	-	-	-	13 ± 7
3	6/74 to 9/84	19	43 - 250	110	8/79	8	<1.1 - 9.7	4.4	3
4	4/67 to 9/84	7	29 - 2,900	1,900	4/67 to 8/79	3	1.6 - <110	16	1.6 ± 0.5

Pit	Radium-226 Samples Dates[c]	Radium-226 Samples Number	Radium-226 Concentration (pCi/L) Range	Radium-226 Concentration (pCi/L) Average	Radium-228 Samples Dates[c]	Radium-228 Samples Number	Radium-228 Concentration (pCi/L) Range	Radium-228 Concentration (pCi/L) Average
1	11/74 to 9/84	6	85 - 1,300	520	-/83	1	3	3
2	3/75 to 9/84	6	<0.4 - 1,600	440	5/78 to -/83	3	0.4 - 15	5.4
3	6/74 to 9/84	9	<0.4 - 330	160	6/74 to -/83	5	<0.4 - 43	25
4	4/67 to 8/84	8	0.9 - 500	77	4/67 to -/83	6	23 - 910	190

[a] Values rounded to two significant figures. A dash means no data available. Where applicable, averages were computed by assuming that upper limits are actual values.

[b] Values given in the literature (Anonymous, undated; Task Force 1967; Taylor et al. 1979; U.S. Geol. Surv. 1984) as mg/L are converted to pCi/L using the conversion factor 1 pCi = 1.38 µg for natural uranium and 1 pCi = 8.8 µg for thorium-232. The uranium data in Bechtel National (1984--Appendix D) are given in units of pCi/L total uranium. The ratios of concentrations of uranium-234, -235, and -238 in the raffinate sludge (Table H.2) are close to the natural uranium ratios. Because these ratios should also apply to uranium isotope concentrations in the water in the raffinate pits, the Bechtel data are reported here in units of pCi/L of natural uranium.

[c] Dates are given as month/year; -/83 means sometime in 1983.

Sources: Anonymous (undated); Task Force (1967); Taylor et al. (1979); Bechtel National (1984--Appendix D); U.S. Geological Survey (1984).

shortly before sampling. For example, samples taken after a heavy rainfall should have low concentrations whereas samples taken after an extended dry period would have high concentrations.

H.1.2 Quarry

The quarry is located in limestone and covers about 3.6 ha (9 acres). The deepest part is filled with water and covers about 0.2 ha (0.5 acres). In 1954, drummed residues containing 3.8% thorium were dumped in the quarry. This material, which is estimated to contain about 4.5 MT (5 tons) of thorium and about 0.5 Ci of radium-228 (equilibrium amount), is probably below the average water level in the quarry (Task Force 1967; Pennak 1975). From 1963 through 1964, about 38,000 m^3 (50,000 yd^3) of uranium- and radium-contaminated rubble from demolition of the Uranium Processing Facilities in St. Louis was added (see Appendix B, Figure B.1). This material is estimated to cover at least 0.4 ha (1 acre) of the quarry floor to a depth of 9 m (30 ft). The total radium-226 contamination is estimated to be less than 10 Ci. About one-third of this material lies below the average water level in the quarry (Task Force 1967; Pennak 1975). In 1966, additional drummed thorium residues containing 3% thorium (or about 12 MT [13 tons] thorium) were dumped in the quarry. This material lies well above the average water level (Task Force 1967; Pennak 1975) and now contains about 1.3 Ci of radium-228. In 1968 and 1969, about 4,200 m^3 (5,600 yd^3) of contaminated process equipment and building rubble from the chemical plant were dumped in the quarry. The radioactivity of this material is not known (Pennak 1975; Berkeley Geosci. Assoc. 1984). Barium sulfate residues from the vicinity of the St. Louis Airport may also have been dumped in the quarry in 1969. Since 1969, no chemically or radiologically contaminated materials have been added to the quarry. However, some of the wastes and rubble are mixed with or covered by rubble contaminated with trinitrotoluene (TNT) resulting from TNT manufacturing operations carried out by the U.S. Army at the former Weldon Spring Ordnance Works (Bechtel Natl. 1983a). TNT-contaminated rubble and soil were dumped in the quarry both prior to the AEC takeover of the quarry in 1958 and in 1965 (Task Force 1967).

Extensive data are available on surface soil concentrations of uranium, radium, and thorium in the quarry; this information is given in reports of Berkeley Geosciences Associates (1984) and Bechtel National (1985). Three sediment samples taken at different locations from the quarry pond had average values of 890 pCi/g uranium-238, 7.0 pCi/g radium-226, 340 pCi/g thorium-230, and 2.3 pCi/g thorium-232 (Bechtel Natl. 1985).

A recent analysis of subsurface samples taken from boreholes drilled into the quarry wastes (Bechtel Natl. 1985) showed large variations in the

concentrations of radioactive species as a function of both depth within a borehole and borehole location. As examples of the variations at different borehole locations, values of uranium-238 concentrations -- averaged over samples taken at different depths within each borehole -- ranged from 4 to 460 pCi/g. Measurements of uranium-234 and uranium-235 concentrations at a few depths and locations had values indicating that these isotopes were present in about their natural activity ratios with uranium-238. Concentrations of thorium-230 and radium-226 ranged from 1.2 to 5,500 pCi/g and from 1.3 to 560 pCi/g, respectively; concentrations of thorium-232 ranged from 1 to 414 pCi/g.

Radon-222 gas concentrations in the atmosphere and in augered holes at several locations in the quarry have also been measured. Atmospheric concentrations at the quarry ranged from 0.8 to 18 pCi/L, averaging about 14 pCi/L. Downhole radon-222 concentrations were more variable, ranging from 120 to 120,000 pCi/L (typical soil gas concentrations are reported to be 200 to 300 pCi/L) (Berkeley Geosci. Assoc. 1984). The values of other measurements at different points along the fence around the quarry in 1985 ranged from an annual average low of 0.2 pCi/L to an annual average high of 1.3 pCi/L. (Each value is an average of values measured at different times at a fixed point along the quarry fence.) The values along the fence are comparable to the background radon-222 gas concentration in this area of 0.5 pCi/L and are below the maximum permissible value of 3 pCi/L for uncontrolled areas (DOE Order 5480.1A--Attachment XI-1) (Bechtel Natl. 1986).

Because of the possible existence of a hydraulic connection between the quarry and adjacent surface water and groundwater, it is necessary to know the concentrations of radioactive species in waters inside and outside of the quarry. Concentrations of uranium, radium, and thorium in water samples taken in the quarry at various dates are given in Table H.4. These data suggest that concentrations of total uranium in the quarry pond increased from 1 pCi/L in 1960 to a peak value of 17,000 pCi/L in 1967 and then decreased to 620 pCi/L in 1985. The buildup from 1960 to 1967 may represent increasing amounts of uranium available for leaching from wastes that were dumped in the quarry during that period. The decline after 1967 may represent a slow leachout and depletion of uranium inventory from the quarry wastes. However, the time period for which data are available is too short to give much support to this possibility. The data also suggest that stratification may occur in the quarry pond; in particular, the high 1967 concentration refers to a sample collected from the bottom of the quarry pond (Task Force 1967) and the much lower 1974 value refers to a sample collected 7.6 cm (3 in.) below the water surface (Pennak 1975). The depths in the pond at which samples were collected are not given in the references for the other data in Table H.4.

Table H.4. Concentrations of Uranium, Radium, and Thorium in Surface Water and Groundwater at the Quarry Area[a]

Sample Location	Sampling Dates	Natural Uranium[b]			Radium			Thorium		
		Number of Samples	Concentration (pCi/L)		Number of Samples	Concentration (pCi/L)		Number of Samples	Concentration (pCi/L)	
			Range	Average		Ra-226	Ra-228		Th-230	Th-232
Surface Water										
Quarry pond	1960-1964	5	1 - 8,400[c]	3,300	-	-	-	-	-	-
	1967[d]	1	-	17,000	1	30	140	1	-	<100
	1974[e]	1	-	1,500	1	1.4	0.45	-	-	-
	1979-1981	6	2,200 - 3,500	2,900	-	-	-	-	-	-
	1984	2	1,200 - 1,500	1,400	2	0.8	<5	1	1.5 ± 0.1	0.5 ± 0.5
	1985	1	-	620	1	<0.2	-	1	<0.3	-
Groundwater[f]										
0.5-ft region	1979-1981	30	<8 - 2,100	430	-	-	-	-	-	-
	1984	2	550 - 2,900	1,700	2	0.25	-	1	-	<0.4
	1985	1	-	190	1	3.5 ± 0.4		1	0.1 ± 0.1	-
7-ft region	1979-1981	6	220 - 8,500	4,800	-	-	-	-	-	-
14-ft region	1979	1	-	2,200	-	-	-	-	-	-
40-ft region	1979	4	2,300 - 8,000	4,800	-	-	-	-	-	-
	1984	2	4,100 - 8,200	6,100	2	0.65	-	2	-	<0.25

Table H.4. Continued

Sample Location	Sampling Dates	Natural Uranium[b]			Radium			Thorium		
		Number of Samples	Concentration (pCi/L)		Number of Samples	Concentration (pCi/L)		Number of Samples	Concentration (pCi/L)	
			Range	Average		Ra-226	Ra-228		Th-230	Th-232
Inside fence at southern edge:										
TW-8 & TW-9	1979-1980	10	4,600 - 8,800	6,700	-	-	-	-	-	-
	1985	2	2,000 - 7,000	4,500	2	0.6	-	2	0.5	-
TW-S	1979-1980	5	<8 - 65	39	-	-	-	-	-	-
	1985	1	-	18	1	0.7 ± 0.1	-	1	0.6 ± 0.5	-

[a] Concentrations are rounded to two significant figures. A dash means either not applicable or no data available.

[b] The uranium data in the references are given in units of mg/L or ppm (Berkeley Geosci. Assoc. 1984; Layne Western 1986; U.S. Geol. Surv. 1984; Task Force 1967); µCi uranium/mL (Task Force 1967); and pCi/L of uranium-234, uranium-235, and uranium-238 (Bechtel Natl. 1985). The data given in units of mg/L or ppm were converted to pCi/L natural uranium using the conversion factor 1 pCi = 1.38 µg. The portion of the Task Force (1967) data given in units of µCi/mL was assumed to refer to total uranium activity. The concentration data of Bechtel National (1985) on the separate uranium isotopes shows that they are present in approximately their natural activity ratios; consequently, total uranium activity is equivalent here to natural uranium activity .

[c] The values are 1 pCi/L (1960); 1,200 pCi/L (1961); 4,700 pCi/L (1962); 8,400 pCi/L (1963); and 2,200 pCi/L (1964).

[d] Sample collected from the bottom of the quarry pond.

[e] Sample collected 7.6 cm (3 in.) from surface.

[f] Regions are those given in Figure 1.6 (Appendix I). The 0.5-ft region includes wells TW-1 through TW-5 and TW-7 (Berkeley Geosci. Assoc. 1984) and two locations sampled by Bechtel National (1985). The 7-ft region contains well TW-6 (Berkeley Geosci. Assoc. 1984). The 14-ft region includes borehole 0-0 (Berkeley Geosci. Assoc. 1984). The 40-ft region includes boreholes 1-5, 2-2, 3-1 and 2-4 (Berkeley Geosci. Assoc. 1984) and two locations sampled by Bechtel National (1985).

Sources: 1960-1967 data, Task Force (1967); 1974 data, Pennak (1975); 1979-1981 data, Berkeley Geosciences Associates (1984); 1984 data, U.S. Geological Survey (1984) and Bechtel National (1985); 1985 data, Layne Western (1986).

The data in Table H.4 indicate that groundwater in the quarry area is also contaminated with uranium. The values, which range up to 8,800 pCi/L, are much higher than the values measured in two boreholes -- one 91 m (300 ft) north and the other 15 m (50 ft) south of the quarry pond -- in each of the five years 1960 through 1964 (Task Force 1967). These 10 values (i.e., one value for each borehole for each of 5 years), which range from 6 to 27 pCi/L and show no evidence of an increase with increasing time, probably represent background values in the immediate quarry area.

Large differences also exist in the groundwater data for uranium concentrations at different nearby holes. For instance, concentrations in TW-8 and TW-9 ranged from 4,600 to 8,800 pCi/L in 1979 to 1980 whereas much lower uranium concentrations of <8 to 65 pCi/L were measured in TW-S located 52 m (170 ft) from TW-9. These differences with hole location may reflect the fact that groundwater movement in the limestone occurs mainly in solution channels and fractures and, thus, different solution channels are sampled by nearby holes.

H.1.3 Chemical Plant

In 1941, the U.S. Army built a TNT manufacturing plant at the Weldon Spring site. The plant occupied about 1,200 ha (3,000 acres) and contained 20 complete production lines for explosives when it closed in 1944. Extensive chemical contamination of the area occupied by the plant is known to have occurred (Niedermeyer 1976; Ryckman & Assoc. 1978; Bechtel Natl. 1984).

In 1955 and 1956, 89 ha (220 acres) of the chemical plant area was transferred to the U.S. Atomic Energy Commission for construction and operation of a plant for processing and sampling uranium and thorium ore concentrates. The transferred area had in the past contained three complete TNT lines and parts of a fourth line (Niedermeyer 1976; Ryckman & Assoc. 1978; Bechtel Natl. 1984).

During its lifetime from June 1957 to plant closure in December 1966, the chemical plant was used primarily to process uranium concentrates in the form of sodium diuranate containing 70% uranium. Small amounts of materials containing depleted and slightly enriched uranium were also processed, and thorium concentrates were processed in 1965 and 1966 (Task Force 1967; Harris 1986). According to a recent materials balance study (Harris 1986), during its lifetime the plant received materials containing a total of 122,000 MT (134,000 tons) of natural uranium, 167 MT (184 tons) of depleted uranium, 842 MT (926 tons) of slightly enriched uranium, and 941 MT (1,040 tons) of natural thorium. The uranium receipts correspond to an annual average of 13,000 MT/yr (14,200 tons/yr) uranium.

Measurements of uranium, radium, and thorium have been made around various pieces of equipment in the chemical plant (e.g., sumps, tanks, furnaces, hoppers). It is reported that about 4 MT (4.4 tons) of recoverable uranium is present at the chemical plant and that radium and thorium contamination was also detected (Ryckman & Assoc. 1978). An estimated 240,000 m^3 (310,000 yd^3) of contaminated soils and rubble, excluding the raffinate pits area, must be removed or decontaminated prior to releasing the area for appropriate use (Rockwell Int. 1979). DOE is planning to undertake more extensive sampling and refinement of this estimate.

The results of radium, uranium, and thorium contamination measurements in soil taken at various locations in the chemical plant area, as well as background values for these radionuclides in this area, are given in Table H.5. The data show the presence of several areas of elevated uranium-238 concentrations, with values ranging from 11 to 140,000 ppm (3.9 to 50,000 pCi/g). Radium-226 concentrations appear to be quite low, 0.7 to 8.3 pCi/g; and thorium-232 concentrations are slightly elevated, 5.4 to 380 ppm (0.6 to 42 pCi/g). Concentrations of uranium-238 and thorium-232 in samples taken in other areas of the plant ranged from 3.7 to 53,900 ppm (1.3 to 19,000 pCi/g) and 3.9 to 178 ppm (0.44 to 20 pCi/g), respectively, and concentrations of radium-226 and radium-228 ranged from 1.1 to 150 pCi/g and 0.6 to 13 pCi/g, respectively (Niedermeyer 1976). For purposes of comparison, the measured background values in soils in the area are < 2.0 to 4.9 ppm (< 0.68 to 1.62 pCi/g) for uranium-238, 8.4 to 13 ppm (0.95 to 1.48 pCi/g) for thorium-232, and 0.55 to 0.98 pCi/g for radium-226 (Boerner 1986). The total volume of soils contaminated with 500 ppm (180 pCi/g) or more of uranium-238 is estimated to be 3,800 m^3 (5,000 yd^3) (Ryckman & Assoc. 1978).

The data in Table H.5 give a limited characterization of radiological contamination at the chemical plant, but they are the best available at this time. Additional characterization activities are scheduled to take place in the near future.

Three surface streams exit the chemical plant area (Figure H.3). One stream flows out of Ash Pond and goes north to Schote Creek, then to Lake 35 in the Busch Wildlife Area, and then into Dardenne Creek; Dardenne Creek empties into the Mississippi River 11 km (7 mi) upstream from St. Louis. Another is an intermittent stream out of Frog Pond that flows north to Lake 36 in the Busch Wildlife Area and then into Schote Creek. A third stream exits the plant area to the south carrying flows from underground drain lines and the plant process sewer drainage, then moves along a drainage ditch to the Missouri River. Prior to 1975, drainage from Pits 1, 2, and 3 entered this stream from the sewers at a rate of 1 to 2 L/min (0.26 to 0.53 gpm) per pit; the drainage from the pits was stopped in 1975 (Niedermeyer 1976).

Table H.5. Concentrations of Uranium, Radium, and Thorium at Various Locations around the Chemical Plant Area[a]

Area	Contaminated Material (m^3)[b,d]	Uranium-238 Concentration (pCi/g) Range	Uranium-238 Concentration (pCi/g) Average	Possible Depths of Contamination (cm)	Number of Sample Locations
Around buildings:	530				
Surface[c]		8.8 - 980	120	Surface	11
Subsurface[d,h]		<18 - 13,000[f]	2,500	20 - 50	6
Ash Pond:					
Sediments[e,h]	760	71 - 140	110	Surface	3
Soil[d,h]	760	<7.9 - 3,400[f]	470	5 - 60	9
Drainageway[e,h]	7.6	14 - 110	72	Surface	7
Frog Pond:					
Sediments[d,h]	160	170 - 320	240	Surface	2
Overflow ditch[e,h]	-	11 - 110	54	Surface	18
Coal Pond soil[d]	7.6	<18 - 47[f]	36	15 - 20	3
North dumps:	1,600				
Surface[e,h]		3.9 - 3,600	360	Surface	66
Subsurface[d,h]		500 - 50,000[f]	14,000	10 - 150	4
Background[i]		<0.68 - 1.6	1.1	-	6
TOTAL	3,800[a]				

Table H.5. Continued

Area	Radium-226			Thorium-232			Thorium-230		
	Number of Samples[g]	Concentration (pCi/g)		Number of Samples[g]	Concentration (pCi/g)		Number of Samples[g]	Concentration (pCi/g)	
		Range	Average		Range	Average		Range	Average
Around buildings:									
Surface[c]	11	0.7 - 5.1	2.2	7	0.6 - 3.6	2.0	11	0.6 - 27	6.0
Ash Pond:									
Sediments[e,h]	3	2.0 - 6	4.7	3	1.7 - 3.7	2.8	-	-	-
Drainageway[e,h]	7	1.2 - 5	2.9	7	0.9 - 1.9	1.7	-	-	-
Frog Pond:									
Sediments[e,h]	16	2.1 - 5.5	3.1	16	2.2 - 42	10	-	-	-
Overflow ditch[e,h]	18	1.6 - 4.0	2.3	18	1.1 - 6	2	-	-	-
North dumps:									
Surface[e,h]	66	0.7 - 8.3	3.8	66	0.8 - 2	1.6	-	-	-
Background[i]	6	0.55 - 0.98	0.79	6	0.95 - 1.5	1.2	-	-	-

[a] Values rounded given to two significant figures. A hyphen means no data available.

[b] Volume of material containing at least 500 ppm (180 pCi/g) uranium-238.

[c] Source: Dickson et al. (1981). Uranium concentrations are given in reference in units of pCi/g uranium-238.

[d] Source: Ryckman & Assoc. (1978). Uranium concentrations are given in reference in units of ppm. Locations of Ash Pond and Frog Pond are shown in Figure H.3. Coal Pond is located to the east of the coal storage area, and the north dumps are located along the western part of the north fence line (see Figure 1.2). These locations are shown explicitly in Figure 4 of the report of Ryckman & Associates (1978).

[e] Sources: EG&G (1977) and Mohr (1985). Uranium concentrations are given in reference in units of net ppm uranium-238 from protactinium-234m activity.

[f] Range is taken over set of maximum values of concentrations, with one maximum value for each borehole location.

[g] Number of samples is also the number of sampling locations.

[h] Uranium and thorium values given in units of ppm are converted to pCi/g by use of the conversion factors 1 pCi = 2.79 µg (uranium-238) and 1 pCi = 8.8 µg (thorium-232).

[i] Source: Boerner (1986).

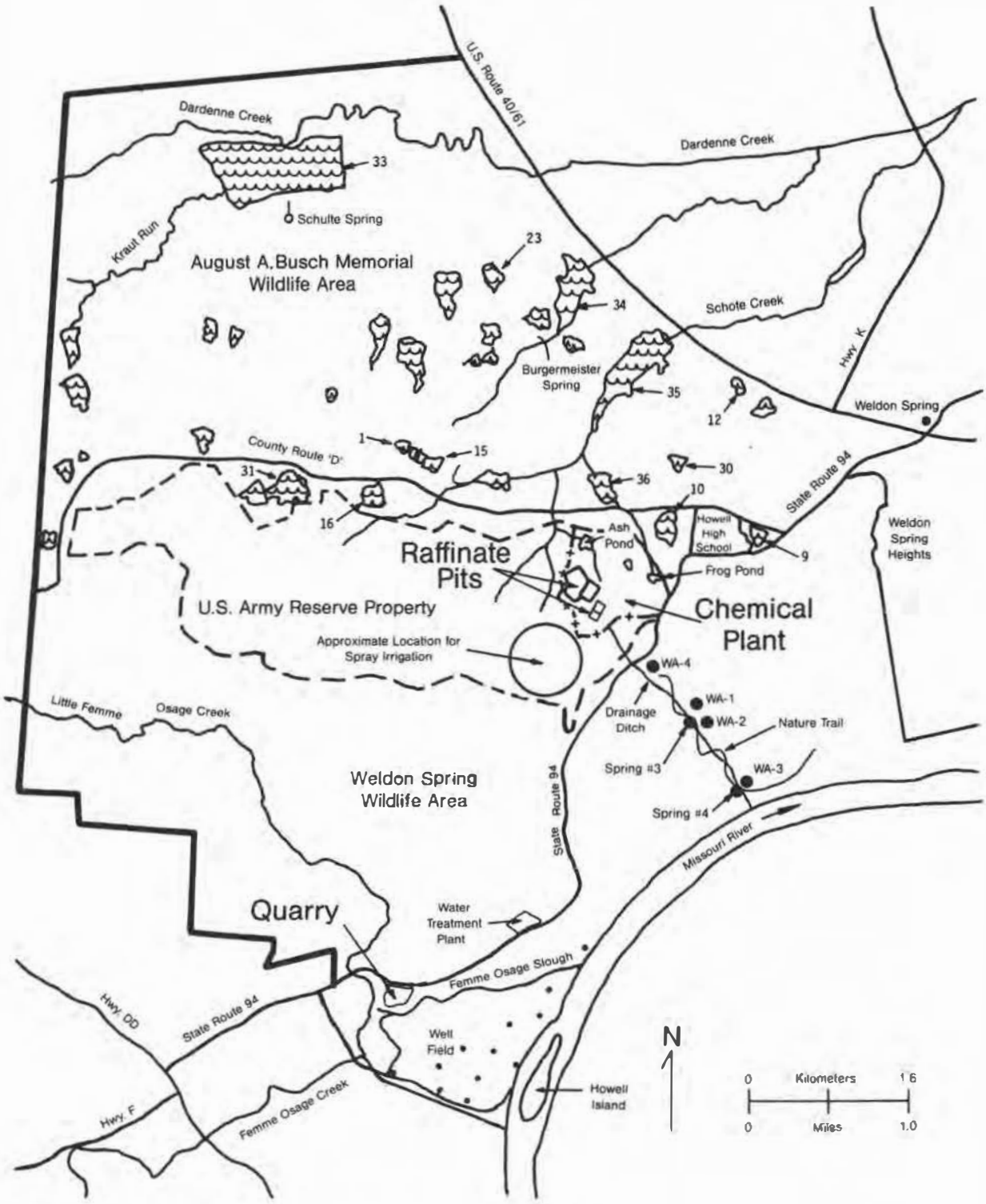


Figure H.3. Map of the Weldon Spring Site and Vicinity. Source: Modified from deRoos (1984) and Stevens (1984).

Concentrations of radioactive species in water samples taken at several locations in the chemical plant area are given in Table H.6. These data show that concentrations of natural uranium and radium-226 in surface waters are above background at several locations, which indicates that some uranium and radium are leaving the plant area by surface water outflow. For example, concentrations in the Ash Pond outflow -- which empties into Schote Creek -- reached values as high as 4,100 pCi/L for natural uranium and 15 pCi/L for radium-226 in 1983 to 1984 (Turnbull 1983-1984; Hansen 1983-1984; Mazur 1983-1984; Smith 1983-1984).

H.1.4 Vicinity Properties

Concentrations of radioactive species in soils and sediments in the vicinity properties are given in Table H.7. Background values for these radionuclides in soil in this area are 0.55 to 0.98 pCi/g for radium-226, < 0.68 to 1.62 pCi/g for uranium-238, and 0.95 to 1.48 pCi/g for thorium-232. For sediments, the background values are 0.35 to 0.92 pCi/g for radium-226, < 0.60 to 1.44 pCi/g for uranium-238, and 0.24 to 1.02 pCi/g for thorium-232 (Boerner 1986).

In sediment samples collected from 34 lakes in the Busch Wildlife Area (see Figure H.3), there was essentially no radium-226 or thorium-232 contamination. The highest value reported for radium-226 was 2.3 ± 0.3 pCi/g, compared to a background range of 0.35 to 0.92 pCi/g); and the highest value reported for thorium-232 was 2.5 ± 0.7 pCi/g, compared to a background range of 0.24 to 1.02 pCi/g. Uranium-238 concentrations slightly higher than background were measured in sediments of Lake 1 (5.96 ± 0.97 pCi/g), Lake 12 (4.66 ± 0.67 pCi/g), Lake 15 (3.65 ± 1.61 pCi/g), Lake 23 (7.00 ± 1.00 pCi/g), Lake 30 (3.12 ± 0.78 pCi/g), and Lake 31 (5.11 ± 0.78 pCi/g) (Boerner 1986). Concentrations of uranium-238 measured in sediments of Lakes 34, 35, and 36 (up to 120 pCi/g) were considerably above background values (Table H.7).

Spotty contamination occurs in the main drainage ditch from the chemical plant along the entire 2.1 km (1.3 mi) length from the U.S. Army Reserve Property fence to the Missouri River (Figure H.3). Concentrations of uranium-238, thorium-232, and radium-226 ranged up to 720 ± 6 pCi/g, 240 ± 3 pCi/g, and 110 ± 1 pCi/g, respectively. Thorium-230 concentrations measured at five locations ranged from 570 ± 10 pCi/g to 10,100 ± 110 pCi/g. Concentrations of uranium-238 and radium-226, which were considerably above background, were found up to 11 m (36 ft) from the ditch centerline and at depths up to 0.5 m (1.6 ft) (Boerner 1986).

Six hot spots and other areas of elevated concentrations of uranium-238 were found by sampling an alluvium area between the quarry and Femme Osage

Table H.6. Concentrations of Uranium, Thorium, and Radium in Water Samples from Various Locations around the Chemical Plant Area

Sample Description	Dates Sampled	Number of Samples	Concentration (pCi/L) Range	Concentration (pCi/L) Average[a]
		Uranium (natural)[b]		
Ash Pond	9/75	8	32 - 120	59
Ash Pond outflow	10/74 - 10/84	20	36 - 4,100	1,000
Frog Pond inlet	9/75, 5/81	4	1,200 - 2,700	2,000
Frog Pond outlet	9/75 - 10/84	18	27 - 340	190
North dump runoff	4/83 - 9/83	3	29 - 2,200	800
Drain, Raffinate Pit 3 (Manhole #12)	3/75, 4/83 - 10/84	9	5.4 - 1,500	350
Drains, Raffinate Pits 1,2 (Manhole #13)	3/75 - 10/84	10	14 - 240	73
Plant sewer drainage	4/83 - 10/84	27	310 - 2,000	730
Building sumps	4/83	3	2,400 - 8,000	4,900
Surface water at plant	10/74 - 4/83	6	51 - 10,000	2,000
Drinking water, chemical plant	4/83 - 10/84	7	<0.27 - 9.4	1.7
Background[c]	1985	9	3 - 4	3.3
		Thorium-232[b]		
Ash Pond	9/75	8	0.022 - 0.27	0.052
Ash Pond outflow	9/75	4	0.022	0.022
Frog Pond inlet	9/75	4	0.044 - 0.733	0.27
Frog Pond outlet	9/75	4	0.022	0.022
North dump runoff	-	0	-	-
Drain, Raffinate Pit 3 (Manhole #12)	-	0	-	-
Drains, Raffinate Pits 1,2 (Manhole #13)	-	0	-	-
Plant sewer drainage	9/75	5	0.022 - 0.60	0.22
Building sumps	-	0	-	-
Surface water at plant	9/75	1	<0.022	<0.022
Drinking water, chemical plant	-	0	-	-

Table H.6. Continued

Sample Description	Dates Sampled	Number of Samples	Concentration (pCi/L) Range	Average[a]
		Radium-226		
Ash Pond	9/75	8	0.2 - 1.4	0.51
Ash Pond outflow	10/74 - 12/84	13	0.1 - 1.6	0.39
Frog Pond inlet	9/75	4	0.3 - 0.9	0.67
Frog Pond outlet	9/75 - 12/84	14	0.09 - 2.4	0.56
North dump runoff	3/83 - 6/83	3	0.20 - 0.31	0.25
Drain, Raffinate Pit 3 (Manhole #12)	11/74 - 6/83	8	0.36 - 153	44
Drains, Raffinate Pits 1,2 (Manhole #13)	11/74 - 6/83	7	0.61 - 1,144	240
Plant sewer drainage	9/77 - 12/84	17	0.53 - 37.6	4.1
Building sumps	3/83	3	0.11 - 0.39	0.25
Surface water at plant	6/74 - 3/83	2	0.10 - 0.9	0.50
Drinking water, chemical plant	3/83 - 12/83	4	0.11 - 0.39	0.25
Background[c]	1985	9	0.1 - 0.6	0.33
		Radium-228		
Ash Pond	-	0	-	-
Ash Pond outflow	6/74 - 12/84	11	0.45 - 15	2.1
Frog Pond inlet	-	0	-	-
Frog Pond outlet	4/80 - 12/84	10	<0.54 - 1.9	0.99
North dump runoff	3/83 - 9/83	3	<0.63 - <1.7	1.0
Drain, Raffinate Pit 3 (Manhole #12)	4/80 - 12/83	5	0.56 - 3.8	1.5
Drains, Raffinate Pits 1,2 (Manhole #13)	4/80 - 12/83	5	<0.48 - 18	5.4
Plant sewer drainage	4/80 - 12/84	14	<0.6 - 3.4	1.8
Building sumps	3/83	3	1.1 - 42	17
Surface water at plant	6/74 - 3/83	2	0.84 - 2.25	1.5
Drinking water, chemical plant	3/83 - 9/83	4	<0.70 - <1.4	1.0
Background[c]	1983, 1984	-	<0.48 - 3.2	-

[a] Averages were computed by reading upper limits as actual values. Averages are rounded to two significant figures.

[b] The data in Niedermeyer (1976) are given in units of µCi/mL natural uranium. The data in the other references are given in units of pCi/L uranium. No information is given in the other references regarding the isotopic composition of the uranium activity. It is assumed here that the data refer to natural uranium.

[c] Data for uranium and radium-226 are results of off-site measurements taken by Bechtel National (1986). Radium-228 background was assumed to be the results of measurements for the Missouri River and the Water Treatment Plant as given in Table H.8.

Sources: Niedermeyer (1976); Turnbull (1983-1984); Hansen (1983-1984); Mazur (1983-1984); Smith (1983-1984); Peterson (1985).

Table H.7. Radioactivity in Soils and Sediments from Various Locations in the Vicinity Properties

Sample Location[a]	Date of Sample Collection	Concentration (pCi/g dry)[b]					
		Uranium-238[c]	Thorium-230	Radium-226	Thorium-232	Gross Alpha[d]	Gross Beta[d]
Dardenne Creek bank, Hwy. DD	3/84	0.21	0.5 ± 0.2	0.3 ± 0.2	0.5 ± 0.2	6.6 ± 5.2	14 ± 5
Lake 9	2/83	0.90	1.1 ± 0.2	1 ± 0.2	1.2 ± 0.2	19 ± 7	35 ± 6
Lake 10	2/83	1.8	0.8 ± 0.2	2.3 ± 0.3	0.7 ± 0.2	16 ± 7	32 ± 6
Lake 34:	1984-1985	8 - 120	-	0.44 - 1.9	0.49 - 2.5	-	-
Spillway area	3/84	7.8	0.6 ± 0.2	1 ± 0.2	0.9 ± 0.2	26 ± 8	45 ± 7
East spring area	3/84	2.4	0.9 ± 0.2	1.4 ± 0.3	0.9 ± 0.2	26 ± 7	34 ± 6
Upper end	3/84	1.6	0.5 ± 0.2	1.3 ± 0.3	0.7 ± 0.2	24 ± 8	28 ± 6
Lake 35	1977, 1983-1985	3.9 - 37	1 ± 0.2	0.85 - 1.7	0.9 - 1.8	54 ± 11	61 ± 7
Burgermeister Spring	3/84	3.5	0.6 ± 0.2	1.2 ± 0.2	0.4 ± 0.1	42 ± 10	36 ± 6
Drainage ditch to Lake 36	2/83	19	1.7 ± 0.2	1 ± 0.2	0.9 ± 0.2	62 ± 11	65 ± 7
Lake 36	1977, 1983	4.7 - 103	3.3 ± 0.5	2.1 ± 0.3	1.9 ± 0.3	150 ± 10	120 ± 10
Weldon Spring Wildlife Area:							
Nature trail hot spot (WA-1)	7/83[e]	61	66 ± 4	62 ± 1	4.4 ± 1.1	360 ± 20	190 ± 10
Nature trail hot spot (WA-2)	7/83[e]	64	91 ± 4	130 ± 10	2.6 ± 0.7	460 ± 20	200 ± 10
Nature trail, east end (WA-3)	7/83[e]	30	19 ± 2	8.0 ± 0.5	1.9 ± 0.6	99 ± 11	81 ± 8
Pool/Creek at St. Rt. 94 (WA-4)	7/83[e]	14	8.8 ± 0.8	8.4 ± 0.5	1.8 ± 0.4	60 ± 9	48 ± 7

Table H.7. Continued

Sample Location[a]	Date of Sample Collection	Concentration (pCi/g dry)[b]					
		Uranium-238[c]	Thorium-230	Radium-226	Thorium-232	Gross Alpha[d]	Gross Beta[d]
Main drainage ditch:							
On Army Reserve Property	1984-1985	<0.76 - 1,000	-	0.70 - 210	0.43 - 69	-	-
From Army Reserve Property to Missouri River	1977, 1984-1985	<0.48 - 720	Up to 10,000	0.58 - 110	0.12 - 240	-	-
Little Femme Osage Creek	2/83	0.21	0.9 ± 0.2	0.7 ± 0.2	0.5 ± 0.1	8.7 ± 5.4	16 ± 5
Femme Osage Slough	3/84	3.3	0.7 ± 0.2	1.6 ± 0.3	0.7 ± 0.2	21 ± 7	36 ± 7
Little Femme Osage Slough	-	10	-	-	-	-	-
Between Quarry and Femme Osage Slough	1979, 1984-1985	<0.14 - 890	-	0.31 - 1.6	<0.08 - 3.0	-	-
Quarry outfall pool	2/83	0.86	1.1 ± 0.2	1.4 ± 0.2	1.2 ± 0.2	17 ± 7	33 ± 6
Army Reserve Property, area in southeast	1984-1985	<0.55 - 30,000	-	<0.13 - 63	0.35 - 450	-	-
Along State Route 94[f]	-	0.8 - 1.3	-	0.81 - 1.2	0.64 - 1.2	-	-
Background[g]	1984-1985	<0.68 - 1.6	-	0.55 - 0.92	0.95 - 1.5	-	-

[a] Sample locations are depicted in Figure H.3.

[b] The error limits refer to counting statistics at the 95% confidence level (2σ). Results are rounded to two significant figures. Data are given as concentration ranges when results are reported for more than one sample.

[c] The uranium data of Ryckman & Associates (1978) (the 1977 data), deRoos (1984) (the 2/83, 7/83, and 3/84 data), and Berkeley Geosciences Associates (1984) (the 1979 data) were given in units of ppm uranium; they were converted to pCi/g uranium-238 by the conversion factor 1 µg = 0.358 pCi.

[d] Gross alpha and gross beta values account for all the alpha and beta particle emissions from the radionuclides present. These come mainly from the various decay products of uranium, thorium, and radium (see Figures H.1 and H.2).

[e] Date samples received.

[f] Source: Berkeley Geosciences Associates (1984). 27 samples were taken at various locations: along State Route 94 from about 2.4 km (1.5 mi) below the quarry up to 0.8 km (0.5 mi) from the raffinate pits area and along the road leading into the well field near the quarry area.

[g] Source: Boerner (1986).

Sources: Ryckman & Associates (1978); deRoos (1984); Berkeley Geosciences Associates (1984); Boerner (1986); Deming (1986).

Slough. Values of surface concentrations ranged up to 890 pCi/g. Analyses of samples collected from shallow boreholes down to 0.75 m (2.5 ft) at four of the hot spots were as follows: there was a decrease in concentration with depth at one location, essentially no change at two locations, and an increase with depth at the fourth location (Boerner 1986). Elevated uranium-238 concentrations were also measured (in 1980) in soil samples taken from several observation wells in the area, extending to depths of 3 m (10 ft) or more (Berkeley Geosci. Assoc. 1984). These results are shown in Figure H.4.

The elevated uranium-238 concentrations in this alluvium area may result from uranium migrating from the quarry toward the slough. There are no zones of elevated radium-226 concentrations in the holes, and the concentrations are within the range of local natural environmental values. These results indicate that radium-226 has not migrated from the quarry toward the slough (Berkeley Geosci. Assoc. 1984).

The uranium-238 concentrations measured in two holes drilled into the alluvium on the south side of the slough were close to background levels at all depths; average concentrations were 3.1 and 4.0 ppm (1.1 and 1.4 pCi/g) (Figure H.4). Thus, radioactive materials have not reached the alluvium where the county well field is located (Berkeley Geosci. Assoc. 1984). Surface concentrations of uranium measured in the area of elevated concentrations shown in Figure H.4 were quite high, up to 890 pCi/g (2,500 ppm) of uranium-238 (Boerner 1986). Measurements along traverses to the slough show that surface concentrations fall off rapidly toward the slough (Berkeley Geosci. Assoc. 1984).

Contaminated areas have also been detected on the U.S. Army Reserve Property (Figure H.3). One area of about 1,520 m^2 (1,820 yd^2) in the southeast portion near the Weldon Spring site has many hot spots, with concentrations ranging up to 29,500 ± 200 pCi/g uranium-238, 40.1 ± 1.3 pCi/g radium-226, and 450 ± 6 pCi/g thorium-232. Uranium contamination was found to a depth of at least 1 m (3.3 ft) (Deming 1986).

Another area of contamination on the Army property is the portion (305 m [1020 ft] long) of the main drainage ditch from the chemical plant to the Missouri River located on the Army property. Elevated concentrations of uranium-238 (up to 1010 ± 17 pCi/g), radium-226 (up to 210 ± 3 pCi/g), and thorium-232 (up to 69.1 ± 30.4 pCi/g) were found, and contamination extends to a depth of at least 0.6 m (2 ft) (Deming 1986).

Other contaminated areas include (1) an area along the Army Railroad #2 (up to 1,350 ± 33 pCi/g uranium-238 and 38.8 ± 1.3 pCi/g radium-226), (2) two drainage ditches on the Army property (up to 62.6 ± 1.6 pCi/g radium-226 in one ditch and up to 123 ± 80 pCi/g uranium-238 in the other ditch), (3) two

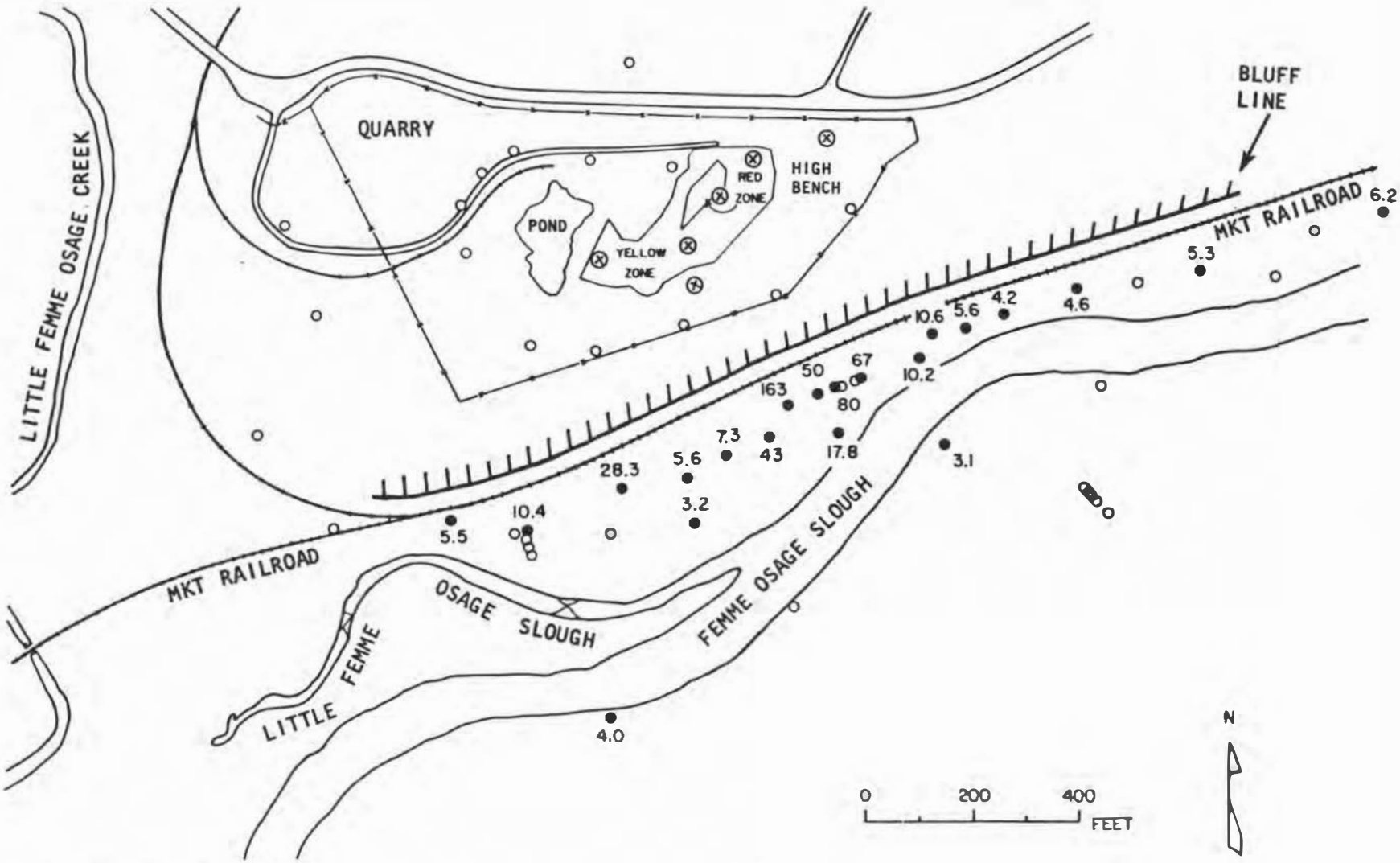


Figure H.4. Average Uranium Concentrations (ppm) in Subsurface Soil Samples at the Quarry Area. Closed circles represent boreholes that were sampled; open circles represent unsampled boreholes; circles enclosing x's represent boreholes used to analyze nonradiological parameters. Conversion Factors: To convert ppm to pCi/g natural uranium, multiply by 0.724; to convert feet to meters, multiply by 0.3048. Source: Modified from Berkeley Geosciences Associates (1984).

isolated hot spots at the Army Railroad #2 loading dock (up to 2,640 ± 10 pCi/g uranium-238 and 4.46 ± 0.78 pCi/g radium-226), (4) an isolated hot spot on Army Reserve Road #1 (up to 1.21 ± 1.40 pCi/g uranium-238 and 215 ± 3 pCi/g radium-226) (Deming 1986), (5) debris along an access road to the radio tower (up to 430 ± 4 pCi/g radium-226 and 80.9 ± 11.4 pCi/g uranium-238), and (6) two small areas on County Route "D" 716 m (2,360 yd) east of the entrance to the Busch Wildlife Area headquarters (up to 3,020 ± 30 pCi/g uranium-238) (Boerner 1986).

The concentrations of radioactive species in water on the vicinity properties are given in Table H.8. There are wide variations among the different locations, especially for gross alpha activity. Of particular note are the elevated (above background) gross alpha activities in and around Lakes 34, 35, and 36; Nature Trail Valley; Burgermeister Spring; and Femme Osage Slough (Figure H.3). Uranium concentrations are also elevated in Little Femme Osage Slough, Femme Osage Slough, Burgermeister Spring, and Lakes 34, 35, and 36. This is based on comparison of the uranium concentrations at these areas to the values for Little Femme Osage Creek and the Missouri River, values that are reported to represent background values for the area (Berkeley Geosci. Assoc. 1984). Radium-226 and radium-228 concentrations given in Table H.8 are all low and close to the range of values reported for the Missouri River.

Groundwater concentrations of uranium were measured by sampling water in boreholes drilled around the quarry. The results (Figure H.5) indicate that elevated concentrations occur, especially in the same areas in which elevated concentrations were found in the subsurface soil samples (Figure H.4). Also, adjacent boreholes can have quite different water concentrations of uranium (e.g., see Figure H.5).

Possible sources of the uranium contamination in the slough include (1) pumping of quarry pond water during 1960 through 1963 into Little Femme Osage Creek, which flowed into Femme Osage Slough until the slough was isolated by levee construction in late May 1961 (Berkeley Geosci. Assoc. 1984); (2) past dumping of contaminated materials; and (3) groundwater movement from the quarry through limestone fractures in the quarry rock into the alluvium in the slough area. The latter has been suggested as the most likely mechanism because of the natural hydraulic groundwater gradient from the quarry toward the slough (Berkeley Geosci. Assoc. 1984).

The area that will be used for spray irrigation to dispose of treated aqueous effluents is expected to be located on the U.S. Army Reserve Property in a 45-ha (110-acre) area south of the raffinate pits (see Figure H.3). It was assumed that an additional buffer zone of 30% of the irrigation area will also be needed during the action period.

Table H.B. Radioactivity in Surface Water from Various Locations in the Vicinity Properties

Sample Location[a]	Dates of Sampling	Concentration (pCi/L)[b]				
		Natural Uranium[c]	Radium-226	Radium-228	Gross Alpha[d]	Gross Beta[d]
Little Femme Osage Slough	5/75, 1979	120 - 220	<0.45	0.45	-	-
Femme Osage Slough[e]	5/75, 11/79	22 - 130	<0.45	0.45 - 0.91	-	-
	1984-1985	56[f] - 94	0.1 - 0.7[f]	-	67 ± 4	16 ± 3
Little Femme Osage Creek	5/75, 11/79	0.72 - 1.4	<0.45	0.45 - 0.91	-	-
Spring at Lake 33	1/84	-	0.2		1.5	2.5
Lake 9	3/84	-	-	-	<1.0	4.2
Lake 10	3/83 - 10/84	<0.47 - 1.2	1.3	2.4	<1.0 - 7.9	3.8 - 5.7
Burgermeister Spring	1/84	-	0.3	-	135 ± 10	36
	3/84	-	0.2	-	77 ± 7	22 ± 4
	1984-1985	140[e] - 220	0.16 ± 0.07	0.45 ± 0.39	260 ± 9	97 ± 3
Lake 34						
Inlet creek	3/84	-	0.1	-	64 ± 6	13
Lake water	3/84	-	0.1	-	31 ± 4	13
	1984 - 1985	35 ± 5	0.26 ± 0.25	-	14 ± 2	18 ± 2
Lake 35						
Inlet creek	3/84	-	0.1	-	16 ± 3	7.8
Lake water	12/83 - 10/84	5.6 - 29	0.2	<1.5	10 - 23	9.9 - 20
Lake 36						
Inlet creek	3/84	-	0.1	-	77 ± 7	6 ± 3
Lake water	4/83 - 10/84	9 - 56	0.1 - 0.62	0.59 - 0.95	32 - 41	17 - 28
Drinking water						
Busch Wildlife Area	3/83 - 10/84	<0.27 - <0.54	0.19 - 0.29	<0.55 - <1.2	<1.1 - <1.8	4.2 - 4.4
Nature trail in valley						
Spring #3	3/84	-	0.9	-	148 ± 9	32 ± 4
Spring #4	3/84	-	0.3	-	53 ± 6	11
Pool/Creek St. Rt. 94 (WA-4)	3/84	-	0.6	-	242 ± 11	44 ± 4
Well No. 5 in well field	9/84	<1.5[f]	0.3[f]	-	<9[f]	-

Table H.8. Continued

Sample Location[a]	Dates of Sampling	Concentration (pCi/L)[b]				
		Natural Uranium[c]	Radium-226	Radium-228	Gross Alpha[d]	Gross Beta[d]
Water Treatment Plant, St. Charles County	3/83 - 10/84[f]	<0.24 - <0.5	0.46 - 2.3	<0.48 - 1.8	<1.9 - <4.1	4.7 - 6.6
Missouri River at confluence with drainage ditch from chemical plant						
Upstream	3/83 - 10/84[g]	<0.27 - 2.9	0.26 - 1.4	<0.56 - 2.0	3.4 - 18	7.4 - 15
Downstream	3/83 - 10/84[g]	<0.27 - 2.1	0.19 - 1.5	<0.79 - 3.2	<3.7 - 13	11 - 15
Background[h]	1983 - 1985	3 - 4	0.1 - 0.6	-	0.48 - 4.1	3.5 - 7.4

[a] Sample locations are depicted in Figure H.3.

[b] Concentrations are given either as ranges or as values. The ranges refer to the range of values obtained on different sample dates. Data are rounded to two significant figures.

[c] Uranium concentrations are given as pCi/L uranium (Turnbull 1983-1984; Hansen 1983-1984; Mazur 1983-1984; Smith 1983-1984); ppm, µg/L, or mg/L uranium (Pennak 1975; Berkeley Geosci. Assoc. 1984; U.S. Geol. Surv. 1984; Layne Western 1986); or pCi/L for uranium-234 and uranium-238 (Boerner 1986). The data were converted to pCi/L natural uranium using the conversion factor 0.724 pCi = 1 µg. The data of Boerner (1986) indicate that uranium-238 and uranium-234 are present in approximately their natural ratio. Uranium-235 was not measured in the water.

[d] Gross alpha and gross beta values account for the alpha and beta particle emissions from all the radionuclides present in the water. These radionuclide emissions come mainly from the various decay products of uranium, thorium, and radium (see Figures H.1 and H.2).

[e] 31 samples were taken in Femme Osage Slough, mostly in 1979. The average concentrations of uranium in the 1979 samples were 0.067 mg/L for samples taken in the upper slough east of the quarry and 0.035 mg/L for samples taken in the lower slough close to and south of the quarry (Berkeley Geosci. Assoc. 1984).

[f] Dissolved concentrations only (U.S. Geol. Surv. 1984).

[g] Sampling data include one radium-226 sample taken in 9/77.

[h] Sources: Uranium and radium-226 data, Bechtel National (1986); gross alpha and gross beta, Boerner (1986). Background values for radium-228 were assumed to be the values given in this table for the Water Treatment Plant and the Missouri River.

Sources: Pennak (1975); Berkeley Geosciences Associates (1984); deRoos (1984); Turnbull (1983-1984); Hansen (1983-1984); Mazur (1983-1984); Smith (1983-1984); U.S. Geological Survey (1984); Boerner (1986); Layne Western (1986).

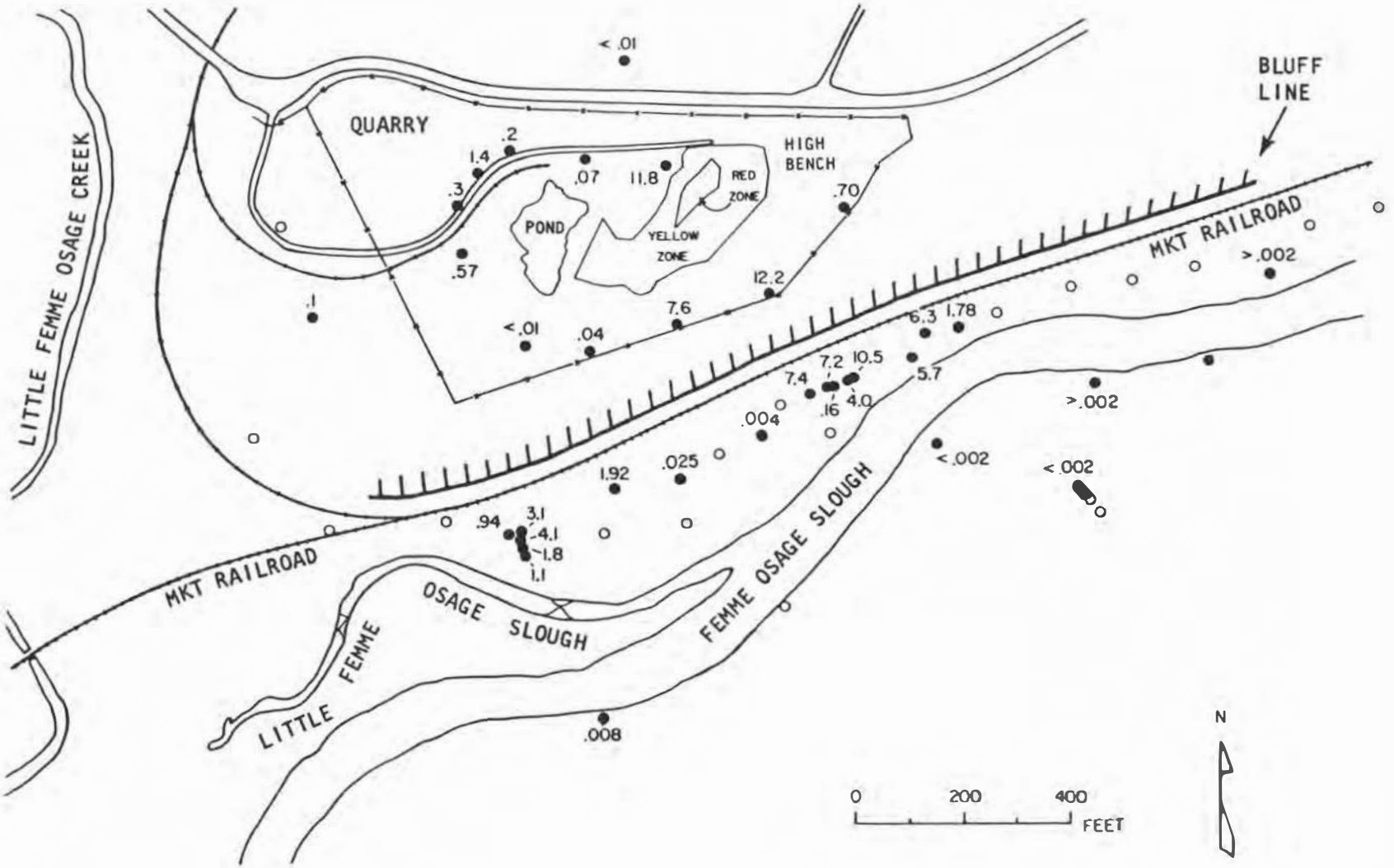


Figure H.5. Uranium Concentrations (mg/L) in Groundwater Samples at the Quarry Area. Closed circles represent boreholes that were sampled; open circles represent unsampled boreholes. Conversion Factors: To convert mg/L to pCi/L natural uranium, multiply by 724; to convert feet to meters, multiply by 0.3048. Source: Modified from Berkeley Geosciences Associates (1984).

H.1.5 Estimated Inventories of Uranium, Radium, and Thorium in the Wastes

The data presented in Sections H.1.1 through H.1.4 were used to estimate the inventories of the major radiological species in the different components of the Weldon Spring wastes. The results are summarized in Table H.9. Because of uncertainties in the data, estimates are reported to one significant figure only. Details are given below.

Inventories of uranium-238, radium-226, thorium-230, and thorium-232 for the raffinate pits were estimated from the data in Tables H.1 and H.2. Inventories for the quarry were taken directly from Appendix I, Table I.11. Details of how the inventory values were obtained and a discussion of sources of uncertainty are given in Appendix I, Section I.4.2.

The values in Table H.5 were used to determine estimates of the radium, thorium, and uranium inventories in the contaminated soils and rubble from the chemical plant. The radium-226 inventory was calculated by assuming that 240,000 m^3 (310,000 yd^3) of soils and rubble, with an average density of 1.9 g/cm^3, is contaminated with an average of 3.3 pCi/g radium-226. This value was obtained by averaging the average values given in Table H.5 for each location. The average was weighted by the number of samples chosen for each location. The thorium-230 and thorium-232 inventories were calculated in the

Table H.9. Estimated Waste Inventory at the Weldon Spring Site[a]

Component	Inventory (Ci)			
	Uranium-238[b]	Thorium-232	Thorium-230	Radium-226
Raffinate pits[c]	30	6	700	20
Quarry[d]	30	3	90	10
Chemical plant	7	1	3	1
Vicinity properties	4	0.2	30	0.5

[a] Because of uncertainties, values are reported to one significant figure. Details of how the estimates were determined are given in the text.

[b] The amounts of uranium-238 and uranium-234 and a smaller amount of uranium-235 were assumed to be present in their natural activity ratio, 238:234:235 = 1:1:0.046.

[c] Errors due to counting statistics are about ±25% for radium-226, ±10% for uranium-238, ±7% for thorium-230, and ±16% for thorium-232 (Bechtel Natl. 1984).

[d] Measurement errors are about ±18% for uranium-238, ±85% for radium-226 and thorium-230, and ±33% for thorium-232 (Bechtel Natl. 1985).

same way using weighted average concentrations of 6 pCi/g and 2.9 pCi/g, respectively, obtained from the data in Table H.5. The inventories are summarized in Table H.9.

The method used to estimate the inventories of thorium-230, thorium-232, and radium-226 was not used for uranium-238 because uranium is unique among the four species in having very large variations in concentrations from samples taken over the chemical plant area. Samples that represent concentrations of 10,000 ppm and up (> 1% uranium) were assumed to represent an extremely small volume fraction of the estimated 240,000 m^3 (310,000 yd^3) of contaminated soils and rubble. As a result, the inventory of uranium-238 was estimated by assuming that the weighted average of 820 pCi/g for uranium-238, computed from data in Table H.5 in the same manner as the weighted averages for radium and thorium, applies to the 3,800 m^3 (5,000 yd^3) of waste material with a density of 1.9 g/cm. This amount was added to the estimated amount (4 MT or 1.4 Ci) found in the buildings (Ryckman & Assoc. 1978), giving a total inventory of 7 Ci of uranium-238.

The total volume of contaminated soils and sediments in the vicinity properties was estimated by Boerner (1986) and Deming (1986) to be 21,000 m^3 (27,000 yd^3); of this, 6,000 m^3 (7,800 yd^3) is distributed among four locations at the U.S. Army Reserve Property and 15,000 m^3 (19,000 yd^3) is distributed mainly among three locations in the remaining vicinity properties. These volumes represent soils and sediments contaminated with uranium, radium, or thorium at levels in excess of DOE Guidelines for Residual Radioactivity (Appendix D). The criterion used for uranium was 60 pCi/g uranium-238 averaged over 100 m^2 (120 yd^2) with an equal amount of uranium-234 assumed to be present (Boerner 1986).

The measured concentrations of uranium-238, radium-226, and thorium-230 at different places in each of the seven locations were averaged together to give average concentrations for each location. The average concentrations were then multiplied by the volume estimate (Boerner 1986; Deming 1986) and soil density for that location, and the results were summed over all locations. An average soil density of 1.8 g/cm^3 was used for all locations.

Because almost all the thorium-230 in the vicinity properties is expected to be in the main drainage ditch (Berger 1986), the vicinity properties inventory of thorium-230 was assumed to be equal to that of the main drainage ditch. The value was obtained by determining the thorium-230/radium-226 ratio for each of the five places in the ditch at which both thorium-230 and radium-226 concentrations were measured and averaging the five ratios together. The average ratio was then multiplied by the radium-226 inventory in the main drainage ditch to obtain the thorium-230 inventory in the ditch. The inventory values are summarized in Table H.9.

The inventory values in Table H.9 are quite uncertain because both the average concentrations and waste volumes and densities are uncertain (see also Section H.1.1). The estimates of these uncertainties are as follows. For uranium in the raffinate pits, other inventory estimates are 100 MT (36 Ci) uranium-238 (Bechtel Natl. 1984); 137 MT (49 Ci) uranium-238 (Natl. Lead Co. Ohio 1977), and 430 MT (150 Ci) uranium-238 (Task Force 1967). A recent historical materials balance study (Harris 1986) estimated that the raffinate pits contain about 155 MT (55 Ci) uranium-238. As discussed in Section H.1.1, the 430 MT value is suspect. The 155 MT value is based on the materials balance reports provided by the contractor for the plant (Mallinckrodt, Inc.) and on annual audits by the former U.S. Atomic Energy Commission rather than on recent measurements of concentrations in the raffinate pits. Thus, the 155 MT value would be expected to be an estimate independent of problems of stratification and material heterogeneities in the raffinate sludge. The above results suggest that the inventory estimate for uranium-238 given in Table H.9 may be low and that the true value could be as much as five times higher.

Other estimates of the thorium-232 inventory in the raffinate pits are 100 MT (11 Ci) (Task Force 1967), 65 MT (7 Ci) (Bechtel Natl. 1984), 64 MT (7 Ci) (Natl. Lead Co. Ohio 1977), and 76 MT (8 Ci) (Harris 1986). These values are all quite close together and suggest that the value of 6 Ci in Table H.9 may be slightly low and that the true value could be as much as 1.5 to 2 times higher.

Other estimates of the thorium-230 inventory in the raffinate pits do not appear to be available. However, the above discussion for uranium-238 and thorium-232 indicates that the estimated inventory value of 700 Ci may be low and that the true value could be as much as 2 to 3 times higher. A similar uncertainty estimate can be made for radium-226 because no estimates are available other than the 1967 value of 240 Ci (Task Force 1967). However, as discussed in Section H.1.1, this value is unreasonably high.

Inventory estimates for the quarry may be assumed to be uncertain by factors of two to three. This is based on the fact that much of the quarry waste was not sampled, i.e., the rubble and any contaminated soil underlying the rubble (about two-thirds of the waste in the quarry floor [40-ft region] is rubble--see Appendix I, Section I.4.2). Also, other estimated values for the quarry are 10 to 21 Ci of radium-226 and a somewhat larger amount of uranium-238 (Berkeley Geosci. Assoc. 1984).

Inventory estimates for the chemical plant are preliminary; they may be high or low by a factor of 10 or more. The inventory estimates for the

vicinity properties are also quite uncertain. One problem is the heterogeneity of contaminant concentrations in each contaminated area and the resultant uncertainty in the averaging process.

The large amount of thorium-230 in the wastes compared to the amount of radium-226 means that the radium-226 inventory in the wastes will increase with time. The growth of the radium-226 inventory as a function of time is shown in Figure H.6. There is a large increase from 32 Ci in 1983 to 300 Ci in 1,000 years; the maximum amount of more than 700 Ci will be reached in 9,100 years. At times greater than 9,100 years, the radium-226 inventory will decrease slowly with the 77,000-year half-life of its parent thorium-230.

H.2 NONRADIOLOGICAL CHARACTERIZATION

H.2.1 Raffinate Pits

The results of measurements of chemical species in the raffinate pits solids for 1967 (Task Force 1967) and 1983 (Bechtel Natl. 1984) are given in Table H.10. The two sets of 1983 data represent independent measurements of the raffinate sludge.

Large differences in concentrations exist in the different sets of data, particularly for the major constituents. Calcium concentrations differ by a factor of at least 100. There are also large differences for iron, manganese, magnesium, sodium, and zinc. Aluminum is an extreme case, with concentrations in the different data sets differing by factors as high as 2,000 to 20,000. These differences may be partially explained by the reported stratification and heterogeneity of the sludge in each pit (Task Force 1967; Natl. Lead Co. Ohio 1977). Consequently, the different data sets may refer to samples of different composition even though the samples taken for both the 1983 data sets are composites. Note that the 1967 data are given as ranges over a factor of 10 (Task Force 1967).

An analysis for organic priority and nonpriority pollutants was carried out on a composite sample of raffinate sludge (Haywood 1984). The sample was analyzed for 82 priority pollutants (19 pesticides, 7 PCBs, and 56 acid and base/neutral compounds) and 13 nonpriority pollutants (including PCB 1262, which is not listed as a priority pollutant [Keith and Telliard 1979]). All concentrations were reported as detection limits only, which varied from 0.1 to 1 ppm for the different individual compounds. No positive results (above detection limits) were reported for any organics. However, organic compounds are expected to be present as minor components because the processing of the uranium materials included a solvent extraction step using tributyl phosphate in hexane (Niedermeyer 1976).

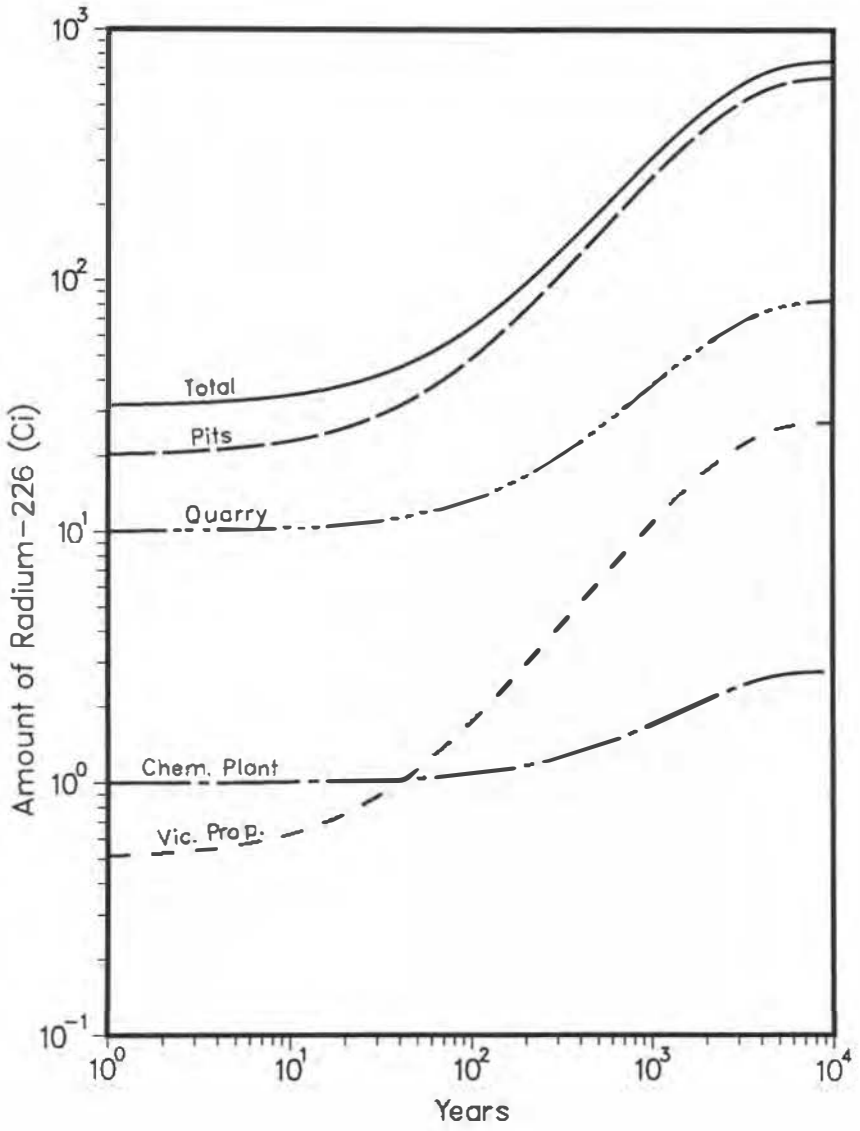


Figure H.6. Radium-226 Inventory in the Weldon Spring Wastes as a a Function of Time.

The raffinate sludge has recently been sampled and analyzed for the four characteristics of RCRA hazardous wastes: ignitability, corrosivity (i.e., pH), reactivity, and EP toxicity. All analysis results were within (pH) or below regulatory limits. An additional analysis was carried out for the presence of PCBs, and no PCBs were detected at concentrations above detection limits (Nemec 1986). DOE is gathering additional data on the chemical characteristics of the raffinate sludge to ensure that appropriate waste stabilization and confinement features are incorporated in the alternative selected.

Table H.10. Concentrations of Chemical Species in the Raffinate Pits Solids[a]

Chemical Species	Concentrations (ppm dry weight)				
	Pit 1			Pit 2	
	1983a[b]	1983b[c]	1967[d]	1983a[b]	1967[d]
Aluminum	4.3	500	10,000 - 100,000	4.1	10,000 - 100,000
Arsenic	130	100	10 - 100	170	10 - 100
Boron	60	>10,000	10 - 100	350	10 - 100
Barium	23	30	100 - 1,000	20	100 - 1,000
Cadmium	9.1	≤20	10 - 100	7.3	10 - 100
Calcium	980	>10,000	>100,000	990	>100,000
Cobalt	9.4	≤10	10 - 100	14	10 - 100
Chromium	90	30	100 - 1,000	60	100 - 1,000
Copper	5.5	100	100 - 1,000	4.9	100 - 1,000
Iron	210	10,000	10,000 - 100,000	200	10,000 - 100,000
Lead	110	≤10	100 - 1,000	140	100 - 1,000
Magnesium	1,800	10,000	10,000 - 100,000	1,700	10,000 - 100,000
Manganese	7.8	500	100 - 1,000	7.8	100 - 1,000
Mercury	1.8	≤10	-	0.75	-
Molybdenum	4,700	5,000	1,000 - 10,000	2,800	1,000 - 1,000
Nickel	30	-	100 - 1,000	27	100 - 1,000
Phosphorus	-	2,000	1,000 - 10,000	-	1,000 - 10,000
Potassium	650	500	-	620	-
Selenium	0.89	-	-	0.5	-
Silicon (total)	10,000	≤20	>100,000	16,400	>100,000
Strontium	84	50	100 - 1,000	220	100 - 1,000
Sodium	10,000	8,000	100 - 1,000	5,000	100 - 1,000
Titanium	1,000	-	100 - 1,000	860	100 - 1,000
Vanadium	5,000	10,000	1,000 - 10,000	800	1,000 - 10,000
Zinc	10	1,000	100 - 1,000	20	100 - 1,000
Zirconium	17,000	20	1,000 - 10,000	14,400	1,000 - 10,000
Phosphate	0.7	-	-	0.6	-
Nitrate	50,000	-	-	18,000	-
Fluoride	23,000	>10,000	-	2,500	-
Chloride	670	1,000	-	230	-
Sulfate	400	10,000[e]	-	200	-
Hydroxide (% $CaCO_3$)	7	-	-	10	-
pH (pH units)	8.1	-	-	8.7	-

Table H.10. Continued

Chemical Species	Concentrations (ppm dry weight)				
	Pit 3			Pit 4	
	1983a[b]	1983b[c]	1967[d]	1983a[b]	1967[d]
Aluminum	6.2	>10,000	10,000 - 100,000	4.0	10,000 - 100,000
Arsenic	54	200	10 - 100	1.0	10 - 100
Boron	50	>10,000	10 - 100	30	10 - 100
Barium	10	400	100 - 1,000	22	10,000 - 100,000
Cadmium	5.1	≤50	10 - 100	2.8	10 - 100
Calcium	990	>10,000	>100,000	980	>100,000
Cobalt	11	≤5	10 - 100	2.1	10 - 100
Chromium	60	100	100 - 1,000	70	100 - 1,000
Copper	5.5	600	100 - 1,000	4.4	100 - 1,000
Iron	130	>10,000	10,000 - 100,000	210	10,000 - 100,000
Lead	220	50	100 - 1,000	1.5	10 - 100
Magnesium	1,700	>10,000	>100,000	860	>100,000
Manganese	9.4	500	100 - 1,000	8.8	100 - 1,000
Mercury	17	≤20	-	2.3	-
Molybdenum	2,500	600	1,000 - 10,000	370	100 - 1,000
Nickel	27		100 - 1,000	22	100 - 1,000
Phosphorus	-	7,000	1,000 - 10,000	-	100 - 1,000
Potassium	220	500	-	310	-
Selenium	1.4	-	-	0.5	-
Silicon (total)	13,000	≤100	>100,000	13,800	>100,000
Strontium	50	50	100 - 1,000	40	100 - 1,000
Sodium	5,000	>10,000	100 - 1,000	4,000	100 - 1,000
Titanium	1,150	-	100 - 1,000	670	100 - 1,000
Vanadium	800	4,000	1,000 - 10,000	300	100 - 1,000
Zinc	10	50	-	10	100 - 1,000
Zirconium	19,000	200	1,000 - 10,000	11,500	10,00 - 100,000
Phosphate	0.8	-	-	0.5	-
Nitrate	22,000	-	-	220	-
Fluoride	107,000	>10,000	-	64,300	-
Chloride	300	700	-	50	-
Sulfate	370	700[e]	-	270	-
Hydroxide (% $CaCO_3$)	10	-	-	11	-
pH (pH units)	9.4	-	-	8.5	-

[a] A hyphen means that no value was given in the reference source.

[b] Source: Bechtel National (1984--Appendix A).

[c] Source: Bechtel National (1984--Appendix H). Entries given here as >10,000 ppm are designated in the source reference as "major". The highest listed concentration in the source reference is 10,000 ppm.

[d] Source: Task Force (1967). Most values are given as ranges.

[e] Reported as ppm sulfur.

Concentrations of nonradiological chemical species in the water standing in the pits -- as determined from samples taken in 1967, 1979, 1983, and 1984 -- are given in Table H.11. Other data for nitrate and chloride ions are available for other years, particularly 1974 and 1975 (Anon., undated). Large variations exist in concentration data for different years, especially for Pit 3 for which the most data are available. For many species, the concentrations appear to decrease with increasing time; that is, the 1967 concentrations are higher than the 1979 concentrations, which are somewhat higher than the 1983 and 1984 concentrations.

In general, variations in concentrations with time are expected because of such factors as the sensitivity of the data to meteorological conditions preceding the sampling. For example, sampling after a heavy rainfall will give lower concentrations, due to dilution and stratification factors, than will sampling in a period of drought.

It should be noted that the total dissolved solids (TDS) are higher in the 1967 samples than in the 1984 samples (Table H.11, footnotes a and c). The 1967 values for each chemical species may be high because the value for each chemical species in each pit was obtained by multiplying the measured concentration range by the TDS concentration (Table H.11, footnote c). One reason that the TDS concentrations are higher in 1967 than in 1984 may result from the fact that water in the pits exists in two phases, free water above the sludge and water in intimate contact and bound to the raffinate materials making a sludge or gel. The water in intimate contact with the raffinate materials is quite likely to have higher dissolved solids concentrations than free water standing over the sludge. The 1967 samples were taken in such a way that both phases were included. It is reported that after stirring the sludge, "a significant quantity of water 'came loose' and rose to the top of the sample" (Task Force 1967). Thus, the TDS concentrations in these samples are expected to be higher than in samples restricted to free water standing over the sludge. The 1979, 1982, 1983, and 1984 data in Table H.11 all appear to represent samples drawn from the surface water layer above the sludge.

If the concentrations of chemical species measured in 1979 and later years in the pit waters are compared to state of Missouri irrigation limits (see Table H.12), the concentrations of all but one of these species are below the limits. (Current plans are to dispose of the water by treating it, as necessary, and then using it for irrigation [Chapter 4, Section 4.1.3.1].) The single exception is arsenic whose concentration in 1983 in Pit 3 exceeded the irrigation limit; the 1983 arsenic values in Pits 1 and 2 are close to or at the limit. The 1984 arsenic values for all four pits are below the state irrigation limit, and the 1984 cadmium value for Pit 3 is below the state irrigation limit.

Table H.11. Concentrations of Nonradiological Chemical Species in the Water in the Raffinate Pits

	Concentrations (mg/L)					
	Pit 1			Pit 2		
Chemical Species	1984[a]	1983[b]	1967[c]	1984[a]	1983[b]	1967[c]
Aluminum	0.03	-	-	0.04	-	-
Arsenic	0.006	0.10	-	0.015	0.090	-
Boron	-	<0.10	0.81 - 8.1	-	<0.10	0.72 - 7.2
Barium	0.090	<0.10	8.1 - 81	0.072	<0.10	7.2 - 72
Cadmium	<0.003	0.0004	-	<0.003	0.0002	-
Calcium	560	-	>8,100	380	-	>7,200
Cobalt	<0.009	0.028	-	<0.009	0.032	-
Chromium	<0.001	<0.001	<0.81	<0.001	<0.001	<0.72
Copper	0.004	0.11	0.81 - 8.1	0.004	0.005	0.72 - 7.2
Iron	0.015	0.012	8.1 - 81	<0.009	0.005	7.2 - 72
Lead	<0.001	<0.001	-	<0.001	0.002	-
Lithium	0.14	-	-	0.14	-	-
Magnesium	26	-	8.1 - 81	66	-	720 - 7,200
Manganese	0.009	0.028	<0.81	0.009	0.033	0.72 - 7.2
Mercury	<0.0001	<0.001	-	<0.0001	0.020	-
Molybdenum	3	-	0.81 - 8.1	7.1	-	7.2 - 72
Nickel	<0.001	<0.010	-	<0.001	0.020	-
Phosphorus	-	-	8.1 - 81	-	-	7.2 - 72
Potassium	48	-	81 - 810	35	-	72 - 720
Selenium	<0.001	0.010	-	<0.001	<0.010	-
Silicon (as Si)	2.5	-	8.1 - 81	1.0	-	7.2 - 72
Sodium	520	-	>8,100	180	-	>7,200
Strontium	1.4	-	8.1 - 81	0.78	-	7.2 - 72
Titanium	-	-	-	-	-	-
Vanadium	3.2	-	8.1 - 81	2.0	-	7.2 - 72
Zinc	0.045	0.070	-	0.025	0.040	-
Zirconium	-	-	-	-	-	-
Chloride	17	15	210	5.7	6	50
Fluoride	2.5	1.1	-	2.7	1	-
Nitrite (as NO_2)	105	-	-	18	-	-
Nitrate (as NO_3)	2,900	3,100	25,000	907	0.44	38,000
Sulfate	400	100	2,300	990	460	3,300
Hydroxide (as $CaCO_3$)	-	-	-	-	-	-
pH (pH Units)	8.9	-	9.2	9.3	-	8.6

Table H.11. Continued

Chemical Species	Concentrations (mg/L)								
	Pit 3					Pit 4			
	1984[a]	1983[b]	1982[d]	1979[e]	1967[c]	1984[a]	1983[b]	1979[e]	1967[c]
Aluminum	0.030	-	0.032	0.031	-	0.010	-	0.027	-
Arsenic	0.004	0.14	<0.06	-	-	0.001	0.020	-	-
Boron	-	<0.10	0.13	0.18	0.55 - 5.5	-	<0.10	0.091	0.37 - 3.7
Barium	0.17	<0.10	0.096	0.19	5.5 - 55	0.10	<0.10	0.12	3.7 - 37
Cadmium	<0.010	<0.0001	<0.004	-	0.003	<0.001	0.003	-	-
Calcium	980	-	890	1,400	>5,500	17	-	15	>3,700
Cobalt	<0.030	0.040	<0.0020	<0.008	-	<0.003	0.013	<0.004	-
Chromium	<0.001	<0.001	0.0048	<0.004	<0.55	<0.001	<0.001	<0.004	<0.37
Copper	0.007	0.011	-	0.027	0.55 - 55	0.001	<0.001	<0.004	0.37 - 3.7
Iron	<0.030	0.011	<0.0050	0.056	5.5 - 55	<0.003	0.074	<0.004	3.7 - 37
Lead	0.011	0.002	<0.0020	-	-	0.017	0.004	-	<0.37
Lithium	0.46	-	2.9	6.2	0.55 - 5.5	0.66	-	0.69	3.7 - 37
Magnesium	320	-	210	480	550 - 5,500	52	-	34	3.7 - 37
Manganese	0.035	0.009	<0.0050	0.093	<0.55	0.007	0.007	<0.004	-
Mercury	<0.0001	<0.001	-	-	-	<0.0001	<0.001	-	-
Molybdenum	3.6	-	2.2	3.9	5.5 - 55	0.67	-	1.0	3.7 - 37
Nickel	<0.001	<0.010	0.0064	0.007	-	<0.001	<0.010	<0.004	-
Phosphorus	-	-	<0.0020	0.047	5.5 - 55	-	-	0.009	37 - 370
Potassium	150	-	95	180	55 - 550	23	-	18	37 - 370
Selenium	<0.001	0.080	-	-	-	<0.001	<0.010	-	-
Silicon (as Si)	1.3	-	0.31	1.7	5.5 - 55	0.7	-	1.0	3.7 - 37
Sodium	1,500	-	1,200	1,200	>5,500	190	-	122	>3,700
Strontium	2.8	-	2.2	5.5	5.5 - 55	0.19	-	0.122	3.7 - 37
Titanium	-	-	<0.0010	0.06	-	-	-	<0.004	-
Vanadium	0.81	-	0.48	0.48	5.5 - 55	0.079	-	0.18	0.37 - 3.7
Zinc	0.066	0.050	<0.0020	-	-	0.003	0.010	-	-
Zirconium	-	-	<0.0010	0.006	-	-	-	<0.004	-
Chloride	25	20	-	37	90	7.7	7	10	90
Fluoride	8.9	2.7	-	6	-	7.8	5.8	13	-
Nitrite	49	-	-	40	-	11	-	<5	-
Nitrate (as NO_3)	8,400	6,600	-	13,000	37,000	410	440	56	21,000
Sulfate	640	268	-	620	2,200	150	70	140	2,200
Hydroxide (% $CaCO_3$)	-	-	-	-	-	-	-	-	-
pH (pH Units)	8.6	-	8.6	7.7	8.5	9.6	-	8.8	10.5

[a] Concentrations reported by U.S. Geological Survey (1984) as dissolved concentrations. Sample collected 9/5/84. Total dissolved solids are: Pit 1, 5.1 g/L; Pit 2, 2.8 g/L; Pit 3, 13 g/L; Pit 4, 1 g/L.

[b] Source: Bechtel National (1984--Appendix O).

[c] Sources: Task Force (1967) and Anonymous (undated). Concentrations are reported as ranges of percents (e.g., 10^{-3} to 10^{-2}%, 10^{-2} to 10^{-1}%, etc.) in the solids remaining after the samples were filtered and dried at 110°C. Sampling date was 3/28/67. Values given in the table were obtained by multiplying the ranges by the concentrations of dissolved solids in the samples. Concentrations of soluble solids in samples from Pits 1, 2, 3, and 4 are 81 g/L, 72 g/L, 55 g/L, and 37 g/L, respectively.

[d] Source: Rudolph (1983). Data, rounded to two significant figures, refer to samples after filtration through a 0.8-μm filter.

[e] Source: Taylor et al. (1979). Date of sampling was 8/2/79.

Table H.12. State of Missouri Regulatory Limits on Concentrations of Chemical Species in Surface Water and Groundwater

Chemical Species	Concentrations (mg/L)		
	Subsurface Water[a]	Irrigation	Drinking Water[b]
Arsenic	0.05	0.1	0.05
Barium	1	-	1
Beryllium	0.1	0.1	-
Boron	0.75	0.75	-
Cadmium	0.01	0.01	0.01
Chromium	0.05	0.1	0.05
Cobalt	1	-	-
Copper	0.02	-	1.0
Iron	-	-	0.3
Lead	0.05	-	0.05
Manganese	0.05	-	0.05
Mercury	0.00005	-	0.002
Nickel	0.1	-	-
Selenium	0.01	-	0.01
Silver	0.05	-	0.05
Zinc	0.1	2	5
Chloride	250	-	250
Cyanide	0.005	-	-
Fluoride	2	-	2
Nitrate (as N)	10	-	10
Sulfate	250	-	250

[a] Values are effluent limitations for released, stored, or discarded waters that enter aquifers either directly or indirectly. These values are at least as stringent as the EPA limits for groundwater protection (40 CFR Part 264.94).

[b] These values are at least as stringent as the EPA primary drinking water standards (40 CFR Part 141.11).

Source: Missouri Water Quality Standards (10 CSR Part 20-7 as amended through June 1, 1983).

H.2.2 Quarry

In general, the following is known about concentrations of nonradiological chemical species in the solid materials in or around the quarry. Based on the known history of materials dumped in the quarry (Task Force 1967; Pennak 1975; Berkeley Geosci. Assoc. 1984), materials contaminated with residues from trinitrotoluene (TNT) manufacture could be present. Prior to 1958, the quarry was used for disposal of materials contaminated with TNT and intermediates produced in the process for making TNT. Other TNT-contaminated materials were also disposed of in 1965 and 1966. Part of this material overlies the drummed residues containing 3% thorium that were deposited in

1966 (Task Force 1967; Pennak 1975). Barium sulfate residues from the vicinity of the St. Louis Airport may also have been dumped in the quarry (Berkeley Geosci. Assoc. 1984). Other chemical species present are those in the various types of building rubble and contaminated materials that were dumped in the quarry.

Additional characterization of the quarry wastes is currently being conducted to better define the chemical characteristics of these wastes. Discussions were held with EPA Region VII concerning this plan prior to initiating field activities. This chemical characterization includes analyses for organics, PCBs, and TNT and DNT and their process intermediates (Nelson 1986).

The ranges and average concentrations of metals and organic compounds in the quarry wastes, as determined from samples taken in 1984 from six boreholes, are presented in Table H.13 (Bechtel Natl. 1985). The results indicate that there is chemical contamination and that concentrations of some metals at one or more locations are higher than those in the dried raffinate sludge (Table H.10). Lead, arsenic, nickel, and copper were detected at concentrations ranging up to 410, 120, 120, and 160 ppm, respectively; and zinc was detected at concentrations up to 870 ppm. The presence of organic pollutants was analyzed, but most were below detection levels. Examples of compounds for which positive results were obtained are γ-benzene hexachloride (lindane) (0.0013 ppm), PCB 1254 (up to 46 ppm), polycyclic hydrocarbons (up to 75 ppm) and diacetone alcohol (up to 14 ppm) (see Table H.13 for complete listing). The presence of PCB 1254 prevented detection of most pesticides (Bechtel Natl. 1985).

Detailed measurements have been made of concentrations of nonradiological chemical species in surface water and groundwater in and around the quarry. Concentrations of chemical species within the quarry fence are presented in Table H.14. Additional results are presented under the discussion of vicinity properties in Section H.2.4. The data do not show any clear trend of higher concentrations in the pond or in the groundwaters of the "yellow zone" (Figure H.4).* Concentrations of barium and fluoride are somewhat higher in the pond than in the groundwater, and concentrations of manganese and nitrates are somewhat lower in the pond. For most other elements, the differences are small. Differences between element concentrations in the yellow zone groundwaters and those along the fence and access road are also small.

*The yellow and red zones in the quarry are areas of relatively high surface alpha activity that were so designated for protection of personnel during surveys of the quarry in 1979 through 1981 (Berkeley Geosci. Assoc. 1984).

Table H.13. Concentrations of Nonradiological Parameters in the Quarry Wastes[a]

Parameter	Concentrations (ppm) Range	Average
Priority Pollutant Metals and Cyanide[b]		
Arsenic	73 - 120	100
Beryllium	0.43 - 0.83	0.62
Cadmium	1.8 - 6.4	2.7
Chromium	19 - 49	30
Copper	38 - 160	100
Lead	130 - 410	280
Mercury	0.18 - 6.3	2.0
Nickel	19 - 120	43
Selenium	17 - 28	23
Silver	<0.2 - 8.3	3.6
Thallium	3.0 - 6.2	4.7
Zinc	68 - 870	340
Cyanide	<0.2 - 0.6	0.35
Organic Priority Pollutants: Pesticides and PCBs[c,d]		
α-Benzene hexachloride[e]	0.0051 - 0.0053	0.0052
δ-Benzene hexachloride[e]	0.019 - 0.095	0.045
γ-Benzene hexachloride (lindane)[e]	0.0013	0.0013
Endrin	<0.011 - 1.1	<0.23
PCB 1254	0.56 - 46	11
Organic Priority Pollutants: Base/Neutral Compounds		
Fluoranthene	<0.06 - 72	28
Benzo(a)pyrene	<0.06 - 68	12
Chrysene	<0.06 - 38	12
Anthracene	<0.06 - 11	2.3
Phenanthrene	<0.06 - 60	22
Pyrene	<0.06 - 63	27
Benzo(a)anthracene	<0.06 - 40	7.2
Benzo(b)fluoranthene Benzo(k)fluoranthene	<0.12 - 75	12
Napthalene	<0.06 - 0.74	0.35
Di-n-butylphthalate	<0.06 - 1.2	0.43
Fluorene	<0.06 - 2.6	0.66
Organic Nonpriority Pollutants		
2-Pentanone-4-hydroxy-4-methyl (diacetone alcohol)	2 - 14	6.2
Dibenzofuran	<0.06 - 1.8	0.53
2-Methylnaphthalene	<0.06 - 0.67	0.34

[a] All compounds that gave one or more positive results (above detection limits) are listed in this table. Concentrations are rounded to two significant figures. Upper limit values are assumed to be actual values in determining averages. Samples were taken in the last quarter of 1984 from six boreholes in the quarry wastes (Figure H.4).

[b] Antimony was the only priority pollutant metal that was not detected at a sensitivity level of 20 ppm.

[c] The 29 volatile priority pollutants measured for were not detected at a sensitivity level of 1 ppb.

[d] PCB 1254 was the only PCB for which concentrations were reported. The presence of this PCB prevented the detection of most pesticides.

[e] Concentrations in borehole samples were reported for only 1 sample of γ-benzene hexachloride, 2 samples of α-benzene hexachloride, and 3 samples of δ-benzene hexachloride.

Source: Bechtel National (1985).

Table H.14. Concentrations of Chemical Species in Surface Water and Groundwater within the Quarry Fence

Chemical Species	Surface Water Concentration (mg/L)[a]		Groundwater Concentration (mg/L)[a]			
	Quarry Pond[b]		Yellow Zone[c]		Inside Quarry Fence and Along Access Road[d]	
	Range	Average	Range	Average	Range	Average
Aluminum	<0.1[f] - 0.08	0.045	0.05 - 0.18	0.13	-	-
Arsenic	<0.001[f] - 0.15	0.075	0.06 - 0.19	0.11	<0.1	<0.1
Boron	0.52 - 0.60	0.54	0.14 - 0.94	0.53	<0.05 - 0.56	0.22
Barium	0.04 - 0.36	0.11	0.033 - 0.096	0.065	0.028 - 0.57	0.11
Cadmium	<0.001 - 0.01	<0.006	<0.01	<0.01	<0.01	<0.01
Calcium	70 - 100	86	62 - 170	120	48 - 290	140
Chromium	<0.001 - 0.02	0.013	<0.01	<0.01	<0.01	<0.01
Copper	<0.001[f], <0.02	<0.01	<0.01	<0.01	<0.01 - 0.08	0.018
Iron	0.003 - 0.33	0.068	<0.02	<0.02	<0.02 - 6.8	0.49
Lithium	<0.01 - 0.036	0.025	0.01 - 0.04	0.02	<0.01 - 0.04	0.016
Magnesium	16 - 26	22	8 - 59	30	4.9 - 62	40
Manganese	0.003 - 0.26	0.07	0.008 - 0.19	0.075	<0.01 - 3.6	0.48
Mercury	<0.0001 - 0.0006	0.0004	-	-	-	-
Molybdenum	<0.01 - 0.07	0.035	0.16 - 0.21	0.19	0.14	0.14
Nickel	<0.001 - 0.02	<0.01	<0.01	<0.01	<0.01 - 0.04	<0.01
Phosphorus (as P_2O_5)	0.5	0.5	0.2 - 0.9	0.4	0.11 - 7.4	0.74
Potassium	11 - 18	15	2.4 - 26	14	2.3 - 18	6.7
Silicon (as SiO_2)	13 - 21	16	10 - 32	20	7.5 - 28	16
Sodium	14 - 29	22	32 - 55	42	1 - 100	26
Strontium	0.37 - 0.54	0.47	0.42 - 0.62	0.51	0.12 - 1.3	0.68
Tin	<0.05	<0.05	<0.05	<0.05	<0.05 - 0.25	0.074
Lead	0.002[f], <0.05	<0.05	<0.05	<0.05	<0.05 - 0.26	0.057
Zinc	0.005 - 0.31	0.068	0.01	0.01	<0.01 - 4.5	0.26
Chloride	14 - 200	44	16 - 58	37	<1 - 130	60
Fluoride	0.9 - 1.1	1.0	0.3 - 1.1	0.7	<0.1 - 1.1	0.49
Nitrate	<1 - 9	3.7	4 - 13	8.5	<1 - 89	15
Sulfate	150 - 240	200	210 - 430	320	11 - 700	190
Bicarbonate	190 - 220	210	-	-	130 - 640	380
pH	7.3 - 8.2	7.7	-	-	6.0 - 7.6	7.0

[a] Concentrations are given to two significant figures. Samples were collected in 1979-1981 and in 1984 and 1985. Only those chemical species are listed for which positive results were obtained. Other species that were analyzed but not detected and the corresponding detection limits (in mg/L) include cobalt (<0.01), beryllium (<0.001), cyanide (<0.02), and selenium (<0.05).

[b] Ten samples were taken on four different dates in 1979, 1980, 1981, 1984, and 1985. Different samples were analyzed for different chemical species. Entries of only one value mean that only one sample was analyzed for the species in question.

[c] Borehole samples were taken from the yellow zone (see Figure H.4) in 1979-1980 at up to five locations in the floor of the quarry (see Chapter 3, Figures 3.4, and 3.5) (Berkeley Geosci. Assoc. 1984). The locations were different than those used to obtain the solid samples for concentrations reported in Table H.13.

[d] Eleven boreholes were sampled at four or five dates to yield 47 samples. The boreholes were located along the southern (4) and eastern (1) perimeter fence and along the access road (6) to the quarry rim (Figure H.5).

[e] Source: National Lead Company (1977). Samples collected in 1976.

[f] One sample collected on 9/5/84 by U.S. Geological Survey and labeled "quarry site". Concentrations reported as dissolved concentrations (U.S. Geol. Surv. 1984).

Sources: National Lead Company of Ohio (1977); Berkeley Geosciences Associates (1984); U.S. Geological Survey (1984); Bechtel National (1985).

Asbestos fibers have been measured in the quarry pond water at a concentration of 1.9×10^6 fibers/liter (Bechtel Natl. 1985). However, no impact from groundwater leaching and transport is expected because asbestos, which exists as particles, is not transported by leaching and groundwater movement. It is reported that clay particles of similar sizes are estimated to move in soil by purely physical processes at a rate of 1 to 10 cm per 3,000 to 40,000 years, depending on soil texture (Fuller 1977).

Based on comparison of the concentrations in the quarry pond with the irrigation regulatory limits (Table H.12), arsenic and cadmium are the only species whose concentrations in one or more samples equal or exceed the limits. For boreholes in the yellow zone or along the fence, the average concentrations of arsenic, lead, manganese, sulfate, and zinc exceed the Missouri subsurface water limits (Table H.12) as applied to water that enters aquifers either directly or indirectly.

An analysis was carried out for the presence of organic pollutants in two samples of quarry pond water (Bechtel Natl. 1985). The pond water was analyzed for 111 priority pollutants (19 pesticides, 7 PCBs, 29 volatiles, and 56 acid and base/neutral compounds) and 13 nonpriority pollutants (including PCB 1262, which is not listed as a priority pollutant [Keith and Telliard 1979]). None of these compounds were detected. Detection limits for most pesticides, priority and nonpriority organic pollutants, and all PCBs were 0.2 µg/L, 1 µg/L, and 2 µg/L, respectively. The maximum detection limit for any of the analyzed organics was 20 µg/L (2-butanone).

H.2.3 Chemical Plant

A few data are available on nonradiological contamination of soils and materials in and around the chemical plant. The plant is contaminated with about 4 MT (4.4 tons) of uranium compounds, some of which is "green salt" (uranium tetrafluoride) that is visible around the plant (Niedermeyer 1976; Ryckman & Assoc. 1978). This gives an indication of the extent of fluoride contamination.

As of 1978, two stainless steel tanks at the plant were partially filled with tributyl phosphate, an organic solvent used to extract uranium (Ryckman & Assoc. 1978). This material should be considered as an organic contaminant present at the plant. There was also an on-site refinery tank farm, and some of the tanks are still present (Henry 1986). Possible contents included nitric and sulfuric acids, caustic soda solution, ether, and hexane. The tank farm area is reported to be heavily contaminated (Rockwell Int. 1979).

Limited soil analyses have been carried out in and around the plant for TNT and related compounds. The measured values ranged from "none detected" to

Table H.15. Concentrations of TNT and Related Intermediates in Surface and Subsurface Samples in and around the Chemical Plant Area[a]

Sample Location	2,6-Dinitrotoluene			2,4-Dinitrotoluene			Trinitrotoluene (TNT)		
	Number of Positive Samples	Concentration (ppb)		Number of Positive Samples	Concentration (ppb)		Number of Positive Samples	Concentration (ppb)	
		Range	Average		Range	Average		Range	Average
Surface									
Near previous explosive process lines	1	<1.0	<1.0	4	3.9 - 38	19	1	11	11
Depressions near areas of previous explosive buildings	2	<1.0 - 1.2	<1.1	4	3.8 - 27	14	1	14	14
Roadbeds	1	1.0	1.0	2	6.5 - 23	15	1	8.3	8.3
Around Ash Pond	2	<1.0 - 2.9	<2.0	1	<2.0	<2.0	1	32	32
Subsurface									
Near previous explosive process lines	1	<1.0	<1.0	1	34	34	1	27	27
Depressions near areas of previous buildings	3	1.2 - 2.0	1.7	3	23 - 56	34	2	<3.0 - 3.2	<3.1
Near roads used by TNT plant	1	2.2	2.2	1	15	15	1	<3.0	<3.0
Around Ash Pond	3	<2.0 - 13	5.8	3	120 - 180	150	2	<3.0 - 290	<147

[a] Concentrations rounded to two significant figures. Samples taken in 11/75. The table gives only those results that were reported either as definite concentration values or as concentrations less than some value (Niedermeyer 1976). The results for most samples were reported as none detected or as not analyzed due to interfering materials.

Source: Niedermeyer (1976).

290 ppb TNT (Table H.15). In general, measured subsurface soil concentrations were higher than surface soil concentrations. Presumably, this is due to the protection of subsurface materials from direct solar radiation, which helps break down some of the organic compounds (Niedermeyer 1976).

Dinitrotoluene (DNT) and TNT were detected in only a few water samples. One sample obtained from standing water had concentrations of 2,4-DNT, 2,6-DNT, and TNT of 40 ppb, 7.1 ppb, and 1.5 ppb, respectively. Results of TNT measurements for most other samples were reported as "none detected" or as < 0.6 ppb. Four samples had TNT concentrations of 1.2, 1.5, 1.6, and 2.2 ppb. The results also indicate that some of the nitrated toluenes have been leached from the soil (Niedermeyer 1976). It is reported that examination of soil excavated from locations containing soil discolored by the presence of various nitrated toluene compounds contained DNT and TNT at concentrations that are below those considered to represent either a safety hazard (explosive or fire) or toxicological hazard to humans or wildlife (Niedermeyer 1976).

H.2.4 Vicinity Properties

No data are available on concentrations of nonradiological parameters in the vicinity properties soils. Concentrations of nonradiological chemical parameters measured in surface water and groundwater are given in Table H.16. The locations of the observation wells (the 08 wells between the quarry and Femme Osage Slough) from which groundwater samples were taken are included in Figure H.5 because uranium was also measured in these wells.

For several elements -- arsenic, lead, manganese, and selenium -- the average concentrations measured in groundwater between the quarry and slough in 1979 through 1981 are at or above the state groundwater limits (Table H.12). (The groundwater limit applies because water in boreholes between the quarry and slough may enter the river alluvium aquifer.) For other species -- barium, copper, zinc, nitrate, and sulfate -- the 1979-1981 averages are below the groundwater limits, but the upper limits of the concentration ranges are above the groundwater limits. For elements such as arsenic, lead, and selenium, the results are not meaningful because the reported detection limits are equal to or greater than the state groundwater limits. Concentrations of arsenic in water samples from the slough exceed Missouri limits for all uses.

According to 1985 data for groundwater in the alluvium between Femme Osage Slough and the quarry, average concentrations of copper, lead, manganese, and zinc are above the groundwater limits (Table H.12). Concentrations of arsenic and selenium are below the Missouri groundwater limits.

However, the 1985 data are for samples collected from only three observation wells and are thus less representative of the near alluvium area than are the 1979-1981 data.

In the 1985 data for groundwater in the alluvium between Femme Osage Slough and the Missouri River, the average concentrations of manganese and zinc are also in excess of state groundwater limits. For arsenic and copper, the average concentrations are below the groundwater limits, but the upper limits of the concentration ranges are at or above the groundwater limits. For the other elements listed in Table H.16, the ranges and averages are below state groundwater limits.

The data in Table H.16 show the presence of contaminants in groundwater at concentration values that are at or above state groundwater limits for the Femme Osage Slough and for the alluvium between the quarry and Femme Osage Slough. It is not clear at present whether these concentrations represent contaminants that are being actively leached from the quarry, are background values, or are a result of the high detection limits (arsenic and selenium). The elevated manganese concentrations (above the groundwater limit) probably represent background values because the groundwater in the river alluvium has elevated concentrations of manganese. One well to the north (upgradient) of the quarry has high concentrations of barium. Also, the same area south of the quarry that has elevated groundwater concentrations of uranium has elevated concentrations of chloride, sulfate, nitrate, sodium, and manganese (Berkeley Geosci. Assoc. 1984). These results suggest that other elements besides uranium are being actively leached and transported from the quarry. However, the complexity of the chemical transport of elements in the fractured limestone and adjacent alluvium is indicated by the fact that two adjacent boreholes have quite different chemical concentrations (Berkeley Geosci. Assoc. 1984).

Very few measurements of concentrations of organics are available for the vicinity properties, including TNT or its process intermediates. The only available data are for water in Wells OB-6 and OB-10 between Femme Osage Slough and the limestone cliffs near the quarry (see Figure I.10 in Appendix I for well locations). The total organic halide concentrations measured in 1985 (in μg/L of chloride) were 36 for Well OB-6 and 71 for Well OB-10. Concentrations of TNT were reported as 377 μg/L in OB-6 and <0.5 μg/L in OB-10. The compounds 2,6-DNT and 2,4-ONT were not found (Hengerson 1985). The TNT concentration in Well OB-6 is above an interim environmental criterion of 44 μg/L for protection of public health (U.S. Dept. Army 1980). It is also documented that two off-site locations (i.e., Schote and Dardenne creeks) were contaminated during operation of the TNT plant (Niedermeyer 1976).

Table H.16. Concentrations of Chemical Species in Surface Water and Groundwater at the Vicinity Properties

Chemical Species	Surface Water Concentration (mg/L)[a]				Groundwater Concentration (mg/L)[a]	
	Little Femme Osage Creek	Femme Osage Slough		Little Femme Osage Slough[b]	Burgermeister Spring[e]	Busch Area Headquarters[e]
		1979[d]	1984[e]			
Arsenic	0.10	0.14	0.002	0.17	<0.001	<0.001
Boron	0.05	0.083	-	0.09	-	-
Barium	-	-	0.17	-	0.15	0.13
Cadmium	-	-	0.001	-	<0.001	0.001
Calcium	33	52	78	73	120	50
Chromium	-	-	<0.001	-	<0.001	<0.001
Copper	-	-	<0.001	-	<0.001	<0.001
Iron	-	-	0.010	-	<0.003	<0.003
Lithium	0.01	0.003	0.012	0.005	0.077	0.008
Lead	-	-	<0.001	-	<0.001	<0.001
Magnesium	8	16	18	21	30	37
Manganese	0.04	0.056	0.39	0.003	0.004	0.008
Molybdenum	0.03	0.04	<0.01	0.05	<0.010	<0.001
Phosphorus (as P_2O_5)	0.2	0.23	-	0.2	-	-
Potassium	2.9	6.3	7.5	6	3.2	1.3
Selenium	-	-	<0.001	-	0.003	<0.001
Silicon (as SiO_2)	12	9.3	5.5	6	11	8.0
Sodium	17	11	9.1	15	47	5.5
Strontium	0.10	0.24	0.29	0.27	0.22	0.15
Tin	-	-	-	-	-	-
Zinc	0.01	0.01	0.014	0.01	0.017	0.040
Chloride	9.3	5.3[b]	6.9	-	37	1.3
Fluoride	-	-	0.3	-	0.20	0.20
Nitrate (as NO_3)	0.55[c]	2.5[c]	<0.44	-	240	<0.44
Nitrite (as NO_2)	-	-	<0.033	-	<0.033	<0.033
Sulfate	45	12[b]	24	-	48	18
Bicarbonate	126	238[b]	-	-	-	-
pH	7.8[c]	7.7[c]	8.2	-	6.9	7.3

Table H.16. Continued

Chemical Species	Groundwater Concentration (mg/L)[a]						
	Alluvium between Quarry and Slough				Alluvium between Slough and Missouri River, 1985[h]		Well No. 5 of Well Field[e]
	1979 - 1981[f]		1985[g]				
	Range	Average	Range	Average	Range	Average	
Arsenic	<0.1 - 0.2	0.11	<0.005 - 0.008	0.006	<0.005 - 0.079	0.022	0.001
Boron	<0.05 - 0.42	0.21	-	-	-	-	-
Barium	0.07 - 3.7	0.46	0.20 - 0.52	0.32	0.22 - 0.93	0.46	0.48
Cadmium	<0.01 - 0.1	<0.01	<0.005	<0.005	<0.005	<0.005	0.001
Calcium	67 - 260	160	140 - 250	200	78 - 166	130	110
Chromium	<0.01 - 0.036	0.012	<0.025	<0.025	<0.025	<0.025	<0.001
Copper	<0.01 - 0.057	0.015	0.02 - 0.28	0.11	0.01 - 0.02	0.011	<0.001
Iron	<0.02 - 12	0.64	2.1 - 20	11	3.0 - 15	8.8	7.0
Lithium	0.01 - 0.53	0.033	-	-	-	-	0.033
Lead	<0.05 - 0.1	<0.05	<0.01 - 0.20	0.082	<0.01	<0.01	<0.001
Magnesium	16 - 49	43	34 - 44	40	20 - 39	27	24
Manganese	0.07 - 4.1	1.2	0.21 - 1.94	1.0	0.6 - 2.6	1.5	0.81
Molybdenum	-	-	-	-	-	-	<0.010
Phosphorus (as P_2O_5)	<0.11 - 3.9	0.32	-	-	-	-	-
Potassium	3.2 - 20	8.4	0.2 - 11	4.4	2.8 - 7.2	5.3	4.1
Selenium	<0.1	<0.1	<0.005	<0.005	<0.005	<0.005	<0.001
Silicon (as SiO_2)	16 - 39	23	-	-	-	-	25
Sodium	11 - 98	37	12 - 80	42	8.0 - 37	16	15
Strontium	0.24 - 0.95	0.56	-	-	-	-	0.58
Tin	0.05 - 0.33	0.13	-	-	-	-	-
Zinc	<0.01 - 0.31	0.044	0.20 - 0.58	0.36	<0.1 - 0.67	0.25	0.009
Chloride	9.5 - 100	34	30 - 72	46	4 - 14	6.9	7.2
Fluoride	0.2 - 0.7	0.4	0.25 - 0.50	0.34	0.16 - 0.28	0.20	0.30
Nitrate (as NO_3)	<1 - 64	7.8	0.40 - 46	16	0.26 - 0.58	0.35	<0.44
Nitrite (as NO_2)	-	-	-	-	-	-	<0.033

Table H.16. Continued

Chemical Species	Groundwater Concentration (mg/L)[a]						
	Alluvium between Quarry and Slough				Alluvium between Slough and Missouri River, 1985[h]		Well No. 5 of Well Field[e]
	1979 - 1981[f]		1985[g]				
	Range	Average	Range	Average	Range	Average	
Sulfate	4 - 540	230	62 - 470	270	<10 - 92	34	42
Bicarbonate	180 - 760	460	-	-	-	-	-
pH	6.4 - 7.9	7.2	7.1 - 7.4	7.2	7.1 - 7.4	7.3	7.1

[a] Concentrations rounded to, at most, two significant figures.

[b] One sample collected 9/79 (Berkeley Geosci. Assoc. 1984).

[c] Source: National Lead Company of Ohio (1977). Samples collected 10/76.

[d] Average of three samples collected 9/79 (Berkeley Geosci. Assoc. 1984).

[e] Source: U.S. Geological Survey (1984). Sample collected 9/5/84. Concentrations reported as dissolved concentrations.

[f] Dates of sampling were 11/79 to 4/81. Many samples were taken from 14 observation wells (the OB series). Other elements for which concentrations were reported at the detection limit or less include nickel (≤0.01 mg/L), silver (≤0.01 mg/L), and cobalt (≤0.01 mg/L). Averages were calculated assuming upper limits are real values.

[g] Source: Hengerson (1985). Samples collected 5/14-15/85 from Wells OB-6, OB-10, and OB-11 before pump test was carried out by Layne Western on Well No. 8 in the county well field. Data are also available on concentrations during and after the pump test.

[h] Source: Hengerson (1985). Samples collected 5/14-15/85 from Wells 1-LW, 2-LW, 3-LW, 6-LW, and 11-LW through 14-LW before pump test was carried out by Layne Western on Well No. 8 in the county well field.

Sources: National Lead Company of Ohio (1977); Berkeley Geosciences Associates (1984); U.S. Geological Survey (1984); Hengerson (1985).

The two values of TNT concentrations obtained for Wells OB-6 and OB-10 are quite different from one another even though the wells are close to each other. This situation also exists for other chemical species in other nearby OB wells, which indicates that the transport of chemical elements in the fractured limestone and adjacent alluvium is complex (Berkeley Geosci. Assoc. 1984).

Some measurements of nitrate and chloride concentrations in surface water have been made at several off-site locations. Elevated concentrations of nitrate, 78 mg/L (in 1981) and 370 mg/L (in 1979 through 1980), have been found in the chemical plant process sewer outfall stream at the head of the main drainage ditch to the Missouri River (Weidner and Boback 1982; Bechtel Natl. 1983b). A value of 16 mg/L (one sample) was obtained close to the point where the ditch enters the Missouri River (Weidner and Boback 1982). This limited amount of data suggests that the nitrate concentration may be decreasing along the ditch with decreasing distance to the river.

Concentrations of nonradiological species in groundwater at other locations reported in Table H.16 are below the Missouri groundwater limits (Table H.12), with the exception of manganese in the water of Well 5 in the county well field and nitrate in Burgermeister Spring. Dye-tracing studies suggest the existence of a subsurface connection between the stream drainage out of Ash Pond and the Burgermeister Spring area (Dean 1985) (see Section 3.1.2.2).

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*In this reference list, (1) the term "personal communication" is used to indicate either a telephone conversation or a face-to-face conversation and (2) all letters and memos cited are on file in the Energy and Environmental Systems Division, Argonne National Laboratory; copies are available upon request from: J.M. Peterson, EES, Bldg. 362, Argonne National Laboratory, Argonne, Illinois 60439.

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APPENDIX I. GROUNDWATER MODELS, INPUT PARAMETERS, AND CALIBRATION OF CONTAMINANT TRANSPORT

I.1 ANALYTICAL MODEL FOR THE RAFFINATE PITS AREA AND THE HANFORD SITE

I.1.1 Model Description

The migration of radionuclides and nonradioactive chemicals near the Weldon Spring raffinate pits and the Hanford site was simulated using a modified solute transport code, AT123D, which was originally developed at Oak Ridge National Laboratory (Yeh 1981). AT123D solves the following solute transport equation analytically for a homogeneous medium under saturated conditions:

$$\frac{\partial C}{\partial t} = \frac{D_{xx}}{Rd}\frac{\partial^2 C}{\partial x^2} + \frac{D_{yy}}{Rd}\frac{\partial^2 C}{\partial y^2} + \frac{D_{zz}}{Rd}\frac{\partial^2 C}{\partial z^2} - \frac{\upsilon}{Rd}\frac{\partial C}{\partial x} - \lambda C + \frac{\dot{M}}{n_e Rd} \qquad (I.1)$$

where C is the dissolved concentration of the solute; D_{xx}, D_{yy}, D_{zz} are the longitudinal, transverse, and vertical dispersion coefficients; Rd is the retardation factor; υ is the average pore water velocity; λ is the radioactive decay constant; $\dot{M}$ is the source release rate; and n_e is the effective porosity. The modifications of AT123D include (1) incorporating a transient waste-leaching option, (2) extending the application of the code to saturated-unsaturated media, and (3) extending the application of the code to macropore/fracture conditions. Detailed descriptions of the modifications follow.

I.1.1.1 Transient Waste Leaching

The waste release rate (source strength) was calculated using a first-order leaching rate (exponential leaching) model (Baes and Sharp 1983). The source strength, $\dot{Q}(t)$, can be written as:

$$\dot{Q}(t) = \lambda_\ell \cdot \rho_b \cdot V \cdot C_d(t) \qquad (I.2)$$

where λ_ℓ is the solute (radionuclide or nonradioactive chemical) leach rate, ρ_b is the waste density, V is the waste volume, and $C_d(t)$ is the solute concentration at time t. The solute leach rate, λ_ℓ, can be calculated as (Baes and Sharp 1983):

$$\lambda_\ell = \frac{I}{\theta_e L_z Rd} \qquad (I.3)$$

where I is the infiltration rate, θ_e is the effective moisture content in the wastes, L_z is the depth of the waste field, and Rd is the solute retardation factor. The effective moisture content, θ_e, in Equation I.3 is the product of

effective porosity, n_e, and the moisture saturation ratio, R_s; R_s is defined as the fraction of the voids in a porous medium that is filled with water. The retardation factor is discussed in Section I.1.1.3. The solute concentration at any time t, $C_d(t)$, is obtained by solving the following two differential equations:

$$\frac{dC_p(t)}{dt} = -(\lambda_p + \lambda_{\ell p})\ C_p(t) \qquad (I.4)$$

and

$$\frac{dC_d(t)}{dt} = -(\lambda_d + \lambda_\ell)\ C_d(t) + \lambda_d\ C_p(t) \qquad (I.5)$$

where $C_p(t)$ is the parent radionuclide concentration, λ_p is the parent decay constant, $\lambda_{\ell p}$ is the parent leach rate, and λ_d is the daughter (radionuclide of interest) decay constant. For the radionuclides in the Weldon Spring wastes, only uranium-238 and radium-226 need to be analyzed (see Section I.1.2). For uranium-238, $C_p(t)$, λ_p, and $\lambda_{\ell p}$ are all zero. For radium-226, the leaching of thorium-230 (the parent radionuclide of radium-226) may be neglected because thorium-230 has a high distribution coefficient of about 60,000 mL/g (Gilbert et al. 1983). Therefore, Equation I.4 reduces to

$$\frac{dC_p(t)}{dt} = -\lambda_p\ C_p(t) \qquad (I.6)$$

Solving Equations I.5 and I.6 with initial source concentrations (pCi/g) of S_p and S_d for parent and daughter radionuclides, respectively, one obtains:

$$C_d(t) = S_d e^{-(\lambda_\ell + \lambda_d)t} + \frac{\lambda_d S_p}{\lambda_\ell + \lambda_d - \lambda_p}\left[e^{-\lambda_p t} - e^{-(\lambda_\ell + \lambda_d)t}\right] . \qquad (I.7)$$

I.1.1.2 Saturated-Unsaturated Media

The transport media near the raffinate pits area and the Hanford site may be modeled as shown in Figure I.1. The AT123D model was modified by a travel-time method described by Gilbert et al. (1983). The waste field was assumed to be a rectangular block with dimensions L_x, L_y, and L_z. The geometry of the contaminated layer was assumed to remain constant during vertical migration in the vadose zone. Precipitation falling on the waste field would infiltrate through the wastes and transport contaminants down through the unsaturated (vadose) zone into the saturated (aquifer) zone. After the contamination reached the water table, it would spread out, be diluted in the groundwater,

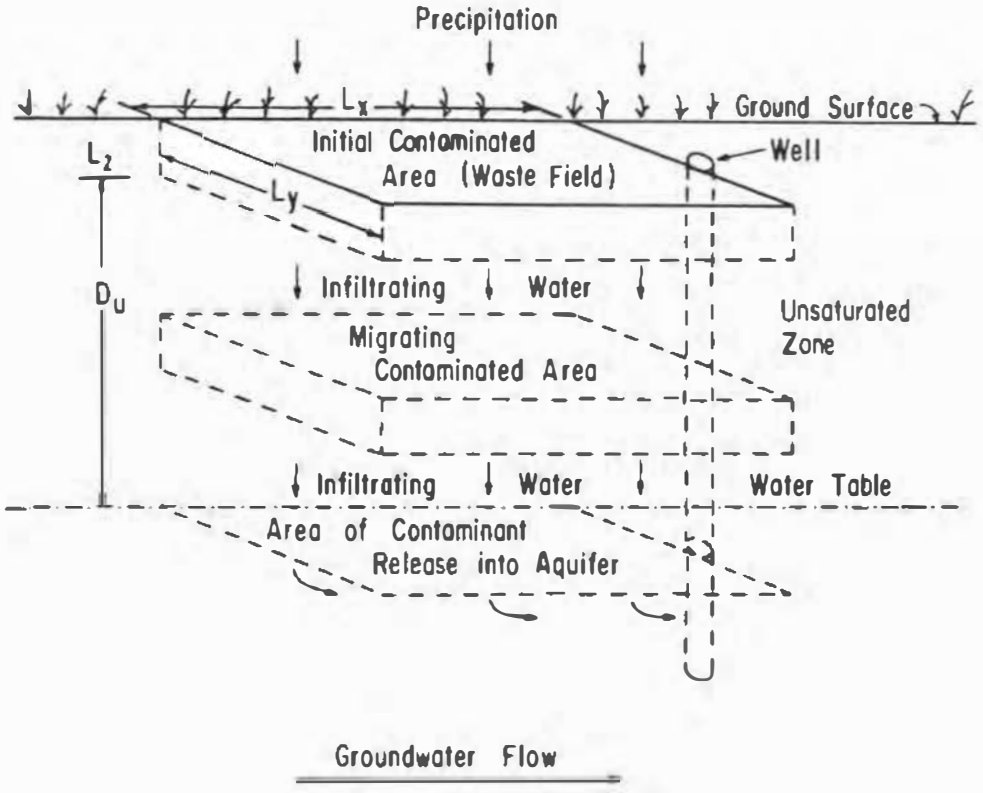

Figure I.1. Schematic Diagram of Model for Existing Contaminant Concentrations in Groundwater near the Weldon Spring Raffinate Pits Area and the Hanford Site. Source: Modified from U.S. Department of Energy (1986a).

and generate a contaminant plume that would move in the downgradient direction.

The travel-time method is extended to handle layered soil conditions in the vadose zone. The travel time, i.e., the time for contaminants to transport through the vadose zone, for an N-layered zone is calculated as

$$T_d = \sum_{j=1}^{N} L_j \theta_{e_j} Rd_j / I \qquad (I.8)$$

where T_d is the travel time, L_j is the jth layer thickness, θ_{e_j} is the jth layer effective moisture content, Rd_j is the jth layer retardation factor, and I is the infiltration rate. The retardation factor is discussed in Section I.1.1.3. For the Hanford site (Alternative 3a), a homogeneous (i.e., N = 1) vadose zone was assumed. The upper 2 m of the vadose zone was assumed to be homogeneous clay with an effective porosity of 5%; the lower 4 m was assumed to be composed of clay and clay till with blocky fractures (Bechtel

Natl. 1984b). For the lower 4 m layer, a lower effective porosity of 0.5% was chosen. This choice provides conservatism to the results because this assumes that water will move faster in the fractured clay than in the homogeneous clay; this was assumed even though water might not move through the fractures under unsaturated conditions.

Radioactive decay during transport through the vadose zone is considered in calculating the source term by multiplying the concentration by $\exp(-\lambda_d T_d)$, where λ_d is the radioactive decay constant and T_d is the travel time through the vadose zone.

I.1.1.3 Macropore and Fracture Conditions

The AT123D model was originally developed for a homogeneous medium. Therefore, modification was required for this analysis to account for the existence of blocky fractures in the clay and clay till overburden materials (see Section 3.1, Table 3.1) and the solution channels in the Burlington-Keokuk Formation limestone (see Section 3.1, Table 3.2) near the raffinate pits area. The modification was based on the Group-Transfer Concept (GTC) developed at the Pennsylvania State University (Yu 1984; Yu et al. 1985a). Application of GTC resulted in modification of the retardation factor used in the AT123D model. The retardation factor is calculated in AT123D using the equation

$$Rd = 1 + \rho_b \, Kd/n_e \qquad (I.9)$$

where ρ_b is the bulk density of the medium (kg/m^3), Kd is the distribution coefficient (see Section I.3), and n_e is the effective porosity. Based on the GTC model, the bulk density in Equation I.9 should be replaced by an effective density, which is defined as the quotient of the solid mass in the effective region* divided by the bulk volume. Because the solid mass in the effective region is always less than or equal to the bulk mass, the retardation factor calculated using Equation I.9 is an overestimate (i.e., nonconservative). An alternative approach is therefore required.

The GTC model defines the retardation factor for an arbitrary group, group i (e.g., the solution channels), as (Yu 1984; Yu et al. 1985a, 1985b, 1986)

$$Rd_i = 1 + \frac{f_i \rho_i}{\theta_i} Kd_i \qquad (I.10)$$

*The solid mass in the effective region is defined in the GTC model as the solid mass associated with the group of interest.

where f_i is the fraction of bulk volume associated with group i, ρ_i is the bulk density of the medium associated with group i, θ_i is the volumetric water content in group i, and Kd_i is the distribution coefficient for group i. It can be shown (Yu 1984; Yu et al. 1985a) that

$$\sum_{i=1}^{n} f_i \rho_i = \rho_b \qquad (I.11)$$

and

$$\sum_{i=1}^{n} \theta_i = \theta_t \qquad (I.12)$$

where n is the total number of groups, ρ_b is the bulk density of the entire medium, and θ_t is the total water content in the medium. Because the application of AT123D to a medium containing macropores or fractures was essentially the same as using the single group option of the GTC model to solve a multigroup (medium containing macropores/fractures) problem,* a set of effective parameters should be chosen to reflect this effect and obtain conservative estimates of contaminant concentrations. For n = 1 (single group), $f_1\rho_1 = \rho_b$ and $\theta_i = \theta_t$. That is,

$$Rd = 1 + \frac{\rho_b}{\theta_t} \quad Kd = 1 + \frac{\rho_b \, Kd}{n_t \, R_s} \qquad (I.13)$$

where n_t and R_s are the total porosity and saturation ratio of the medium, respectively.

When using Equation I.1 to estimate contaminant concentrations in a medium containing macropores or fractures, Equation I.13 is a more conservative way of calculating the retardation factor than Equation I.9. Equation I.13 has been incorporated in the modified version of AT123D. The modified AT123D will be referred to as MAT123D in the remaining discussions.

I.1.2 Source Term

The source term for the raffinate pits area is a major factor in determining the radionuclide concentrations in the groundwater system. Because of the relatively low thorium-232 source concentration, the high retardation for thorium, and the relatively short half-life of radium-228 and other radionuclides in the wastes (see Appendix H, Section H.1), it was

*The AT123D model is a special case of the GTC model with the number of groups, n, set equal to 1. The GTC model can solve macropore/fracture problems by setting n equal to 3 (multigroup) with the first group simulating the fracture zone, the second group simulating the homogeneous zone, and the third group simulating the dead-end zone (Yu 1984; Yu et al. 1985a).

concluded that the only radionuclides that needed to be analyzed individually to estimate radiological impacts were uranium-238 and radium-226. The ingrowth of radium-226 from thorium-230 was considered in the source leaching term (see Section I.1.1.1). The concentration of uranium-234 was assumed to be equal to the concentration of uranium-238.

The concentrations of radioactive and chemical contaminants in the wastes that were used in the MAT123D model are discussed in Section I.4.1 and Appendix H. Distribution coefficients are discussed in Section I.3. The source strength (contaminant release rate) was calculated using the method described in Section I.1.1 (see further discussion in Section I.4).

I.1.3 Hydrologic Parameter Values

The hydrogeologic parameter values used in the MAT123D model for calculation of contaminant migration at the raffinate pits area and the Hanford site are listed in Table I.1.* These parameters were selected to generate conservative estimates of contaminant concentrations in groundwater. For example, the infiltration rate for Alternative 4 was calculated from annual precipitation and evapotranspiration with the conservative assumption that surface runoff is negligible. For Alternatives 1 and 2a, the surface runoff coefficient is conservatively estimated to be 0.053 based on the conceptual design of the cell covers (Section 2.1, Figures 2.1 and 2.2). Hence, the infiltration rate for Alternatives 1 and 2a was calculated to be 0.05 m/yr. The infiltration rate for Alternative 3a, i.e., 0.01 m/yr, was adapted from a recent report (U.S. Dept. Energy 1985a) because the cell design for this alternative was also adapted from the same report (see Section 2.1, Figure 2.6). The hydrologic parameters such as precipitation and evapotranspiration were obtained from site-specific reports (Brown and Isaacson 1977; U.S. Nucl. Reg. Comm. 1981; Bechtel Natl. 1984b; U.S. Dept. Energy 1986b) or from the Water Atlas of the United States (Geraghty et al. 1973).

The groundwater flow characteristics -- such as hydraulic gradient, hydraulic conductivity, effective porosity, total porosity, unsaturated zone

*Only Alternatives 1, 2a, and 4 were modeled for the raffinate pits area. Groundwater impacts for Alternatives 2b and 3c are expected to be less than those predicted for Alternatives 1 and 2a because of (1) the lead sheet in the disposal cell cover for Alternative 2b (which will reduce infiltration for a period of time and therefore delay leaching of the wastes) and (2) the lower source concentration at the raffinate pits area for Alternative 3c (in which the raffinate sludge and quarry sludge will be removed to a uranium processing site).

Table I.1. Parameters Used in the MAT123D Model

Parameter	Symbol	Unit[a]	Raffinate Pits Area		Hanford Site
			Alt. 1 and 2a	Alt. 4	Alt. 3a
Annual precipitation	P	m/yr	0.94	0.94	0.16
Evapotranspiration rate	E	m/yr	0.84	0.84	0.10
Infiltration rate	I	m/yr	0.05	0.1	0.01
Total porosity:	n_t	-			
Wastes			0.23	0.88	0.23
Unsaturated zone			0.42	0.42	0.40
Saturated zone			0.30	0.30	0.40
Effective porosity:	n_e	-			
Wastes			0.23	0.88	0.23
Unsaturated zone[b]			0.005 (0.05)	0.005 (0.05)	0.2
Saturated zone			0.0015	0.0015	0.2
Dry bulk density:	ρ_b	kg/m^3			
Wastes[c]			1,700	320	2,100
Unsaturated zone			1,800	1,800	1,600
Saturated zone			2,400	2,400	1,600
Hydraulic gradient	i	-	0.0095	0.0095	0.002
Hydraulic conductivity	K	m/yr	1.2×10^2	1.2×10^2	6.0×10^4
Longitudinal dispersivity	α_L	m	10	10	3.0
Transverse dispersivity	α_T	m	1.0	1.0	0.3
Vertical dispersivity	α_V	m	1.0	1.0	0.3
Unsaturated zone depth	D_u	m	6	6	65
Saturation ratio:	R_s	-			
Wastes			1.0	1.0	1.0
Unsaturated zone			0.5	0.5	0.2
Saturated zone			1.0	1.0	1.0
Distribution coefficient[d]:	Kd	mL/g			
Uranium--					
Wastes			370	370	370
Unsaturated zone			370	370	3.7
Saturated zone			0	0	3.7
Radium--					
Wastes			1,200	1,200	1,200
Unsaturated zone			1,200	1,200	10
Saturated zone			100	100	10

[a] A hyphen means that the parameter is dimensionless.

[b] The effective porosities for the clay overburden under the pits area are assumed to be 0.05 for the top 2 m and 0.005 for the bottom 4 m to account for the blocky fractures that occur in the clay and clay till (see Section 1.1.1).

[c] The waste density for Alternatives 1 and 2a is the average density of the stabilized sludge, quarry wastes, and vicinity properties wastes. For Alternative 4, the density is that obtained by removing the water from the sludge and keeping the volume fixed at the value for the wet sludge. For Alternative 3a, the density is the average of the values for the dried sludge, the quarry wastes, and the vicinity properties wastes.

[d] The distribution coefficients for chemicals used in the MAT123D model are discussed in Section 1.3.2.1.

depth, and various dispersivities -- were obtained primarily from site-specific reports (Brown and Isaacson 1977; Exxon Nucl. Co. 1977; Routson et al. 1981a, 1981b; Bechtel Natl. 1983, 1984a, 1984b; Berkeley Geosci. Assoc. 1984; U.S. Dept. Energy 1986b). When site-specific data were not available, generic values were used (Till and Meyer 1983; Natl. Counc. Radiat. Prot. Measure. 1984; Sheppard et al. 1984). The MAT123D model requires values for hydraulic gradient, hydraulic conductivity, and various dispersivities only for the saturated zone.

The waste-field dimensions used in MAT123D for the analysis of the raffinate pits area and the Hanford site are shown in Table I.2. For analyzing the radioactive contaminants, the waste field was assumed to be a rectangular block with a total volume equal to the sum of the volumes of the raffinate sludge, quarry wastes, and vicinity properties wastes. For analyzing the chemical contaminants, the total volume was taken to be the sum of the volumes of the raffinate sludge and quarry wastes. Chemical plant wastes were excluded from the radiological contaminant analysis because average concentrations of uranium-238 (15 pCi/g) and radium-226 (2 pCi/g) are much lower than the average concentrations in the other waste components. Also, these values are not sufficiently higher than background to have an appreciable impact. Chemical plant and vicinity properties wastes were excluded from the chemical contaminant analysis because data on the chemical composition of these wastes are not available. However, it is expected that the concentrations of chemicals will be less than in the raffinate sludge and quarry wastes because most of the chemical contaminants present in the vicinity properties and chemical plant wastes are expected to result from spillage whereas the raffinate sludge and some of the quarry wastes are waste products from chemical processing. Predicted concentrations of contaminants in groundwater using a smaller waste field of more concentrated wastes are not expected to be significantly different from concentrations using a larger waste field for which average concentrations may be somewhat lower.

I.1.4 Modeling Results

Inputs to the MAT123D model were the hydrogeologic parameters listed in Table I.1, waste-field dimensions listed in Table I.2, and source concentration data. The results and sensitivity analyses of some key parameters are presented in this section.

I.1.4.1 Travel Times

The water and radionuclide travel times from the bottom of the wastes through the unsaturated (vadose) zone to the water table, as determined by use

Table I.2. Dimensions of the Waste Field Used in the MAT123D Model[a]

Alternative	L_x Length (m)	L_y Width (m)	L_z Depth (m)
Raffinate Pits Area			
Alternative 1[b]:			
Radioactive contaminants	240	240	5.7
Chemical contaminants	230	230	5.7
Alternative 2a[b]:			
Radioactive contaminants	200	200	8.3
Chemical contaminants	190	190	8.3
Alternative 4[b]:			
Radioactive contaminants	240	240	3.0
Chemical contaminants	240	240	3.0
Hanford Site			
Alternative 3a[b]:			
Radioactive contaminants	240	240	2.4
Chemical contaminants	230	230	2.4

[a] Values are reported to two significant figures.

[b] Waste volumes are different for the different alternatives because of the properties of the raffinate sludge. For Alternative 4, the volume is the original wet volume (Appendix H, Table H.1); for Alternatives 1 and 2a, the raffinate volume is larger because stabilizer has been added; for Alternative 3a, the raffinate volume is smaller because the sludge has been dried for transportation to Hanford.

Conversion Factor: To convert meters (m) to yards (yd), multiply by 1.0936.

of parameters listed in Table I.1, are shown in Table I.3. The water travel times were calculated using Equations I.8 and I.13, with Rd_j equal to unity. The calculated water travel time for the raffinate pits area is 1.2 years for the action Alternatives 1 and 2a and 0.6 years for the no-action Alternative 4. The water travel time for the Hanford site, Alternative 3a, is 260 years based on the parameters listed in Table I.1. The calculated radionuclide travel time for all alternatives is greater than 1,900 years for uranium-238 and greater than 6,300 years for radium-226. The radionuclide travel time is greater than the water travel time due to the sorptive properties of soil.

Table I.3. Water and Radionuclide Travel Times[a] for the Raffinate Pits Area and Hanford Site

Alternative[b]	Travel Time (yr)		
	Water	Uranium-238	Radium-226
Raffinate Pits Area			
1	1.2	3,900	13,000
2a	1.2	3,900	13,000
4	0.6	1,900	6,300
Hanford Site			
3a	260	20,000	52,000

[a] Results rounded to two significant figures.

[b] Parameters used are listed in Table I.1.

Radionuclide travel time is a function of several parameters including distribution coefficient, infiltration rate, and moisture content. The most sensitive parameter among these parameters is the distribution coefficient (see Section I.3). Hence, a sensitivity analysis was carried out to show to what extent the travel times will decrease if the distribution coefficients are decreased. The results of the sensitivity analysis indicate that the travel times for uranium and radium in the unsaturated zone at the raffinate pits area are sensitive to the Kd values used in the analysis. If the lowest reported Kd value of 12 mL/g for uranium in clayey soil in the raffinate pits area (Seeley and Kelmers 1985b) is used rather than the mean value of 370 mL/g (see Section I.3.1.1), uranium is predicted to reach the groundwater in limestone within 130 years for Alternatives 1 and 2a and within 64 years for the no-action Alternative 4 (Table I.4). The resulting uranium concentrations in groundwater are discussed in Section I.1.4.2. If the lowest reported value of 660 mL/g for radium (Seeley and Kelmers 1985b) is used instead of the mean value of 1,200 mL/g (see Section I.3.1.1), radium is still predicted to remain within the unsaturated zone for more than 1,000 years for Alternatives 1, 2a, and 4 (Table I.4).

For the Hanford site, the sensitivity analysis for uranium was conducted using a Kd value of 18 mL/g for uranium in the wastes instead of 370 mL/g (18 mL/g is the same Kd value used for analysis of leaching from the quarry wastes--see Section I.3.1.2) and a Kd value of 0 mL/g for uranium in the

Table I.4. Sensitivity Analysis Results: Radionuclide Travel Times Using Lower Distribution Coefficients[a]

Alternative[b]	Travel Time (yr)	
	Uranium-238	Radium-226
Raffinate Pits Area	Kd = 12 mL/g (wastes and vadose zone)	Kd = 660 mL/g (wastes and vadose zone)
1	130	6,900
2a	130	6,900
4	64	3,500
Hanford Site	Kd = 18 mL/g (wastes) 0 mL/g (vadose zone)	
3a	260	-[c]

[a] Results are rounded to two significant figures.

[b] Parameters used (except Kd) are listed in Table I.1.

[c] No sensitivity analysis was conducted for radium at the Hanford site (see text).

unsaturated sandy soils instead of 3.7 mL/g. These are very conservative assumptions. For example, a Kd of 0 mL/g is the most conservative assumption that could be made if negative adsorption (anion exclusion) can be neglected (Jester et al. 1985; Yu et al. 1986). The results indicate that uranium would be predicted to reach the groundwater in 260 years (Table I.4). No sensitivity analysis was done for radium at the Hanford site because the Kd of 10 mL/g that was used for the sandy soils is already very conservative (Section I.3.1.1) and the predicted travel time (52,000 years) is very long.

I.1.4.2 Radionuclide Concentrations

The time and spatial variations of radium-226 and uranium-238 concentrations in groundwater were calculated using the MAT123D model. Based on model simulations using the parameter values given in Section I.1.3, the concentrations of radium-226 and uranium-238 in groundwater would be negligible at both the Weldon Spring raffinate pits area (for all alternatives) and the Hanford site (Table I.5). The groundwater in the hypothetical on-site well at the waste-field boundary would not be contaminated in 1,000 years. Uranium would have the shortest travel time; but, even under the no-action Alternative 4, the predicted mean travel time is 1,900 years (see Table I.3). Thus, the contaminated region would remain in the unsaturated zone for at least 1,000 years. For all alternatives, the maximum concentrations of

Table I.5. Radionuclides in Hypothetical On-site Wells[a]

Alternative[b]	Uranium-238 Maximum Concentration (pCi/L)[c]	Uranium-238 Time to Reach Maximum Concentration[d] (yr)	Radium-226 Maximum Concentration (pCi/L)	Radium-226 Time to Reach Maximum Concentration[d] (yr)
Raffinate Pits Area				
1	0.22	3,900	0.0004	22,000
2a	0.18	3,900	0.0004	22,000
4	2.1	1,900	0.12	14,000
Hanford Site				
3a	0.18	20,000	0.00005	61,000

[a] Wells are assumed to be located next to the waste-disposal area.

[b] Parameters used are listed in Table I.1.

[c] To obtain natural uranium concentration, multiply uranium-238 concentration by 2.046.

[d] Includes the travel times for nuclides to move from the bottom of the waste field to the water table (see Table I.3 for radionuclide travel times).

radium-226 and uranium-238 in on-site wells are predicted to occur thousands of years in the future and are predicted to be considerably below DOE limits for uncontrolled areas, 100 pCi/L for radium-226 and 600 pCi/L for uranium-238 (U.S. Dept. Energy 1986c). Although the EPA drinking water standards do not apply, the concentrations would also be below the standard of 5 pCi/L for radium-226 (40 CFR Part 141.15) and the recommended standard of 10 pCi/L for uranium (including uranium-234 and -238) in drinking water (Cothern et al. 1983).

A travel time sensitivity analysis (Section I.1.4.1) that uses lower distribution coefficients for uranium results in a travel time of less than 1,000 years. Using the lowest Kd values, the maximum uranium-238 concentrations would be below the DOE limit of 600 pCi/L for all alternatives (Table I.6).

I.1.4.3 Chemical Concentrations

The predicted chemical concentrations are discussed in Section 4.1.2.4. The contributions of chemical species from the chemical plant and vicinity properties wastes were not specifically addressed because the concentrations

Table I.6. Sensitivity Analysis Results: Uranium-238 Concentrations Using Lower Distribution Coefficients[a]

Alternative[b]	Distribution Coefficient, Kd (mL/g)			Maximum Uranium-238 Concentration (pCi/L)	Time to Reach Maximum Concentrations (yr)
	Waste	Vadose Zone	Aquifer		
Raffinate Pits Area					
1	12	12	0	6.7	130
2a	12	12	0	5.4	130
4	12	12	0	61	64
Hanford Site					
3a	18	0	0	2.7	260

[a] Results rounded to two significant figures.

[b] Parameters used (except Kd) are listed in Table I.1.

of chemicals in these wastes are not available. The sensitivity of the results to neglecting these potential sources was tested by doing calculations for Alternative 1 using the conservative assumption that concentrations of chemicals in the chemical plant and vicinity properties materials were the same as the average concentration for the stabilized raffinate sludge and quarry wastes. For Kd = 3 mL/g (see Section 1.3.2.1), predicted peak concentration contributions of arsenic, selenium, and cadmium in the limestone groundwater are about 35% higher but still below the EPA and Missouri state groundwater limits (see Appendix H, Table H.12). Thus, the prediction that regulatory limits for the groundwater beneath the pits would be met for 1,000 years is probably not sensitive to the lack of data on chemicals in the vicinity properties and chemical plant wastes. DOE will be gathering additional data characterizing the chemical properties of the chemical plant wastes to support detailed engineering activities.

The model results are sensitive to the choice of Kd values in the raffinate sludge. The values chosen here (see Section I.3.2) are based on literature values because no experimental data are available for the raffinate sludge. However, for Kd values greater than zero, if the actual Kd values were lower by a factor of two, the predicted peak concentration contributions would increase by a factor of two.

I.2 NUMERICAL MODEL FOR THE QUARRY AREA

I.2.1 Model Description

The numerical model used for the quarry is a pseudo-three-dimensional, finite-element groundwater model. This model was originally developed by Tracy and Carlton (1982) and has been modified and used to simulate the groundwater flow system and contaminant migration patterns at various locations (Ertec Western 1982; Tsai et al. 1985). Basically, the model employs the Galerkin finite-element method to solve the following time-dependent groundwater flow and solute transport equations.

$$[S_y + S_s\,(h-z_1)]\,\frac{\partial h}{\partial t} - \nabla \cdot [K\,(h-z_1) \cdot \nabla h] = Q + q \qquad \text{(I.14)}$$

and

$$\theta b\,\frac{\partial C}{\partial t} - (T \cdot \nabla h) \cdot \nabla C - \nabla \cdot (\theta b D \cdot \nabla C) + Q_1 C = Q_1 C' - b\,\rho_s (1-\theta)\,Kd\,\frac{\partial C}{\partial t} + \theta b \lambda C \qquad \text{(I.15)}$$

where S_y is the specific yield; S_s is the specific storage; h is the hydraulic head; z_1 is the bottom of the aquifer unit; K is the hydraulic conductivity tensor; Q is the volumetric recharge per unit area from above; q is the recharge from leakage from below; t is the time; θ is the aquifer porosity; b is the aquifer thickness; C is the concentration of the substance in water; T is the transmissivity tensor; D is the dispersion tensor; Q_1 is the volumetric flux per unit area, which includes the leakage term; C' is the concentration of the substance in the source; ρ_s is the density of the solid in aquifer matrix; Kd is the distribution coefficient; and λ is the radioactive decay constant. Equations I.14 and I.15 were derived based on the principles of conservation for mass and momentum and on some basic assumptions for the movement of groundwater through the modeled aquifer (Tracy and Carlton 1982). The equations were solved numerically by the Galerkin finite-element method (Zienkiewicz 1971). Detailed procedures for numerical solution are presented by Tracy and Carlton (1982).

The model selected for the current study was used to simulate several unique hydrogeologic features of the quarry area, including (1) leakage between the alluvium and limestone aquifers, (2) discharge from well field pumpage, (3) interaction of the aquifer with the slough and the Missouri River flow systems, and (4) variable properties and thickness of the alluvium. For this study, an aquifer system covering an area about 2.6 km × 2.0 km

(1.6 mi × 1.3 mi) was selected (Figure I.2). This area extends in the east-northeast direction. The longitudinal boundary of the study area (in the flow direction) was oriented such that the area is in approximate alignment with the principal groundwater flow direction and, therefore, simplifies the input data for the parameters that define the aquifer properties.

The selected area was then subdivided into an assemblage of discrete elements. The grid spacings chosen were 80.5 m (264 ft), and the entire study area was discretized into 832 elements with 858 nodal points. The groundwater flow system of the quarry area was modeled as a water-table aquifer with transient flow conditions. The simulation considered the potential leakage between the aquifers in both the alluvium and limestone. Recharge from precipitation was also taken into account.

I.2.2 Boundary Conditions

The boundary conditions imposed on the modeled area are indicated in Figure I.3. Because of the orientation of the grid system for the study area, a no-flux condition exists at portions of the boundaries. The in-flow boundaries were assumed to be a specified flux-type boundary. The flux was determined based on available data for flow conditions. The specified flow rates at the northern boundary vary from 2×10^{-5} to 8.5×10^{-6} m^3/s (7.2×10^{-4} to 3×10^{-4} cfs) from west to east; the flow rate at the western boundary varies from 0 to 8.5×10^{-3} m^3/s (0 to 0.3 cfs) from north to south. A constant-head condition was assumed for the Missouri River boundary. The surface water elevations in the river vary from about 137.2 to 136.3 m (450 to 447 ft) MSL over the modeled distance of about 2.6 km (1.6 mi). Other parameters used for modeling and their sources are discussed in the Section I.2.3.

I.2.3 Hydrogeologic Parameter Values

The hydrogeologic parameters used for modeling the groundwater flow system and contaminant migration patterns at the quarry area are summarized in Table I.7. Selection of the values for model parameters and the data sources are briefly discussed below.

The elevation of the land surface in the modeled area was obtained from topographic maps for the Defiance and Weldon Spring quadrangles (U.S. Geol. Surv. 1972, 1974). The topography surrounding the quarry area is rugged and heavily wooded, and -- with the exception of a floodplain of the Missouri River to the south -- is characterized by deeply dissected hills and deep ravines (Chapter 3, Section 3.1.1). The ground surface elevations around the modeled area range from about 137 m (450 ft) MSL at the Missouri River

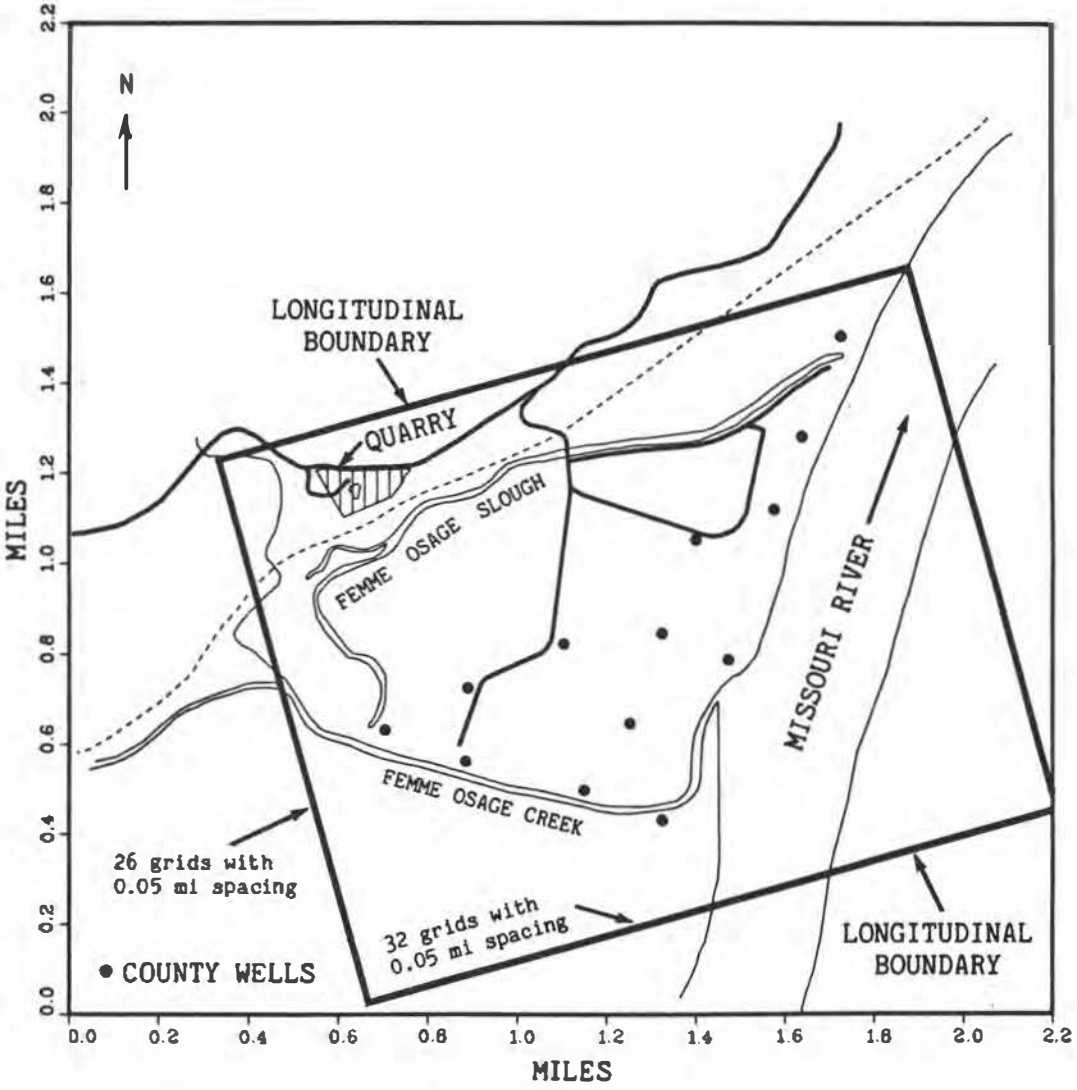

Figure I.2. Selected Area for Groundwater Flow Simulation.

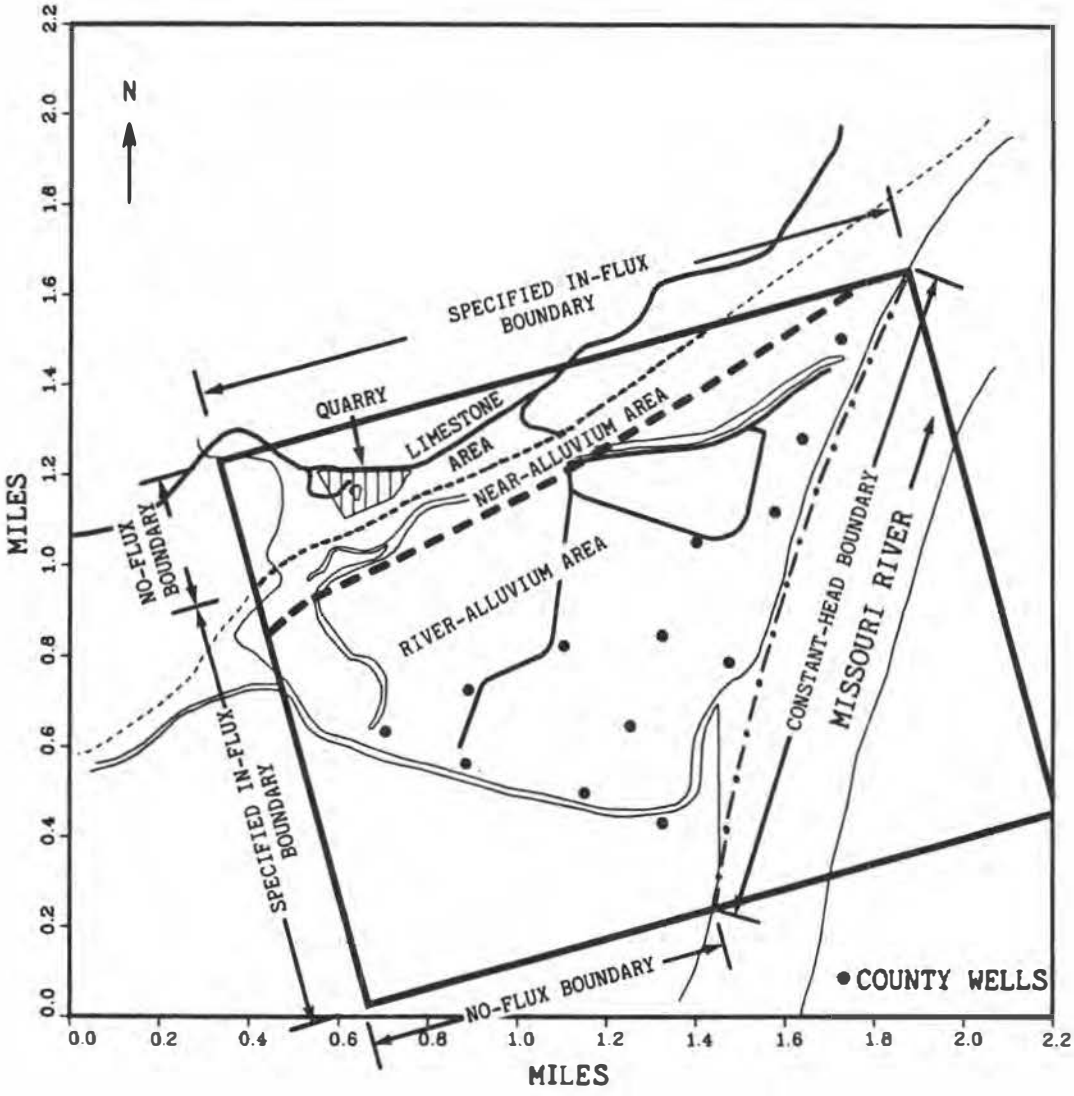

Figure I.3. Specified Boundary Conditions and Aquifer Regions for Quarry Site Modeling.

Table I.7. Data Requirements for Model Parameters at the Quarry Area

Parameter	Input Data	Unit	Data Source
Land surface elevation	a/	m MSL	U.S. Geol. Surv. (1972, 1974)
Starting groundwater elevation	Figure 1.4	m MSL	Berkeley Geosci. Assoc. (1984); Kleeschulte and Emmett (1986); Layne Western (1986)
Aquifer base elevation	Figure 1.5	m MSL	Berkeley Geosci. Assoc. (1984); Layne Western (1986)
Hydraulic conductivity for near alluvium[b]	2.3×10^{-4}	m/s	Calculated[b]
Hydraulic conductivity for river alluvium[b]	1.1×10^{-3}	m/s	Calculated[b]
Transmissivity for limestone[c]	4.25×10^{-5}	m^2/s	Berkeley Geosci. Assoc. (1984)
Effective porosity for river alluvium	0.3	-[d]	Berkeley Geosci. Assoc. (1984)
Effective porosity for near alluvium[b]	0.23	-	Baes and Sharp (1983)
Effective porosity for limestone	1.5×10^{-3}	-	Berkeley Geosci. Assoc. (1984)
Permeability for limestone[e]	4×10^{-13}	m^2	Berkeley Geosci. Assoc. (1984)
Hydraulic conductivity for limestone	3.88×10^{-6}	m/s	Calculated[f]
Hydraulic gradient for limestone	0.006	-	Calculated[b]
Storage coefficient for alluvium	0.3	-	Todd (1980); Berkeley Geosci. Assoc. (1984)
Storage coefficient for limestone	1.5×10^{-3}	-	Todd (1980); Berkeley Geosci. Assoc. (1984)
Longitudinal dispersivity	66	m	Berkeley Geosci. Assoc. (1984)
Transverse dispersivity	6.6	m	Berkeley Geosci. Assoc. (1984)
Annual precipitation	94	cm	Berkeley Geosci. Assoc. (1984)
Annual evapotranspiration	84	cm	Geraghty et al. (1973)

[a] See text for description.

[b] See text for details.

[c] Value is average of measured steady-state values at observation Wells OB-13, OB-14, and OB-15 (Berkeley Geosci. Assoc. 1984).

[d] A hyphen means that the parameter is dimensionless.

[e] Derived from measurements in the fractured limestone (Berkeley Geosci. Assoc. 1984).

[f] The hydraulic conductivity K is related to the permeability k by $K = k\rho g/\mu$ where ρ is the density of water, μ is the dynamic viscosity of water, and g is the acceleration of gravity (Freeze and Cherry 1979).

floodplain to about 168 m (550 ft) MSL at the quarry. The quarry floor is at an elevation of about 136 m (446 ft) MSL.

Groundwater levels in wells of the county well field and in observation wells around the quarry were measured as follows: on September 5, 1980, by Berkeley Geosciences Associates (1984); on October 30, 1984, by the U.S. Geological Survey (Kleeschulte and Emmett 1986); and on May 28, 1985, by Layne Western (1986). Examination of the measured results indicate that none of the water-level data covered the entire modeled area. Berkeley's measurement was limited to the quarry limestone area whereas the other two measurements were performed primarily for the river alluvium area. Steady-state calibration of the groundwater flow model for this analysis is based on Layne Western's water-level data, adjusted by U.S. Geological Survey data and by Berkeley data for the quarry area. The calibrated results with four wells pumping were compared with the test results performed by Layne Western for similar pumping conditions, and the results were similar. Therefore, the calibrated results (Figure I.4) were used as the initial flow condition for subsequent modeling.

For modeling purposes, the alluvium was divided into two components: (1) the near alluvium lying between Femme Osage Slough and the limestone cliffs and (2) the river alluvium lying between the slough and the Missouri River (Figure I.3). This division was based on measurements showing that the near alluvium component is less permeable than the river alluvium (Berkeley Geosci. Assoc. 1984; Layne Western 1986). The hydraulic conductivity of the near alluvium, 2.3×10^{-4} m/s (7.5×10^{-4} ft/s), was calculated from the measured transmissivity of 1.15×10^{-3} m^2/s (8,000 gpd/ft) for alluvium (Layne Western 1986) and an average depth of 5 m (16 ft) (Berkeley Geosci. Assoc. 1984). The average hydraulic conductivity of the river alluvium, 1.1×10^{-3} m/s (3.6×10^{-3} ft/s), was calculated from the measured transmissivity and saturated alluvium thickness at various test wells (Layne Western 1986). A review of the test hole logs indicated that the alluvium thickness is about 30 m (100 ft) in the middle portion of the well field and decreases to the north toward Femme Osage Slough (Layne Western 1986). The spatial variation of the alluvium thickness was considered in the model. The alluvial aquifer base elevations are shown in Figure I.5.

Contaminated groundwater from the quarry area is expected to flow in the Kimmswick Formation (Chapter 3, Section 3.1.1) and eventually discharge into the near alluvium, with minimal infiltration into the underlying Decorah Formation. The characteristics of the Decorah Formation are not completely known, although it is believed that solution channels and fractures may not exist in this formation (Layne Western 1986). For modeling purposes, various

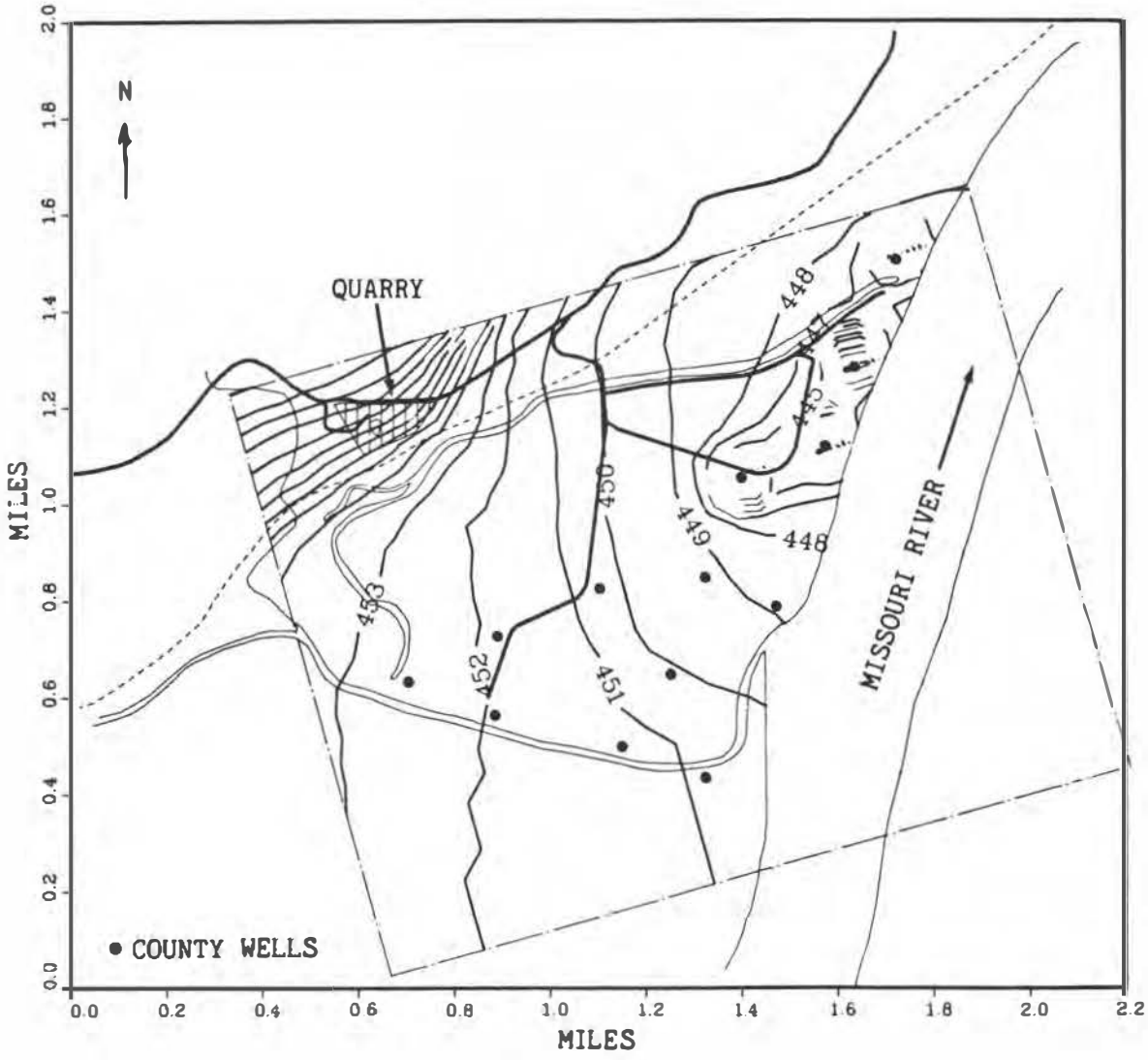

Figure I.4. Calibrated Groundwater Levels (ft above MSL) with Four County Wells Pumping. Conversion Factor: To convert feet to meters, multiply by 0.3048.

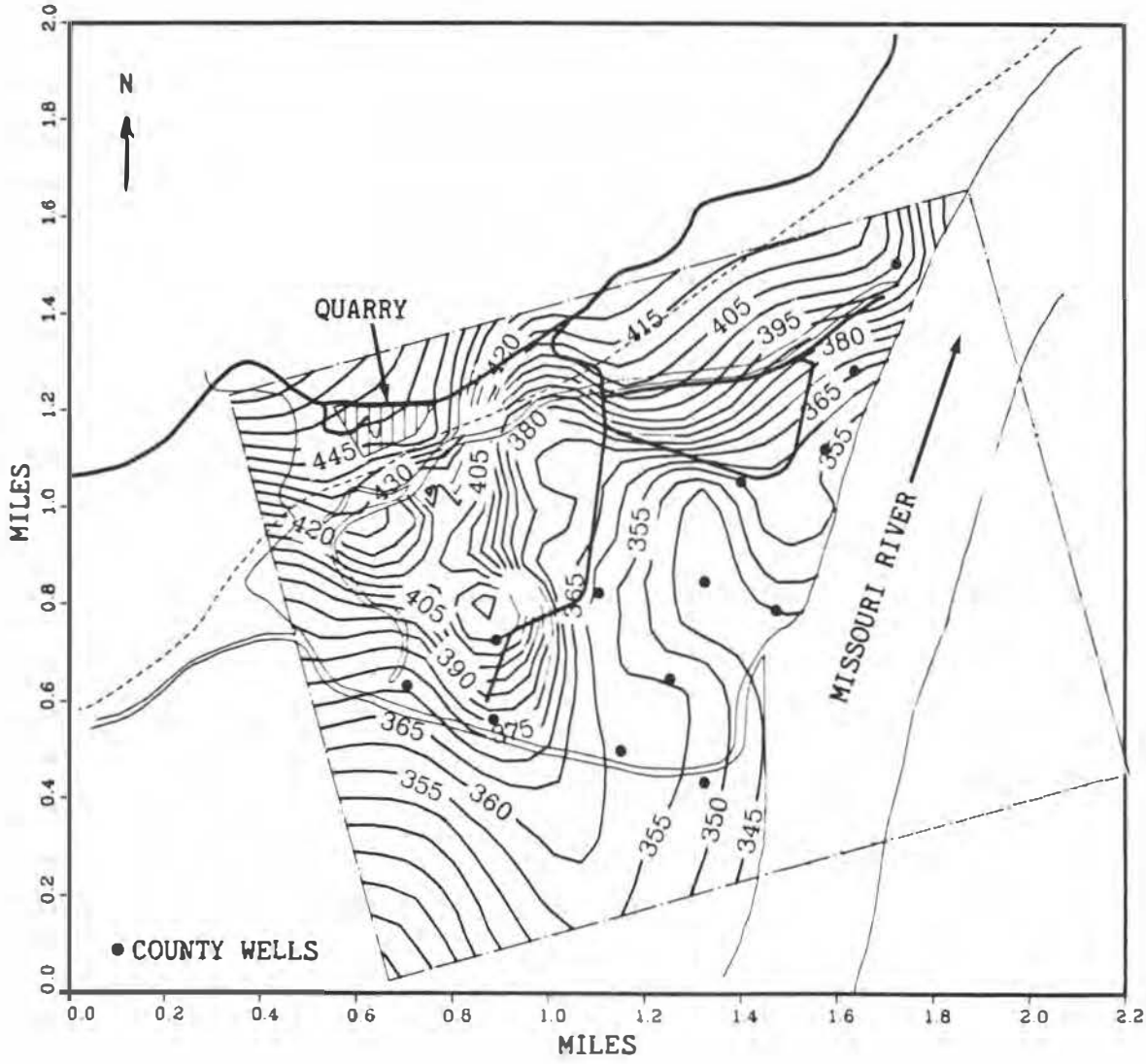

Figure I.5. Aquifer Base Elevations (ft above MSL) at the Quarry Area. Conversion Factor: To convert feet to meters, multiply by 0.3048.

ratios of discharge from the Kimmswick Formation into the alluvium to discharge from the Kimmswick Formation into the deeper limestone were considered (100/0, 90/10, and 50/50). The discharge into the deeper limestone would migrate and disperse in the Plattin Formation (Chapter 3, Section 3.1.1) underlying the river alluvium. Flow in the alluvium and Plattin Formation might interchange depending on the piezometric head difference in the two strata. The potential head in the limestone aquifer in the Plattin Formation is assumed to be the same as the initial head in the alluvial aquifer.

The results indicate that, for different discharge ratios, the predicted contaminant concentrations in the county well field reflect mostly the direct discharge into the alluvium and do not show significant influence from the contaminants being transported in the underlying bedrock. The contaminant interchange between the alluvial and underlying bedrock aquifers appears to be minimal because (a) the contaminant plume generally moves with groundwater flow and stays mostly in the northwestern region of the well field and (b) pumpings at the well field do not seem to significantly influence the water table in the plume area or the head differences in the adjacent aquifers. For modeling purposes, a conservative and perhaps more realistic discharge ratio of 90/10 was used to predict spatiotemporal concentration distributions in the well field area.

Femme Osage Slough, which is about 150 m (500 ft) south of the quarry, is connected to the Missouri River by a drain pipe at the eastern end of the slough. Previous studies (Natl. Lead Co. Ohio 1977; Berkeley Geosci. Assoc. 1984; Kleeschulte and Emmett 1986) indicated that the slough is also hydraulically connected to the quarry through the areal groundwater system. Groundwater flow in the area occurs under a natural hydraulic gradient from the quarry toward the slough through fractures in the limestone and then through the near alluvium between the quarry and the slough. Water levels in the slough may be affected by occasional flooding, but they generally reflect the groundwater table. For modeling, the slough was not considered to be a significant recharge or discharge boundary.

Flooding in the well field area occurs about once every 3 to 5 years, and the depth of inundation is about 1 m (3 ft) (Hovatter 1986a). The flooded areas require about 1 to 2 months to dry out. Drainage occurs through a 40-cm (16-in.) drain pipe connecting the slough and the Missouri River. During the dryout period, the standing water can infiltrate into the alluvium, resulting in some dilution of the contaminant plume. The effect of flooding on contaminant plume migration was evaluated. The results indicate that the contaminant concentrations for which the additional infiltration caused by flooding is considered would be only about 2% less than the contaminant

concentrations for which flooding is not considered. Therefore, flooding was neglected in the long-term modeling.

The values of the various transport properties for each of the aquifer components were based on data available from the literature. The values chosen for the transmissivities and hydraulic conductivities are discussed above and in footnotes to Table I.7. Measured values of effective porosity vary from 0.27 to 0.32 for the river alluvium and from 1.02×10^{-3} to 2.15×10^{-3} for limestone (Berkeley Geosci. Assoc. 1984); values of 0.3 and 1.5×10^{-3} were selected for the river alluvium and limestone, respectively. The effective porosity of the near alluvium, 0.23, was selected to be equal to the average field capacity for all soils (Baes and Sharp 1983). This value, which is somewhat lower than the porosity value of the river alluvium, was also selected for the quarry wastes. The hydraulic gradient in the limestone near the quarry, estimated from groundwater elevation contours (Berkeley Geosci. Assoc. 1984; Kleeschulte and Emmett 1986), is about 0.006. The longitudinal and transverse dispersivities were assumed to be 66 m (220 ft) and 6.6 m (22 ft), respectively, and to be constant at all locations (Berkeley Geosci. Assoc. 1984). For an unconfined aquifer, the storage coefficient equals the effective porosity (Todd 1980).

The climatological conditions in the study area are discussed in Chapter 3, Section 3.1.3. Monthly precipitation and evapotranspiration values were used in the model simulation. However, only annual data are presented in Table I.7.

Several county wells are located within the modeled area (Figures I.2 and 3.9 [Section 3.1.2]), and the effect of these pumping wells on contaminant plume migration was considered in the model. Based on information obtained from Hovatter (1985), it was assumed for modeling purposes that each well has a pumping capacity of 126 L/s (2,000 gpm) and that Wells 1, 2, 3, and 5 have been operating since wastes were placed in the quarry in 1963 and that Wells 6 through 12 were placed in service in 1986 to satisfy increased demand. Wells 4 and 13 were assumed to be unusable. Recently (summer 1986) changes were made so that Wells 3, 4, 6, and 7 are in service, and Wells 1, 8, and 9 are available for backup. Wells 2 and 5 will be available for backup after refurbishing (Hovatter 1986b). The difference in predicted chemical and radiological concentrations based on the revised well operating system and on the operating system assumed for the model is expected to be negligible because Well 4 and Wells 10 through 13 are close to the Missouri River and are thus expected to draw essentially all their water from the river with almost none coming from the quarry area.

I.2.4 Source Term

The source term for the quarry is quite complex; it is described in Section I.4.2. The schematized source used in the model was based on the distribution of wastes in the quarry in terms of both the depth of the wastes and the amount and location of wastes above and below the water table. The regions of different waste depths in the quarry, as determined by surface and subsurface borehole measurements, are shown in Figure I.6.

For each of the waste regions shown in Figure I.6, concentrations of radioactive contaminants were averaged together and the average was assumed to apply everywhere in the region. This is a major simplifying assumption because the borehole measurements sampled only the contaminated soils lying above the rubble (borehole augers were driven to refusal on the rubble or bedrock, whichever occurred first). Neither the rubble nor any contaminated waste below the rubble was sampled (Bechtel Natl. 1985; Hickey 1986). The concentrations of radioactive contaminants in each region were multiplied by the appropriate volumes and densities of material in each region (see Section I.4.2) and summed to obtain the total quarry inventory of radioactive materials. The results are presented in Appendix H, Table H.9.

For modeling purposes, it was necessary to know the average concentrations of radioactive contaminants in each region (Figure I.6) at the time of waste emplacement in 1963. For almost all radioactive contaminants and regions, the relevant leach times are long compared to the 22-year time interval between waste emplacement (1963) and measurement (1985) (Bechtel Natl. 1985); therefore, the measured concentrations can be taken to apply to the time of emplacement. The one exception is uranium in the 0.5-ft deep region, which has a leach time of 51 years. For this region, the measured uranium concentration was multiplied by a factor of 1.54 (given by exp [22/51]) to obtain the concentration in 1963. Details are presented in Section I.4.2. The simulated results and the potential impacts of waste release from the quarry to the well field are presented and discussed in Section 4.1.2.3.

I.2.5 Sensitivity Analyses

The results of groundwater modeling in the county well field area are presented in Sections 4.1.2.3 and 4.1.2.4. Some sensitivity analyses were carried out for these model predictions by repeating the calculations using different values for some of the input parameters. For example, calculations were carried out for selenium and arsenic using the more conservative Kd value of 3 mL/g for the quarry wastes (see Section I.3.2.1) instead of the respective values of 10 mL/g and 50 mL/g.

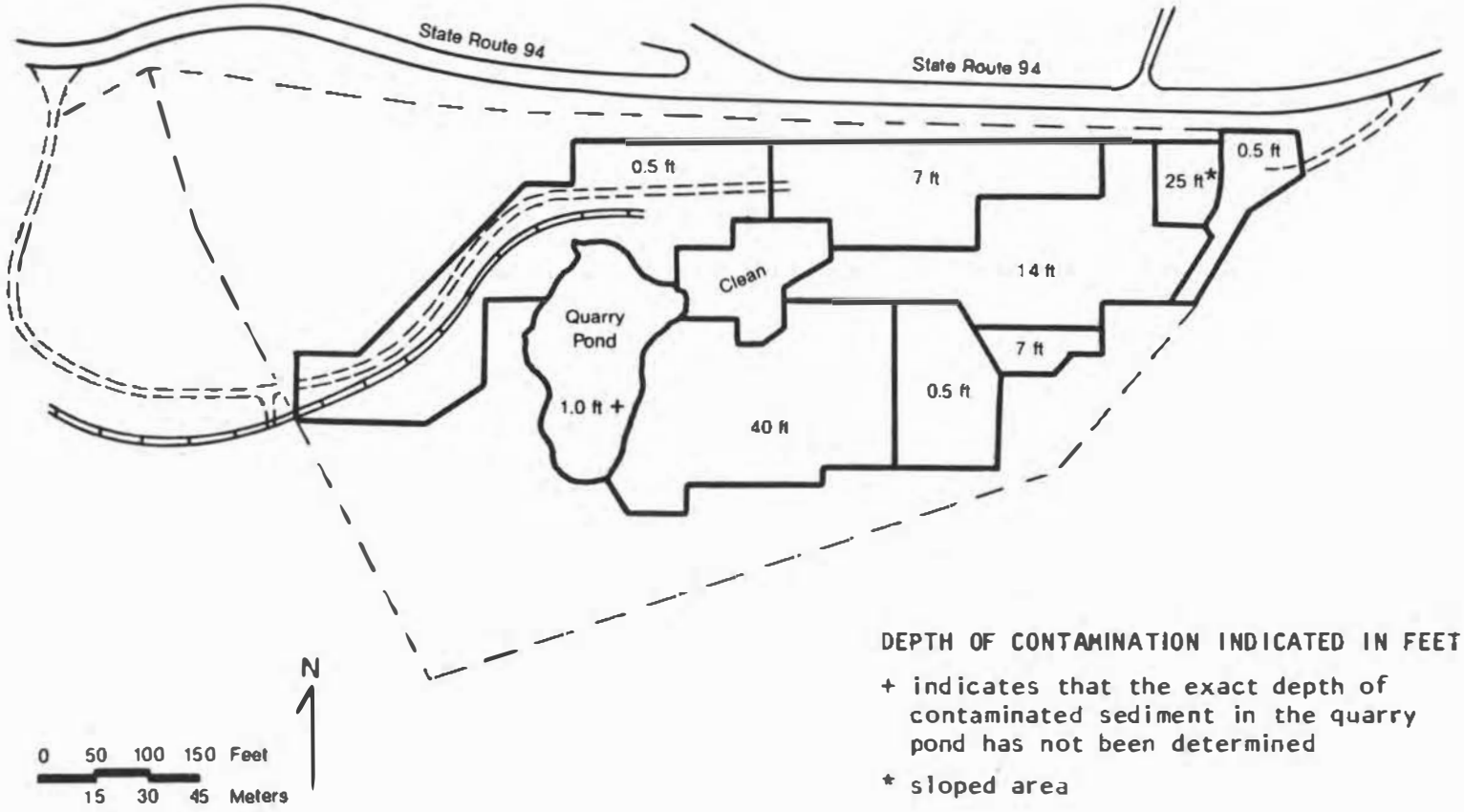

Figure I.6. Areas of Subsurface Contamination in the Quarry.
Source: Modified from Bechtel National (1985).

The results showed that, as would be expected, a decrease in the Kd value increases the predicted peak concentrations in Well 8. For selenium, the peak concentration would be increased from 1.2×10^{-7} mg/L (for Kd = 10 mL/g) to 1.8×10^{-7} mg/L (for Kd = 3 mL/g). For arsenic, the increase would be from 2.7×10^{-7} mg/L (for Kd = 50 mL/g) to 7.6×10^{-7} mg/L (for Kd = 3 mL/g).

Calculations were also carried out for arsenic and selenium with the less conservative assumption of equal retardation factors for the waste and limestone instead of Kd = 0 mL/g for the limestone. The results showed a small decrease in peak concentrations in Well 8 of about 3% for selenium and 4% for arsenic; this indicates that model predictions of peak concentrations are quite insensitive to which of the two assumptions are used for the limestone.

Calculations were also carried out under the assumption that the source term concentrations of arsenic and selenium were higher by a factor of 10.* This increased the peak concentration of selenium in the water in Well 8 by a factor of 10 from 1.2×10^{-7} mg/L to 1.2×10^{-6} mg/L and the peak concentration of arsenic from 2.7×10^{-7} mg/L to 2.7×10^{-6} mg/L.

Combining all of the above changes in the most conservative way (i.e., Kd = 3 mL/g for the wastes, Kd = 0 mL/g for the limestone, and source term concentrations higher by a factor of 10) gives peak concentrations in Well 8 of 7.9×10^{-6} mg/L for arsenic and 1.3×10^{-6} mg/L for selenium. However, these concentrations, as well as the ones described above, are still several orders of magnitude below the regulatory limits (Appendix H, Table H.12). These results can be extended to all contaminants listed in Table H.13 because of the linear dependence of predicted model concentration contributions in the well water on the average concentration of the chemical species in the quarry wastes. The sensitivity analyses support the prediction that concentration contributions of the various chemical species at the county well field would be negligible.

I.3 DISTRIBUTION COEFFICIENTS

One of the parameters that is important in determining the contaminant concentrations in groundwater is the distribution coefficient, Kd. This coefficient represents the equilibrium concentration ratio of the radionuclides or chemicals sorbed on the solid soil or rock matrix to the concentration in water in the soil or in the pores and fractures in the rock. The

*An increase in the source term concentration by a factor of 10 should be more than enough to account for the uncertainties in the source term concentrations. These uncertainties arise from such factors as the limited number of samples taken (7) and the fact that much of the waste in the 40 ft region (Figure I.6) was not sampled.

values of Kd vary depending on soil/rock properties, chemical content and pH of the water, and sorptive properties of the radionuclide or chemical being considered.

I.3.1 Distribution Coefficients for Uranium and Radium

I.3.1.1 Raffinate Pits Area and Hanford Site

Seeley and Kelmers (1985b) determined the distribution coefficients (Kd values) for uranium and radium for the raffinate pits area using batch contact methods. They report that the Kd values range from 660 to 18,000 mL/g for radium, with a mean value of 1,200 mL/g; the Kd values for uranium range from 12 to 1,300 mL/g, with a mean value of 370 mL/g. The mean values were used in this analysis (see discussions in Section I.1.4.1 regarding sensitivity analyses using the lower values). For the raffinate pits area, the wastes and overburden materials were assumed to have the same Kd value (Table I.1). For the underlying limestone aquifer, the Kd used for uranium was 0 mL/g and that for radium was 100 mL/g because of the mobilization effect of carbonate on uranium in the groundwater (see Section I.3.1.2).

For the Hanford site, the Kd values used for uranium and radium in the wastes were the same as those used for the Weldon Spring raffinate pits area (Table I.1). Kd values for uranium and radium for the specific soils at the Hanford site are not available (Routson et al. 1981a, 1981b; Serne 1986). A value of 3.7 mL/g for uranium was used in this analysis. This value is one-hundreth of that used for the clayey soils at the Weldon Spring site. This is probably a conservative value because Sheppard et al. (1984) reported a mean value of 8.1 mL/g for uranium in sandy soils. However, the value of 3.7 mL/g is greater than the value of 0 mL/g that was used in a recent environmental impact statement on waste management at the Hanford site (U.S. Dept. Energy 1986b). A sensitivity analysis using a Kd value of 0 mL/g for uranium is discussed in Section I.1.4.

For radium, a value of 10 mL/g was used for the sandy soils at Hanford. This value is probably conservative because it is 10 times less than the smallest value reported by Sheppard et al. (1984) for sandy soils. The value of 10 mL/g is consistent with that used in a recent environmental impact statement that evaluated disposal of similar types of wastes at the Hanford site (U.S. Dept. Energy 1986a).

I.3.1.2 Quarry Wastes, Limestone, and Alluvium

The Kd values for uranium and radium in the quarry wastes (18 mL/g and 14,000 mL/g) were determined as geometric means of the ratios of concentrations in the solid wastes to concentrations in the groundwater measured in

three boreholes in the quarry wastes (Bechtel Natl. 1985). The Kd values for uranium in the near alluvium and river alluvium were also selected to be the same value, 18 mL/g. The choice of 18 mL/g for the river alluvium is supported by the fact that Kd values that are averages of measured values for clay/silt/loam soils in the St. Louis area range from 50 to 500 mL/g (Seeley and Kelmers 1985a). The Kd value for the river alluvium, which is mainly sand, is assumed to be about 10% of the value for soils and clay (Nucl. Saf. Assoc. 1980); 10% of 50-500 mL/g is 5-50 mL/g, which includes the selected value for uranium in the river alluvium. The Kd value of 18 mL/g was also selected for the near alluvium. This choice is conservative because the near alluvium, which is less sandy than the river alluvium, is expected to have a higher Kd value that is closer to that for soils, i.e., about 50-500 mL/g.

The use of a different Kd value for radium for the quarry wastes than for the raffinate wastes is supported by the fact that the value of 14,000 mL/g gives (by use of data in Tables I.11 and I.12) radium-226 concentrations in the groundwater at the quarry of 2 to 4 pCi/L. This compares favorably with recent measured values of up to 1.9 pCi/L (Bechtel Natl. 1986). Use of a Kd value of 1,200 mL/g, which was used for the raffinate wastes (Table I.1), gives groundwater concentrations of radium-226 at the quarry of 30 to 60 pCi/L. These values are much higher than the measured values noted above, which suggests that Kd = 1,200 mL/g for radium-226 in the quarry wastes is too low.

The Kd value for uranium in the fractured limestone was selected to be 0 because of two effects: (1) the mobilizing effect of carbonates and (2) the saturation effect resulting from transport in fractures. Concentrations of carbonate or bicarbonate ions are high in groundwater moving in limestone, and uranium is mobilized by forming soluble complexes with these ions.

It is known that as the concentration of uranium (or any ion) becomes sufficiently high, the Kd value decreases dramatically (Seeley and Kelmers 1985b). This effect, which is due to saturation of available sorption sites, is expected to occur at much lower concentrations for transport in fractures and solution channels than for transport in soils or sand. The reason is that the average surface area available for sorption per unit volume of water transported is expected to be much smaller for transport in fractures and channels than for transport in a porous medium. The concentration at which uranium occurs in groundwater at the quarry indicates that the saturation effect may be occurring, with the result that uranium is fairly mobile in the rock fractures around the quarry (Appendix H, Section H.1.4).

The Kd values for radium in the near alluvium and river alluvium were selected to be 100 mL/g, which is about 10% of the average values measured by Seeley and Kelmers (1985b). This lower value was used because the Kd value of

1,200 mL/g reported by Seeley and Kelmers (1985b) was for the raffinate pits area, which has a much higher clay content. This value was also used for radium in the fractured limestone because radium does not form soluble carbonate complexes with carbonate ions and the saturation effect is not expected for radium due to the much lower concentration of radium in the groundwater at the quarry (measured in units of mass unit volume) relative to the uranium concentration. The value selected for radium for the near alluvium is conservative because the near alluvium contains a smaller fraction of sand than the river alluvium.

I.3.2 Distribution Coefficients for Chemicals

I.3.2.1 Raffinate Pits Area and Hanford Site

Literature values of distribution coefficients for the chemical elements that are regulated in groundwater by either the EPA or the state of Missouri (Appendix H, Table H.12) are given in Table I.8 for soils, clays, and sand. Geometric standard deviations, σ_g, were calculated for these Kd values by use of literature data (Baes and Sharp 1983). The geometric standard deviations, which are uncertainty factors for the mean Kd values, range from 1.8 for arsenic to 15 for manganese. For purposes of analysis, the elements were grouped into six groups with similar Kd values (values other than zero are given in Table I.8) and a representative Kd value was chosen for each group. This simplification is reasonable because, for all the elements, the average Kd values are within one geometric standard deviation of the assigned values.

For the raffinate pits area, model calculations were done for all alternatives using the representative Kd values for soils/clays in Table I.8 for the wastes and for the clay under the wastes. This was done because no Kd values have been measured for the raffinate pits wastes or for any other waste components (e.g., quarry wastes, chemical plant wastes, and vicinity properties soils). All chemical species were assumed to be mobile (Kd = 0) in the limestone under the clay because no measured values are available. This assumption is conservative because the saturation effect discussed for uranium (Section I.3.1.2) depends on the concentration of the chemical species in groundwater and may not be significant for chemicals present at very low concentrations. Also, the mobilizing effect of carbonate and bicarbonate ions is not applicable for most chemical species.

For the sandy soil at the Hanford site, there are some Kd values available for some chemical elements but not for the chemical elements of concern in the Weldon Spring wastes (Routson et al. 1981a, 1981b; Serne 1986). Therefore, for purposes of analysis, it was assumed that the Kd values for

Table I.8. Distribution Coefficients for Various Chemical Elements

Element	Average Distribution Coefficient,[a] Kd (mL/g)		Geometric Standard Deviation	Assigned Representative Distribution Coefficient Kd (mL/g)	
	Soils and Clays[b]	Sand[b,c]		Soils and Clays[b]	Sand[b,c]
Arsenic	3.3	0.33	1.8	3	0.3
Barium	50	5	-[d]	50	5
Beryllium	50	5	-	50	5
Boron	-	-	-	3[e]	0.3[e]
Cadmium	6.7	0.67	2.4	3	0.3
Chromium (+6)	37	37	9.0	25	2.5
Cobalt	1,000	100	-	1,000	100
Copper	22	2.2	3.0	25	2.5
Fluoride	-	-	-	3[e]	0.3[e]
Iron	1,000	100	-	1,000	100
Lead	99	9.9	5.5	100	10
Manganese	150	15	15	100	10
Mercury	100	10	-	100	10
Nickel	1,000	100	-	1,000	100
Selenium	2.7	0.27	1.9	3	0.3
Silver	110	11	3.7	100	10
Zinc	16	1.6	6.7	25	2.5

[a] Data for arsenic, cadmium, chromium, cobalt, copper, lead, manganese, selenium, silver, and zinc are from Baes and Sharp (1983); the values of Kd are the geometric means of the literature data (see also Gilbert et al. [1983--pp. 3-57 to 3-60]). Data for other elements are from Nuclear Safety Associates (1980).

[b] The values for soils and clays are used for assessment of impacts at the Weldon Spring site whereas the values for sand are used for Hanford and the river alluvium.

[c] The values were assumed to be 10% of the values of soils and clays (Nucl. Saf. Assoc. 1980).

[d] A hyphen means that relevant data are not available.

[e] See Section I.3.3.

chemical elements for the soil at the Hanford site are the representative values for sand as listed in Table I.8, which are 10% of the values for soils and clays.

I.3.2.2 Quarry Area

For arsenic, selenium, cadmium, and nickel, Kd values are available for soil containing appreciable amounts of carbonate and having pH values from 6.3 to 7.6 (Wangen et al. 1982). These values are appropriate to use for the Weldon Spring quarry area because the groundwater at the quarry comes from limestone and contains appreciable concentrations of bicarbonate (Appendix H, Table H.14). The Kd values, as geometric means of reported results from batch adsorption studies, are 480 mL/g for cadmium, 450 mL/g for nickel, 60 mL/g for arsenic, and 10 mL/g for selenium. The respective geometric standard deviations are 8, 6, 2, and about 1.4. Selecting representative values, as was done in Table I.8, the values used for the quarry wastes were 500 mL/g for cadmium and nickel, 50 mL/g for arsenic, and 10 mL/g for selenium.

For the near alluvium, Kd values for each species were assumed to be the same as those used for the quarry wastes. Because the river alluvium has a large sand component, representative Kd values for sand were used -- which are 10% of the values used for the quarry wastes (Nucl. Saf. Assoc. 1980). The same Kd values in the limestone were used here as were used in model calculations for the limestone under the raffinate pits, i.e., Kd = 0 for all chemical species. This is based on the fact that groundwater movement in the limestone occurs in solution joints and fractures rather than as diffusion in a porous medium.

Organic priority pollutants -- i.e., pesticides, PCB 1254, and base/neutral compounds (Keith and Telliard 1974) -- and organic nonpriority pollutants are also present in the quarry wastes (see Appendix H, Table H.13). These compounds are known to bind strongly to the total organic carbon (TOC) in soils. The TOC content of the quarry wastes has not been measured. However, literature values range from 0.04% for sand to more than 2% for loamy soils, with values for clay soils of about 0.1 to 0.5% (Griffin et al. 1976; Reinbold et al. 1979; Wangen et al. 1982). A value of 0.5% was assumed here for the quarry wastes and the near alluvium. Rough estimates of Kd values for the quarry wastes were obtained by combining 0.5% with values of K_{OC}, the partition coefficient for organic carbon. Values of K_{OC} were obtained either directly from the literature or from aqueous solubility data and a relation between K_{OC} and aqueous solubility (Hassett et al. 1983; Kirk Othmer 1981). The resulting Kd values for the quarry wastes and the near alluvium are 13 mL/g for lindane (γ-benzene hexachloride) and 100 mL/g for endrin (as examples of organics with the lowest regulatory limits in

groundwater--40 CFR Part 264.94 and 40 CFR Part 141.12), and 200 mL/g for PCB 1254. For modeling purposes, representative Kd values for the quarry wastes and near alluvium were selected to be 10 mL/g for lindane, and 100 mL/g for endrin and PCB 1254. Kd values for the sandy river alluvium were assumed to be 10% of those for the wastes. Kd = 0 was assumed for the fractured limestone.

I.3.3 Mobility of Boron and Fluoride

Estimates of Kd values in the literature for both boron and fluoride are Kd = 0 (Nucl. Saf. Assoc. 1980). However, these values are probably not applicable in the Weldon Spring area. For fluoride, the low solubility of calcium fluoride and the high concentrations of calcium in the water in the raffinate pits result in low concentrations of fluoride in the water in the pits in spite of high concentrations of fluoride in the solid raffinate sludge. This effect also would suppress the movement of fluoride in the limestone groundwater because the water would be expected, in general, to have appreciable concentrations of calcium ions in solution as calcium bicarbonate.

The distribution coefficient is defined as the ratio of the concentration of a chemical in the solid matrix divided by the concentration in the interstitial pore water. Concentrations of chemical species have been measured in dry raffinate sludge (see Appendix H, Table H.10). Because of stratification, dilution by precipitation, and other factors, the concentrations in the water in the raffinate pits (Table H.11) probably do not represent concentrations in interstitial water. However, the data are useful for comparative purposes because the solid/water concentration ratios for chemical species such as nitrate, sulfate, and chloride -- species for which it is reasonable to assume that Kd = 0 -- can be compared with the solid/water concentration ratios for boron and fluoride. These ratios, calculated using average solid and average water concentrations with the averages weighted over the four pits, are: nitrate, 4.0; sulfate, 1.7; chloride, 20; boron, 4,200; and fluoride, 6,100. The results indicate that the Kd values for fluoride and boron may be appreciably greater than zero because the fluoride and boron ratios are much larger than the ratios for nitrate, sulfate, and chloride.

For purposes of analysis, a Kd of 3 mL/g for the raffinate sludge and underlying clays was used in the analysis of transport of fluoride and boron in the raffinate pits area. Model calculations of leaching and transport for boron and fluoride were not carried out for the quarry area because data on concentrations of these two species in the quarry wastes are not available. As a result, the problem of appropriate Kd values for boron and fluoride for the quarry area does not arise. For the reasons outlined above, Kd = 3 mL/g was also used for boron and fluoride in the wastes buried at the Hanford

site. For purposes of analysis, 10% of this value, or 0.3 mL/g, was used for transport in the sandy soil underlying the wastes at the Hanford site (Table I.8). A sensitivity analysis shows that the main effect of use of a more conservative value of Kd = 0 mL/g in the sandy soils is to decrease the delay time required for boron and fluoride to reach the groundwater. Concentrations in the groundwater are about the same as those obtained for the value of Kd = 0.3 mL/g.

I.4 RELEASE OF CONTAMINANTS

I.4.1 Release of Contaminants from the Raffinate Pits Area

The ion-exchange model (Gilbert et al. 1983) was used to determine the contaminant release rate from the source. This model tends to overestimate the release rates by assuming that the contaminants are localized on the surface of the waste particles and thus do not have to diffuse out from the interiors of the particles before they are mobilized.

The average concentrations of radium-226, thorium-230, and uranium-238 in the wastes used to model release of contaminants from the raffinate pits area are given in Table I.9. For Alternatives 1 and 2a, the average concentrations of each radionuclide were obtained by dividing the total inventory of the raffinate sludge, quarry wastes, and vicinity properties wastes by the sum of the masses of the stabilized raffinate sludge, quarry wastes, and vicinity properties wastes. For Alternative 3a, the mass of the raffinate sludge dried

Table I.9. Radium-226, Thorium-230, and Uranium-238 Source Concentrations Used in the MAT123D Model

	Source Concentration (pCi/g)[a]		
Alternative	Radium-226	Thorium-230	Uranium-238
Raffinate Pits Area			
1	56	1,500	120
2a	56	1,500	120
4	350	13,000	550
Hanford Site			
3a	110	2,800	220

[a] Chemical plant wastes were neglected because the concentrations of the radioactive species in these wastes are less than in the other components.

for shipment was used instead of that of the stabilized raffinate sludge. For Alternative 4, the inventory of the raffinate sludge was divided by the mass of the dried sludge to obtain the average concentrations.

For Alternatives 1, 2a, and 2b, the average concentration of each chemical species in the source material was determined as a weighted average of the concentrations in the stabilized raffinate sludge and the quarry wastes. The weighting factors are the mass fractions of the stabilized raffinate sludge and quarry wastes in the wastes (excluding chemical plant and vicinity properties wastes).

The average concentrations of chemical species in the quarry wastes were taken to be those given in Appendix H, Table H.13. For chemical species for which measured values are not available, it was assumed that the concentrations are the same in the quarry wastes as in the stabilized raffinate sludge. The concentration of each chemical in the stabilized raffinate sludge was determined as a weighted average of the average concentration in the wet raffinate sludge and in the stabilizer (assumed to consist of 80% by weight fly ash and 20% by weight portland cement). The weighting factors (weight fractions of sludge and stabilizer in the stabilized sludge) were obtained from these values, data in Appendix H, Table H.1, and the assumption that the sludge would be stabilized by mixing 1 kg stabilizer with 1.2 L of wet sludge (Bechtel Natl. 1984a). The average concentration of each chemical species in the wet sludge was obtained as an average, weighted over the mass fractions in each pit (Table H.1), of the concentrations in each pit given by the 1983 data in Table H.10. For Pits 1 and 3, geometric means of the 1983a and 1983b data (as given in Table H.10) were used. A conversion factor from dry to wet sludge was obtained from the data in Table H.1.

The average concentration of each chemical species in the fly ash component of the stabilizer was taken as the geometric mean of the high and low ends of the corresponding concentration range reported in the literature for fly ash from coal combustion (Saguinsin et al. 1981). Contributions from the cement were neglected because it consisted of only 8% (by weight) of the final stabilized sludge.

I.4.2 Release of Contaminants from the Quarry Area

The model of the source term for the quarry was based on the recent survey of the quarry carried out by Bechtel National (1985). The survey showed that the contaminated wastes can be divided into regions of different depths as shown in Figure I.6. The elevation contours in the quarry, including the elevation of the pond water at about 140 m (470 ft) MSL, are shown in Figure I.7. The pond water elevation is assumed to be the elevation of the groundwater surface in the quarry. All regions of wastes in the quarry

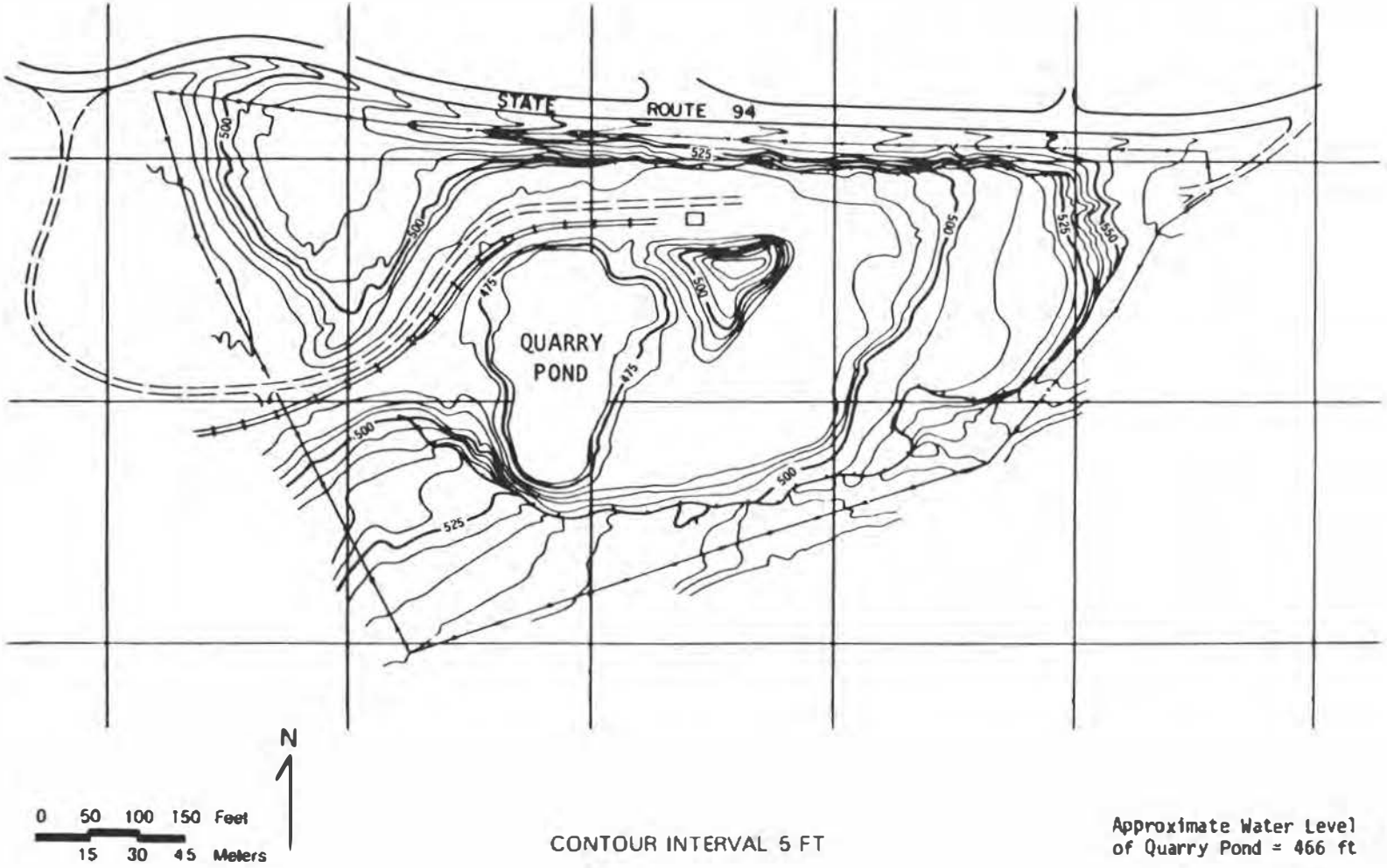

Figure I.7. Topographic Map of the Weldon Spring Quarry (all elevations are given in feet above MSL). Conversion Factor: To convert feet to meters, multiply by 0.3048. Source: Modified from Bechtel National (1985).

are above the groundwater table except for the wastes under the pond and in the 40-ft deep region (Figures I.6 and I.7). Based on an average surface elevation in this region (from Figure I.7) of about 150 m (480 ft), approximately 5 m (16 ft) of this region is above the water table and the rest is below.

The total volumes of each region were obtained from the depth and area of each region, as estimated from Figure I.6; the values are given in Table I.10. The volume of the 40-ft region so obtained was increased by 6,900 m^3 (9,000 yd^3) of material to give a total waste volume of 72,600 m^3 (95,000 yd^3) (Bechtel Natl. 1985). It is believed by staff of Bechtel National that 72,600 m^3 (95,000 yd^3) is a better estimate than the sum of volumes, 65,700 m^3 (86,000 yd^3), because the amounts of material under the pond and in the 40-ft region are uncertain (Hickey 1986).

Most of the rubble is believed to be on the quarry floor in the 40-ft region (Hickey 1986). The density of the wastes in this region, which includes contributions from cement and steel rubble, was assumed to be

Table I.10. Physical Characteristics of the Regions of Quarry Wastes

Region[a]	Depth (m)	Area[b] (m^2)	Volume (m^3)	Density (g/cm^3)	Weight (MT)
40-ft	12.2	3,460	49,100[c]	2.61	128,000
Pond	0.305	1,860	567	2.61	1,500
25-ft	7.62	418	3,105	1.84	5,860
14-ft	4.27	3,090	13,180	1.84	24,300
7-ft	2.13	2,630	5,600	1.84	10,300
0.5-ft	0.152	6,250	950	1.84	1,750
			72,600	2.37[d]	172,000

[a] See Figure I.6.

[b] From Figure I.6.

[c] Increased by 6,900 m^3 (9,000 yd^3) as described in the text.

[d] Average density of quarry wastes.

Conversion Factors: To convert meters (m) to feet (ft), multiply by 3.281; to convert square meters (m^2) to square feet (ft^2), multiply by 10.76; to convert cubic meters (m^3) to cubic yards (yd^3), multiply by 1.308; to convert metric tons (MT) to short tons (t), multiply by 1.102.

2.61 g/cm^3 (Bechtel Natl. 1984a). The density of the wastes in the other regions was assumed to be 1.84 g/cm^3, which is the value given for soil, clay, and sludge in the quarry (Bechtel Natl. 1984a--Appendix F). The physical characteristics of the quarry waste components are summarized in Table I.10.

Average concentrations of radionuclides in the various regions of quarry wastes were determined from measurements at different locations and depths in the quarry (Bechtel Natl. 1985). These average concentrations were combined with the data in Table I.10 to give estimates of the inventories of radionuclides in the quarry. The results are given in Table I.11. The average concentrations are approximate because the measurements apply only to samples taken of contaminated soils that either lie above the rubble or occur at rubble-free locations. They do not apply to the rubble or contaminated wastes underlying the rubble because sampling techniques (auger drilling to the point of first refusal on rubble or bedrock) did not provide samples of these materials (Bechtel Natl. 1985; Hickey 1986). This is especially important for the 40-ft region, which is believed to contain most of the estimated 31,000 m^3 (41,000 yd^3) of rubble (Bechtel Natl. 1984a; Hickey 1986).

Also, the number of locations is limited in the deeper (> 0.5 ft) regions of wastes for which radiological depth profiles were obtained. In particular, sample locations were chosen as follows: 8 locations in the 40-ft region (6 were analyzed for thorium-230); 10 locations in the 14-ft region (9 were analyzed for uranium-238 and 1 for thorium-230); 8 locations in the 7-ft region (7 were analyzed for uranium-238 and 3 for thorium-230); no locations in the 25-ft region; and 88 locations in the 0.5-ft region (78 were analyzed for uranium-238 and 32 for thorium-230).

The data of Bechtel National (1985) were used here because they are the best data available. At each location, the depth profile values for a particular radionuclide were first averaged; these average values were then averaged over the locations in each region to obtain an average value for each region. The regional values, which are given in Table I.11, were assumed to apply to all wastes in the region, including the rubble and the wastes underlying the rubble.

For modeling purposes, the quarry and surrounding regions were divided into grid cells 80.5 m × 80.5 m (264 ft × 264 ft). The areas of the different regions are such that the area of the 0.5-ft-deep region covers one grid cell and the sum of the areas of the 7-ft, 14-ft, and 25-ft regions (Figure I.6) also covers one grid cell. The 40-ft region can be represented as a rectangle with one dimension equal to 80.5 m (264 ft) and the other dimension, 43.0 m (141 ft) chosen to give the correct area, 3,460 m^2 (37,200 ft^2), of the region

Table I.11. Average Concentrations and Inventories of Radionuclides in the Quarry Wastes

Region[a]	Average Concentration (pCi/g)[b]			
	Uranium-238	Radium-226	Thorium-230	Thorium-232
40-ft[c]	190	64	475	3.0
25-ft[d]	155	54	400	73
14-ft	155	54	400[e]	73
7-ft	77	90	1860[f]	24
0.5-ft	20	7.9	29	3.4

Region[a]	Inventory (Ci)			
	Uranium-238	Radium-226	Thorium-230	Thorium-232
40-ft[c]	24.7	8.3	61.7	0.39
25-ft[d]	0.9	0.3	2.3	0.43
14-ft	3.8	1.3	9.7	1.77
7-ft	0.8	0.9	19.1	0.24
0.5-ft	0.03	0.01	0.05	0.006
	30	11	93	2.8

[a] See Figure I.6.

[b] Determined from data in Bechtel National (1985).

[c] Contribution from wastes under the pond are included here.

[d] No subsurface samples were taken in this region. Consequently, the average concentrations were assumed to be the same as in the 14-ft region.

[e] One sample location only.

[f] Average of values for three sample locations. One value was very high, 5500 pCi/g; the other two were less than 100 pCi/g.

as shown in Figure I.6. The wastes beneath the pond water and the additional 6,900 m^3 (9,000 yd^3) (see Section I.4.2) are included at the bottom of the 40-ft region to give this region a total depth of 14.4 m (47.2 ft).

The average bulk groundwater flow velocity (Darcy) in the fractured limestone can be determined from the values of the parameters in Table I.7. The Darcy flow velocity is estimated to be 0.73 m/yr (2.4 ft/yr) through the quarry wastes. The average linear velocity of groundwater in the limestone is estimated to be 490 m/yr (1,600 ft/yr).

The representation of the quarry source along with the directions of groundwater flow and of infiltration are shown in Figure I.8a. The aquifer depth b is also shown. From the values of hydraulic conductivity and transmissivity in the limestone (Table I.7), the value of b was calculated to be 10.9 m (35.8 ft). Contaminants are leached out of the waste in grid cell 1 (40-ft region) both by precipitation infiltrating through the top 4.9 m (16 ft) and by groundwater moving through the bottom 9.5 m (31 ft). The other two cells are leached by infiltration only because the wastes are above the water table.

The ion-exchange model (Gilbert et al. 1983) was used to represent the leaching of contaminants from the waste. This model is conservative in that it tends to overestimates the leach rate by assuming that contaminants are absorbed at the surface of the substrate and are sorbed and released by ion-exchange mechanisms. The possibility that contaminants may be found in the interior of substrate particles and must diffuse slowly to the surface before they can be mobilized is not considered in this model because it gives a much lower leach rate.

Concentrations of a contaminant in the groundwater flowing through a vertical section of the aquifer at the downstream face of cell 1 (or downstream edge of cells 2 and 3) were modeled as a sequence of steps or ramps, each with a characteristic leach time (see Figure I.8b). The relative heights and widths of the steps in Figure I.8b approximately represent the relative concentration levels and leach times for the cells.

The use of steps or ramps instead of exponential release rates tends to be conservative because it assumes that all the contaminant in a section of wastes is completely released at a uniform rate within an appropriate leach time rather than spreading the leach rate out with an exponential decrease over all time.

Cell 3 has a short leach time because it is quite thin, 0.5 ft. The first step in cell 2 has the highest concentration, C_1, because it includes

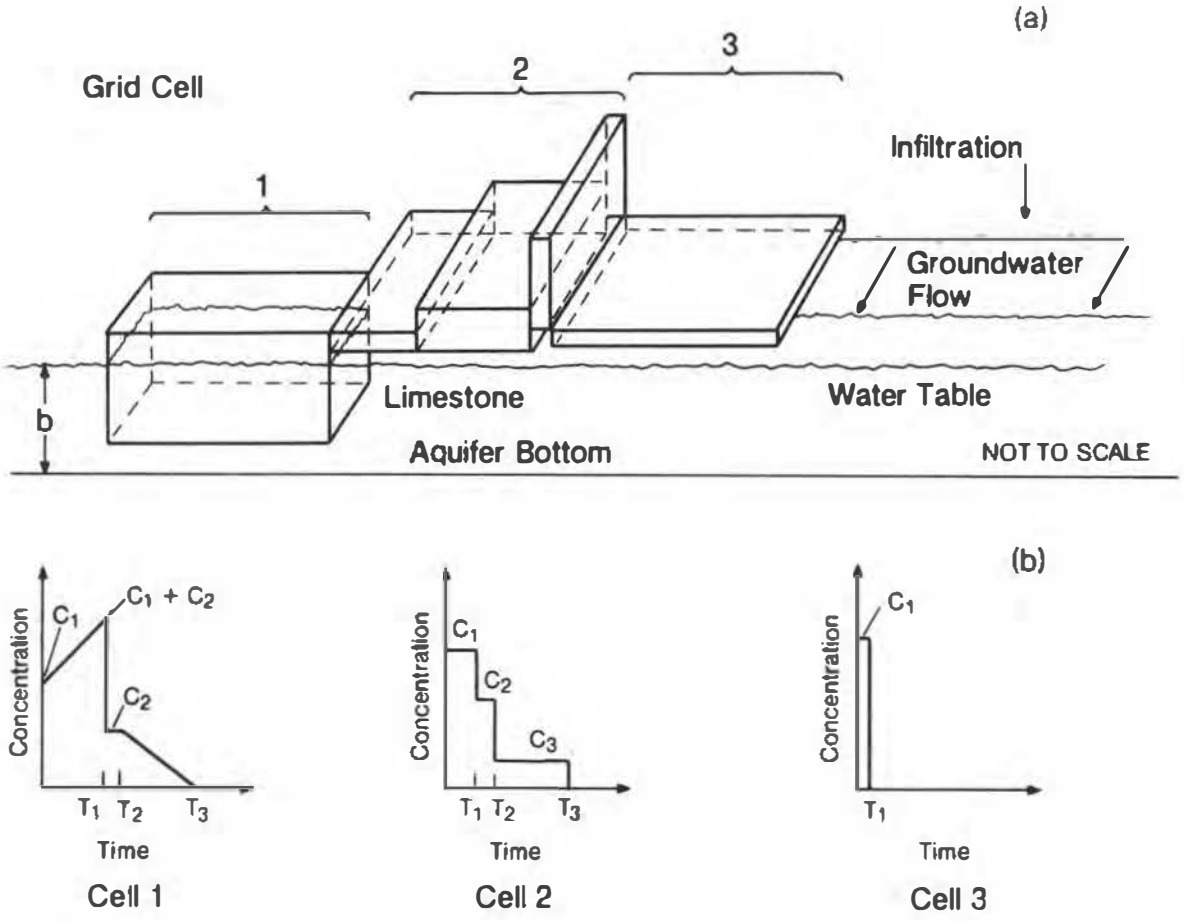

Figure I.8. Representation of the Quarry Source Term for the Model Calculations. In part (a), the different regions are shown as components of three grid cells. Cell 1 has components above and below the water table. Components of cells 2 and 3 are above the water table. In part (b), groundwater concentrations of a contaminant as a function of time are shown for each of the three grid cells as a sequence of concentration ramps or steps. Further details are given in the text.

contributions from all three regions (7-ft, 14-ft, and 25-ft). After time T_1, the 7-ft region is leached out and no longer contributes, so the concentration drops to C_2. When the 14-ft region is leached out at time T_2, the concentration drops to C_3 where it remains until the small 25-ft region (Figure I.6) is leached out in time T_3.

Cell 1 is different in that the first step consists of a linear buildup of the concentration from C_1 to $C_1 + C_2$ over the time period T_1, which is the leach time for the lower layer in the groundwater. At T_1, the bottom layer is depleted of its initial inventory so the concentration drops to C_2 and remains at C_2 until time T_2, the leach time of the upper layer above the groundwater. At this time, the concentration decreases linearly with time to 0 at time $T_3 = T_1 + T_2$ when all the contaminant that has leached into the lower layer from the upper layer is leached out of the lower layer. The linear increase and decrease of concentrations with time occurs because the upper layer is spread out in the direction of groundwater flow and it takes an appreciable time for contaminants leached from the top layer to be transported by the groundwater through the saturated wastes.

In modeling contaminant releases from the quarry wastes, it was assumed that the times for movement of a contaminant in the limestone -- vertically through the unsaturated zone (cells 2 and 3 only) to the bottom of the aquifer (to depth b), and horizontally for a grid cell length (80.5 m [264 ft]) -- is short compared to leach times through the waste. This assumption is valid because the porosity of the limestone aquifer is so low, 0.0015 (Table I.1). For example, the transit time for the groundwater to move horizontally in the limestone one grid cell length is only 0.15 yr. This is shorter than the time of 0.35 yr required for infiltrating water to move down through the thinnest source (cell 3) and much shorter than the other waste leach times. For all the contaminants of interest here, the Kd values and porosities are such that the contaminants move much faster in the fractured limestone than in the wastes, so the assumption remains valid.

The leach times and concentration factors calculated for each of the three cells shown in Figure I.8b are given in Table I.12. The concentration factors combine relevant waste parameters and groundwater parameters in such a way that the groundwater concentration contribution of leachate from the cell can be calculated by multiplying the factors by the average concentration in the quarry wastes. For cell 2, the factors give the combined contributions of the 7-ft, 14-ft, and 25-ft regions (F_1); the contributions of the 14-ft and 25-ft regions (F_2); and the contributions of the 25-ft region (F_3). For example, the average concentration of arsenic in the quarry wastes (Appendix H, Table H.13) is 100 μg/g; using the appropriate entries from

Table I.12. Leach Times and Concentration Factors for the Quarry Waste Components

Kd Value (mL/g)[a]	Leach Times (yr)			Concentration Factors (g/L)		
	T_1	T_2	T_3	F_1	F_2	F_3
Cell 1						
0	10.0	11.2	21.2	6,650	3,050	-
10	1,130	1,260	2,390	58.8	26.9	-
18	2,000	2,290	4,290	33.2	14.9	-
25	2,840	3,190	6,030	23.4	10.6	-
50	5,660	6,350	12,000	11.7	5.37	-
100	11,300	12,700	24,000	5.88	2.69	-
500	56,500	63,300	120,000	1.18	0.539	-
1,000	113,000	127,000	240,000	0.588	0.269	-
14,000	1,580,000	1,770,000	3,350,000	0.042	0.019	-
Cell 2						
0	4.9	9.8	17.5	5,420	3,100	491
10	396	793	1,420	56.9	38.2	6.05
18[b]	710	1,420	2,540	37.6	21.5	3.39
25	980	1,960	3,500	27.0	15.4	2.43
50	1,960	3,920	7,000	13.6	7.74	1.22
100	3,920	7,840	14,000	6.78	3.87	0.613
500	19,600	36,200	70,000	1.35	0.774	0.122
1,000	39,200	78,400	140,000	0.68	0.386	0.0613
14,000[b]	548,000	1,110,000	1,960,000	0.048	0.027	0.0044
Cell 3						
0	0.350	-	-	3,940	-	-
10	28	-	-	48.6	-	-
18	51	-	-	27.4	-	-
25	70	-	-	20.0	-	-
50	140	-	-	9.9	-	-
100	280	-	-	4.92	-	-
500	1,400	-	-	0.99	-	-
1,100	2,800	-	-	0.49	-	-
14,000	39,200	-	-	0.035	-	-

[a] The Kd values are either the representative Kd values described in Section I.3 or are the Kd values in the wastes for uranium (18 mL/g) and radium (14,000 mL/g).

[b] Concentrations of uranium and radium in cell 2 were calculated directly for each cell component because the measured concentrations are different in each component.

Table I.12 for cell 2 for Kd = 50 mL/g, one obtains C_1 = 1.4 mg/L, C_2 = 0.77 mg/L, and C_3 = 0.12 mg/L. For chemical species for which concentrations in the quarry wastes are known to be different in the different regions (e.g., uranium as shown in Table I.11), one must take -- for cell 2 -- differences of the concentration factors to obtain the individual factors for each region and then sum the results. For example, for uranium (Kd = 18 mL/g) for cell 2 (Tables I.11 and I.12), $C_1 = 77 \times (F_1-F_2) + 155 \times (F_2-F_3) + 155 \times (F_3) \approx$ 4,600 pCi/L.

The leach times for contaminants in the waste components above the water table at the quarry were calculated (Gilbert et al. 1983) from

$$T = \frac{nRdD}{v_I} \tag{I.16}$$

where n is the waste porosity, v_I is the infiltration rate (Table I.1), Rd is the retardation factor, and D is the thickness of the layer being considered. Calculation of the leach times for different contaminants for the lower layer of cell 1 in the water is more complex because the water velocity increases in the direction of flow. This occurs because infiltration over the top layer must be accommodated and the aquifer thickness b is assumed to be fixed. In this case, the leach time is given by

$$T = nRd \int_{a-\ell}^{a} \frac{dx}{v_d + \frac{v_I x}{b}} = \frac{nRdb}{v_I} \ln \left[\frac{v_d + \frac{v_I a}{b}}{v_d + v_I \frac{a-\ell}{b}} \right] \tag{I.17}$$

where n, Rd, v_I, and b are as defined above, v_d is the (Darcy) groundwater velocity at the upstream face of the grid cell of length a, and ℓ is the path length of the groundwater in the wastes. The integration limits are given by the position of the wastes in the grid cell (Figure I.8). The value ℓ = 52.8 m (173 ft) was determined from the actual geometry of the 40-ft region (Figure I.6) and an assumed groundwater flow direction to the southeast (150° compass direction).

The source term concentrations calculated as described above refer to the concentrations in the wastes at the time they were put in the quarry, i.e., in 1963. However, concentrations of radioactive and nonradioactive contaminants as given in Table I.11 and Table H.13 (Appendix H) were measured as average concentrations in boreholes with varying depths in 1985, 22 years later (Bechtel Natl. 1985). For most contaminants, this difference may be ignored because the leach times appropriate for the boreholes are much greater than 22 years. However, this is not the case for mobile species with Kd = 0 or species with Kd $\leq$ 18 mL/g for boreholes in the 0.5-ft region (Figure I.6).

A convenient way to correct for this is to assume that the rate of source leaching depends exponentially on time. In this case, it can be shown that the average concentration down to depth D at time t, C(D,t), is related to the average concentration at time 0, C(0,0), by $C(0,0) \approx C(0,t)\ e^{t/T_D}$ where T_D is the appropriate leach time for a contaminant in a section of wastes of depth D. All measurements of uranium in cell 3 were composites over a depth of 0.5 ft. For these measurements, T_D = 51 years. Consequently, the correction factor to be applied to the measured concentration of uranium in cell 3 (Table I.12) is exp (22/51) = 1.54.

No correction is necessary for selenium (Kd = 10 mL/g) because no measurements of selenium concentrations were made in the 0.5-ft region.

For a species such as cyanide, with Kd = 0 (Theis and West 1986; Fuller 1977), the corrections must be applied individually to each borehole measurement, to obtain appropriate concentrations in 1963, and the results averaged. This is possible because, for each of the six boreholes that provided a composite sample for analysis (Bechtel Natl. 1985), the depth of the borehole was available (Hickey 1986). These depths were used to obtain separate leach times and exponential correction factors for each borehole. Based on corrected data for each borehole for which a positive result was obtained, the measured average concentration of 0.35 ppm cyanide in 1985 corresponds to an initial average concentration in the quarry wastes in 1963 of 4.0 ppm.

I.4.3 Transport of Lead from the Disposal Cell Cover

The presence of a lead sheet in the cover for Alternative 2b would result in lead being corroded from the lead sheet and being transported from the cover to the drainage ditch surrounding the containment cell. The transport time for lead was estimated as follows. In one conceptual design, the cover for the containment cell consists of four triangular sections covering the upper parts of a four-sided pyramid. The lead sheet extends throughout each triangle and ends at the edge of the waste pile in the cell. Details are given in Chapter 2, Figure 2.5. A representative cover triangle is illustrated in Figure I.9. In the long dimension, the cover slope equals 2.2% and L equals 260 m (850 ft). In the short dimension, the cover slope equals 5% and L equals 120 m (390 ft).

It was assumed that flow above the lead sheet would be confined to the sand layer above the sheet. If Z is the percolation rate and r the thickness of the sand layer, then the velocity of water moving off the lead sheet at

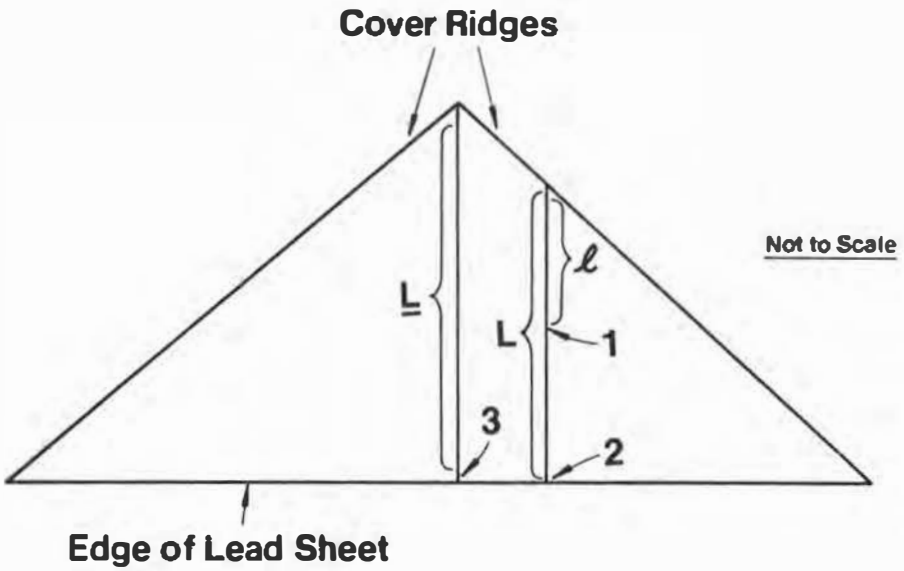

Figure I.9. A Representative Cover Triangle from a Top View of the Disposal Cell. The cover consists of four such triangles covering the upper parts of a four-sided pyramid.

point 1 located at a distance ℓ from a cover ridge (shown in Figure I.9) is given by $v(\ell) = \ell Z/r$.* The maximum velocity, $v(\underline{L}) = \underline{L}Z/r$, is reached at point 3 (Figure I.9).**

The time it would take for the water to flow off the lead sheet from point 1 to point 2 (Figure I.9) is given by

$$T(\ell) = \int_{\ell}^{L} d\ell/v(\ell) = (r/Z)\ln(L/\ell) \quad . \qquad (I.18)$$

*The velocity, $v(\ell)$, is calculated from volume-preserving considerations. Consider a small distance Δ at point 1 parallel to the triangle base. The cross-sectional area through which the water must move is given by Δr. The volume flow rate through Δr is $\ell\Delta Z$. Therefore, $v(\ell) = \ell\Delta Z/\Delta r = \ell Z/r$.

**Hydraulic conditions require that the Darcy velocity cannot be greater at any one point than the product of the gradient and the hydraulic conductivity. This condition is satisfied for the values of $\underline{L}$, r, and Z, and the values of the gradient or cover slope given here. The value of the hydraulic conductivity is taken to be 0.19 cm/s (Table I.1). (It is assumed that the hydraulic conductivity of the sand in the cap is the same as that for the sandy soil at the Hanford site.)

The time it would take the water to move off the sheet, averaged over points on the straight line from the ridge through point 1 to point 2, is given by

$$\langle T \rangle = (1/L) \int_0^L T(\ell) d\ell = r/Z \quad . \tag{I.19}$$

Because this is independent of L, <T> is also the time averaged over the whole cover triangle, and thus over the whole cover, for water to flow off the lead sheet.

The lead would be sorbed onto sand and soil particles and, therefore, would move more slowly than the water. The average time it would take the lead to move in the sand to the edge of the lead sheet can be given by $(Kd\rho_s r)/Z$ where Kd is the distribution coefficient for lead in sand and ρ_s is the density of sand (Gilbert et al. 1983). Using Kd = 10 mL/g (Table I.8), ρ_s = 2 g/cm^3, r = 0.15 m (Figure 2.5), and Z = 0.13 m/yr, it is estimated that the lead would reach the edge of the lead sheet in about 23 years (average value). The additional time it would take lead to move from the edge of the lead sheet to the drainage ditch (Figure 2.5) should be much less than 23 years because of the relatively short distance and steep slope of the clay dike. Thus, 23 years can be assumed to be an average transport time for corroded lead to reach the drainage ditch. Lead corroded from the edge of the lead sheet would reach the drainage ditch before 23 years.

The transport of lead off the cover by other means such as erosion of the cover with the lead bound to the eroding soil particles is not expected to be significant because the cover will be maintained to prevent erosion.

Another potential problem with the lead sheet is the development of holes by pit corrosion. If a sufficient number of holes formed in this manner, the sheet would leak and water would infiltrate the buried wastes. A rough estimate of the possible impact of pit corrosion can be obtained as follows. The maximum pit depth P in a unit area of the sheet would depend on the exposure time t by

$$P = kt^n \tag{I.20}$$

where k and n are constants (Uhlig 1948). Data obtained by burial of samples of lead pipes in different types of soils for 11 years showed maximum pit depths ranging from 0.15 mm (0.006 in.) to 2.7 mm (0.1 in.), with the depths depending on the composition of the lead and the soil type (Amistadi 1985). At low corrosion rates of 0.0025 mm/yr (0.0001 in./yr), pit depths ranged from 0.15 to 0.87 mm (0.006 to 0.034 in.), with an average depth of 0.46 mm (0.018 in.). At the highest corrosion rate of 0.01 mm/yr (0.0004 in./yr), pit

depths ranged from 1.2 to 2.7 mm (0.046 to 0.107 in.), with an average depth of 2.2 mm (0.087 in.).

In studies on the burial of steel in different types of soil for varying time periods, values of the exponent n ranged from 0 to 0.9, where the higher values refer to poorly aerated soils and the lower values to well aerated soils (Uhlig 1948). For soils with good to poor aeration, n ranged from about 0.33 to 0.5.

Using an average maximum pit depth of 0.46 mm (0.018 in.) in 11 years and n = 0.5 (a conservative assumption), the estimated maximum pit depth in 1,000 years for the low corrosion rate is 4.3 mm (0.17 in.) per unit area.* Because this is less than the sheet thickness of 6.3 mm (0.25 in.), pits would probably not break through the lead sheet for 1,000 years following site closure. Differential rates of settlement of the underlying wastes and cover materials could, however, result in stresses on the lead sheet with tears developing that would allow water to infiltrate sooner. At the highest corrosion rate, holes would occur in the sheet in only 90 years, resulting in leakage at that time.

I.5 CALIBRATION OF CONTAMINANT TRANSPORT

I.5.1 Raffinate Pits Area

No on-site data were available for calibration of the groundwater transport model. However, off-site data are available for the Shaw Well No. 2 in the Busch Wildlife Area, located about 1.6 km (1 mi) in a downgradient direction from the raffinate pits area. This well is shallow (15 m [50-ft]) and therefore probably draws water from the upper limestone aquifer that was modeled in this analysis. Predicted peak concentration contributions (exclusive of background) are 2.7 mg/L nitrate (as N), 0.055 mg/L chloride, and 0.082 mg/L sulfate. The corresponding measured values are 26 mg/L nitrate (as N), 39 mg/L chloride, and 35 mg/L sulfate (Mo. Div. Health 1983). The measured values are all higher than the predicted values. However, a direct comparison is not useful because background values appropriate for water in

*The size of the unit area is equal to the average area of the samples of lead used in the pit corrosion study (Amistadi 1985). The area is not known, but it is not likely to be more than 0.093 m^2 (1 ft^2). The reason that the maximum pit depth is considered per unit area rather than for the whole sheet is that the amount of water that passes through the lead sheet per unit time depends on the density per unit area of sheet perforations and not on whether or not one pit breaks through somewhere in the sheet. Also, the statistical nature of the distribution of pit sizes, depths, and spacings must be considered.

the upper limestone aquifer are not available. Comparisons for other less mobile chemicals are not useful because of the long delay times for movement of contaminants away from the raffinate pits area. Concentrations of nitrate, sulfate, and chloride measured in other nearby wells are lower than in Shaw Well No. 2: < 2 mg/L chloride, 0.1 mg/L nitrate (as N), and 26 mg/L sulfate for the Cassidy well; and 3 mg/L chloride, < 1 mg/L nitrate (as N), and 28 mg/L sulfate for Shaw Well No. 1 (Mo. Div. Health 1983). However, these wells are much deeper, 110 m (Cassidy well) and 79 m (Shaw Well No. 1). As a result, water quality data from these wells are not useful as background values for water in the upper limestone aquifer. For similar reasons, water quality data measured for other wells in St. Charles County (Mo. Div. Health 1983) cannot be used to obtain relevant background data.

There are several possible sources of the high nitrate and chloride concentrations in the water in Shaw Well No. 2. These include contamination from the Weldon Spring site, abandoned septic tanks, and fertilizer runoff and seepage. Without more data it is not possible to determine conclusively the source of contamination of this well. One argument against the Weldon Spring site as being the source of the high nitrate concentrations is the following. The Shaw Well No. 2 water contains gross alpha and gross beta concentrations that are in the range of values found in a survey of 25 private wells in St. Charles County (Mo. Div. Health 1983). The water in Burgermeister Spring, which may have an underground connection to the Ash Pond outflow, is different in that it has high gross alpha and gross beta concentrations in addition to a high nitrate concentration (see Appendix H, Tables H.8 and H.16).

I.5.2 Quarry Area

Comparison of model predictions with measured values of concentrations of chemical and radiological species in groundwater is useful only at locations where the predicted values are considerably above background values. This limits the comparison to regions at or near the quarry. Comparisons in the river alluvium, at the locations of most of the LW wells where good measured values are available, are not useful because the predicted values of concentrations contributed by the quarry wastes for the years for which measured data are available are very much smaller than background values.

Measured and predicted values of concentrations of selected chemical species at locations in the quarry are given in Table I.13. The locations are shown in Figure I.10. For most species, the predicted values are higher than the measured values. For example, the predicted value for arsenic is higher than the measured values at the quarry pond by factors ranging from 5 to > 700

Table I.13. Measured and Predicted Values of Concentrations of Some Chemical Species in Groundwater at the Quarry[a]

Chemical Species	Quarry Pond[d]					TW-6[d]		
	Measured[b]			Predicted 1985	Predicted/ Measured[c]	Measured[b]	Predicted 1985	Predicted/ Measured[c]
	1980-1981	1984	1984-1985			1980-1981		
Arsenic	< 0.1-0.15 (5)	<0.001	<0.04	0.7	5->700	<0.1 (7)	0.9	> 9
Antimony	<0.1 (6)	-	-	-	-	<0.1 (7)	-	-
Beryllium	<0.001 (3)	<0.0005	<0.001	-	-	-	-	-
Cadmium	<0.01-0.01 (6)	<0.001	<0.007	-	-	<0.01 (7)	-	-
Chromium	<0.01 (6)	<0.001	0.02	0.5	25->500	<0.01-0.01 (7)	0.6	≥60
Copper	<0.01 (6)	<0.001	<0.01	1.7	>170->1700	<0.01-0.028 (7)	2.1	75->210
Lead	<0.05 (6)	0.002	<0.05	1.5	>30-750	<0.05-0.07 (7)	1.8	25->36
Mercury	-	<0.0001	<0.0005	0.01	>20->100	-	0.012	-
Nickel	<0.01-0.02 (6)	<0.001	<0.04	-	-	<0.01-0.04 (7)	-	-
Selenium	<0.01 (6)	<0.005	<0.005	0.9	>90->180	<0.1 (7)	1.0	>10
Silver	<0.01-0.01 (3)	<0.001	<0.003	-	-	<0.01 (3)	-	-
Zinc	0.01-0.31 (6)	0.005	0.02	5.8	19-580	<0.01-18 (7)	7.1	0.4->710
Cyanide	-	-	<0.02	0.13	>6.5	-	-	-
Uranium	2200-2800 (3)	1500	620-1200 (2)	5800	2.1-9.4	220-8500 (4)	5800	0.7-26

Table I.13. Continued

Chemical Species	TW-8[d] Measured[b] 1980-1981	TW-8[d] Predicted 1985	TW-8[d] Predicted/ Measured[c]	TW-9[d] Measured[b] 1980-1981	TW-9[d] Predicted 1985	TW-9[d] Predicted/ Measured[c]
Arsenic	<0.1-0.1 (6)	0.5	≥5	<0.1 (6)	0.6	>6
Antimony	<0.1 (6)	-	-	<0.1 (6)	-	-
Beryllium	<0.001 (2)	0.01	>10	<0.001	0.01	>10
Cadmium	<0.01 (6)	-	-	<0.01 (6)	-	-
Chromium	<0.01-0.02 (6)	0.3	15->30	<0.01 (6)	0.4	>40
Copper	<0.01-0.02 (6)	1.1	55->110	<0.01-0.08 (6)	1.4	18->140
Lead	<0.05-0.2 (6)	0.8	4->16	<0.05-0.06 (6)	1.1	18->22
Mercury	-	-	-	-	-	-
Nickel	0.01-0.04 (5)	-	-	<0.01-0.028 (6)	-	-
Selenium	<0.1 (6)	0.7	>7	<0.1 (6)	0.7	>7
Silver	<0.01 (2)	-	-	<0.01	-	-
Zinc	<0.01-0.30 (6)	3.9	13->390	<0.01-0.39 (6)	4.6	12->460
Cyanide	-	-	-	-	-	-
Uranium	7000[e]-8100 (5)	1500[f]	0.17-0.21	2000[e]-5800 (6)	1500[f]	0.26-0.75

[a] Values rounded to, at most, two significant figures. A dash means no data available (measured) or not calculated. Calculated values are given for only those species that had sufficiently high mobilities and concentrations in the quarry wastes to give meaningful values for comparison.

[b] The values in parentheses are the number of samples collected. The number is not given when only one sample was collected.

[c] Values are given as a range, with the lowest and highest values equal to the predicted value divided by the highest or lowest measured value, respectively. If the measured value is an upper limit, then the predicted/measured ratio is given as a lower limit -- i.e., greater than (>).

[d] Concentrations are mg/L for all species except uranium; uranium concentrations are pCi/L (natural uranium).

[e] Low value is 1985 value reported by Layne Western (1986).

[f] Value predicted for 1980.

Sources: 1980-1981 data, Berkeley Geosciences Associates (1984); 1984 data, U.S. Geological Survey (1984); 1984-1985 data, Bechtel National (1985) and Layne Western (1986).

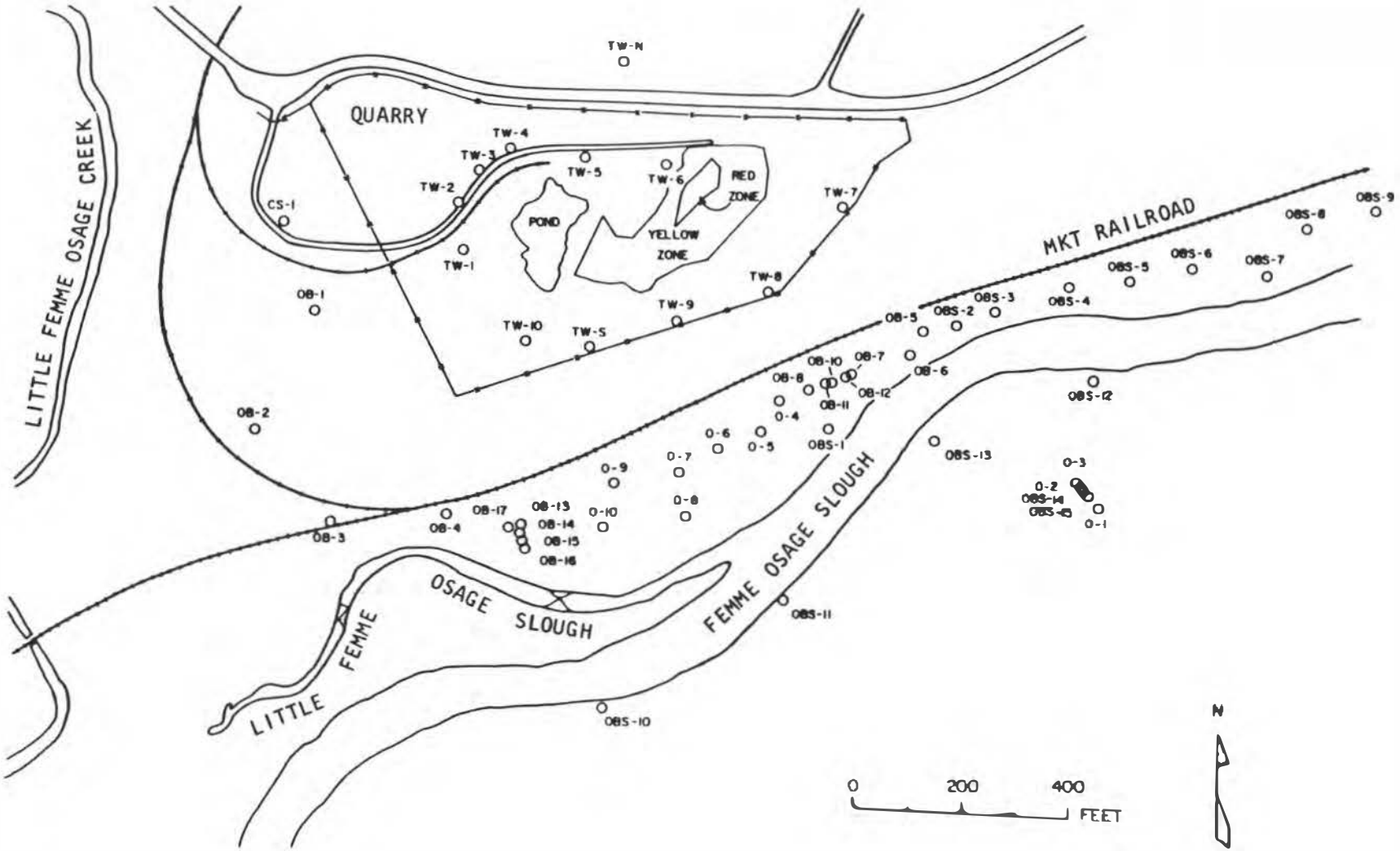

Figure I.10. Locations of Boreholes Outside the Quarry Floor in the Vicinity of the Weldon Spring Quarry Area. Source: Modified from Berkeley Geosciences Associates (1984).

(5 is obtained by dividing the predicted value 0.7 by the largest measured value, 0.15; > 700 is obtained by dividing 0.7 by the smallest measured value which is an upper limit, < 0.001). For Wells TW-6, TW-8, and TW-9, the predicted values are higher than measured values by factors of more than 9, 5 or more, and more than 6, respectively. These low factors are not too meaningful, however, because the measurement detection limit of 0.1 mg/L is so high. For the same reason, the comparison between predicted and measured values at TW-6, TW-8, and TW-9 for selenium is not too meaningful. The same problem exists for copper, lead, and chromium because most measured values are upper limits.

In general, the predicted values are in reasonable agreement with measured values for uranium and possibly zinc. For uranium, the agreement is good. For zinc, the agreement is good for TW-6 (values measured in 1981 ranged from 5.2 to 18 mg/L) and fair to poor for the other test well locations. Zinc and uranium are also the only species for which most of the measured values are real values and are not detection limits. For each species, measured values are generally lower in the pond than at the TW-6, TW-8, and TW-9 locations whereas predicted values are similar at all locations. The lower pond values may reflect problems in sampling a possibly heterogeneous and stratified body of water.

A similar comparison between measured and predicted concentrations can be made for samples collected in 1985 from Wells 08-6, 08-10, and 08-11 in the alluvium between the quarry and slough (measured data are given in Appendix H, Table H.16 and Figure H.5). The results show general agreement for some species, with the predicted/measured ratio ranging from 0.16 to 18 for uranium, 1 to 20 for copper, and 0.5 to > 15 for lead. The ratios for selenium (> 40) and arsenic (from 20 to > 40) are higher. This comparison in the near alluvium is problematic because a background value should be subtracted from the measured values before comparison with the predicted values, and background is likely to be a large component of the measured values in the near alluvium in 1985 for most chemical species.

Comparisons between predicted and measured values are further complicated by the large variability in measured values for the different 08 wells. For example, measured concentrations in Well 08-10 were 5,200 pCi/L uranium, 0.20 mg/L lead, 0.28 mg/L copper, and 280 mg/L sulfate. In the adjacent Well 08-11, the corresponding concentrations were 120 pCi/L uranium, 0.032 mg/L lead, 0.04 mg/L copper, and 62 mg/L sulfate (Hengerson 1985; Layne Western 1986). These large differences may reflect the strong dependence on location of the flow in the fractured limestone.

Calibration of the model by adjusting the input parameters to give agreement between prediction and measurement at the quarry (TW wells and boreholes

in the quarry wastes) is complicated by the following aspects. One cannot adjust parameters that affect the predicted results for all species in the same way without jeopardizing the reasonable agreement between prediction and measurement for uranium. Parameters that fall into this category include such things as the velocity of groundwater at the quarry, the thickness of the limestone aquifer, and the conservative assumption that 90% of all chemical species leaching from the quarry enter the near alluvium from the limestone and 10% remain in the limestone and move under the near alluvium and river alluvium. This problem may be avoided by adjusting parameters whose values can be chosen individually for each chemical species. Parameters in this category include both Kd values and concentrations of different chemical species in the quarry wastes and Kd values in the limestone.

The Kd values, which were chosen as described in Section I.3, are not based on values measured for the quarry wastes; consequently they can be adjusted to calibrate the model. However, adjusting the Kd values for each chemical species to give good agreement between prediction and measurement at the quarry requires the use of Kd values that, for many chemical species, are either at the upper end of or are above the range of measured values given in the literature. For example, Kd values of > 4,800 mL/g copper, > 590 mL/g arsenic, and > 4,900 mL/g lead are required to give agreement between prediction and measurement. The values for copper and arsenic are outside the range of reported measured values, which range up to 60 mL/g for arsenic and up to 560 mL/g for copper (Wangen et al. 1982; Baes and Sharp 1983; Sheppard et al. 1984). The range of measured values reported for lead is 4.5 to 7,600 mL/g (Baes and Sharp 1983; Sheppard et al. 1984), which includes the required value close to the upper end of the range.

Concentrations in the quarry wastes are based on samples of soil taken from boreholes drilled to refusal in rubble or bedrock, whichever came first (Bechtel Natl. 1985; Hickey 1986). The average concentrations were obtained assuming that, for each chemical parameter, the concentrations in the rubble and in any contaminated soil below the rubble are the same as they are in the sampled material. Because the rubble is a major component of the wastes, this assumption introduces uncertainty into the averages. The predicted concentrations in groundwater will vary linearly with the concentrations in the wastes. Thus, if the average concentration of a specific contaminant in the wastes is higher by a factor of two, then the predicted groundwater concentrations will be higher by a factor of two.

Another aspect of the calibration problem is that the predicted values at or near the quarry are uncertain because the grid size used in the model calculations, which is appropriate for predictions at the county well field, is too large for accurate model prediction at or near the quarry. However,

uncertainties contributed by this source are not expected to be as large as those related to Kd values.

Other sources of uncertainties include model assumptions such as the use of step-function source term concentrations (Section I.4.2) and the ion-exchange model (Gilbert et al. 1983) to estimate contaminant release rates. For mobile species such as cyanide, additional uncertainties are involved in (a) extrapolating concentrations measured in the wastes in 1985 back to 1963 to obtain the appropriate source term concentration and (b) assuming that 1963 is the year in which wastes containing cyanide were placed in the quarry.

The above discussion illustrates the problems associated with attempts to calibrate the model calculations with existing data. The availability of more measured data would justify calculations that are more accurate for the quarry and the near alluvium region than those described in Section I.3. In particular, Kd values and more accurate measurements of concentrations are needed for chemical species in the quarry wastes (see Section I.4.2). The availability of these additional data would then justify model calculations in which a smaller grid size was used.

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APPENDIX J. ESTIMATION OF AIRBORNE RADIOACTIVE RELEASES

In this document, the calculation of airborne radioactive releases from the Weldon Spring wastes was based on estimating the releases from three types of sources: (1) continuous releases of radon gases from the wastes, (2) temporary, or "puff", releases of radon gases from disturbance of the wastes (excavation, transport, and reburial), and (3) releases of resuspended particulates into the atmosphere from exposed waste surfaces and from movement of contaminated materials.

J.1 CONTINUOUS RELEASES OF RADON GASES

Radon gases are one source of radioactive releases from the Weldon Spring wastes. Radon gases are decay products of radium, which is a naturally occurring radioactive element produced by the decay of uranium and thorium. Radon-222 is a decay product of radium-226 and has a half-life of 3.8 days; radon-220 is a decay product of radium-224 and has a half-life of 55 seconds (see Appendix H, Figures H.1 and H.2). The radon gas produced by radioactive decay of radium is released into the spaces between the grains of waste material and diffuses toward the surface; much of it undergoes radioactive decay enroute. The radon gas that reaches the surface escapes into the atmosphere and is transported by wind to the surrounding environs where it can result in human exposure to radiation. Because the Weldon Spring wastes contain radionuclides from both the uranium and thorium decay series, releases of both radon isotopes will occur.

In the current configuration of wastes at the Weldon Spring site, both radon isotopes could present a hazard. However, after remedial actions have been completed, radioactive gaseous release to the atmosphere would consist mainly of radon-222. The amount of radon-220 escaping through the engineering cover would be very small because its short half-life (55 seconds) would result in almost complete decay before the gas could reach the ground surface.

The axial radon gas concentration and flux with multiple layers of wastes and cover materials was calculated using diffusion theory. The one-dimensional diffusion equation for estimating the diffusion of radon gas (U.S. Nucl. Reg. Comm. 1983) is:

$$\frac{\partial C}{\partial t} = \frac{D}{p}\frac{\partial^2 C}{\partial x^2} - C\lambda + \beta \qquad (J.1)$$

where: C = radon gas concentration (pCi/cm^3 of pore space); D = effective bulk diffusion coefficient (cm^2/s); λ = radon gas radioactive decay constant (s^{-1}); β = radon gas source ($pCi/s \cdot cm^3$ of pore space); t = time (s); x = thickness of cover (cm); and p = void fraction or porosity. The ratio of the

effective bulk diffusion coefficient to the void fraction or porosity (D/p) is referred to simply as the diffusion coefficient.

The diffusion coefficient, D/p, can be evaluated as:

$$D/p = 0.106e^{-0.261M} \tag{J.2}$$

where: M = weighted percentage of soil moisture.

The radon gas source term, β, can be expressed as:

$$\beta = \lambda \left(\frac{\varepsilon}{p} \cdot [Ra] \cdot \rho\right) \tag{J.3}$$

where: ε = emanating power (0.2 for radon-222; 0.1 for radon-220); [Ra] = radium-226 or radium-224 concentration (pCi/g); and ρ = waste density (g/cm^3). The emanating power is the fraction of radon gas that migrates from the waste particles into the surrounding void space.

Assuming steady-state conditions, Equation J.1 can be rewritten as:

$$-\frac{d}{dx}\left(\frac{D}{p} \cdot \frac{dC}{dx}\right) + \lambda C = \beta \tag{J.4}$$

A finite-difference scheme is used to solve Equation J.4 for the multilayer case, applying boundary conditions that preserve continuity of radon gas concentration and flux between the layers.

One important parameter in estimating the diffusion coefficient is soil moisture, which can be evaluated empirically (Kalkwarf et al. 1984) as follows:

$$m = \left[0.124P^{1/2} - 0.0012E - 0.04 + 0.156f_{cm}\right]\left[1 - \left(\frac{0.7 + f_{cm}}{H}\right)^2\right] + \left(\frac{0.7 + f_{cm}}{H}\right)^2 \tag{J.5}$$

where: m = moisture saturation; P = annual precipitation (in.); E = annual evaporation (in.); f_{cm} = fraction of soil passing a U.S. Standard Sieve No. 200; and H = distance from the ground surface to the water table (ft).

The moisture saturations for soil were calculated to be 0.83 at the Weldon Spring site and 0.46 at the Hanford site. The physical properties of the "Nearby Site" were assumed to be same as those of the Weldon Spring site. Hence, the moisture saturation for the "Nearby Site" is also 0.83. Using an average waste density of 2 g/cm^3 (ranging from 1.8 to 2.2 g/cm^3 for

different alternatives) and a porosity of 0.3, the weighted percentages of soil moisture were calculated to be 13% at the Weldon Spring site and "Nearby Site" and 6.9% at the Hanford site. The estimated diffusion coefficients of radon gas in soil were based on the conceptual design of the disposal cell cover, which is summarized in Table J.1.

Table J.1. Conceptual Design of the Disposal Cell Cover for the Weldon Spring Wastes

	Thickness (m)/Moisture (%)/ Diffusion Coefficients (cm^2/s)	
Material	Weldon Spring Site and "Nearby Site"	Hanford Site
Topsoil	0.46/6.9/0.0040	1.2/6.9/0.018
Sand/gravel	0.30/6.9/0.018	Not applicable
Riprap	0.90/0.0/0.10	0.90/0.0/0.1
Clay or soil	1.5/13/0.040	0.90/6.9/0.018

Due to the current activity imbalance between radium-226 and thorium-230, there will be radium-226 ingrowth with time (see Appendix H, Figure H.6), which is important for long-term management. The ingrowth kinetics of radium-226 can be described by the following formula:

$$A = \left[\lambda A_{p0}(e^{-\lambda_p t} - e^{-\lambda t})/(\lambda - \lambda_p)\right] + A_0 e^{-\lambda t} \qquad (J.6)$$

where: A = activity concentration of radium-226 (pCi/g); λ = radioactive decay constant of radium-226 (s^{-1}); λ_p = radioactive decay constant of thorium-230 (s^{-1}); A_0 = initial activity concentration of radium-226 (pCi/g); A_{p0} = initial activity concentration of thorium-230 (pCi/g); and t = time (s). A similar situation exists for radium-224 and its parent thorium-228.

These equations were used to estimate the time-dependent release of radon-222 and radon-220 gases for the alternatives considered in this EIS.

J.2 TEMPORARY GASEOUS RELEASES FROM DISTURBANCE OF THE WASTES

Gaseous releases of radioactivity can also occur due to disturbance of the wastes from activities such as excavation, transfer, or reburial. When the wastes are disturbed and exposed to the air (e.g., when they are excavated or unloaded at a disposal site), radon-222 and radon-220 gases that have built

up in the interstitial void spaces in the wastes may be released in "puffs". For this analysis, it was assumed that the total amount of radon gas available for release -- i.e., 20% of the radon-222 and 10% of the radon-220 in the wastes -- will be released in puffs. These percentages are the same as those used in the calculation of steady radon gas releases and represent the amount of gases in the interstitial void spaces of the wastes. The remainder of the gases will be trapped inside individual particles and clumps of wastes and will not be available for release but will decay to solid products before they can escape to the air. After the wastes are again covered with clay, soil, etc., gaseous releases (especially radon-220 releases) will be markedly reduced.

J.3 RELEASES OF RESUSPENDED PARTICULATES

There will also be releases of radioactive particulates associated with exposed waste surfaces and with activities involving movement of the contaminated materials. It is expected that control methods (such as periodic watering and minimizing exposed surfaces) will be used at the Weldon Spring site and/or at an alternative disposal site. A release rate of 0.2 g/kg of material excavated was used to estimate particulate release rates during waste-retrieval activities. This release rate is based on previous experience with earth-moving activities as reported by the U.S. Environmental Protection Agency (1977) and Argonne National Laboratory (1982). This release rate was modified by factors accounting for the specifics of waste-retrieval activities (e.g., use of water sprays and other methods to reduce particulate emissions) and a weather correction factor that accounts for local meteorological effects (principally, precipitation and evaporation). Use of water sprays to control dust emissions was assumed to reduce the release of radioactive particulates by a factor of two. A weather correction factor of $0.85/(PE/100)^2$ was used in this analysis, where PE is the annual precipitation-evaporation (PE) index (which is an indicator of average surface moisture). A national map showing PE values is available to facilitate the calculation of the weather correction factor (U.S. Environ. Prot. Agency 1977). The weather correction factors were calculated to be 0.8 for the Weldon Spring site and "Nearby Site" and 14 for the Hanford site (Argonne Natl. Lab. 1982).

Wind erosion on the exposed waste surfaces can also result in airborne releases. The release rate due to wind erosion at a typical site is 0.5 kg/1,000 m^2/d (U.S. Environ. Prot. Agency 1977). This emission rate was reduced by a factor of 10 at the Weldon Spring site and "Nearby Site" to account for the reduction afforded by the lush vegetation. Such a reduction was not used for the Hanford site.

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APPENDIX K. PHYSICAL AND BIOLOGICAL EFFECTS ON CONTAINMENT SYSTEMS

K.1 BIOTIC EFFECTS ON LONG-TERM INTEGRITY OF CONTAINMENT SYSTEMS

K.1.1 General Effects

Biota can affect the long-term integrity of disposal cells both beneficially and adversely. For cells that have multilayered covers with vegetated soil as the outermost layer, beneficial biotic effects include soil stabilization and erosion control. Adverse biotic effects include plant and animal intrusion into the wastes followed by mobilization and dispersal of contamination via both physical and biological pathways.

Disturbance of the cover by burrowing animals, the creation of channels by plant roots, and the formation of soil aggregates by microorganisms all result in decreased bulk density of the soil and increased number of interconnected voids. Such voids increase water infiltration and provide routes for radon gas escape. Also, penetration of roots through the cover can lead to direct contact of the wastes by rainwater (McKenzie et al. 1982). Increased water infiltration enhances the likelihood of slippage or slumping of the cover materials, especially in the more steeply sloped flanks. Additional undesirable effects of increased infiltration are more water moving through the cover layers into the wastes, accelerated leaching, and potential contamination of groundwater. These effects can be balanced by an enhancement of water storage capacity of soils by biological activity, as well as by the ability of vegetation to absorb and transpire large amounts of soil moisture back to the atmosphere.

Root intrusion into buried radioactive wastes has been reported (Cline and Uresk 1979; Fitzner et al. 1979; Breedlow et al. 1982; Yamamoto 1982). When the riprap and other layers of cell covers have been breached by deep-rooted plant species, some waste constituents may be taken up and transported to above-ground or near-surface plant organs (Knight 1983). Waste constituents can then be dispersed either directly to the atmosphere (as in the case of radon gas) or to food webs via a variety of herbivores (e.g., mammals, reptiles, insects). Physical dispersal of contaminated plant parts by wind and water can also occur. Prevention of intrusion by deep-rooted vegetation (e.g., trees) will be part of long-term site maintenance.

The cover thickness for the various action alternatives is about 3 m (10 ft). This thickness of cover is sufficient to largely prevent burrowing of mammals into the wastes (see Burt and Grossenheider [1964] and Schwartz and Schwartz [1959]). Insect species such as harvester ants can also burrow to depths ranging from 2.4 to 3.0 m (7.9 to 9.7 ft) (Willard 1964; Headlee and Dean 1968). In moister areas (e.g., Weldon Spring and "Nearby Site"),

earthworms can be abundant and are capable of burrowing as deep as 0.9 to 1.8 m (3 to 6 ft) (Smith 1974).

Burrowing into buried radioactive wastes has been reported for rodents, carnivores, ants, and termites (Fitzner et al. 1979). Tunneling by animals causes soil pulverization, transfer of materials between layers of the disposal cell structure, and creation of voids. Any wastes brought to the surface by burrowing would be subject to accelerated erosion and dispersal in the environment. The potential exists for the thickness of the cover to be reduced after a few years, rendering the waste more readily available for transport by physical or biological processes (McKenzie et al. 1982).

In addition to the effects of burrowing, rodents, insects, and native mammalian herbivores may jeopardize cover integrity by grazing on the cover vegetation (Whicker 1978). Grazing can cause a decrease in vegetative cover, leading to increased erosion and a decreased number of the more palatable plants -- primarily the fibrous-rooted species with the most soil-holding capacity. The shallow- and tap-rooted species therefore increase, reducing the overall soil holding capacity of the vegetation (Young 1943; Gates 1974).

Impacts from the biomodification and biointrusion mechanisms presented above are summarized in the following sections, grouped according to similarity of environments and results.

K.1.2 Weldon Spring Site and "Nearby Site"

The disposal cell covers at both the Weldon Spring site and the "Nearby Site" could be invaded by roots and animals. Initially, the effective rooting zone would be only about 0.61 m (2 ft) (0.46 m [18 in.] topsoil plus 0.15 m [6 in.] sand), precluding the development of most mature native trees (Spurr and Barnes 1973) and limiting many other deep-rooting plants. Shrubs and trees would germinate and become established in this soil depth, but their growth would slow when the roots encountered the clay or riprap interface.

Tree development on the covers would be controlled by long-term maintenance. The effectiveness of controlling tree growth would depend on both the species that invade the covers and the schedule for tree removal. For example, bur oak taproots may penetrate to depths of 1.5 m (5 ft) after one growing season, 2.1 m (7 ft) after two growing seasons, and 2.9 m (9.5 ft) after three growing seasons (Weaver and Kramer 1932). Although these depths occur under optimal conditions of cultivated soil, the penetration of roots to the riprap layer within one season is possible because roots would not be

extracted in tree removal and they could either send up suckers (for some species) or decay and leave channels through which other plant roots could follow.

Extensive lateral rooting can be assumed once the riprap layer was reached, resulting in increased water infiltration, decreased soil bulk density, and other conditions that compromise cover integrity. The covers would eventually be breached as soil settling and rock fragmentation filled the interstices between the riprap with particles and provided a soil pathway for plant roots (Hakonson et al. 1983). However, the 1.5 m (5 ft) clay layer would afford protection from root invasion for some period of time because impermeable soil layers and hardpan reduce soil aeration and curtail root growth (Kramer and Kozlowski 1960). Cover repair could be conducted in areas where root penetration is most serious.

Plant succession on the covers will be regularly interrupted by maintenance activities aimed at precluding or destroying large, deep-rooted plants. If maintenance ceased, cover communities might initially "stagnate" in a grass/forb/small-shrub/small-tree stage, with a few individuals of larger tree species usually present. These young trees might not reach maturity because they (and some of the other plants) would die out during periods of drought. However, some deep-rooted species that are drought resistant, such as oaks and chokeberry, could become established regardless of soil moisture conditions (Yeager 1935). Eventually, deep-rooted mature trees would develop. Continued soil modification, development, and settling combined with vegetation succession would result in the establishment of a climax forest typical of each area.

The clay liner and the riprap layer in the cell cover would initially deter burrowing animals from intruding into the wastes. However, as root growth modified and disrupted these layers, mammals, ants, and other insects could intrude into the contaminated materials because the materials would be within the known tunneling depths of these organisms (Cline et al. 1982). The total volume of soil brought to the surface could be significant when all potential burrowers are considered over a period of years. For example, at a reference eastern low-level radioactive waste burial site, an average of six woodchucks occur per hectare; they were estimated to bring 0.9 m^3 (32 ft^3) of soil to the surface in the first year. A much larger volume was moved by the 10 moles that occur per hectare, with a value of 6.5 m^3 (230 ft^3) of soil (McKenzie et al. 1982).

K.1.3 Hanford Site

The Hanford site will have covers with an outermost native soil layer 1.2-m thick, underlain by a graded riprap layer 0.9-m thick over 0.9 m of

native soil. No clay layer is included in these covers. Native shrub-steppe grassland -- consisting of grasses, forbs, and shrubs adapted to arid conditions -- would develop on the covers. These plants are mostly deep-rooted, with high levels of root growth activity when soil moisture is available (Fitzner et al. 1979). Most of the plant biomass in these species occurs underground. Typical root-to-shoot biomass ratios are 9:1 (Caldwell and Fernandez 1975).

The disturbed soil cover of the completed cover would probably be initially colonized by weedy annual plants such as Russian thistle, kochia, and cheatgrass. The upper 1.2 m of soil would be adequate to support a nearly normal native plant community that would become reestablished early in the long term. Because the riprap layer would initially be a barrier to most penetrating roots, the deep-rooted shrubs and forbs could experience moisture stress during periods of drought early in their development. Loose rock has been experimentally demonstrated to be an effective barrier to burrowing mammals and insects (Cline et al. 1980); however, the experimental time in this study was only one summer for the mammal (pocket mouse) and two years for the insect (harvester ant). Soil settling, rock fragmentation, and the soil-pulverizing activities of burrowing animals and ants would eventually fill some of the interstices between the rocks of the riprap with soil particles and provide a pathway for root extension through the layer.

Once through the riprap layer, the plant roots would quickly penetrate through the 0.9 m (3 ft) of native soil underlying the riprap and would intrude into the buried wastes. Continued penetration of the riprap layer by new roots and the death of old roots would permit small rodents (mice and pocket gophers), ants, and other insects to gain access to the wastes via root channels because the deepest wastes are well within known tunneling depths of these organisms in the loose sandy soils at Hanford (Cline et al. 1982). These events would likely occur early in the maintenance and monitoring period. Fitzner et al. (1979) found harvester ant colonies on waste-burial sites at the Hanford site to be five times more dense than ant colonies occurring on undisturbed soil at control sites of equivalent size.

Because of the sandy soils and the greater below-ground activity of both plant roots and burrowing animals at Hanford, biointrusion into the buried wastes and residues would probably occur sooner than at the Weldon Spring site or the "Nearby Site".

K.1.4 Mitigative Measures

The disposal cell covers, particularly at the Weldon Spring site and the "Nearby Site", have design features to delay biointrusion that are equivalent to design criteria listed by Cline et al. (1980). The features include

(1) maintenance of a dry burial zone beneath the cover (through the use of a leachate collection system under the wastes for Alternatives 2a, 2b, and 3b; and sand drainage layers in the cover for Alternatives 1, 2a, 2b, 3b, and 3c), (2) placement of adequate soil on the cover to retain the annual precipitation, (3) establishment of vegetative growth on the cover to remove soil moisture by evaporation and plant transpiration, and (4) use of layers in the cover that would prevent burrowing animals from creating channels. Measures to mitigate biointrusion can be of two types, correction and delay.

K.1.4.1 Corrective Measures

During the long term, maintenance would preserve cover integrity if the activity of plants and animals reached a level of concern. Such measures could include removal of large, deep-rooted plants; herbicide application to eliminate and prevent regrowth of such plants; maintenance of enclosure fences; trapping and/or poisoning of burrowing animals; insecticide application to control ants and other insects; mowing; and additions to and recompaction of the soil layer. However, such measures do not reduce the long-term potential for biointrusion (McKenzie et al. 1982) and would have to be reapplied periodically.

K.1.4.2 Delaying Measures

The covers for all alternatives would be affected by biomodification and biointrusion, with the potential for resulting dispersal of radionuclides to the environment. Although these events cannot be prevented, the cover designs could be modified to further delay biomodification and biointrusion. For example, the outer soil layer could be eliminated, leaving only rock on the outer surface, or a chemical "biobarrier" could be built into the cover to delay establishment of deep-rooted plants.

Rock Cover. At the Weldon Spring site and "Nearby Site", the outer 0.6 m (2 ft) of soil and sand could be eliminated, leaving the 0.9 m (3 ft) of riprap outermost. At the Hanford site, the trenches could be backfilled almost entirely with soil and the 0.9 m (3 ft) of riprap could be placed on top. Such rock armoring of the cover surface would provide adequate protection against water and wind erosion (Breedlow et al. 1982) and would greatly delay biointrusion. Over the long term, sediments might fill the riprap interstices and provide sites for vegetation. However, the resulting sparse community would have a much lower biomodification and biointrusion potential than a soil layer. A rock outer layer would also eliminate the increased soil erosion that would otherwise occur when vegetation on a soil outer layer was reduced because of drought, disease, fire, herbivory, or agricultural use.

An outer layer of rock is feasible at all alternative sites, but consideration would have to be given to its effect on water management. A rock outer layer would eliminate the water storage capability of the soil cover and the transpiration potential of the vegetation. Both of these processes can prevent a large portion of infiltrated precipitation from moving downward through the cover and into the wastes, leading to leaching and groundwater contamination (Hakonson et al. 1983). Transpiration by vegetation is especially important at the arid Hanford site. At the humid sites, water transport through the clay layer would be very slow, and most precipitation entering the cover would probably flow along the clay/sand interface and out of the cover at the periphery. This type of cover would be constructed so that the clay, sand, and riprap layers toe out at ground level with provision for erosion control in the cover perimeter areas.

Chemical Biobarriers. Chemical biobarriers have been developed to prevent root intrusion through soil-covered, vegetated covers (Cline et al. 1982). The most effective barrier consists of polymer pellets impregnated with an herbicide that is released slowly into the soil. The pellets would be placed in a layer during cover construction, probably at the sand/clay interface in the Weldon Spring site or "Nearby Site" covers. The purpose of this procedure would be to prune plant roots. The estimated effective lifetime of the chemical biobarrier is 100 years, which would help reduce maintenance costs during that period but would not reduce biointrusion and biomodification impacts in the long term.

K.2 GEOLOGICAL EFFECTS ON DISPOSAL CELLS

K.2.1 Erosion

Because DOE intends to continue monitoring the chosen disposal site and to take any remedial actions dictated by the monitoring, the effects of erosion should be counteracted by maintenance. Nevertheless, the erosion of disposal cells is discussed here as part of the analysis of a hypothetical loss of institutional control. Erosion rates were calculated for the Weldon Spring site and the "Nearby Site" using the Universal Soil Loss Equation (Section K.2.1.1). Predictions of erosion rates (mainly from wind) at the Hanford site and the uranium processing site are discussed in Section K.2.1.2.

K.2.1.1 Sheet and Rill Erosion at the Weldon Spring Site and "Nearby Site"

A formula for calculating the rate of erosion on slopes with variable steepness and/or variable vegetation cover (Wischmeier and Smith 1978), known as the Universal Soil Loss Equation (USLE), can be used to calculate the

average rate of sheet and rill erosion on a slope per unit area. It cannot estimate gully erosion or the rate of erosion at a given point on a slope. The equation is useful in estimating short-term soil erosion rates (years). It was used here for estimating erosion losses from containment cell covers over hundreds to thousands of years because it is the major analytical tool available. It is useful primarily for comparing the relative durability of containment covers with respect to erosion losses. The erosion rates calculated using this equation should not be interpreted as being definitive.

According to the USLE, the average rate of erosion is given by

$$A = RKLSCP \tag{K.1}$$

where A = erosion rate (tons/acre·year), R = rainfall factor, K = erodibility, L = slope length factor, S = slope steepness factor, C = vegetation cover factor, and P = support practice factor.

The slope length factor L is given by

$$L = \left(\frac{\lambda}{72.6}\right)^m \tag{K.2}$$

where λ = length of slope in feet and m = 0.5 for slopes ≥5%.

The slope steepness factor is given by

$$S = 65.41 \sin^2\theta + 4.56 \sin\theta + 0.065 \tag{K.3}$$

where θ = slope angle.

The rate of erosion can be converted to cm/yr using

$$E\ (cm/yr) = \frac{A \cdot F}{D} \tag{K.4}$$

where E = erosion rate (cm/yr), A = erosion rate (tons/acre·year), D = density of soil (lb/ft^3), and F (conversion factor) = (2,000 lb/ton × 12 in./ft × 2.54 cm/in.) ÷ 43,560 ft^2/acre = 1.40 $\frac{\text{acre-lb-cm}}{\text{ton-ft}^3}$. The parameters used for the erosion calculations are given in Table K.1.

To demonstrate possible erosion rates if site maintenance and land-use controls should cease, annual erosion rates were calculated using the USLE and the following land uses: (1) mature forest, (2) shrub forest consisting of grasses and shrubs, (3) row crops, (4) hay, and (5) grasses. The estimated erosion rates are presented in Table K.2.

Table K.1. Parameters Used to Calculate Erosion Rates of Topsoil at the Weldon Spring Site and "Nearby Site"

Parameter	Value[a]	Vegetation	Land Use
R	220	-[b]	-[b]
K	0.35	-	-
C	0.0001	Mature forest	Natural succession
	0.003	Shrub forest	Natural succession
	0.35[c]	Row crops	Agriculture
	0.01	Hay	Agriculture
	0.003	Grasses	Maintained
	1	None	Severe erosive events
P	1.0 for unmaintained slopes (shrub/forest and mature forest)	-	-
	0.5 for maintained slopes (row crops, hay, and grasses)	-	-
D	1.55 g/cm^3 (97 lb/ft^3)	-	-
λ	Actual slope lengths used as estimated from Figures 2.1, 2.2, and 2.5	-	-

[a] Source: Wischmeier and Smith (1978).

[b] A hyphen means not applicable.

[c] Assumes 4-year crop rotation of wheat, meadow, and corn (grown in two successive years), using good soil management practice (e.g., contour planting) (U.S. Dept. Energy 1986).

Table K.2. Estimated Erosion Rates and Times Required to Erode Topsoil and the Entire Cover at the Weldon Spring Site and "Nearby Site"[a]

Alternative	Land Use or Vegetation	Erosion Rate (cm/yr)[b]	Time to Erode Topsoil (yr)[c]	Time to Erode Entire Cover (yr)[c]
1	Mature forest	0.00035	130,000	900,000
	Shrub forest	0.011	4,300	30,000
	Row crops	0.61	73	520
	Hay	0.018	2,600	18,000
	Grasses	0.0053	8,500	60,000
2a, 3b	Mature forest	0.00068	66,000	470,000
	Shrub forest	0.020	2,200	16,000
	Row crops	1.2	38	270
	Hay	0.034	1,300	9,300
	Grasses	0.010	4,400	31,000
2b	Mature forest	0.00068	66,000	470,000
	Shrub forest	0.020	2,200	16,000
	Row crops	1.2	38	270
	Hay	0.034	1,300	9,300
	Grasses	0.010	4,400	31,000

[a] The cover consists of 0.46 m topsoil, 0.15 m sand, 0.9 m riprap, 0.15 m sand, and 1.5 m clay. It was assumed that the erosion rates are the same for all these materials (see text). The top of the cover has a 5% slope and the flanks a 20% slope. For these calculations, the average slope of the cover was used as estimated from Figures 2.1, 2.2, and 2.5.

[b] Erosion rates were calculated according to the parameters in Table K.1.

[c] The times to erode the topsoil and entire cover should be used for comparative purposes only. Because of limitations on use of the Universal Soil Loss Equation (see text) and major simplifying assumptions (i.e., constant climatological conditions), the calculated times are not definitive.

Conversion Factors: To convert convert centimeters per year (cm/yr) to inches per year (in./yr), multiply by 0.3937; to convert meters (m) to feet (ft), multiply by 0.3048.

After the topsoil and the top layer of sand have eroded away, the riprap will still protect the underlying clay. Some of the sand and soil will settle into the riprap. The effects of erosion on the riprap will depend on the material selected -- i.e., how it is affected by freezing and thawing and by chemical weathering in air and soil. Limestone, which is a common type of rock found in the Weldon Spring area, is highly susceptible to solution erosion, particularly if it is thinly bedded or fractured. If limestone riprap is used, it should be high-grade and free of fractures and bedding planes. Other materials such as granite or other silicate rocks that are less susceptible to dissolution could also be used as riprap to extend the lifespan of the cover. The erosion rate for the riprap layer was conservatively assumed to be the same as that for the soil layer.

The specific chemical and physical properties of the clay minerals in the clay layer of the containment cell cover will affect the durability of the cover. The clay minerals used in the clay layer should be selected to have properties that will minimize the infiltration of water into the underlying wastes. With respect to the potential erosion of the clay layer (after removal of the overlying riprap and soil layers), it was assumed that the clay layer is similar to the soil layer for purposes of calculating the erosion rate for the entire cover.

The time required to erode the topsoil and the entire cover at the assumed erosion rates for the various land-use patterns and alternatives is shown in Table K.2. For a natural succession from grass to mature forest, the estimated lifetime of the topsoil is very long. On the other hand, if the site were used for row-crop farming, erosion could be very fast. Row-crop farming could lead to very rapid erosion of the mounds whereas forest cover would lead to very slow erosion. The covers might not be suitable for row-crop agriculture because of the relatively thin topsoil layer over the riprap and clay. The cultivation of hay instead of row crops would lead to moderately slow erosion.

K.2.1.2 Wind Erosion at the Hanford Site and Uranium Processing Site

Hanford Site (Alternative 3a). Land use would be the primary factor affecting erosion rates at the Hanford site, so erosion was estimated for both an erosive and a nonerosive land use. The erosive land use was assumed to be overgrazing of rangeland, and the nonerosive land use was assumed to be the growth of native, undisturbed vegetation (i.e., a sagebrush community). These land uses represent two extremes because the burial trenches at Hanford are not expected to provide particularly good habitat for either native vegetation or rangeland. It was assumed that 30 years will be required to develop a sagebrush community.

For estimating wind erosion, it was conservatively assumed that the burial site is in an area of net erosion rather than deposition. The soil at the Hanford site was assumed to be a sandy loam (Uresk et al. 1977) with an undisturbed bulk density of 1.5 g/cm^3. It was also assumed that the riprap will be changed into soil by physical and biological weathering processes and will have the same rate of erosion as the soil whereas, in fact, it may erode much more slowly than soil.

Because soil losses at the Hanford site are estimated to be 100 to 200 times greater due to wind erosion than due to water erosion, only the wind erosion losses are reported here. Erosion losses at Hanford due to wind were computed according to the Wind Erosion Equation (WEE) (Woodruff and Siddoway 1965; Skidmore and Woodruff 1968; Skidmore 1983).

Erosion rate (tons/acre·year) due to wind is expressed as

$$\text{Erosion rate} = f(I' K' C' L' V) \qquad (K.5)$$

where I' = erodibility, K' = surface roughness factor, C' = climatic factor, L' = field length along the prevailing wind erosion direction, and V = vegetative cover factor. The parameters used to compute average wind erosion loss are presented in Table K.3.

Table K.3. Estimated Wind Erosion Rates for the Hanford Site

Land Use	Stage of Community Development	Length of Stage (yr)	WEE Factors[a]					Erosion Rate (cm/yr)
			I'	K'	C'	L'	V	
Sagebrush used for rangeland	Overgrazed	800	98	1.0	0.19	10,000	730	0.20
Native sagebrush community	Developing	30	98	1.0	0.19	10,000	730	0.20
	Mature	770	98	1.0	0.19	10,000	1,740	0.12

[a] I' = Erodibility factor. Computed assuming 20% of the soil particles of the site's soil is greater than 0.84 mm in diameter.
K' = Surface roughness factor. Assumes top of the burial trench is smooth.
C' = Climatic factor. Calculated according to Woodruff and Siddoway (1965).
L' = Open-field length. Assumes no attenuation of erosion due to limited length of field.
V = Vegetative cover factor. Assumes productivity of developing and overgrazed community to be 41 g/(m^2·yr) and mature community to be 80 g/(m^2·yr) (from Uresk et al. 1977).

The estimated erosion rates vary from 0.12 to 0.20 cm/yr (Table K.3). At these rates, at least 1,500 years would be required to erode the cover down to the wastes (3 m [10 ft]). However, assuming there is no redeposition of eroded materials over the site area once the 1.2 m (4 ft) of soil was eroded from the top, the riprap would probably shelter the soil somewhat from further erosion. Because wind erosion of the disposal site cover materials is likely to be concurrent with deposition of eroded material from off-site (primarily via wind erosion), the estimated rates of erosion for the disposal site should be much less. Thus, the estimated rates probably represent upper limits, and it can be expected that the covers will last even longer, possibly for several thousand years.

Uranium Processing Site (Alternative 3c). The impacts related to the operation and post-operation of a uranium processing site in the Four Corners area of the southwestern United States are described in detail in a report on uranium milling (U.S. Nucl. Reg. Comm. 1980).

The major geologically related impacts that need to be considered in the design of the disposal area are the potential salinization of land resulting from seepage from tailings and potential wind erosion of the tailings. These impacts are of particular importance in this arid/semiarid region where the rate of soil formation is very slow and the potential for wind erosion is high. The design of the tailings containment system will have to be considered on a site-specific basis if this alternative is selected.

K.2.1.3 Severe Erosion

Loss or reduction of ground cover -- whether from fire, disease, or drought -- would lead to increased rates of erosion at any of the disposal sites. The limited depth of the soil, compared to normal soil horizons, would increase the susceptibility to drought.

The loess-derived soils in the Weldon Spring area are highly subject to gully erosion if exposed. The clay layer would be even more subject to gully erosion if exposed because it has very low permeability and small grain size. Thus, in case of heavy loss of vegetation, severe gully erosion of the topsoil or any exposed clay would probably result. If gullies cut through the clay layer, the wastes would be susceptible to erosion.

A severe drought or fire at the Hanford site could destroy the vegetative cover over the trenches, and subsequent high winds could increase the rate of soil loss. The 0.9-m (3-ft) riprap layer might serve to reduce such losses by acting like "desert pavement" (a stone-covered surface resistant to wind erosion that is naturally present in deserts).

K.2.2 Floods

The Weldon Spring site is not on a floodplain, and there appears to be no danger of flooding. Any "Nearby Site" would also be selected to avoid any threat of flooding.

In arid climates, water erosion can be the dominant geomorphic factor. Sudden, heavy rains ("flash floods") may be responsible for substantial sheet erosion and possibly gully erosion of the trench covers. At the Hanford site, the flat topography of the plateau, the low relief of the trench covers, and the distance of the disposal site from major surface water drainage systems should minimize the potential for gully erosion. The flooding potential at a uranium processing site would depend on site-specific conditions.

K.2.3 Settling and Subsidence

Settling and subsidence of the wastes after emplacement in disposal cells at the Weldon Spring site or "Nearby Site" could lead to irregularities in the slope of the covers and result in concentration of runoff in some areas. Concentration of runoff would cause increased gully erosion rates in those areas. The rate of infiltration of water would also increase in areas of no slope. The effects of settling would be minimized by compacting the wastes and filling the voids with grout.

In the Weldon Spring area, the underlying limestone bedrock contains numerous large solution cavities and enlarged joint and bedding planes (Bechtel Natl. 1984b). However, geotechnical studies to date do not indicate the presence of large cavities beneath the raffinate pits or chemical plant areas that could cause extensive collapse (Bechtel Natl. 1984b). Geotechnical studies are currently being carried out to better characterize the underlying bedrock. Preliminary data from these latest studies do not indicate the presence of any large cavities (Blank 1986). If the "Nearby Site" (Alternative 3b) were located in an area underlain by shale bedrock, it would not be subject to this potential failure mode.

Eventually, large cavities could develop in the limestone under the Weldon Spring site because of the humid climate, the presence of the highly fractured limestone, and the flow of groundwater through the limestone. The collapse of a solution cavity, if located under a disposal cell, could result in the release of contaminants to the local groundwater and surface water systems and/or damage to the cover system. Cover system damage would increase infiltration of water to the wastes or erosion of the cover system. Such a collapse would be most likely following a change in the groundwater regime (e.g., due to prolonged drought or local excavation and construction projects).

K.2.4 Earthquake Damage

Even though the Weldon Spring site and the "Nearby Site" lie within the tectonically quiet Central Stable Region for earthquake activity (Bechtel Natl. 1983), it can be expected that seismic forces could be experienced in the future at these sites. Based on historic seismic activity within the region, the maximum expected seismic intensity for the two sites could range from Modified Mercalli VII to VIII or Richter magnitudes of 5.3 to 5.8, and expected maximum horizontal accelerations could be 0.1g to 0.2g (Algermissen et al. 1982; Bechtel Natl. 1983; Hopper et al. 1983).

During historic earthquakes of the New Madrid fault zone, most of the damage occurred on unconsolidated alluvium (Hopper et al. 1983). At the Weldon Spring site, the unconsolidated sediments consist of topsoil, clay, loess, and till over limestone bedrock. The sediments beneath the raffinate pits area are about 3 and 15 m (9 and 46 ft) thick and are dry (Bechtel Natl. 1984b). The area has good topographic drainage, and the groundwater table is in the underlying limestone (Bechtel Natl. 1984b). At present, there is water in and around the pits because the pits are not drained. However, once the wastes are stabilized and covered, the sediments beneath the wastes will not be saturated with water. As long as the materials beneath the wastes are not saturated with water, the potential for major failure of the containment features as a result of earthquakes is expected to be minimal.

Preliminary engineering evaluations of the effect of gravity and seismic forces on the engineered cover (Bechtel Natl. 1984a) indicate that the slopes will be stable and not subject to failure. As long as controls continue, repair will be possible if there is failure of the slope (presumably along a sand layer). To ensure structural integrity of the engineered containment system during seismic events, in-depth engineering evaluations -- similar to those done for DOE's Niagara Falls Storage Site (Bechtel Natl. 1985) -- will be done for the containment system during the design phase.

Damage to the disposal cell during severe earthquakes would be less likely to occur at the Hanford site than at the Weldon Spring site or "Nearby Site" because the tops of the disposal trenches would be nearly flush with the ground surface (e.g., no unstable side slopes) and no vulnerable low-permeability layer is planned as part of the containment.

The stability of the containment system for the tailings at the uranium processing site will depend on where the site is located, the specific characteristics of the tailings, and the specific containment system design.

K.2.5 Performance of the Clay Beneath the Wastes

At the Weldon Spring site or "Nearby Site", the ability of the "existing ground" or specially constructed clay liner beneath the wastes to retard the migration of contaminants will depend on the specific types of clays used. Interaction between the wastes and the clays could affect the permeability of the clays with respect to the various contaminants. The physical and chemical properties of clays are variable and have to be considered with respect to the specific chemical species that may be in the wastes and in the infiltrating water.

Detailed information on the specific types of clays that are in the "existing ground" or that would be in a specially constructed clay liner are not yet available. DOE will be gathering information such as the specific clay mineralogy, concentrations of organic contaminants in the wastes, and leachability of the stabilized sludges. These data will then be used to develop the detailed engineering design for the disposal cell. Additional clay will be transported to the raffinate pits area as needed and installed and compacted to appropriate permeability characteristics to ensure adequate confinement.

K.2.6 Mitigation of the Effects of Natural Forces on Disposal Cell Integrity

The following mitigative measures will be considered at the detailed design stage for the chosen alternative:

1. Adding gravel or rock to the cover, particularly on steeper slopes, to reduce the potential for gully erosion and wind erosion.

2. Using terracing or other measures to slow down and divert runoff water to reduce the rate of erosion and the possibility of gully erosion.

3. Designing the covers at the Weldon Spring site or "Nearby Site" so that the gently sloping parts will extend well beyond the outer edge of the internal dikes and/or placing the thickest part of the cover above the dikes to decrease the potential for erosion at the edge of the wastes.

4. Carefully selecting the type and thickness of topsoil and type of vegetation to minimize the potential loss of vegetation during prolonged droughts.

5. Placing riprap on the trench covers and/or on the ground surface between the trenches at the Hanford site to provide more protection against wind erosion.

6. Carefully evaluating the slope on top of the disposal cell for Alternative 2b at the Weldon Spring site to ensure that the potential for slippage on the lead sheet is minimized.

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APPENDIX L. GLOSSARY

ABSORBED DOSE -- The amount of energy absorbed in any material from incident radiation. Measured in rads, where 1 rad equals 100 ergs of energy absorbed in 1 gram of matter.

ACTION PERIOD -- For this EIS, the period (approximately 10 years) during which physical actions such as excavation, transportation, and stabilization will take place.

ACTIVITY -- A measure of the rate at which radioactive material is undergoing radioactive decay, usually given in terms of the number of nuclear disintegrations occurring in a given quantity of material over a unit of time. The unit of activity is the curie (Ci).

ALARA -- Acronym for "As Low As Reasonably Achievable." This refers to the DOE goal of keeping releases of radioactive substances to the environment and exposures of humans to radiation as far below regulatory limits as "reasonably achievable".

ALLUVIUM -- All material deposited permanently or in transit by streams.

ALPHA PARTICLE -- A particle emitted from the nucleus during radioactive decay of certain radionuclides. It consists of two protons and two neutrons bound together and is identical to the nucleus of a helium-4 atom.

ANION -- A negatively charged ion.

AQUIFER -- A water-bearing layer of permeable rock or soil that will yield water in usable quantities to wells. Confined aquifers are bounded on top and bottom by less-permeable materials. Unconfined aquifers are bounded on top by a water table.

ARTESIAN AQUIFER -- An aquifer that is confined so that its hydraulic head rises above the top of the aquifer unit; thus, an artesian water body is one that is confined under hydraulic pressure.

ATOM -- The basic component of all matter, i.e., the smallest unitary constituent of a chemical element having all the properties of that element. Atoms are made up of protons and neutrons (in the nucleus) and electrons (surrounding the nucleus).

ATOMIC MASS -- The mass of an atom relative to the mass of a neutral carbon-12 atom, on a scale in which the atomic mass of a carbon-12 atom is precisely 12.

ATOMIC NUMBER -- The number of protons in the nucleus of an atom.

BACKGROUND RADIATION -- In this EIS, includes both natural and man-made (e.g., fallout) radiation in the human environment. It includes cosmic rays and radiation from the naturally radioactive elements that occur both outside and inside the bodies of humans and animals. For persons living in the United States, the individual dose from background radiation ranges from about 80 to 200 millirem per year.

BEDROCK -- A solid rock formation usually underlying one or more other unconsolidated formations.

BERM -- A bench of soil or rock built on an earthen structure. It may serve various purposes such as a dike, an encasement for a drainage system, a weight for structural stabilization of an embankment, or an erosion-control structure.

BETA PARTICLE -- An electron emitted from the nucleus during radioactive decay. Beta particles are easily stopped by a thin sheet of metal or plastic. Large amounts of beta radiation may cause skin burns, and beta emitters are harmful if they enter the body.

BITUMEN -- An asphalt material (usually obtained from petroleum or coal-tar refining) that can be used in immobilization of radioactive wastes.

BREAKTHROUGH CURVE -- A graphical relationship between the tracer concentration in a liquid stream and the elapsed time since introduction of the tracer into that stream.

CATION -- A positively charged ion.

CEMENT -- Any mixture of fine-ground lime, alumina, and silica that will set to a hard product by mixing with water, which combines chemically with the other ingredients to form a hydrate.

CENTRIFUGATION -- A solids/liquids phase separation technique utilizing the force inherent in rotating bodies that impels material outward from the center.

CHERT -- A common quartz mineral that occurs as beds, nodules, lenses, or fragments within various types of bedrock; also referred to as flint.

CLAY -- A composition of particles of very fine grain size, usually very plastic.

COAGULATION -- Irreversible combination or aggregation of semisolid particles into clumps.

CONSERVATIVE ASSUMPTION -- An assumption that results in predicted values for which the resulting impacts are greater than would be expected to occur under actual conditions (e.g., choice of parameter values or type of model leading to higher predicted concentrations of contaminants and therefore greater impacts than would actually occur).

CONTAINMENT -- Confining the radioactive wastes within prescribed boundaries, e.g., within a waste package or a disposal cell.

CONTAMINATION -- The inclusion of foreign substances in or on the surfaces of soils, structures, areas, objects, or personnel.

CUMULATIVE RADIATION DOSE -- The total dose resulting from repeated radiation exposures of the same organ or the whole body over a period of time.

CURIE -- A measure of the rate of radioactive decay. One curie (Ci) is equal to 37 billion disintegrations per second (3.7×10^{10} dis/s), which is approximately equal to the decay of one gram of radium-226.

DECAY CHAIN (DECAY SERIES) -- The nuclides in the sequence of radioactive decay from one nuclide to another until a stable (nonradioactive) nuclide is reached.

DECAY PRODUCTS -- Isotopes that are formed by the radioactive decay of some other isotope. In the case of radium-226, for example, there are 10 successive decay products, ending in the stable isotope lead-206.

DECOMMISSIONING -- The removal of an installation from service and the reduction or stabilization of contamination.

DECONTAMINATION -- The selective removal of radioactive material from a surface or from within another material.

DEMOGRAPHY -- The study of human population, i.e., size, density, distribution, and vital statistics (e.g., age, sex, and ethnicity).

DEWATER -- Removal of water from a suspension or sludge.

DIKE -- A wall or mound built around a low-lying area to prevent flooding (see also BERM).

DISCHARGE -- In groundwater hydrology, the rate of flow (usually from a well) at a given instant in terms of volume per unit time.

DISPERSAL -- Act or result of scattering a material in the environment.

DISPERSION -- The continuous variation in the concentration of a pollutant or tracer as it moves through a medium. Refers to the rate of mixing and transport of the pollutant in the medium (e.g., atmosphere, groundwater).

DISTRIBUTION COEFFICIENT (Kd) -- Ratio of the concentration of a constituent sorbed on the solid matrix of soil or rock to the concentration of the dissolved constituent in water in the pores of the solid matrix.

DOLOMITE -- A rock composed principally of the mineral dolomite (calcium-magnesium carbonate). Dolomites are similar to limestones except for their magnesium content.

DOSE -- Total radiation delivered to a specific part of the body, or to the body as a whole.

DOSE COMMITMENT -- The dose that an organ or tissue would receive during a specified period of time (e.g., 50 or 100 years) as a result of intake (as by ingestion or inhalation) of one or more radionuclides from a given release.

DOSE EQUIVALENT -- A term used to express the amount of effective radiation when modifying factors have been considered. It is the product of absorbed dose (rads) multiplied by a quality factor and any other modifying factors. It is measured in rem (roentgen equivalent man).

DOSE RATE -- Radiation per unit time (i.e., rem per minute, rem per hour) as it is being delivered to the body.

DRAINAGE DIVIDE -- The rim of a drainage basin or watershed and the boundary between adjacent basins or watersheds.

EASEMENT -- A right held by an individual, company, or government body to use land owned by someone else; a right-of-way.

EFFLUENT -- Liquid, gaseous, or solid discharges into the environment generated by a process or procedure.

ELECTROMAGNETIC RADIATION -- A traveling wave motion resulting from changing electric or magnetic fields. Familiar electromagnetic radiations range from X-rays and gamma rays with short wavelength -- through the ultraviolet, visible, and infrared regions -- to radar and radio waves with relatively long wavelength.

ELECTRON -- An elementary particle with a unit negative charge and a mass 1/1837th that of the proton. Electrons surround the positively charged nucleus and determine the chemical properties of the atom.

ELEMENT -- Any one of the 103 known chemical substances that consist of atoms of only one kind.

EMULSION -- A stable mixture of two or more immiscible liquids.

EPHEMERAL STREAM -- A stream that contains water for only short periods of the year, usually following precipitation events; sometimes referred to as an intermittent or "wet weather" stream.

EPICENTER -- The point on the surface of the earth above the focus of an earthquake.

EROSION -- The process in which soil or rock materials are loosened and carried away by the action of wind or water.

EVAPORATION -- The change of a substance from the liquid phase to the gaseous or vapor phase.

EVAPOTRANSPIRATION -- The combined loss of water from soil by evaporation and by transpiration (loss through pores) from the surfaces of plant structures.

EXPOSURE, RADIATION -- The amount of ionization produced in air by X-rays or gamma rays, measured in roentgens (R). A person exposed for one hour to a one-roentgen-per-hour (1 R/h) radiation field of X-rays or gamma rays will incur a dose equivalent of about 1 rem.

EXTERNAL RADIATION -- Radiation from a source located outside the body.

FAULT -- A fracture or fracture zone along which there has been displacement of the sides relative to one another, parallel to the fracture.

FILTRATION -- The separation of suspended solids from a liquid or gas by forcing the mixture through a porous membrane.

FLOCCULATION -- A combination or aggregation of suspended colloidal particles in such a way that they form clumps.

FLOODPLAIN -- The portion of a river or stream valley that periodically is inundated. The 100-year floodplain is the area that is likely to be inundated once in 100 years.

FLY ASH -- A very fine ash produced mainly by the combustion of coal.

FRACTURE -- Breaks in rock formations due to structural stresses. Fractures may occur as faults, shears, joints, or planes of cleavage.

GAMMA RADIATION -- Penetrating high-energy, short-wavelength, electromagnetic radiation (similar to X-rays) emitted during radioactive decay. Gamma rays are very penetrating and require dense materials (such as lead or uranium) for shielding or to be stopped.

GENETIC EFFECTS OF RADIATION -- Effects of radiation on subsequent generations as a result of damage to the genetic material of the exposed individual.

GEOHYDROLOGY -- The study of the character, source, and mode of occurrence of underground water.

GEOMEMBRANE -- Impervious synthetic material used in waste-disposal facilities to minimize water infiltration and seepage.

GROUNDWATER -- Usually considered to be subsurface water within the zone of saturation.

GROUT -- Mortar combined with liquid to provide a matrix for sealing an area.

HALF-LIFE -- The time it takes for half the atoms of a quantity of a particular radioactive element to decay into another form. Half-lives of different isotopes vary from millionths of a second or less to billions of years.

HEAD -- The elevation to which water will rise at a given point as a result of reservoir pressure, the water-level elevation in a well, or the elevation to which the water of a flowing artesian well will rise in a pipe extended high enough to stop the flow.

HEALTH EFFECTS -- Effects of radiation on exposed individuals, specifically referred to in this EIS as the induction of fatal cancers and genetic defects (see also GENETIC EFFECTS OF RADIATION and SOMATIC EFFECTS OF RADIATION).

HEALTH PHYSICS -- The science concerned with recognition, evaluation, and control of health hazards from ionizing radiation.

HOT SPOT -- Specific location where the radiation dose rate is much higher than in the general surrounding area.

HYDRAULIC CONDUCTIVITY -- The rate of flow of water through a unit cross-sectional area under a unit hydraulic gradient at a specific temperature.

HYDRAULIC GRADIENT -- The change in head per unit of lateral distance in a given direction.

HYDROLOGIC -- Pertaining to study of the properties, distribution, and circulation of water on the surface of the land, in the soil and underlying rocks, and in the atmosphere.

IMMOBILIZATION -- Treatment and/or emplacement of the wastes so as to impede their movement.

IMPERMEABLE -- Having a texture that does not permit water to move through it under the head differences ordinarily found in subsurface water.

INDIVIDUAL DOSE -- The radiation dose received by an individual.

INFILTRATION RATE, SOIL -- The rate at which water enters the surface layer of soil.

IN-SITU -- In place.

INSTITUTIONAL CONTROL -- Management by any governmental or other organized body. Institutional controls may include recordkeeping, limitations on land ownership and use, maintenance and security activities, monitoring, or other enforced restrictions.

INTERMITTENT STREAM -- See EPHEMERAL STREAM.

INTERNAL RADIATION -- Radiation from radioactive substances within the body.

INTRUSION -- For this EIS, persons, plants, or animals breaking through the barriers that contain wastes in a disposal cell.

INVENTORY -- For this EIS, the total amount of a chemical or radioactive species contained in a waste volume.

ION -- An atom or molecule from which one or more electrons have been removed (positively charged ion) or to which an electron has become attached (negatively charged ion).

ION EXCHANGE -- Replacement of ions sorbed on a solid (such as a clay particle) or exposed at the surface of a solid by ions in solution (usually water). The phenomenon is known to occur when water moves through clays, zeolitic rocks, and other materials in the earth's crust.

ION-EXCHANGE MEDIUM -- Resin or zeolite material used in an ion-exchange process.

IONIZATION -- The creation of ions by the process of adding electrons to, or removing electrons from, atoms or molecules.

IONIZING RADIATION -- Any radiation capable of displacing electrons from atoms or molecules, thereby producing ions. Examples include alpha particles, beta particles, gamma rays, X-rays, protons, and neutrons.

ISOTOPE -- Nuclides having the same atomic number but different mass numbers.

JOINT -- A fracture or crack in bedrock, usually more or less vertical to the bedding, along which no appreciable movement has occurred. Joints may be enlarged by groundwater dissolution of the surrounding rock.

KARST -- A type of topography formed over limestone, dolomite, or gypsum by dissolution, and characterized by closed depressions or sinkholes, caves, and underground drainage.

LEACH -- To remove or separate soluble components from a solid by contact with water or other liquids.

LEACHATE -- The water and dissolved constituents leached from a given material.

LIMESTONE -- A rock composed of calcium carbonate and formed by either organic or inorganic sedimentary processes.

LOESS -- A soil material, relatively uniform in texture and appearance, that is transported and deposited by wind. It consists predominantly of silt-sized particles of sand and clay, and often stands in stable, vertical bluffs.

LONG-TERM -- For this EIS, any time after completion of the action period (10 years). Cumulative impacts over 1,000 years are assessed in this EIS.

LOSING STREAM -- A stream or river that loses its flow due to the permeability of the bedrock or soil over which it flows.

LOW-SPECIFIC-ACTIVITY (LSA) RADIOACTIVE MATERIALS -- Radioactive materials that present a relatively low hazard because of their low concentration of radioactive substances. Examples of LSA radioactive materials include uranium or thorium ores and physical or chemical concentrates of these ores, unirradiated natural or depleted uranium, and unirradiated natural thorium. Regulations governing the transportation of LSA radioactive materials are given in 49 CFR Part 173.425.

MAINTENANCE/MONITORING -- Under institutional control, monitoring the environment of the disposal area for the release of radioactivity; and maintaining the condition of the waste cover, drainage systems, surfaces, and vegetative cover.

MANAGEMENT, RADIOACTIVE WASTE -- The act of managing or controlling radioactive waste. Because the radionuclides in the Weldon Spring wastes have long half-lives and the potential hazard will not diminish appreciably for thousands of years, there will be a continuing need for management of these wastes.

MASS NUMBER -- The number of protons plus neutrons in the atom. For example, uranium-238 has a mass number of 238, i.e., 92 protons plus 146 neutrons.

MAXIMALLY EXPOSED INDIVIDUAL -- A hypothetical member of the general public whose location and activities result in his/her receiving the largest dose that any member of the general public would potentially incur under a given set of conditions.

MODIFIED MERCALLI SCALE -- An arbitrary scale designed to measure the ground-shaking intensity of an earthquake. The more familiar Richter Scale measures the amount of energy released by an earthquake.

Scale		Description
I	Instrumental	Detected only by seismographs
II	Feeble	Noticed only by sensitive people
III	Slight	Resembling vibrations caused by heavy traffic
IV	Moderate	Felt by people walking; causes rocking of freestanding objects
V	Rather strong	Sleepers awakened; bells ring; widely felt
VI	Strong	Trees sway; some damage from overturning and falling objects
VII	Very strong	General alarm; cracking of walls; etc.
VIII	Destructive	Chimneys fall; some damage to buildings
IX	Ruinous	Ground begins to crack; houses collapse and pipes break
X	Disastrous	Ground badly cracked; many buildings destroyed; some landslides
XI	Very disastrous	Few buildings stand; bridges, railways destroyed; water, gas, electricity, telephones, etc., out of action
XII	Catastrophic	Total destruction; objects thrown in air; much heaving, shaking, and distortion of surface

MOLECULE -- A group of atoms held together by chemical forces. A molecule is the smallest unit of a compound that can exist by itself and retain all its chemical properties.

NEUTRON -- Elementary atomic particle with a mass slightly greater than that of the proton but with no electric charge. It is found in the nucleus of every atom heavier than ordinary (light) hydrogen.

NUCLEAR RADIATION -- Particles and electromagnetic energy given off due to disintegrations occurring in the nucleus of an atom.

NUCLEUS -- The small, central, positively charged region of an atom that carries essentially all the mass. Except for the nucleus of ordinary (light) hydrogen, which has a single proton, all atomic nuclei contain both protons and neutrons. The number of protons determines the total positive charge, or atomic number; this is the same for all the atomic nuclei of a given chemical element. The total number of neutrons and protons is called the mass number.

NUCLIDE -- A general term referring to all known isotopes, both stable (279) and unstable (about 500), of the chemical elements.

OCCUPATIONAL DOSE -- Amount of radiation received by those occupied with the operation of an activity involving the handling of radioactive material.

OLD-FIELD SUCCESSION -- The progressive changes in vegetation and animal species structure and community processes that follow the abandonment of cropland or pasture.

ORGAN DOSE -- The radiation dose to a specific organ.

ORGANIC CHEMICAL -- Carbon compounds, especially those in which hydrogen is attached to carbon, whether derived from living organisms or not.

OVERBURDEN -- All material (loose soil, sand, gravel, etc.) that lies above bedrock.

PARTICULATES -- Fine solid or liquid particles dispersed in air or water.

PENETRATING RADIATION -- Forms of radiation that are capable of passing through significant thicknesses of solid materials, e.g., gamma rays, X-rays, and neutrons.

PERCOLATION -- Downward movement of water through openings in soil or rock.

PERMEABILITY -- The relative ease with which a porous medium can transmit a liquid under a hydraulic gradient. In hydrology, the capacity of rock, soil, or sediment for allowing the passage of water.

PERSON-REM -- The average dose per person multiplied by the number of persons exposed. For example, a thousand people each exposed to one millirem (1/1000 rem) would have a collective dose of 1 person-rem.

pH -- A measure of the relative acidity or alkalinity of a solution; a neutral solution has a pH of 7, acids have a pH below 7, and bases have a pH above 7.

PHOTON -- A quantum (or packet) of energy emitted in the form of electromagnetic radiation. Gamma rays and X-rays are examples of photons.

PIEZOMETRIC SURFACE -- The surface defined by the levels to which groundwater will rise in wells that tap an aquifer.

PINNACLE -- An upward projection of bedrock, usually cone-shaped, that forms on a dissected or corroded bedrock surface.

PLANT COMMUNITY -- Any assemblage of plant populations living in a prescribed area or physical habitat. An organized unit having characteristics additional to its individual and population components.

POLYMER -- A large molecule formed by the repetitive combination of many copies of smaller or simpler molecules.

POPULATION DOSE -- Summation of the doses received by all individuals in a specified population.

POROSITY -- That property of a rock or soil that enables the rock or soil to contain water in voids or interstices, usually expressed as a percentage (the void volume divided by the total volume).

PRECIPITATION -- In solution chemistry, solids separating out of solution and usually settling by gravity; otherwise rain, snow, etc.

PRIORITY POLLUTANT -- Any one of the toxic pollutants defined for priority regulation by the U.S. Environmental Protection Agency under the Clean Water Act.

PROTON -- An elementary nuclear particle with a positive electric charge located in the nucleus of an atom.

QUALITY FACTOR -- A factor used to indicate the relative biological significance of various forms of radiation and to calculate the dose equivalent (in rem) from the absorbed dose (in rads). For example, the quality factors are 1 and 20 for gamma rays and alpha particles, respectively. This means that one unit of absorbed dose from alpha radiation presents the same biological hazard as 20 units of absorbed dose from gamma rays.

RAD -- Unit of absorbed dose; acronym for radiation absorbed dose (see ABSORBED DOSE).

RADIATION -- A very general term that covers many forms of particles and energy, from sunlight and radio waves to the energy that is released from inside an atom. Radiation can be in the form of electromagnetic waves (gamma rays, X-rays) or particles (alpha particles, beta particles, protons, neutrons).

RADIOISOTOPE -- An unstable isotope of an element that spontaneously loses particles and energy through radioactive decay.

RADIOLOGICAL CONVERSION FACTORS --

	SI Units	Conventional Units
Dose	Gray (1 Gy = 1 J/kg = 100 rad)	rad
Dose equivalent	Sievert (1 Sv = 100 rem)	rem
Activity	Becquerel (1 Bq = 1 dis/s = 2.703×10^{-11} Ci)	Ci (Curie)

RADIONUCLIOE -- An unstable nuclide that undergoes radioactive decay.

RAFFINATE -- A waste product from a refining process, i.e., that portion of a treated liquid mixture that is not dissolved and not removed by a selective solvent.

RECHARGE -- In hydrology, the addition of water to the zone of saturation; infiltration of precipitation is a form of natural recharge.

RECLAMATION -- The restoration of disturbed land and/or structures to allow for future productive use.

REGENERATION -- Restoration of a material to its original condition after it has undergone physical or chemical modification, as in regenerating a filter or resin by backflushing with water and/or various chemicals.

REGULATION -- A law promulgated by an administrative agency or regulatory commission. Federal agencies and commissions obtain their power to promulgate laws from the U.S. Congress; state agencies and commissions obtain such power from their respective state legislatures.

REM (ROENTGEN EQUIVALENT MAN) -- A quantity used in radiation protection to express the effective dose equivalent for all forms of ionizing radiation. It is the product of the adsorbed dose in rads and factors related to relative biological effectiveness (see also DOSE EQUIVALENT).

REMEDIAL ACTION -- Activities conducted to reduce potential radiation exposure to humans and potential harm to the environment from radioactive and/or chemical contamination in the environment.

RESIN -- Solid or semisolid product of synthetic origin used in ion-exchange processes for purification of liquids.

RETARDATION FACTOR (Rd) -- A factor that accounts for the holdup of specific contaminants in soil relative to groundwater flow.

RIPRAP -- An assemblage of broken stones often used to protect against erosion.

RISK -- A measure of the hazard associated with the occurrence of an event, taking into account the severity of the event and the probability of its occurrence.

ROENTGEN (R) -- Unit of exposure. One roentgen is the amount of gamma rays or X-rays required to produce one electrostatic unit (esu) of charge of one sign (either positive or negative) in one cubic centimeter of dry air under standard conditions.

RUNOFF -- All precipitation that is not retained in impoundments, that does not soak into the ground, does not evaporate immediately, or is not used by vegetation, and hence flows over the land surface.

SATURATED ZONE -- The subsurface zone in which all interconnecting voids or pores are filled with water.

SCOPING -- The process of determining the actions, alternatives, and impacts to be considered in an EIS.

SECULAR EQUILIBRIUM -- In a radioactive decay series, the state that prevails when the activities of successive members of the series are equal and all remain constant over time.

SEEPAGE -- Any water or liquid effluent that flows through a porous medium (e.g., water lost through the bottom of a containment area).

SEISMIC -- Having to do with the geology of earthquakes and extending to prediction of earthquake frequency and severity.

SHALE -- A thinly laminated rock type composed predominantly of clay and silt particles.

SHIELDING -- A material interposed between a source of radiation and humans for protection against the danger of radiation. Commonly used shielding materials are soils, concrete, water, and lead.

SILT -- A rock fragment, mineral, or detrital particle in soil having a diameter of 0.004 to 0.06 mm (i.e., smaller than fine sand and larger than clay).

SINKHOLE -- A funnel-shaped depression in the land surface, generally circular or subcircular in outline. Sinkholes originate in a number of ways, all related to the underground dissolution of bedrock, especially limestone.

SLOPE -- A land surface deviation from the level horizontal plane. It is measured in percentage, i.e., units vertical drop per 100 horizontal units; for example, a slope of 15% has 15 ft of vertical drop for each 100 ft of horizontal distance.

SLUDGE -- For this EIS, the raffinates currently being stored in the four pits at the Weldon Spring raffinate pits area and the smaller amount being stored in the quarry area.

SLURRY -- A thin watery mixture of a fine insoluble material such as clay, cement, or soil.

SOIL -- Unconsolidated material, several feet thick, formed by environmental factors acting on geologic materials over time and conditioned by relief, to produce a sequence of layers or horizons that occupy predictable and mappable parts of the landscape. Includes all loose or unconsolidated material overlying bedrock, regardless of the origin or thickness of the material.

SOLUTION CAVITIES/CHANNELS -- Cavities or channels formed by water in carbonate rocks (such as limestone and dolomite) caused by the slow dissolving of the rock along fractures, joints, etc.

SOMATIC EFFECTS OF RADIATION -- Effects of radiation that are limited to the exposed individual, as distinguished from genetic effects that may also affect subsequent unexposed generations.

SORPTION -- A general term used to encompass the processes of absorption, adsorption, ion exchange, ion retardation, chemisorption, and dialysis.

SORPTIVE CAPACITY -- The measure of a material's ability to sorb specific constituents from a liquid as it passes through the material.

SOURCE TERMS -- The quantity of radioactive material (or other pollutant) released to the environment per unit time at its point of release (source).

SPECIFIC ACTIVITY -- The activity per unit volume or mass of a substance (see ACTIVITY).

SPRING -- An issue of water from the earth, flowing away as a small stream or standing as a pool or small lake, or the place of such an issue.

STABILITY CATEGORIES, ATMOSPHERIC -- Classification of atmospheric dispersion into six Pasquill categories, A through F. Stability Class A represents the most unstable atmospheric conditions with the greatest mixing (often occurring in the afternoon). Stability Class F represents the case of least mixing, often occurring during nighttime or early morning.

STABILIZATION -- For this EIS, conversion of the sludge to a form that has greater physical stability than the original sludge.

STORAGE COEFFICIENT -- The amount of water an aquifer releases from storage per unit surface area of the aquifer per unit change in head.

STORATIVITY -- See STORAGE COEFFICIENT.

STRATUM -- Bed or layer, regardless of thickness, that consists of approximately the same kind of rock material.

SUBSIDENCE -- Gradual or sudden sinking of the ground surface below natural grade level due to slow decay and compression of material or collapse of a large void space.

SURFICIAL MATERIAL -- Unconsolidated and residual, alluvial, or glacial deposits overlying bedrock or occurring on or near the earth's surface; corresponds with the engineering use of the term "soil" and includes that portion.

TECTONIC -- Of, pertaining to, or designating the process causing, and the rock structures resulting from, deformation of the earth's crust.

TIERING -- A method (see 40 CFR Part 1508.28) for preparing a network of environmental documents branching off from a generic, broad EIS to optimize use of support documentation.

TILL -- Unstratified glacial deposits consisting of clay, sand, gravel, and boulders intermingled.

TOPOGRAPHY -- The shape of the earth, including the size and shape of hills, valleys, and other physical features.

TRACE ELEMENTS -- Chemical elements that normally occur in minute (trace) quantities. Includes elements such as chromium, zinc, cadmium, copper, selenium, boron, and arsenic.

TRANSMISSIVITY -- The rate at which water is transmitted through a unit width of aquifer under a unit gradient (1 unit vertically for each 1 unit laterally). Mathematically, it is the product of hydraulic conductivity and aquifer thickness.

TRENCH, SHALLOW-LAND BURIAL -- A long, narrow excavation with unsupported walls, into which solid radioactive wastes are emplaced and covered with excavated earth.

UNSATURATED ZONE -- See VADOSE ZONE.

URANIUM (NATURAL) -- A naturally occurring radioactive element that consists of 99.2830% by weight uranium-238, 0.7110% uranium-235, and 0.0054% uranium-234. The activity ratio of uranium-238:uranium-234:uranium-235 in natural uranium is 1:1:0.046.

VADOSE ZONE -- The unsaturated region of soil between the ground surface and the water table.

VICINITY PROPERTIES -- For this EIS, areas in the vicinity of the raffinate pits, chemical plant, and quarry areas -- but outside of the current boundaries -- that are radioactively contaminated above current criteria as a result of previous activities.

VITRIFICATION -- Conversion, by heat and fusion, of materials into glass or glassy substances.

WATERSHED -- An area of land that drains into a water body. Watersheds are separated by drainage divides.

WATER TABLE -- The upper surface of the zone of water saturation at which the pressure is equal to atmospheric pressure, i.e., the upper surface of an unconfined aquifer.

WELL YIELD -- The rate at which a well yields water.

WHOLE-BODY DOSE -- The radiation dose to the entire body.

WORKING LEVEL (WL) -- Any combination of radon-222 decay products in 1 liter of air that will result in the ultimate emission of 0.21 erg of alpha energy is defined as 1 WL. It is based on the 0.21 erg of alpha energy that would be emitted by the decay products of 100 pCi of radon-222 in 1 liter of air, where the decay products are in radioactive equilibrium with the parent.

X-RAYS -- Penetrating electromagnetic radiation having a wavelength that is much shorter than that of visible light. It is customary to refer to rays originating in the nucleus as gamma rays and to those originating in the electron field of the atom as X-rays.

ABBREVIATIONS

AEC	Atomic Energy Commission
ALARA	As Low As Reasonably Achievable
ANL	Argonne National Laboratory
BEAR	Biological Effects of Atomic Radiation
BEIR	Biological Effects of Ionizing Radiation
BNI	Bechtel National, Inc.
CEQ	Council on Environmental Quality
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
DNR	Department of Natural Resources (Missouri)
DNT	Dinitrotoluene
DOE	U.S. Department of Energy
DOT	U.S. Department of Transportation
EIS	Environmental Impact Statement
EPA	U.S. Environmental Protection Agency
FUSRAP	Formerly Utilized Sites Remedial Action Program
ICC	Interstate Commerce Commission
ICRP	International Commission on Radiological Protection
LSA	Low-specific-activity
MED	Manhattan Engineer District
MKT	Missouri-Kansas-Texas (Railroad)
MM	Modified Mercalli
MSL	(Above) Mean Sea Level
NAAQS	National Ambient Air Quality Standards
NCRP	National Council on Radiation Protection and Measurements
NEPA	National Environmental Policy Act of 1969
NOI	Notice of Intent
NRC	U.S. Nuclear Regulatory Commission
PCB	Polychlorinated biphenyl
RCRA	Resource Conservation and Recovery Act
SCCAHW	St. Charles Countians Against Hazardous Wastes
SFMP	Surplus Facilities Management Program
SMSA	Standard Metropolitan Statistical Area
TDS	Total dissolved solids
TNT	Trinitrotoluene
TOC	Total organic carbon
TSP	Total suspended particulates
UDAD	Uranium Dispersion and Dosimetry (Computer Code)
UNSCEAR	United Nations Scientific Committee on the Effects of Atomic Radiation
USLE	Universal Soil Loss Equation
WEE	Wind Erosion Equation

SYMBOLS

bbl	Barrels
Btu	British thermal units
cfs	Cubic feet per second
Ci	Curies
cm	Centimeters
cm^2	Square centimeters
cm^3	Cubic centimeters
dB	Decibel
dis/s	Disintegrations per second
ft	Feet (foot)
ft^2	Square feet
ft^3	Cubic feet
g	Grams
gal	Gallons
gpd	Gallons per day
gpm	Gallons per minute
h	Hours
ha	Hectares
in.	Inches
$in.^2$	Square inches
$in.^3$	Cubic inches
Kd	Distribution coefficient
kg	Kilograms
km	Kilometers
km^2	Square kilometers
K_{oc}	Partition coefficient for organic carbon
kW	Kilowatts
kWh	Kilowatt-hours
L	Liters
lb	Pounds
m	Meters
m^2	Square meters
m^3	Cubic meters
mg	Milligrams
mi	Miles
mi^2	Square miles
mL	Milliliters
mm	Millimeters
mph	Miles per hour
mrem	Milliroentgen-equivalent man
MT	Metric tons
MWh	Megawatt-hours
pCi	Picocuries
pH	Common logarithm of the reciprocal of the hydrogen ion concentration
ppb	Parts per billion
ppm	Parts per million
R	Roentgens
Rd	Retardation factor
rem	Roentgen-equivalent man
s	Seconds
t	Short tons
V:H	Ratio of vertical distance to horizontal distance
WL	Working level
wt	Weight
yd	Yards
yd^2	Square yards
yd^3	Cubic yards
yr	Years
α	Alpha
β	Beta
γ	Gamma
µg	Micrograms
µm	Micrometers (microns)
µR	Microroentgens
°C	Degrees Celsius
°F	Degrees Fahrenheit

APPENDIX M. ENGLISH/METRIC - METRIC/ENGLISH EQUIVALENTS

Multiply	By	To obtain
Acres	0.4047	Hectares (ha)
Cubic feet (ft^3)	0.02832	Cubic meters (m^3)
Cubic yards (yd^3)	0.7646	Cubic meters (m^3)
Degrees Fahrenheit (°F) - 32	0.5555	Degrees Celsius (°C)
Feet (ft)	0.3048	Meters (m)
Gallons (gal)	3.785	Liters (L)
Gallons (gal)	0.003785	Cubic meters (m^3)
Inches (in.)	2.540	Centimeters (cm)
Miles (mi)	1.609	Kilometers (km)
Pounds (lb)	0.4536	Kilograms (kg)
Square feet (ft^2)	0.09290	Square meters (m^2)
Square yards (yd^2)	0.8361	Square meters (m^2)
Square miles (mi^2)	2.590	Square kilometers (km^2)
Tons, short (t)	907.2	Kilograms (kg)
Tons, short (t)	0.9072	Tons, metric (MT)

Multiply	By	To obtain
Centimeters (cm)	0.3937	Inches (in.)
Cubic meters (m^3)	35.31	Cubic feet (ft^3)
Cubic meters (m^3)	1.308	Cubic yards (yd^3)
Cubic meters (m^3)	264.2	Gallons (gal)
Degrees Celsius (°C) + 17.78	1.8	Degrees Fahrenheit (°F)
Hectares (ha)	2.471	Acres
Kilograms (kg)	2.205	Pounds (lb)
Kilograms (kg)	0.001102	Tons, short (t)
Kilometers (km)	0.6214	Miles (mi)
Liters (L)	0.2642	Gallons (gal)
Meters (m)	3.281	Feet (ft)
Square kilometers (km^2)	0.3861	Square miles (mi^2)
Square meters (m^2)	10.76	Square feet (ft^2)
Square meters (m^2)	1.196	Square yards (yd^2)
Tons, metric (MT)	1.102	Tons, short (t)