FOSSIL ENERGY

DOE/BC/W-31-109-ENG-38-8
(OSTI ID: 13061)

AN ASSESSMENT OF THE DISPOSAL OF PETROLEUM INDUSTRY NORM IN NONHAZARDOUS LANDFILLS

Final Report
September 1999

By
Karen P. Smith
Deborah L. Blunt
Gustavious P. Williams
John J. Arnish
Manjula Pfingston
Jennifer Herbert
Rebecca A. Haffenden

Date Published: October 1999

Work Performed Under Contract No. W-31-109-ENG-38

Argonne National Laboratory
Environmental Assessment Division
Lakewood, Colorado

National Petroleum Technology Office
U. S. DEPARTMENT OF ENERGY
Tulsa, Oklahoma

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Karen P. Smith, Deborah L. Blunt, Gustavious P. Williams,
John J. Arnish, Manjula Pfingston, Jennifer Herbert, and Rebecca A. Haffenden
Argonne National Laboratory
Environmental Assessment Division

prepared for
U.S. Department of Energy
National Petroleum Technology Office
Tulsa, Oklahoma
under Contract W-31-109-Eng-38

and

American Petroleum Institute
Washington, D.C.

September 1999

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ACKNOWLEDGMENTS

The research for this study was funded jointly by the U.S. Department of Energy, National Petroleum Technology Office and the American Petroleum Institute (API). Direct oversight of the project was provided by John Ford, National Petroleum Technology Office, Tulsa, Oklahoma; Glenda Smith, API, Washington, D.C.; and Lewis M. Cook, Chevron Research and Technology Company, Richmond, California (current chair of the API's NORM Issues Group).

A number of other individuals provided valuable support to the project by serving as sources of information and reviewers. In addition to the individuals cited in the References section of this report, the authors wish to acknowledge the assistance of Kevin J. Grice, Texaco, North American Production, Houston, Texas; David Minnaar, Michigan Department of Environmental Quality, Radiological Protection Section, Lansing, Michigan; Andy Radford, API, Washington, D.C.; Bob Skowronek, Michigan Department of Environmental Quality, Radiological Protection Section, Lansing, Michigan; and Thomas Wodtke, Waste Management, Inc., Cincinnati, Ohio. In addition, the authors would like to thank Halil Avci, S.Y. Chen, and David Tomasko, Argonne National Laboratory, Argonne, Illinois, and John A. Veil, Argonne National Laboratory, Washington, D.C., for their valuable input and critical reviews.

The authors apologize if anyone has been omitted from these acknowledgments and are grateful to all who supported this research from inception through completion.

NOTATION

The following is a list of acronyms, abbreviations, and initialisms (including units of measure) used in this document.

ACRONYMS, ABBREVIATIONS, AND INITIALISMS

ANL	Argonne National Laboratory
API	American Petroleum Institute
CCR	*Colorado Code of Regulations*
CDPHE	Colorado Department of Public Health and Environment
CEDE	committed effective dose equivalent
CESQG	conditionally exempt small quantity generator
CFR	*Code of Federal Regulations*
COGCC	Colorado Oil and Gas Conservation Commission
DCF	dose conversion factor
DEQ	Department of Environmental Quality
DOE	U.S. Department of Energy
E&P	exploration and production
EDE	effective dose equivalent
EIB	Environmental Improvement Board
EPA	U.S. Environmental Protection Agency
HDPE	high-density polyethylene
HELP	Hydrologic Evaluation of Landfill Performance
ICRP	International Commission on Radiological Protection
KDHE	Kansas Department of Health and Environment
LAC	*Louisiana Administrative Code*
LDEQ	Louisiana Department of Environmental Quality
LDNR	Louisiana Department of Natural Resources
LLRW	low-level radioactive waste
MDEQ	Michigan Department of Environmental Quality
MDH	Mississippi Department of Health
NMAC	*New Mexico Administrative Code*
NORM	naturally occurring radioactive material
NOW	nonhazardous oilfield waste
OAC	*Oklahoma Administrative Code*
OCC	Oklahoma Corporation Commission
OCD	Oil Conservation Division
PCB	polychlorinated biphenyl
RCRA	Resource Conservation and Recovery Act
RRCT	Railroad Commission of Texas
TAC	*Texas Administrative Code*
TDH	Texas Department of Health

TNRCC	Texas Natural Resource Conservation Commission
TSD	treatment, storage, and disposal
WCS	Waste Control Specialists, LLC

RADIONUCLIDES

Pb-206	lead-206
Pb-210	lead-210
Ra-226	radium-226
Ra-228	radium-228
Rn-222	radon-222
Th-232	thorium-232
Th-228	thorium-228
U-238	uranium-238

UNITS OF MEASURE

cm	centimeter(s)
d	day(s)
ft	foot (feet)
g	gram(s)
gal	gallon(s)
h	hour(s)
in.	inch(es)
kg	kilogram(s)
L	liter(s)
m	meter(s)
μR	microroentgen
μm	micrometer(s)
mi	mile(s)
mCi	millicurie(s)
mg	milligram(s)
mL	milliliter(s)
mm	millimeter(s)
mrem	millirem
min	minute(s)
nCi	nanocurie(s)
pCi	picocurie(s)
ppm	part(s) per million
rem	roentgen equivalent man
s	second(s)
yd	yard(s)
yr	year(s)

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M. Pfingston, J. Herbert, and R.A. Haffenden

SUMMARY

S.1 BACKGROUND

In the past few years, many states have established specific regulations for the management of petroleum industry wastes containing naturally occurring radioactive material (NORM) above specified thresholds. These regulations have limited the number of available disposal options for NORM-containing wastes, thereby increasing the related waste management costs. In view of the increasing economic burden associated with NORM, the industry and its regulators are interested in identifying cost-effective disposal alternatives that still provide adequate protection of human health and the environment. One such alternative being considered is the disposal of NORM-containing wastes in landfills permitted to accept only nonhazardous wastes.

The Michigan Department of Environmental Quality (MDEQ) has issued guidelines allowing the disposal of materials contaminated with radium-226 (Ra-226) in landfills that are designed and permitted to receive nonhazardous municipal wastes. These guidelines are applicable to radium-bearing NORM wastes generated by the petroleum industry. Other states that have developed NORM regulations or guidelines, however, do not allow this type of disposal.

In this study, the disposal of radium-bearing NORM wastes in nonhazardous landfills in accordance with the MDEQ guidelines was modeled to evaluate potential radiological doses and resultant health risks to workers and the general public. In addition, the study included an evaluation of the potential doses and health risks associated with disposing of a separate NORM waste stream generated by the petroleum industry — wastes containing lead-210 (Pb-210) and its progeny. Both NORM waste streams are characterized in Section 3 of this report.

Section 5 provides a detailed discussion of the assessment methodologies, including descriptions of the scenarios, exposure pathways, source concentrations, and exposure assumptions. For both types of NORM wastes, a variety of scenarios were considered to evaluate the potential effects associated with the operational phase (i.e., during landfill operations) and future land use. Doses were calculated for the maximally exposed receptor for each scenario. For the radium-bearing wastes, the base-case analyses assumed that the disposal action involved 2,000 m^3 of waste containing an average Ra-226 concentration of 50 picocuries

per gram (pCi/g). For the lead-bearing wastes, it was assumed that the disposal action involved 20 m^3 of wastes containing an average Pb-210 concentration of 260 pCi/g.

For the operational phase worker, the primary exposure pathway evaluated in this study was external irradiation. A second pathway — inhalation of contaminated particulates — also was considered for the worker involved in placing the wastes in the landfill when the wastes were not containerized. For the general public living next to or in the vicinity of the landfill (i.e., within a 50-mi radius) during the disposal action, the primary exposure pathway was determined to be inhalation of contaminated particulates; for completeness, the external irradiation, ingestion of contaminated particulates, and ingestion of contaminated foodstuff pathways also were evaluated.

A variety of future land use scenarios — including on-site residential, industrial, and recreational and off-site residential scenarios — were considered. For all of the on-site scenarios, the primary exposure pathways were assumed to be external irradiation and inhalation of indoor and outdoor radon-222. Depending on the scenario, other less likely pathways (e.g., inhalation of contaminated particulates, inadvertent ingestion of contaminated soil, and ingestion of foodstuffs grown on the property) also were considered. For the off-site residential scenario, the only exposure pathways evaluated were ingestion of contaminated groundwater and inhalation of radon.

The study also included reviews of (1) the regulatory constraints applicable to the disposal of NORM in nonhazardous landfills in several major oil and gas producing states (Section 2) and (2) the typical costs associated with disposing of NORM, covering disposal options currently permitted by most state regulations as well as the nonhazardous landfill option (Section 4).

S.2 CONCLUSIONS

Regulatory constraints are reviewed in Section 2 of this report. It was found that the disposal of NORM-impacted wastes in nonhazardous municipal landfills is not explicitly allowed in any states except Michigan. In a few of the states reviewed in this study, NORM wastes may be allowed in other types of nonhazardous landfills, or in municipal landfills by special approval only. In other states, there seems to be less latitude both in the state regulations and on the part of individual regulators.

The NORM disposal cost study is discussed in Section 4 of this report. This study concluded that the disposal of regulated NORM wastes in nonhazardous landfills could be one of the most cost-effective disposal options available to the petroleum industry if approved on a widespread basis. However, because disposal costs depend on a number of factors (e.g., volume, radium content, requirements for waste analyses, competition for market share), they are quite variable. As a result, it is difficult to single out the least expensive disposal option for the petroleum industry; this determination must be made on a case-by-case basis. Neverless, one could conclude that an increase in the number of available disposal options would most likely reduce NORM disposal costs for the industry.

The results of the radiological dose and risk assessments are presented in Section 6 of this report. On the basis of these results, the following conclusions can be drawn regarding the disposal of 2,000 m^3 of radium-bearing NORM containing an average Ra-226 concentration of 50 pCi/g:

- Potential radiological doses and resultant health risks for workers actively involved in landfill operations would be negligible.
- Potential doses to an individual living adjacent to the landfill during the NORM disposal action and to the general population living within a 50-mi radius would be negligible.
- Potential doses to future industrial and recreational users of the landfill property would be negligible.
- Potential doses to hypothetical future residential users of the landfill property are most sensitive to depth of the NORM waste layer and integrity of the landfill cap. These doses would be negligible on the basis of the assumption that (1) the NORM wastes would be placed at a depth greater than approximately 10 ft below the cap and (2) the landfill cap would not be breached during construction of the home.
- Provided the NORM wastes are placed deeper than approximately 10 ft below the landfill cap, the Michigan policy allowing wastes containing up to 50 pCi/g to be disposed of in Type II landfills is protective of human health.
- Increasing the total volume would increase the worker doses linearly and could increase the potential doses to the off-site resident via the groundwater pathway. However, it is estimated that doses for these receptors would be negligible, and increasing the volume probably would not change this overall conclusion. Radiological doses to the future-use receptors would not be affected by increasing the total volume; doses to these receptors are primarily affected by changes in the location of the NORM waste within the landfill.

Regarding the disposal of lead-bearing NORM wastes, the results of this assessment indicate that the risk to workers or to the general public associated with the disposal of 20 m^3 of wastes containing an average Pb-210 concentration of 260 pCi/g would be negligible. Increasing the disposal volume would not significantly change this overall conclusion. Worker doses would increase linearly with volume, but doses to future users of the property would still be zero because once the waste is buried, a complete exposure pathway to a future receptor does not exist.

S.3 RECOMMENDATIONS

On the basis of the conclusions presented above, the following recommendations are suggested:

- It may be feasible for other states besides Michigan to consider issuing regulations allowing the disposal of NORM wastes containing up to 50 pCi/g of Ra-226 in municipal, nonhazardous landfills. In approving of this type of disposal, regulators should consider the total volume of radium-bearing wastes that are disposed of in a single landfill and cell, as well as the depth of the NORM waste layer within the landfill. Property records denoting that a landfill was in operation at that location should also note that radium-bearing wastes were disposed of therein.
- Regulators should consider allowing the disposal of NORM wastes containing radium in concentrations greater than 50 pCi/g on a case-by-case basis.
- States should also consider regulations governing the disposal of wastes containing Pb-210 in municipal, nonhazardous landfills. As they should for radium-bearing wastes, the regulations should consider the allowable concentrations of Pb-210 and the total volume that can be disposed of in a single landfill.
- States may want to consider allowing NORM wastes to be disposed of in other categories of nonhazardous landfills, provided the requirements for deed restrictions and protection of the landfill cap are equivalent to those for municipal landfills.

1 INTRODUCTION

1.1 BACKGROUND

Oil and gas production and processing operations sometimes cause naturally occurring radioactive materials (NORM) to accumulate at elevated concentrations in by-product waste streams. The sources of most of the radioactivity are isotopes of uranium-238 (U-238) and thorium-232 (Th-232), which are naturally present in the subsurface formations from which oil and gas are produced. NORM generated by the petroleum industry may be divided into two general categories: (1) wastes containing radium isotopes and their progeny and (2) wastes containing only lead-210 (Pb-210) and its progeny.

For the radium-bearing wastes, the primary radionuclide of concern is radium-226 (Ra-226), of the U-238 decay series. Radium-228 (Ra-228), of the Th-232 decay series, also occurs in these NORM wastes but is usually present in lower concentrations. Other radionuclides of concern include those that form from the decay of Ra-226 and Ra-228; these decay progeny are shown in Figures 1 and 2, which depict the decay chains of U-238 and Th-232, respectively. The production waste streams most likely to be characterized by elevated radium concentrations include produced water (i.e., the water produced along with the hydrocarbons), scale, and sludge. Radium, which is slightly soluble, can be mobilized in the liquid phases of a subsurface formation and transported to the surface in the produced water stream. Dissolved radium either remains in solution in the produced water or, if the conditions are right, precipitates out in scales or sludges.

A separate category of NORM wastes exists. This category includes wastes that do not contain any radium but do contain Pb-210, which is a decay product of Ra-226, and its progeny (Figure 1). Typically, these wastes accumulate inside gas processing equipment from the decay of radon-222 (Rn-222). The Pb-210 may be present in elemental form, as a chemical precipitate, or as an integrated constituent of the equipment metal.

Many states have established specific regulatory programs that define what materials must be managed as regulated NORM. One effect of the state-level NORM regulations has been increased management and disposal costs for NORM wastes. Disposal options currently allowed under most state NORM programs include (1) burial at a licensed NORM or low-level radioactive waste (LLRW) disposal facility, (2) encapsulation downhole inside the casing of well about to be plugged and abandoned, and (3) underground injection into a subsurface formation. Two states (Texas and New Mexico) also allow some radium-bearing NORM to be disposed of by landspreading, a practice that entails spreading the waste over the land surface and mixing it into the top few inches of soil. In Louisiana, regulated NORM containing up to 30 pCi/g of radium also may be treated and disposed of at commercial disposal facilities that are permitted to receive petroleum industry wastes. Only one state, Michigan, explicitly allows radium-bearing wastes, including petroleum industry NORM, to be disposed of in municipal, nonhazardous landfills. In Michigan, the Department of Environmental Quality (MDEQ) has issued guidelines allowing the disposal of soil and debris having an average Ra-226 concentration of ≤50 pCi/g in landfills designed and permitted to receive only nonhazardous wastes (MDEQ 1996).

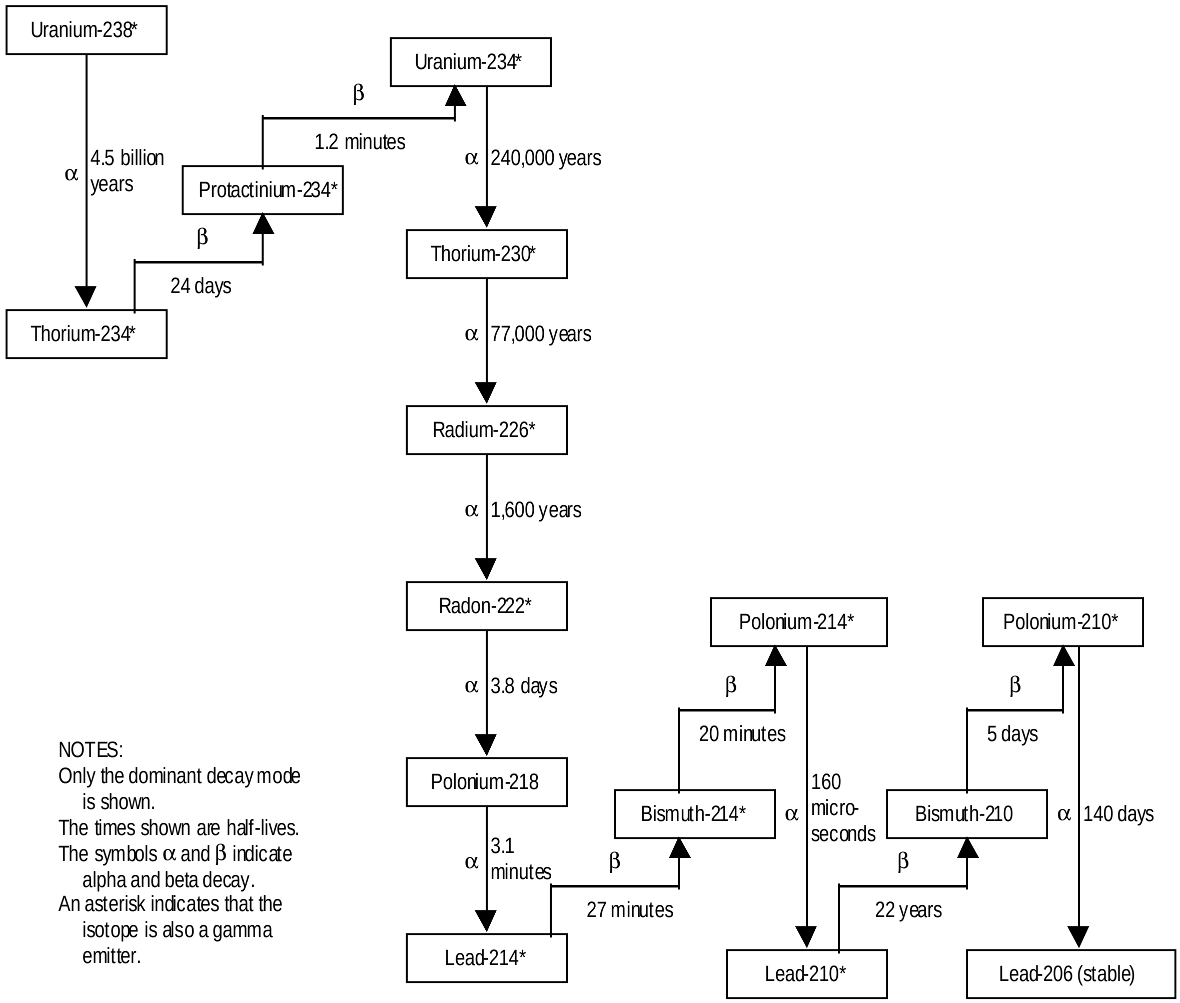

FIGURE 1 Uranium-238 Decay Series

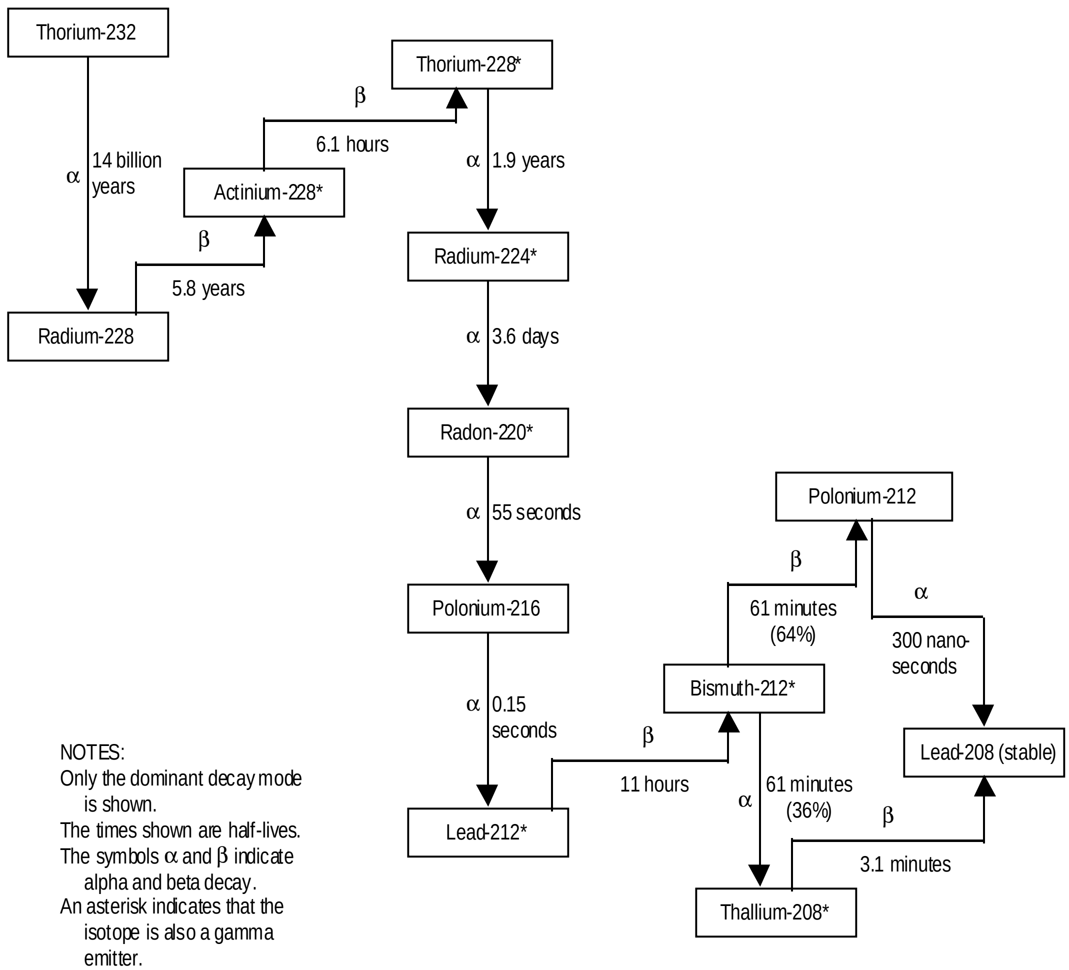

FIGURE 2 Thorium-232 Decay Series

1.2 OBJECTIVES AND SCOPE OF STUDY

The primary objective of this study was to assess the disposal of petroleum industry wastes containing NORM in nonhazardous landfills. Nonhazardous landfills are defined as landfills that are permitted under Subtitle D of the Resource Conservation and Recovery Act (RCRA) to receive only those wastes that are defined by RCRA as nonhazardous (Section 2).

Specifically, this study included (1) an analysis of existing federal and state laws to identify regulatory impediments to disposing of NORM via this nonhazardous landfill option; (2) a comparison of the costs associated with a variety of NORM disposal options, including those currently permitted by most state programs as well as the nonhazardous landfill option; (3) a study of the potential radiological doses and related health risks associated with the disposal of radium-bearing wastes in a nonhazardous landfill in accordance with the MDEQ guidelines (MDEQ 1996); (4) an assessment of the disposal of wastes containing Pb-210 in a nonhazardous landfill; and (5) a sensitivity analysis of key parameters that could affect potential radiological doses. The study does not include an evaluation of the potential risks associated with disposing of other contaminants or constituents of concern in nonhazardous landfills.

The study on radium-bearing NORM evaluated potential doses associated with the radium isotopes and their decay progeny. It assumed that (1) the waste stream consisted of soil mixed with barite scale containing radium and its progeny; (2) the average concentration of radium in the waste met the Michigan guideline constraints (i.e., 50 pCi/g Ra-226); and (3) the wastes were disposed of at a licensed, nonhazardous landfill located in Michigan. A variety of receptors and scenarios — including operational phase and future use scenarios — were considered. The assessment on wastes containing only Pb-210 and its progeny assumed that the waste stream was disposed of independently of the radium-bearing NORM wastes. Therefore, the assessments evaluated potential doses associated with disposal of either the radium-bearing wastes or the Pb-210 wastes, not both. The same receptors and scenarios evaluated in the case study were considered in the Pb-210 waste stream assessment. If both radium-bearing wastes and Pb-210 wastes were placed in the same landfill, the cumulative effects on the receptors could be conservatively estimated by summing the impacts estimated separately for each waste stream.

2 REGULATORY SETTING

2.1 SUMMARY OF LANDFILL REGULATIONS

The design and operation of all permitted solid waste landfills are governed by requirements contained in RCRA. Subtitle D of RCRA contains requirements applicable specifically to the management and land disposal of nonhazardous solid waste. Under RCRA, nonhazardous solid waste includes any discarded, abandoned, recycled, or inherently waste-like material that is not listed as a hazardous waste, does not exhibit any of four hazardous characteristics (i.e., ignitability, corrosivity, reactivity, and toxicity), and is not otherwise exempted. Radioactivity in solid wastes currently is not regulated as hazardous under RCRA.

Parts 257 and 258 in Title 40 of the *Code of Federal Regulations* (40 CFR Parts 257 and 258) provide regulatory standards for the location, design, construction, operation, monitoring, closure, and post-closure care of solid waste disposal facilities, including landfills. Landfills that are subject to these standards may be referred to as Subtitle D landfills or, more simply, nonhazardous landfills. In terms of the applicable regulatory requirements, nonhazardous landfills may be categorized on the basis of the types of solid wastes they are permitted to receive (e.g., household, industrial, or commercial). Part 257 establishes standards to minimize the risk presented to human health and the environment by all solid waste disposal units, except landfills permitted to accept "municipal solid wastes." A municipal solid waste landfill is any landfill that receives household waste. Municipal landfills, which also may receive other types of RCRA Subtitle D wastes, are regulated under 40 CFR Part 258, which establishes minimum national criteria for their location, operation, design, closure, and post-closure care. These regulations generally are more rigorous than those found in Part 257.

Individual states that have been granted authority to operate the Subtitle D permitting program within their borders may adopt the federal regulations by reference or they may promulgate their own regulations. State regulations must be at least as stringent as the federal regulations before approval of the permit program will be granted. Classification schemes for nonhazardous landfills dictating the type of nonhazardous solid waste that may be disposed of in a given facility vary from state to state and often establish more distinctions between types of facilities than are included in 40 CFR Parts 257 and 258.

With respect to the federal standards for nonhazardous landfills, the requirements are most stringent for municipal solid waste disposal landfills. This stringency is required, in part, because the universe of waste streams that may be disposed of in this type of landfill is rather broad. The primary waste stream going into these municipal landfills is household waste including garbage, trash, and sanitary wastes collected in septic tanks that are derived from homes, hotels, most other types of living quarters, and recreation areas, such as campgrounds and picnic grounds. Municipal solid waste landfills also may accept nonhazardous commercial and industrial wastes, nonhazardous sludges, and hazardous wastes generated by conditionally exempt small quantity generators (CESQGs).[1]

Location restrictions contained in Subpart B of 40 CFR Part 258 (§258.10–258.15) address minimum standards for citing municipal solid waste landfills near airports and in floodplains, wetlands, fault areas, seismic zones, and other unstable areas. Operating criteria specified in Subpart C (§258.20–258.29) include (1) requirements for screening incoming loads to ensure that hazardous wastes and polychlorinated biphenyl (PCB) wastes do not enter the landfill; (2) minimum standards for applying clean cover material; and (3) standards addressing disease vector control, explosive gas control, air emissions, site access restrictions, control of stormwater run-on and run-off, discharges to surface water, receipt of liquid wastes, and recordkeeping. Pursuant to Subpart D (§258.40), design plans must be adequate to ensure that drinking water standards (i.e., the maximum concentration levels) for specific chemicals will not be exceeded in the shallowest groundwater aquifer as a result of landfill operations, or the landfill must be constructed with a composite liner and leachate collection system. Subpart E (§258.50–258.58) specifies requirements for groundwater monitoring programs, groundwater contamination detection and assessment, and corrective actions. Requirements for closure and post-closure care, contained in Subpart F (§258.60–258.61), specify that after closure, the owner must record a notification on the deed to the landfill property (or on some other instrument that normally would be examined during a title search). The notification must state that (1) the property was used as a landfill facility and (2) its future use is restricted to the terms incorporated in the post-closure plan to ensure that the integrity of the final cover, liner, containment system, and monitoring system will not be compromised. This deed restriction must be maintained in perpetuity.

Municipal solid waste landfills meeting all of the requirements contained in 40 CFR Part 258 also meet many of the substantive requirements that have been developed for landfills that receive hazardous waste. Hazardous waste landfills are regulated under Subtitle C of RCRA, and the federal standards for locating, designing, constructing, operating, and monitoring permitted Subtitle C landfills are codified in 40 CFR Part 264. Administrative requirements for hazardous waste landfills — such as recordkeeping, reporting, contingency planning, and manifesting requirements — also are contained in Part 264; however, these are more extensive and rigorous than the administrative requirements for nonhazardous waste landfills. The major differences between the substantive requirements for hazardous and nonhazardous landfills include requirements for an additional top liner, an additional leak detection system, a broader point of compliance, and more frequent groundwater sampling.

2.2 LANDFILL DISPOSAL OF EXPLORATION AND PRODUCTION WASTES

In 1988, the U.S. Environmental Protection Agency (EPA) determined that many of the waste streams generated by petroleum industry exploration and production (E&P) activities,

[1] In accordance with 40 CFR Part 261.5, a CESQG is defined as someone who generates no more than 100 kg of hazardous waste or 1 kg of acutely hazardous waste in a given month. CESQGs are exempt from most of the RCRA hazardous waste generator requirements, including requirements to dispose of wastes in a permitted hazardous waste disposal facility.

including scales and sludges, should be exempt from regulation as hazardous waste (EPA 1988a). Consistent with this regulatory determination, most states that have primary authority for the regulation of hazardous wastes also exempt these waste streams from regulation as hazardous, regardless of the fact that they may contain hazardous constituents or exhibit hazardous characteristics.

In many states, E&P wastes, although exempt from regulation as hazardous, are considered to be distinct from other nonhazardous solid wastes, such as household waste. The EPA solid waste regulations specifically exclude oil and gas wastes from the definition of industrial solid waste (40 CFR Part 258.2); however, not all states have adopted this exclusion into their solid waste programs. Therefore, states may classify E&P wastes as "special wastes," "industrial wastes," "exempt E&P wastes," or "nonhazardous oilfield wastes (NOW)." Under these various classification schemes, the disposal of E&P wastes in landfills may be restricted to specific types of nonhazardous landfills, or their disposal in municipal solid waste landfills may be allowed by approval only. Approval to accept E&P wastes may be necessary from both state and local agencies. Documentation and recordkeeping requirements vary from state to state.

2.3 SUMMARY OF NORM REGULATIONS

Currently, no federal regulations specifically address the handling and disposal of NORM wastes. The EPA has issued guidelines for the disposal of NORM wastes generated by drinking water treatment processes (Section 2.3.1); however, these guidelines are not applicable to petroleum industry NORM.

In the absence of federal regulations, NORM regulation is the responsibility of individual states. A number of states have determined that their existing radiation protection programs already contain adequate regulations that are applicable to NORM. Other states have responded by developing specific regulatory programs for NORM. These programs have been evolving rapidly over the last few years. In most instances, the states that have promulgated NORM regulations have a significant level of oil and gas production. Although the primary emphasis of these regulations is on oil and gas wastes containing NORM, their scope typically covers NORM wastes generated by other industries.

The existing state regulatory programs establish standards for (1) NORM exemption or action levels; (2) the licensure of parties possessing, handling, or disposing of NORM; (3) the release of NORM-contaminated equipment and land; (4) worker protection; and (5) NORM disposal. The action levels defining when wastes must be managed as NORM wastes vary from state to state. These levels typically are expressed in terms of radionuclide activity concentrations (in pCi/g), exposure levels (in μR/h), surface contamination levels (in disintegrations per minute per 100 cm^2, and radon flux (in $pCi/m^2/s$). Materials exceeding any one of these state-prescribed levels become regulated NORM materials within that state.

With respect to radium-bearing wastes, typically, the radionuclide activity levels defining regulated NORM are established for Ra-226 or Ra-228. This level varies from state to state but generally is set at 5 or 30 pCi/g. In most states, this level excludes background concentrations of radium. Several states have established two action levels dependent upon the radon emanation rate of the waste. In these states, the action level is 5 pCi/g of radium if the radon emanation rate exceeds 20 pCi/m^2/s and 30 pCi/g of radium if the radon emanation rate is below that level.

With respect to Pb-210, most existing state-level NORM regulations specify a limit of 150 pCi/g of any NORM radionuclide other than Ra-226 or Ra-228. This standard would apply to Pb-210 wastes. In addition, materials impacted by Pb-210 fall under the category of regulated NORM if the Pb-210 concentration is high enough to cause an exceedance of the limit set for either external exposure or surface contamination.

The disposal options currently approved by most state NORM programs include (1) burial at a licensed NORM or LLRW disposal facility, (2) encapsulation downhole inside the casing of a well about to be plugged and abandoned, and (3) underground injection into a subsurface formation. A couple of states (Texas and New Mexico) also allow some radium-bearing NORM wastes to be disposed of by landspreading. Louisiana also allows some regulated NORM wastes to be treated and disposed of at commercial NOW disposal facilities.

Currently, none of the states with specific NORM programs approve of the disposal of regulated NORM wastes in commercial nonhazardous landfills. However, although Michigan does not have a NORM regulatory program, it has issued guidelines for the disposal of materials containing Ra-226, including petroleum industry NORM, in nonhazardous landfills (Section 2.3.2). Other states may allow the disposal of petroleum industry NORM or other wastes containing low concentrations of radioactivity in commercial landfills; however, such disposal may be undocumented or may be on a limited, case-by-case basis only (Section 2.4).

2.4 LANDFILL DISPOSAL OF NORM

2.4.1 Drinking Water Treatment Wastes Containing NORM

In June 1994, the EPA issued its *Suggested Guidelines for the Disposal of Drinking Water Treatment Wastes Containing Radioactivity* (EPA 1994). The purpose was to help water treatment facilities safely and responsibly manage their wastes containing radionuclides at concentrations in excess of background levels. Although these guidelines are not applicable to petroleum industry NORM wastes, the radionuclides addressed by the guidelines include Ra-226 and Ra-228, and some of the water treatment NORM wastes are similar in generation and concentration levels to the petroleum industry's NORM wastes. As a result, the risk-based disposal guidelines for radium-bearing wastes have some relevance to disposal issues facing the petroleum industry.

Under the suggested guidelines, water treatment wastes containing less than 3 pCi/g of total radium (i.e., Ra-226 plus Ra-228) may be placed in a Subtitle D, nonhazardous landfill. These wastes should be dewatered and, when combined with other radioactive materials placed in the landfill, should make up only a small fraction of the material in the landfill.

The EPA recommends that wastes containing between 3 and 50 pCi/g of total radium be disposed of by methods that "provide reasonable assurance that people will be protected from radon releases from the undisturbed waste and that the waste will be isolated to reduce the risk of disturbance or misuse" (page 29, EPA 1994). Specifically, the EPA recommends that the construction of a building on a disposal site containing Ra-226 wastes be avoided or, at least, any such buildings should not be used for residential or commercial purposes. The guidelines do not specify the measures needed to achieve these goals, but they do indicate that the disposal facility should at least be in compliance with RCRA Parts 257 and 258, (i.e., the regulations governing design, construction, operation, and monitoring of nonhazardous landfills) and that requirements for hazardous waste facilities (such as those contained in RCRA Part 264) be considered to ensure adequate groundwater protection. Under these guidelines, the disposal of NORM-impacted water treatment wastes would be allowed in both nonhazardous and hazardous landfills.

For wastes containing between 50 and 2,000 pCi/g of total radium, the EPA recommends that disposal decisions be made on an individual basis. At a minimum, the EPA recommends disposal of these wastes in a RCRA hazardous waste unit. This recommendation is made on the basis that nonhazardous landfills generally do not provide the appropriate degree of assurance that (1) intruders will not be endangered, (2) groundwater and ambient air pathways are adequately controlled, (3) the disposal site is adequately secure against natural disturbances, and (4) effective institutional controls are in place to prevent future misuse of the disposal site. In addition, the EPA recommends that wastes within this concentration range be considered for disposal at a licensed NORM or low-level radioactive waste disposal facility. When the concentrations exceed 2,000 pCi/g of total radium, the EPA recommends disposal in accordance with the provisions of the Atomic Energy Act for source materials.

Colorado, which does not have a NORM regulatory program, has adopted regulations specifically governing the disposal of drinking water treatment wastes containing NORM. In Colorado, water treatment sludge containing up to 40 pCi/g total alpha activity may be disposed of at a nonhazardous, solid waste disposal facility, provided there are no free liquids present in the sludge and its pH is ≤ 6 (in Title 6 of the *Colorado Code of Regulations* (CCR), 1007-2, Section 12). The regulations further stipulate operating and monitoring requirements for any landfill receiving these sludges, including requirements for a liner, adequate cover and surface drainage, access control, groundwater monitoring, and distribution of the sludge within the landfill. At this time, no other states appear to have adopted similar regulations.

2.4.2 Landfill Disposal of NORM Wastes in Michigan

Michigan's *Cleanup and Disposal Guidelines for Sites Contaminated with Radium-226* (MDEQ 1996) were promulgated by the MDEQ's Drinking Water and Radiological Protection Division. These guidelines address the remediation of sites located in Michigan that are contaminated with Ra-226 and its associated decay series. They establish acceptable levels of residual contamination for the release of facilities, equipment, or land for unrestricted use. The guidelines allow bulk wastes contaminated with Ra-226 (e.g., contaminated soil or debris) to be disposed of in a Type II solid waste landfill, provided the Ra-226 concentration does not exceed

50 pCi/g averaged over any single shipment and the maximum Ra-226 concentration within any single shipment does not exceed 100 pCi/g.

The MDEQ Waste Management Division regulates all solid waste disposal facilities under Part 115 of the Natural Resources and Environmental Protection Act of 1994 and the Part 115 Administrative Rules. Part 4 of these rules are specific to Type II landfills. Under these regulations, Type II landfills are municipal solid waste landfills that are permitted to accept only nonhazardous household wastes, municipal solid wastes, incinerator ash, sewage sludge, commercial wastes, and industrial wastes. Hazardous waste generated by CESQGs may be disposed of in some Type II landfills; however, these wastes typically would make up only a small fraction of the total volume of wastes received at a Type II landfill.

The regulations contained in Part 4 establish requirements for the location, design, operation, monitoring, closure, and post-closure care of Type II landfills in Michigan. In general, these requirements are equivalent to or more substantive than the EPA's requirements for municipal landfills contained in 40 CFR Part 258. Areas for which Michigan's regulations are more stringent include location restrictions, planning and reporting requirements for the design and construction of a landfill, and groundwater protection systems for units for which groundwater impacts cannot be determined by using a groundwater monitoring program.

As in Subpart F of 40 CFR Part 258, Rule 447 of the Type II landfill regulations stipulates that the operator of a municipal waste landfill must develop a post-closure plan that ensures that future use of the property will not disturb the integrity of the final cover, liner or liners, or any other components of the landfill's containment system. Furthermore, Section 11518 of Part 115 requires that the land owner(s) of any sanitary landfill file an instrument imposing a restrictive covenant upon the land involved. This covenant must state that the land described in the covenant has been or will be used as a landfill and that no future user of the property shall fill, grade, excavate, drill, or mine on the property during the first 50 years following completion of the landfill without authorization of the MDEQ.

2.5 APPLICABLE REGULATIONS IN OTHER STATES

Regulatory control over the disposal of NORM-impacted wastes is largely a function of individual state regulations. The regulations of states other than Michigan were reviewed to evaluate how regulators in these states view the potential disposal of NORM wastes in landfills. The review was limited to a few states in which oil and gas production activities are prominent.

In several of the states included in the review, regulated NORM wastes may be disposed of in various types of landfills; however, this type of disposal is generally considered on a case-by-case basis only, and broad provisions have not been promulgated in the state regulations. In other states, there seems to be less latitude in the regulations or on the part of individual regulators. In addition, it appears that, where specifically addressed in the states' regulations, the rules focus on radium isotopes and do not mention lead. A summary of these states' regulations governing the management of NORM-impacted wastes, E&P wastes, and other solid wastes are provided below. When pertinent, the interpretations of individual state regulators are presented.

2.5.1 Colorado

Specific NORM regulations have not been promulgated in Colorado. The on-site management and disposal of E&P wastes falls under the jurisdiction of the Colorado Oil and Gas Conservation Commission (COGCC); the 900 series of the COGCC rules and regulations addresses these wastes. Under Section 907.b, E&P wastes transported off-site for treatment or disposal must be sent to disposal facilities approved to receive E&P wastes by the Colorado Department of Public Health and Environment (CDPHE), which has jurisdiction over all solid waste disposal facilities, along with local governing bodies (usually the county or municipality where the facility is located).

Colorado's solid waste disposal regulations (6 CCR 1007-2) specify that solid waste does not include any materials regulated pursuant to the Colorado Radiation Control Act. Diffuse NORM wastes, however, are not regulated under the Radiation Control Act regulations (6 CCR 1007-1). The CDPHE regulations explicitly allow drinking water treatment sludges having a total alpha activity of ≤40 pCi/g of dry sludge to be disposed of in a nonhazardous, solid waste disposal facility, provided there are no free liquids present in the sludge and its pH is ≥6 (6 CCR 1007-2, Section 12.3). By general interpretation, this radioactivity concentration limit of 40 pCi/g total alpha activity is also applied to other solid wastes (Mallory 1998). Under state regulations, therefore, it is possible that any radioactive wastes falling below this threshold, including NORM-impacted E&P wastes, may be disposed of in nonhazardous landfills; approval for this disposal would be on a case-by-case basis only (Mallory 1998).

Each solid waste disposal facility in Colorado must be in compliance with the state regulatory requirements and obtain a Certificate of Designation from the local governing body. These certificates do not specifically permit or exclude NORM or other low-concentration radioactive wastes. Although not required to screen for radioactive materials, most large facilities voluntarily do so (Mallory 1998). As a result, it is unlikely that radioactive wastes exceeding 40 pCi/g total alpha activity end up in nonhazardous landfills located in Colorado.

2.5.2 Kansas

At this time, Kansas has not promulgated any specific NORM regulatory program. The Kansas Corporation Commission regulates the disposal of E&P wastes at the site of generation (KAR 82-3). All solid waste disposal facilities are regulated by the Kansas Department of Health and Environment (KDHE). At one time, the KDHE had regulations for oil field waste disposal; however, they were revoked on May 10, 1996 (KAR 28-41-1 to 28-41-9). Off-site disposal of E&P wastes is subject to the KDHE's solid waste disposal regulations.

Under the KDHE regulations, sanitary landfills may accept nonhazardous and industrial solid wastes (KAR 28-29-23). Municipal landfills, which are one category of sanitary landfill, may receive household waste and other nonhazardous wastes, including commercial solid waste, sludge, and industrial solid waste (KAR 28-29-101). Industrial solid wastes, which are defined as all solid wastes resulting from manufacturing and industrial processes (KAR 28-29-3), also may be disposed of in industrial landfills.

Under the KDHE regulations, it appears that E&P wastes are considered to be “special wastes,” not industrial wastes. Special wastes are defined as solid wastes that, because of their physical, chemical, or biological characteristics, present concerns regarding handling, owner or operator safety, management, or disposal and require special management standards. The KDHE considers any waste that is not a household or commercial waste, including E&P waste, to be a special waste (Cronin 1998). Special wastes can only be disposed of at a permitted municipal solid waste landfill in accordance with a special waste disposal authorization. Each application for a special waste disposal authorization must designate the process that produced the waste and the physical characteristics of the waste, including laboratory analysis to determine if the waste is a listed hazardous waste or a waste that exhibits the characteristics of a hazardous waste. There is no requirement, however, that the waste be analyzed for radioactive contamination.

The KDHE regulations require the owners of all municipal landfills to implement a program at the facility for detecting and preventing the disposal of regulated hazardous wastes and PCB wastes (KAR 28-29-108). There is no requirement for screening incoming waste loads for radioactive contamination; however, some facilities maintain radiation monitoring equipment (Cronin 1998).

Under this regulatory scenario, it appears that the disposal of NORM-impacted E&P wastes in nonhazardous landfills is not addressed explicitly by any Kansas regulations. It is possible that regulators would consider this disposal on a case-by-case basis; authorization or approval would have to be obtained from the KDHE.

2.5.3 Louisiana

The Louisiana Department of Environmental Quality (LDEQ), Radiation Protection Division, has issued NORM regulations that provide exemption levels for regulated NORM and establish regulatory criteria for management and disposal of all regulated NORM. These requirements, contained in Title 33 of the *Louisiana Administrative Code* (LAC), Part XV, Chapter 14, define regulated NORM, with respect to radium, as any material containing >5 pCi/g of Ra-226 or Ra-228 above background. The Louisiana Department of Natural Resources (LDNR), Office of Conservation, has regulatory oversight for the management and disposal of petroleum industry E&P wastes, including commercial NOW disposal facilities. All other solid waste disposal facilities in Louisiana are regulated by the LDEQ, Solid Waste Division.

State regulations prevent the disposal of any regulated radioactive wastes, including NORM, in nonhazardous landfills (Peterson 1999). Furthermore, all E&P wastes are excluded from the LDEQ’s definition of industrial solid waste (LAC33:VII, Chapter 1) and are prohibited from being disposed of in nonhazardous landfills. Instead, these wastes must be disposed of in NOW disposal facilities licensed by the LDNR. Under the LDEQ solid waste regulations (LAC33:VII, Chapter 7), permitted nonhazardous landfills must be equipped with a device or method to determine quantity, source, and type of incoming waste to ensure exclusion of prohibited wastes. Requirements for these devices or methods are not specified.

Pursuant to a memorandum of understanding between the LDEQ and the LDNR, regulated NORM wastes containing up to 200 pCi/g Ra-226 or Ra-228 may be disposed of in LDNR-licensed NOW disposal facilities, provided the facility obtains a specific license from the LDEQ, all operational procedures are adhered to, and the facility has a satisfactory compliance record (LAC33:XV, 1412.B.4). Although, at this time, there are no commercial NOW facilities operating under this provision, under a separate provision in the LDEQ's NORM regulations (LAC33:XV.1412.B.3), several LDNR-permitted, commercial NOW disposal facilities are allowed to accept E&P waste containing up to 30 pCi/g of Ra-226 or Ra-228 under a LDEQ general license. In Louisiana, commercial NOW disposal facilities generally fall into four separate categories: transfer stations, solids/liquids separation facilities, landfarms, and saltwater/produced water underground injection wells (Talbot 1999). The LDEQ has issued NORM general licenses to each of these commercial disposal facility categories, except the saltwater/produced water injection wells. No NORM acceptance criteria are imposed on underground injection wells because the LDEQ's NORM regulations specifically exempt produced waters. Like the State of Texas, a future category may arise — solids/slurry NOW/NORM injection wells. The NORM acceptance criteria for this category has yet to be determined (Talbot 1999).

In 1998, the LDNR issued a NOW disposal facility permit to an existing LDEQ-permitted Subtitle C hazardous waste landfill. At the same time, in a letter of approval, the LDEQ authorized this facility to accept NORM wastes containing up to 150 pCi/g of any NORM radionuclide. The facility's E&P waste acceptance criteria are regulated under the LDNR, whereas the NORM acceptance criteria are regulated under LDEQ. All environmental controls and monitoring of the E&P waste disposed into the landfill are regulated under the LDEQ.

2.5.4 Mississippi

In Mississippi, the regulation of NORM has been divided between two agencies. The Mississippi Department of Health (MDH), Division of Radiological Health, has established regulations for the extraction, mining, beneficiating, processing, use, transfer, storage, and disposal of NORM in the *Regulations for Control of Radiation in Mississippi*, Part 801, Section N. These regulations define regulated NORM, with respect to radium, as any material containing ≥5 pCi/g of Ra-226 or Ra-228 above background or ≥30 pCi/g of Ra-226 or Ra-228 if the radon emanation rate averaged over any 100 m^2 is ≤20 pCi/m^2/s. Jurisdiction over the control and disposal of regulated petroleum industry NORM falls under the Mississippi Oil and Gas Board (Board). The Board has issued two rules (Rules 68 and 69) governing these wastes. The Board's Rule 68, which is specific to the disposal of NORM, is applicable only to on-site disposal activities.

Solid waste disposal facilities are regulated by the Mississippi Department of Environmental Quality (DEQ), Office of Pollution Control. In accordance with the requirements for solid waste management facilities established by the Mississippi DEQ (contained in the *Nonhazardous Solid Waste Management Regulations and Criteria*), E&P wastes are specifically exempted from solid waste regulation, unless they are disposed of or processed in a commercial oil field E&P waste disposal facility. Under the state's solid waste regulations, each municipal solid waste management facility must have a program for detecting and preventing the disposal of hazardous waste and PCB waste; these same restrictions are included in permits issued to commercial E&P disposal facilities. For hazardous waste and PCB wastes, the facility must conduct random inspections of incoming loads and train facility personnel to recognize regulated hazardous waste and PCB wastes. There are no specific requirements for screening for radioactive contamination.

Under its regulations, the MDH stipulates that transfers of waste containing regulated NORM for disposal may be made only to a person specifically authorized to receive such waste. Each facility disposing of regulated NORM must have been issued a specific license by the MDH. Given this specific language, and the absence of any provisions in other regulations, it appears unlikely that regulated NORM wastes may be disposed of in nonhazardous landfills in Mississippi.

2.5.5 New Mexico

In New Mexico, the regulation of NORM has been divided between two agencies. The New Mexico Environment Department, Environmental Improvement Board (EIB), regulates the possession, use, transfer, and storage of NORM under Title 20 of the *New Mexico Administrative Code* (NMAC) Chapter 3.1400. These regulations define regulated NORM, with respect to radium, as any material containing >30 pCi/g of Ra-226 above background. The New Mexico Oil Conservation Division (OCD), a branch of the New Mexico Energy, Minerals, and Natural Resources Department, regulates the disposal of petroleum industry NORM. The NORM disposal regulations, promulgated by the OCD, are contained in 19 NMAC 15.I.714.

Any person disposing of regulated NORM is subject to the regulations of both the EIB and the OCD. Under the OCD regulations, regulated NORM may be disposed of by a number of different methods, including underground injection, downhole disposal in a plugged and abandoned well, and disposal at a commercial or centralized surface waste management facility. The OCD rules do not provide for the disposal of NORM in landfills.

The EIB, which also has jurisdiction over solid waste disposal facilities (20 NMAC 9.101 et seq.), explicitly prohibits the disposal of petroleum industry wastes in any solid waste disposal facility, including landfills. Furthermore, the solid waste management regulations prohibit the disposal of any radioactive waste in a solid waste disposal facility. No minimum regulatory contaminant threshold is established for the radioactive wastes. All solid waste facilities must implement a plan approved by the EIB to inspect loads in order to detect and prevent the disposal of unauthorized wastes; however, there are no specific requirements for screening for radioactivity.

2.5.6 Oklahoma

At this time, Oklahoma does not have any specific NORM regulations. Although a set of proposed regulations has been drafted by the Oklahoma Radiation Management Advisory

Council, efforts to finalize these regulations have been suspended. Nonradioactive E&P wastes generated by the petroleum industry are regulated by the Oklahoma Corporation Commission (OCC). Waste management regulations issued by the OCC (contained in the *Oklahoma Administrative Code* (OAC), Title 165, Chapter 10) make no reference to potential radioactive components in the E&P wastes.

The Oklahoma DEQ regulates the disposal of solid waste. Any person operating any type of solid waste disposal site must have a permit from the DEQ. Solid waste disposal sites may accept nonhazardous industrial solid waste, provided the generator and the disposal facility comply with OAC 252:520, Subchapter 2, and written approval is obtained from the DEQ (OAC 252:520-1-6). Nonhazardous industrial solid waste is defined as waste generated by a manufacturing or industrial process, a spill of a commercial product, or unusable industrial or chemical products. The generator is responsible for properly identifying its waste through analysis and/or knowledge of the process to determine whether such waste is hazardous or nonhazardous. Nonhazardous industrial waste must be identified specifically by the generator. All generators of this waste must provide a specific notification to the DEQ, including a description of the process that generated the waste and the physical features of the waste. It must identify if the waste is radioactive (OAC 252:520-2-2). If the wastes is identified as radioactive, it cannot go to a nonhazardous landfill or any other solid waste disposal site in Oklahoma; it must be disposed of as radioactive waste under the jurisdiction of the Radiation Management Section, Waste Management Division of the DEQ (Broderick 1998).

Each solid waste disposal facility must submit to the DEQ an operational plan that outlines procedures for excluding the receipt of hazardous, radioactive, regulated PCB, and regulated infectious wastes (OAC 252:510-17-5) and other wastes not consistent with its permit (OAC 252: 520-5-3). There is no requirement specifically for screening for radioactivity of incoming wastes.

2.5.7 Texas

In Texas, regulatory oversight of NORM is split among three agencies. The Texas Department of Health (TDH), Bureau of Radiation Control, has jurisdiction over the possession, use, transfer, transport, or storage of NORM, including establishing the definition of regulated NORM. The TDH regulations specific to NORM are contained in Title 25 of the *Texas Administrative Code* (TAC), Chapter 289.259. Under these regulations, most regulated NORM, with respect to radium, is defined as any material containing >30 pCi/g of Ra-226 or Ra-228, provided the radon emanation rate is <20 pCi/m^2/s, or >5 pCi/g of Ra-226 or Ra-228, if the radon emanation rate is ≥20 pCi/m^2/s. For E&P wastes, however, the level is >30 pCi/g of Ra-226 or Ra-228, regardless of the radon emanation rate. The Railroad Commission of Texas (RRCT) has responsibility for regulating the disposal of E&P NORM and the Texas Natural Resource Conservation Commission (TNRCC) has jurisdiction over all other types of NORM. The RRCT's NORM disposal regulations, contained in 16 TAC 3.93, do not explicitly allow the disposal of regulated NORM in nonhazardous landfills.

All solid waste disposal activities are regulated by the TNRCC and the local county. The definition of solid waste does not include E&P wastes; however, these wastes meet the definition of "special wastes" when they are to be processed, treated, or disposed of at a permitted solid waste management facility. Under the TNRCC regulations, all solid waste management facilities are required to monitor incoming loads of waste to ensure the facility is operated in compliance with its permit. Although there are no requirements for the use of radiation detection devices, most large facilities use sodium iodide detectors to prevent the unknowing acceptance of radioactive waste (Cooksey 1998). Special permission may be granted to a solid waste disposal facility to accept E&P wastes containing NORM above regulated levels on a one-time basis or for 6-month or 1-year intervals, provided analytical data, including the Ra-226 and Ra-228 concentrations, are provided to the TNRCC (Bolmer 1998). If the concentration of NORM in the waste exceeds an acceptable concentration, it must be disposed of in a facility regulated by the U.S. Nuclear Regulatory Commission or an Agreement State.

2.6 RADIATION DOSE STANDARDS

Under existing regulations for workers classified as radiation workers by state or federal law, doses are required to be as low as reasonably achievable, not to exceed an annual dose of 5 rem/yr, as specified in 10 CFR Part 20.[2] This limit would apply to workers who handle NORM only if they were classified as radiation workers by state regulations; otherwise, NORM workers are subject to dose limits that apply to the general public. The currently accepted public dose limit recommended by the International Commission on Radiological Protection (ICRP 1991) is 100 mrem/yr from all sources.[3] In addition, the Conference of Radiation Control Program Directors, Inc. has recommended a public dose limit of 100 mrem/yr from all licensed sources, including NORM, in its Part N suggested guidelines for regulation and licensing of NORM (1999).

[2] The unit "rem" stands for roentgen equivalent man. It is a unit of radiation dose that incorporates both the amount of ionizing radiation absorbed by tissue and the relative ability of that radiation to produce particular biological change. The unit is frequently applied to total body exposure for all types of ionizing radiation.

[3] A millirem is equal to one thousandth of a rem.

3 CHARACTERIZATION OF NORM WASTES

3.1 RADIUM-BEARING NORM WASTES

Numerous surveys have been conducted by industry and state agencies to characterize the occurrence and distribution of radium-bearing NORM wastes. Unfortunately, most of the data from these surveys are not readily available, primarily because they were collected by private companies and are considered proprietary. This limited accessibility, coupled with the fact that the available data have not been aggregated to a national level, makes it difficult to fully characterize radium-bearing NORM wastes, particularly with respect to calculating average NORM activity levels.

For the radium-bearing wastes, available data are adequate to determine that, in general, NORM concentrations are greatest in the scales and sludges that form in water-handling equipment and that activity levels decrease with distance from the wellhead. Total radium concentrations depend on the amount of radium present in the subsurface formation, formation water chemistry, extraction processes, treatment processes, and age of production. In general, radium solubility increases in water that has (1) a high saline content and (2) either low or high pH values. Radium precipitation rates increase with decreasing temperature and pressure conditions, such as those encountered when subsurface fluids are brought to the surface. Extraction processes (e.g., water floods, steam floods, chemical floods) and treatment processes that alter a formation's water chemistry or cause temperature or pressure changes may increase or decrease radium mobility. Most radium is brought to the surface in solution in produced water. As a result, a higher water production rate, such as is characteristic of older fields, can result in increased NORM concentrations.

Scales and sludges that accumulate on pieces of equipment are the primary production waste streams of concern. When radium-bearing produced water, scales, or sludges are released to the ground, soils can become contaminated with regulated concentrations of radium. Under most state-level NORM rules, soils containing radium above the state's exemption levels must be remediated. NORM-impacted soils, which also may be candidates for disposal in nonhazardous landfills, are not discussed separately in this report. It is assumed that the descriptive information provided below for NORM scales is applicable to most NORM-impacted soils.

3.1.1 Scale

NORM contamination of scale can occur when dissolved radium in produced water coprecipitates with barium, strontium, or calcium sulfates. Typically, these sulfates form hard, insoluble deposits on the inside of piping, filters, brine disposal/injection wells, and other water handling equipment. Over time, scale deposits thicken and may need to be removed to ensure that equipment continues to operate properly. In Michigan, sulfate scales also may form on the outside of downhole casing as a result of the completion practice that allows intermediate casing to come in contact with formation waters. These scales are cleaned off the casing when it is removed from the borehole, or they may simply fall off on their own as the pipe is handled and stored.

Total radium concentrations in scale typically range from undetectable levels to several thousand pCi/g (Baird et al. 1990). However, concentrations as high as 410,000 pCi/g have been reported by the EPA (1993). The density of scale is approximately 2.6 g/cm^3 (EPA 1993).

The amount of Rn-222 emanating out of the pore spaces is relatively small when compared to the emanation fraction for a typical soil (e.g., 0.25). A recent study measuring the Rn-222 emanation fraction in scale found values ranging from 0.02 to 0.06 (Rood and Kendrick 1996).[4] Another recent study measured radon emanation from NORM-contaminated pipe scale, soil, and sediment at three sites located in Texas, Wyoming, and Oklahoma. (White and Rood 1998). Radon emanation fractions from pipe scale averaged 0.037 for Texas and 0.087 for Oklahoma. Measurements taken from soil and sediment averaged 0.22 for Oklahoma and 0.10 for Wyoming.

3.1.2 Sludge

Sludge deposits consist of accumulations of heavy hydrocarbons, tight emulsions, produced formation sand, and minor amounts of corrosion and scaly debris that settle out of suspension in some oil field equipment. NORM accumulates in sludge inside piping, separators, heater/treaters, storage tanks, and any other equipment where produced water is handled. It occurs when the radium coprecipitates with barium, strontium, or calcium in the form of insoluble sulfates or in the form of slightly more soluble silicates and carbonates.

Typically, NORM concentrations range from undetectable levels to 300 pCi/g (Baird et al. 1990), although sludge samples with Ra-226 concentrations as high as 700,000 pCi/g have been documented (Fisher and Hammond 1994). The density of sludge is approximately 1.6 g/cm^3 (Baird et al. 1990; EPA 1993). The amount of Rn-222 emanating from sludge is higher than the amount emanating from scale, primarily because sludge is more granular. The typical Rn-222 emanation fraction for sludge used in other studies is 0.2 (Baird et al. 1990; EPA 1993; Smith et al. 1996); actual measurements of radon emanation from sludge have not been taken.

3.2 WASTES CONTAINING LEAD-210

When compared with published information on radium-bearing NORM wastes, very little information has been published characterizing petroleum industry wastes containing Pb-210. In part because NORM regulations do not specifically address this radionuclide, the industry has not accumulated much information regarding this waste stream. However, as noted above, wastes containing Pb-210 in concentrations sufficient to exceed external exposure levels or

[4] The radon emanation fraction is the ratio of the amount of radon escaping into the internal porosity of a material to the total amount of radon produced by the decay of Ra-226 within the material.

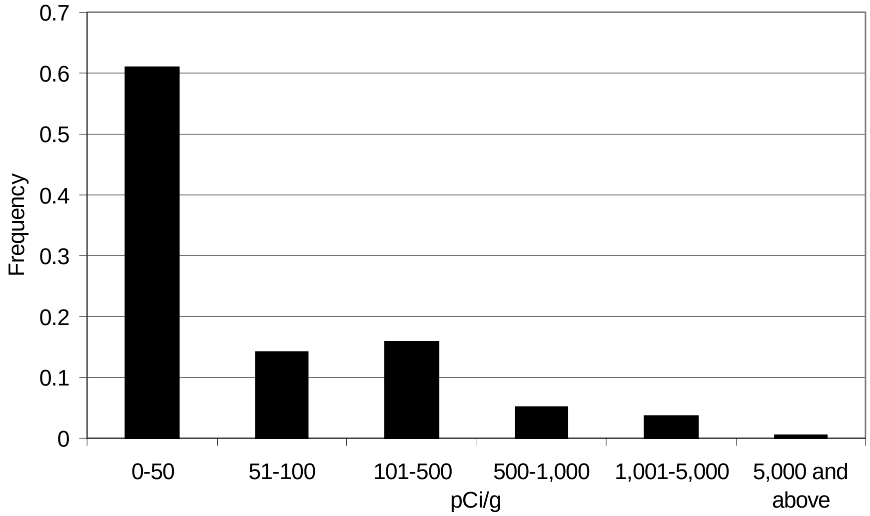

FIGURE 3 Distribution of Pb-210 Activity Concentration

surface contamination levels become a regulated material. Similarly, if the Pb-210 concentrations exceed other state-level radiation regulatory limits, these wastes present a waste management issue to the generator.

For the purposes of this assessment, a limited data set describing one company's inventory of Pb-210 wastes was provided by the American Petroleum Institute (API) (Beckstrom 1999). The data provided by API covered 509 samples of various types of waste containing Pb-210. The waste types included filters, residues, tank bottom solids, and propane vessel deposits. The bulk activity concentration for a drum of waste was estimated by multiplying the activity concentration of the sample by the fraction of the material in the sample that contained Pb-210. For example, if an unused filter weighs 50 g and a used filter, which contains material having Pb-210 contamination, weighs 100 g, the fraction of the material that contains Pb-210 would be 0.5. This methodology resulted in bulk activity concentrations ranging from less than 1 pCi/g to more than 10,000 pCi/g. A frequency distribution plot of the Pb-210 activity concentration is provided in Figure 3. The average and median Pb-210 bulk activity concentrations were estimated to be 200 pCi/g and 30 pCi/g, respectively.

The total activity for one truckload of Pb-210 material was estimated by multiplying the bulk activity of the total weight of material contained in all the drums for each waste type. One truckload was assumed to contain 96 55-gal drums of lead-bearing wastes, equal to 20 m^3. This process captures both physical and radiological characteristics of the material slated for disposal. The drum-weighted total activity for a representative truckload of 96 drums containing Pb-210-impacted waste was estimated to be 4.6 mCi, which corresponds to a drum-weighted bulk activity concentration of 256 pCi/g.

4 DISPOSAL COST STUDY

Currently, the number of options for disposing of regulated NORM wastes available to the petroleum industry is limited. NORM wastes can be disposed of either on-site or at commercial disposal facilities. Options explicitly provided for in existing state-level NORM regulations include (1) burial at a commercial, licensed NORM or LLRW disposal facility; (2) encapsulation downhole inside the casing of well about to be plugged and abandoned; (3) underground injection into a subsurface formation via a Class II well[5]; (4) landspreading (Texas and New Mexico only); and (5) commercial landfarms (Louisiana only).

Estimating the costs associated with these disposal options is difficult. The cost components that must be considered include those associated with the disposal activity, waste analyses, transportation, permitting, and container decontamination. For the most part, cost comparisons are truly accurate only when they are made on a case-by-case basis because cost is often determined by waste volume or radionuclide concentrations. For example, certain disposal options may not be cost effective unless a large volume of NORM waste is involved. Similarly, certain disposal options may not be cost effective for NORM wastes containing high concentrations of radium.

In 1996, the API published a report summarizing survey results on the petroleum industry's NORM disposal costs (API 1996). A more recent compilation of NORM disposal costs, as reported by Veil et al. (1998) and Veil and Smith (1999), provides a summary of 1998 costs. Brief descriptions of each of the existing NORM disposal options and the associated costs are provided below. For simplicity, transportation costs are not calculated on the basis of the assumption that they are largely a function of distance and are not otherwise a determining factor in choosing one disposal option over another. Unless otherwise noted, the cost information is taken from the 1998 disposal costs reported by Veil et al. (1998) and Veil and Smith (1999). For comparison purposes, the potential costs that might be associated with disposal of regulated NORM in a nonhazardous landfill also are discussed.

Table 1 presents a summary of the disposal costs for all of the options discussed below. On the basis of this information, it appears that disposal of regulated NORM in nonhazardous landfills could be one of the most cost-effective disposal options available to the petroleum industry if approved on a widespread basis. As Table 1 reflects, costs associated with any given disposal option can be quite variable. Some of the existing alternatives to the nonhazardous landfill option, such as on-site injection or landspreading, may be comparable in cost under specific operating conditions. NORM disposal costs are heavily influenced by market factors and are adjusted frequently as competition dictates. If disposal of NORM in nonhazardous landfills were to become an option in a number of states, fees charged by some of the commercial NORM disposal facilities potentially would decrease. An increase in the number

5 Class II injection wells are a specific category of injection wells used by the petroleum industry to dispose of saltwater produced in conjunction with oil or gas, to inject fluids to enhance oil recovery, or to store hydrocarbon liquids. Class II injection wells are regulated at either the state or federal level in accordance with requirements of the Underground Injection Control program, Part C of the Safe Drinking Water Act of 1974.

TABLE 1 Summary of NORM Disposal Costs[a]

Disposal Method	Disposal Cost ($/55-gal) 1992[b]	Current
NORM/LLRW landfill		
Envirocare of Utah, Inc.	300–700	Variable[c]
U.S. Ecology	395–730	500–550[c]
Chemical Waste Management, Inc.	NA	60–90[d]
Waste Control Specialists, LLC	NA	27–515[e]
Downhole encapsulation	792–3,333	No longer in use
Underground injection		
On-site	151–2,300	20–393[c]
Newpark Environmental	49–1,000	196[c]
Lotus, LLC	NA	132[c]
On-site landspreading	NA	45–56
Commercial landfarms	100–325[f]	12–39[g]
Nonhazardous landfill		
Company A, Michigan	NA	10–14
Company B, Michigan	NA	5–6

[a] NA indicates data were not collected and are not available.
[b] API (1996.)
[c] Veil et al. (1998); Veil and Smith (1999).
[d] Grout (1999).
[e] Dornsife (1999).
[f] Based on fees charged at NOW facility permitted to accept up to 200 pCi/g Ra-226 or Ra-228; this facility is no longer in operation.
[g] Based on fees charged at facilities permitted to accept up to 30 pCi/g Ra-226 or Ra-228 (Spencer 1999).

of disposal options as well as the number of commercial disposal facilities would most likely reduce NORM disposal costs for the petroleum industry.

4.1 BURIAL AT A LICENSED NORM OR LLRW DISPOSAL FACILITY

Currently, there are two commercial licensed NORM or LLRW disposal facilities that can accept regulated NORM generated by any industry. One is a mixed waste landfill operated by Envirocare of Utah, Inc., in Clive, Utah. The other is an LLRW landfill operated by U.S. Ecology on the U.S. Department of Energy (DOE) Hanford site in southeastern Washington. Neither of these facilities currently receives sizeable quantities of petroleum industry NORM wastes. In part, this situation may be a result of the physical location of these facilities; neither is located centrally to most of the nation's oil and gas production. In addition, it may be because of the higher costs associated with these two facilities.

The base disposal costs for NORM wastes at U.S. Ecology range from $500 to $550/55-gal drum or from $67 to $73/ft^3, depending on volume. All shipments are subject to a minimum disposal fee of $2,500. Because Washington State does not exempt oil and gas E&P wastes from regulation as hazardous waste, each waste stream must be analyzed for both hazardous waste characteristics and radionuclide content; the base cost does not include the associated analytical costs. In addition, the base costs do not include the cost of obtaining a site-use permit required by the Washington Department of Ecology.

A standard disposal cost was not provided by Envirocare because the company establishes disposal costs on a case-by-case basis. Envirocare recognizes that its costs may not be competitive for small quantities of NORM wastes because of flat overhead charges for all shipments of radioactive wastes, but the company believes it is competitive when large quantities of NORM waste are involved.

A third landfill, located near Lake Charles, Louisiana, is a Subtitle C hazardous waste landfill that is permitted to receive regulated NOW/NORM wastes. This landfill is operated by Chemical Waste Management, Inc., under a NOW disposal facility permit from the LDNR and a letter of approval issued by the LDEQ (Fontenot 1999). Regulated NORM containing up to 150 pCi/g of any NORM radionuclide may be disposed of at this facility. The base disposal costs for NORM wastes at this facility range from $60 to $90/55-gal drum, depending on total radionuclide content (Grout 1999).

A fourth landfill is anticipated to be able to receive regulated NORM wastes in the future. This facility, which currently operates under a RCRA Subtitle C permit and a Toxic Substances Control Act permit as a toxic and hazardous waste landfill, is operated by Waste Control Specialists, LLC (WCS), near Andrews, Texas. Under its current regulatory status, WCS can accept petroleum industry wastes containing NORM only at concentrations below Texas's exemption level for regulated NORM. The facility, however, intends to apply for a permit as the LLRW disposal facility for the Texas Low-Level Waste Disposal Compact within the next year.[6] If this permit is issued, WCS would be able to accept NORM wastes containing up to 100 nCi/g of radium for disposal. Currently, WCS has established a sliding scale for its disposal costs, providing discounts for larger volumes of waste. For large shipments of wastes containing nonregulated NORM, WCS charges $100 to $200/yd^3. For a shipment of only one drum, WCS would charge approximately $70/ft^3. These fees translate to approximately $27 to $55/55-gal drum for a large shipment and approximately $515 for a single 55-gal drum. At this time, WCS does not anticipate increasing this fee significantly for disposal of regulated NORM (Dornsife 1999).

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[6] The Texas Low-Level Waste Disposal Compact, which is an agreement between Texas (the host state), Maine, and Vermont, needs to locate and develop a facility for the disposal of LLRW generated within the borders of all member states. In October 1998, the TNRCC, which would issue a license to the Compact's LLRW facility, voted not to locate the LLRW disposal facility in Hudspeth County, Texas. Legislation has been introduced in Texas to locate the LLRW facility in Andrews County. If this legislation is adopted, the WCS facility in Andrews, Texas, could become the Texas Compact LLRW disposal facility (Dornsife 1999).

4.2 DOWNHOLE ENCAPSULATION

Although rarely used to dispose of regulated NORM wastes anymore, downhole encapsulation once was a very commonly employed NORM disposal option. The process entails disposing of NORM wastes inside the casing of a well that is about to be plugged and abandoned. Sometimes the wastes are sealed inside one or more strings of tubing that are placed in the wellbore; in other instances, the NORM-impacted wastes are placed in the wellbore in bulk. This disposal option requires access to a well that is about to be plugged and abandoned. The interior integrity of the borehole must be sufficient to allow placement of the NORM wastes. The volume of NORM-impacted wastes that can be disposed of in this fashion is limited by the number of suitable wells and the total depth of these wells.

This practice is no longer widely employed by the petroleum industry because other disposal options are more cost effective (Wimberley 1999). According to the API (1996), the cost of disposing of NORM wastes via downhole encapsulation ranged from $792 to $3,300/55-gal drum in 1992.

4.3 UNDERGROUND INJECTION

Underground injection may be performed in private wells or at commercial NORM disposal facilities. Currently there are two commercial NORM injection facilities. One, operated by Newpark Environmental Services, Inc. (Newpark), is located near Winnie, Texas. The other, operated by Lotus LLC, is located near Andrews, Texas. Both facilities crush, mill, and slurry the NORM wastes before injection into permitted Class II wells. Newpark charges approximately $196/55-gal drum; this cost includes inspection and verification of contents as well as necessary analytical costs. Lotus LLC charges approximately $132/55-gal drum plus approximately $100 per sample for gamma spectroscopy analysis.

On-site injection of NORM into a Class II injection well is allowed on a case-by-case basis under many state NORM regulations. Many companies currently provide services in which the company will come to an operator’s lease site, process the NORM wastes into an injectable slurry, and inject the wastes into a Class II well. The operator must seek approval from its regulators for this disposal action; in some cases, this may require a permit modification, including a public notification process. The fees charged by these companies vary greatly, ranging from a projected cost of $13 to $18/55-gal drum from one company up to $131 to $392/55-gal drum from another company.

4.4 ON-SITE LANDSPREADING

Landspreading is a long-standing waste disposal option for exempt E&P wastes. It is not a labor-intensive process and requires little more than a suitable tract of land and some basic earth-moving equipment. For very small landspreading activities, a shovel may be all that is required. Landspreading for the disposal of NORM wastes is allowed only in Texas and New Mexico under specific circumstances.

In Texas, landspreading of NORM waste is allowed without a permit on the lease site where the waste was generated, provided the resultant total radium concentration in the soil is ≤5 pCi/g above background levels (16 TAC 3.94(e)(2)(A)). Off-site surface disposal of NORM is allowed in Texas, provided the same dilution standards are met and a permit is obtained (16 TAC 3.94(g)). The permit application must describe the physical nature, volume, and activity level of the waste; background activity level; and dust control measures, and it must include written permission from the surface owner.

In New Mexico, in accordance with requirements contained in the NORM regulations promulgated by the Environment Department (20 NMAC 3.1.1407(A)), on-site surface disposal of NORM-contaminated soils is allowed provided a general license is obtained, a Subpart 13 permit is obtained, and the operator complies with the requirements of OCD Rule 711 that govern surface waste management facilities. Under this regulation, general licensees may blend or disk NORM-contaminated soil in place, provided the soils at the site were contaminated with NORM prior to promulgation of the regulation (i.e., August 3, 1995) and provided the exemption standard for Ra-226 in soil of 30 pCi/g above background is not exceeded. Under 19 NMAC 15.I.714(c)(1), the NORM disposal rules promulgated by the OCD, the disposal of NORM is allowed at centralized surface waste management facilities, provided it is disposed of in a manner that is protective of the environment, public health, and fresh waters and provided the centralized facility operates under a Rule 711 permit.

The costs associated with landspreading regulated NORM have not been well quantified by operators. The base cost for any landspreading activity will always include the labor involved and any costs associated with obtaining the earth-moving equipment (e.g., rental costs, cost to transport equipment to the site). Cost associated with obtaining required permits and confirming that state criteria for landspreading have been met also need to be factored in. One case study, published by Landress (1997), reported that the disposal of 40 m^3 (equal to 192 55-gal drums) of wastes, containing an average Ra-226 concentration of 120 pCi/g, by landspreading costs between $2,900 to $3,600 per day. The total cost for this project was not provided; however, from the effort described to survey the site, conduct the landspreading, and conduct confirmatory sampling, one could assume the work spanned at least three days. Under that assumption, the total project cost could have ranged from $8,700 to $10,800, equal to costs ranging from $45 to $56/55-gal drum.

4.5 COMMERCIAL LANDFARMS

In Louisiana, there are several commercial NOW landfarms that are allowed to accept regulated NORM wastes containing up to 30 pCi/g of Ra-226 or Ra-228 for treatment and disposal. Under the provisions of the state NORM regulations (LAC33:XV.1412.B.3), these facilities must treat the NORM wastes so that the radium concentrations do not exceed 5 pCi/g above background levels. These NOW/NORM landfarms are all operated by U.S. Liquids of Louisiana, LTD, at several different locations throughout the state. The gate rates charged for disposal of NOW wastes at these landfarms are determined in part on the basis of hydrocarbon and chloride content of the waste (Spencer 1999). The disposal fee is $8.75/barrel (equal to about $11.50/55-gal drum) for wastes containing <5% hydrocarbons and <10,000 ppm chlorides and \$10.75/barrel (equal to about \$14/55-gal drum) for wastes exceeding these levels. When the NOW wastes contain regulated concentrations of NORM, an extra charge of \$0.75 per pCi/g of radium is accessed for each barrel. With this fee added in, the gate rate charged to dispose of regulated NORM wastes ranges from approximately \$12 to \$39/55-gal drum. For large volumes of waste, rate reductions may be negotiated.

4.6 DISPOSAL AT A COMMERCIAL NONHAZARDOUS LANDFILL

As discussed in Section 3, in Michigan, wastes containing up to 50 pCi/g of radium can be disposed of in municipal, nonhazardous landfills. To assess the cost of this potential disposal option, two companies operating municipal landfills in Michigan were contacted. Neither company wanted to be identified by name and are referred to only as Company A and B in this report.

Company A currently charges \$16 to \$25/yd^3 to dispose of bulk wastes at its Type II municipal landfill. When asked to speculate what the company might charge to dispose of radium-bearing wastes in accordance with Michigan's policy, the contact at this company speculated that the cost would increase to at least \$35 to \$50/yd^3, a cost that equals approximately \$10 to \$14/55-gal drum. The contact reiterated that this number was purely speculative because the company had not assessed all the potential liability and risk issues that could be associated with disposing of radioactive wastes.

In comparison, Company B, which has received radium-bearing wastes under the Michigan policy, estimates that it would charge approximately \$20/yd^3 of radium-bearing wastes. This cost equals approximately \$5 to \$6/55-gal drum.

5 ASSESSMENT METHODOLOGY

5.1 ESTIMATION OF RADIOLOGICAL DOSES AND CARCINOGENIC RISKS

Radiation exposure pathways can be separated into external and internal components. External exposure, which occurs when the radioactive material is outside the body, is a concern primarily only for gamma radiation because it can easily penetrate tissue and reach internal organs. Internal exposure occurs when the radioactive material is taken into the body through inhalation or ingestion. For internal exposures, alpha and beta particles are the dominant concern because their energy is almost completely absorbed in adjacent cells, potentially causing biological harm.

Exposure to internally deposited radioactive contaminants is expressed in terms of the 50-year committed effective dose equivalent (CEDE). This concept, developed by the ICRP (1977), represents the weighted sum of the dose equivalent in various organs. The CEDE considers the radiosensitivity of different organs, biological effectiveness of different types of radiation, and variable retention times in the body for different radionuclides. For external pathways, no long-term residence of radionuclides in the body occurs, and the measure of dose is the effective dose equivalent (EDE). Both CEDE and EDE are expressed in units of rem.

The major radiological health concern from exposure to NORM is potential induction of cancer. The development of radiation-induced cancer is a stochastic process and is considered to have no threshold dose (i.e., the probability of occurrence, not the severity of effect, increases with dose, and there is no dose level below which the risk is zero). The relationship between radiation dose and development of cancer is well characterized for high doses of most types of radiation, but for low doses, it is not well defined and is subject to a large degree of uncertainty. Low levels of radiation exposure may present a health risk, but it is difficult to establish a direct cause-and-effect relationship because of the lack of data and the presence of compounding environmental stresses. In the absence of definitive data, the risk from low levels of radiological exposure are estimated by extrapolating from data available for increased rates of cancers observed at higher doses. For this assessment, radiation doses were converted to carcinogenic risks by using risk factors identified in ICRP Publication 60 (ICRP 1991). The ICRP risk factor is 5×10^{-7} per mrem for the public and 4×10^{-7} per mrem for workers. Risks are expressed as the increased probability of fatal cancer over a lifetime.

As a point of reference, radiation exposures from natural sources of radiation result in an annual dose of about 300 mrem/yr — approximately 200 mrem/yr from exposures to Rn-222 and its short-lived decay progeny and 100 mrem/yr from exposures to other natural sources of radiation (National Council on Radiation Protection and Measurements 1987). By applying the ICRP risk factor for the public, the risk of fatal cancer over a lifetime from background radiation is 2×10^{-4}/yr.

5.2 IDENTIFICATION OF SCENARIOS AND EXPOSURE PATHWAYS

For this study, the disposal of NORM-impacted wastes in a specific nonhazardous landfill was modeled to evaluate potential doses and health risks to workers and the public. Evaluations were performed for a variety of potential receptors who could be exposed as a result of waste placement activities (i.e., operational phase scenarios), future use of the property following closure of the landfill, or future consumption of contaminated groundwater (i.e., future use scenarios). Dose calculations were conducted for the maximally exposed receptor for each scenario. Collective doses were also estimated for the off-site population who could be exposed during waste placement operations.

5.2.1 Operational Phase Scenarios

Operational phase scenarios were evaluated to estimate potential exposures resulting from waste placement activities to on-site workers and to members of the general public living in the vicinity of the landfill. Workers considered to have potential exposures included the workers involved in landfill operations and the single worker involved in leachate management.

Landfill Operators. Two types of landfill operators were evaluated: a driver and a waste-placement operator. The driver was assumed to conduct all activities from inside a truck cab, so the only potential route of exposure would be external irradiation. The waste-placement operator would be involved with activities related to receiving and sampling the waste (e.g., reviewing the manifest, weighing the truck, and inspecting the shipment). The operator also would direct placement of the waste while standing in the vicinity of the truck. Potential routes of exposure for the waste-placement operator would include external irradiation and inhalation of contaminated particulates. In instances in which the wastes would be disposed of in containers, inhalation of particulates would not be considered to be a potential pathway of exposure. Exposure times were estimated on the basis of the volume of material disposed of and typical handling procedures at the landfill.

Leachate Worker. It was assumed that the leachate worker would be responsible for pumping the leachate generated at the landfill into and out of a truck. The only pathway of exposure would be external irradiation. Doses were estimated for transfer of a total volume of 2×10^6 ft^3 of leachate per year. The time required for handling this volume was estimated to be 30 minutes.

Off-Site Residents. Radiological doses and health risks resulting from potential airborne emissions generated during waste placement also were evaluated for the population living near the landfill. Exposures were estimated for the maximally exposed member of the public (i.e., an individual living adjacent to the landfill) and the collective population dose (i.e., the population living within a 50-mi radius of the landfill). The primary pathway of exposure would be inhalation of contaminated particulates. External irradiation, incidental ingestion of contaminated particulates, and ingestion of contaminated foodstuff were also evaluated for completeness. The residents were assumed to spend 24 hours per day at the residences, 365 days a year.

5.2.2 Future Use Scenarios

Four scenarios were considered to evaluate the potential radiological doses and health risks associated with future use of the landfill property or future use of groundwater underlying the landfill. Three of these scenarios evaluated potential doses to on-site receptors, including an on-site resident, industrial worker, and recreational visitor. The fourth future use scenario evaluated potential doses to an individual living next to the landfill after its closure who consumed groundwater.

On-Site Resident. On-site residential use of the property after closure of the landfill was evaluated as the most conservative scenario (i.e., the scenario expected to result in the greatest risk). Under this scenario, it was assumed that an individual lived on the site and produced most of his or her food on site, including vegetables, milk, meat, and fish. This scenario may not represent a realistic future use of a landfill; however, it was evaluated to represent a maximally exposed individual. These residential land use assumptions are commonly used by risk assessors to evaluate the potential dose to a maximally exposed individual.

It was assumed that the resident lived in a home built on a slab. Consistent with landfill requirements, it was assumed that the integrity of the landfill cap would be maintained so that construction of a home with a basement would not be possible. The resident was assumed to spend 18 hours each day on site (12 hours spent indoors), 365 days per year. The likely exposure pathways for the on-site resident would include external irradiation and inhalation of indoor and outdoor radon. Although unlikely, given that the integrity of the landfill cap would be maintained, the following pathways of exposure were also evaluated: inhalation of contaminated particulates; inadvertent ingestion of contaminated soil; and ingestion of crops, milk, and meat grown on the contaminated property. It was assumed the resident's water supply was from an unaffected off-site source, such as a municipal drinking water system.

On-Site Industrial Worker. The on-site office worker scenario considered potential exposures to an individual working inside a building constructed over the landfill. The building was assumed to be built on a slab. The receptor was assumed to work on site eight hours per day, five days per week. Exposure time was assumed to consist of six hours spent indoors and two hours spent outdoors. The exposure pathways evaluated included external irradiation, inhalation of indoor and outdoor radon, inhalation of contaminated particulates, and inadvertent ingestion of soil. As they did for the residential scenario, only external irradiation and inhalation of radon would represent complete pathways of exposure if the integrity of the landfill cap was maintained. Again, it was assumed that the on-site worker's water supply was from an unaffected off-site source.

Recreational Visitor. The recreational land use scenario evaluated potential doses to an individual who visited the former landfill site for recreational use. It was assumed that the recreational visitor made 20 one-hour visits to the site each year. This scenario represents the most likely land use, given that many closed landfills are converted to park districts and used for a variety of recreational purposes (e.g., golf courses). The exposure pathways evaluated for the recreational visitor included external irradiation and inhalation of outdoor radon. Inhalation of contaminated particulates and ingestion of soil also were evaluated; however, these pathways would not be likely, given that the landfill would be capped with a thick layer of compacted clay, soil, and gravel. Any water used by recreational visitors was assumed to come from an unaffected, off-site supply.

Off-Site Resident. The off-site resident scenario evaluated potential doses resulting from future impacts to the underlying aquifer associated with disposal of NORM waste in the landfill. It was assumed that the resident lived adjacent to the former landfill property and that all of his or her drinking water was retrieved from a residential well located 1,000 ft from the landfill cell containing the NORM waste. The off-site resident was assumed to drink 2 L of water each day, 365 days per year. The only exposure pathway to this receptor would be ingestion of groundwater. Inhalation of radon via volatilization during showering also was considered as a potential pathway of exposure; however, this pathway would not be a major contributor to dose because groundwater transport calculations predicted low Rn-222 concentrations in groundwater (Section 6.1.3.1), exposure time for showering would be minimal, and resultant radon concentrations in air would be diluted via mixing with air in the house. Therefore, potential doses were not calculated for this pathway.

5.3 LANDFILL DESIGN

The landfill modeled in the case study has a disposal area of approximately 75 acres and a total capacity of 9.6 million yd^3. The landfill contains nine disposal cells of varying size. In this assessment, it was assumed that the cell that would receive the NORM waste would have an area of approximately 513,000 ft^2, encompassing about 14% of the total landfill area. When the landfill is completed, it will be approximately 80 ft thick.

Figure 4 illustrates the general landfill design. The landfill cap is composed of Layers 1 through 3. Layer 3, which is placed directly over the landfilled wastes once a cell is full, consists of a 2-ft thick layer of compacted clay, having a hydraulic conductivity of less than 1.8×10^{-1} ft/d. Layer 2, the middle layer of the cap, consists of a 1.5-ft thick gravel layer designed to reduce infiltration into the landfill. The final layer of the cap, Layer 1, is a 1.5-ft thick layer of topsoil designed to support plant growth and thereby slow erosion. Below the cap is Layer 4, the municipal wastes placed in the landfill. At the end of each day, the landfill operator places a soil cover on the compacted waste lifts; these are not shown in Figure 4. The landfilled waste is placed directly over a 3-ft thick lateral drainage layer composed of gravel (Layer 5). The gravel drainage layer is underlain by a 0.39-in. thick flexible membrane liner made of high-density polyethylene (HDPE) (Layer 6), a 0.24-in. thick layer of bentonite (Layer 7), a 0.24-in. thick drainage net made of man-made materials (Layer 8), another 0.39-in thick HDPE liner (Layer 9), and a 3-ft thick compacted clay layer (Layer 10).

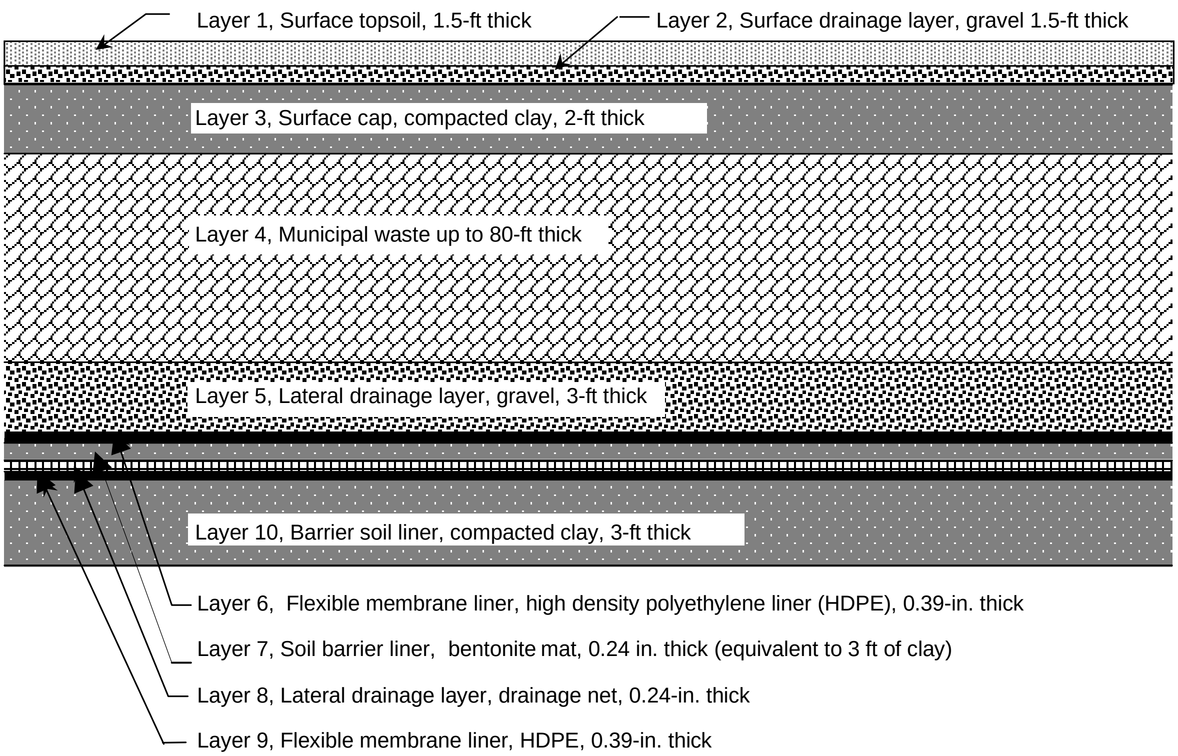

FIGURE 4 Landfill Design

5.4 DETERMINATION OF EXPOSURE POINT CONCENTRATIONS

5.4.1 Source Concentrations

This study evaluated two separate waste streams — radium-bearing NORM wastes and waste containing Pb-210. Because the origins of these waste streams are different and because these wastes would thus be managed and disposed of separately, they are treated independently in this analysis. Dose calculations were performed for each waste stream for the principal radionuclides in the decay series. The term "principal" refers to those radionuclides in the decay series with half-lives of more than one year; these include Ra-226, Pb-210, Ra-228, and thorium-228 (Th-228) for the radium-bearing NORM wastes, and Pb-210 for the wastes containing lead. The chain of decay products of a principal radionuclide (i.e., the associated radionuclides) extending to (but not including) the next principal radionuclide were assumed to be in secular equilibrium[7] with the principal radionuclide. Secular equilibrium also was assumed between Ra-228 and Th-228.

[7] Secular equilibrium refers to the relationship established between a radioactive element that has a long half-life and a decay product that has a much shorter half-life. For example, Ra-226 has a half-life of about 1,600 years. As this element decays and emits radiation, Rn-222, which has a half-life of about 3.8 days, is produced. Over time (after seven progeny half-lives), an equilibrium is established between the concentrations of these two elements (disregarding the mobility of the radon gas) such that the activity of each element is equal.

For most of the receptors and exposure pathways, the source concentration was defined by the case study assumptions. For those receptors exposed to either leachate or groundwater, leachate and groundwater transport calculations (Section 5.4.2) were made to define the source concentrations.

5.4.1.1 Radium-Bearing NORM Wastes

For the case study, it was assumed that 2,000 m^3 of NORM waste having a Ra-226 concentration of 50 pCi/g was disposed of in the nonhazardous landfill. Although Ra-228 is also commonly present in NORM waste, concentrations are usually lower, and with a half-life of 5.8 years, Ra-228 does not present a long-term hazard. Even so, the contribution from Ra-228 was addressed in the analysis, assuming a 3:1 ratio of Ra-226 to Ra-228 (i.e., 50 pCi/g Ra-226 in addition to 12.5 pCi/g Ra-228). Ingrowth of Pb-210, which has a longer half-life (22 years), was assumed for 10 years at the start of analysis.

5.4.1.2 Waste Containing Lead-210

Disposal of lead-containing NORM waste was evaluated on the basis of a limited data set provided by API, describing one company's inventory. Discussions with API indicated that lead waste would be disposed of in 55-gal drums and that disposal in any single landfill would be limited to one truckload per year. One truckload was assumed to contain 96 55-gal drums, equal to 20 m^3. This analysis evaluated disposal in drums as well as the possibility of bulk disposal. Estimates provided in this report are based on a barrel-weighted average activity concentration of 260 pCi/g of Pb-210.

Lead-210 is the only principal radionuclide of interest for this waste stream. Radioactive decay of Pb-210 results in bismuth-210, which has a half-life of about 5 days. Bismuth-210 decays by beta emission to polonium-210, which has a half-life of approximately 140 days. Polonium-210 decays by alpha particle emission to stable lead-206 (Pb-206).

5.4.2 Leachate and Groundwater Transport Calculations

Three separate models were used individually and in tandem to evaluate leachate and groundwater transport. These models are discussed briefly in this section; details of their implementation are provided in Appendix A. Figure 5 illustrates the main inputs required to run each model, the outputs generated by each model, and the interdependencies of the models' computations.

To estimate potential doses to the off-site groundwater receptor, the mobilities of Ra-226 and Pb-210 within the landfill and beyond the landfill's containment system were evaluated. Ra-226 is generally in the form of radium/barium sulfate, which is relatively insoluble, having a solubility limit of 2×10^{-6} g/L. It was assumed that, as the radium was exposed to leachate moving through the landfill, it would dissolve instantly to its solubility limit. In reality, as the

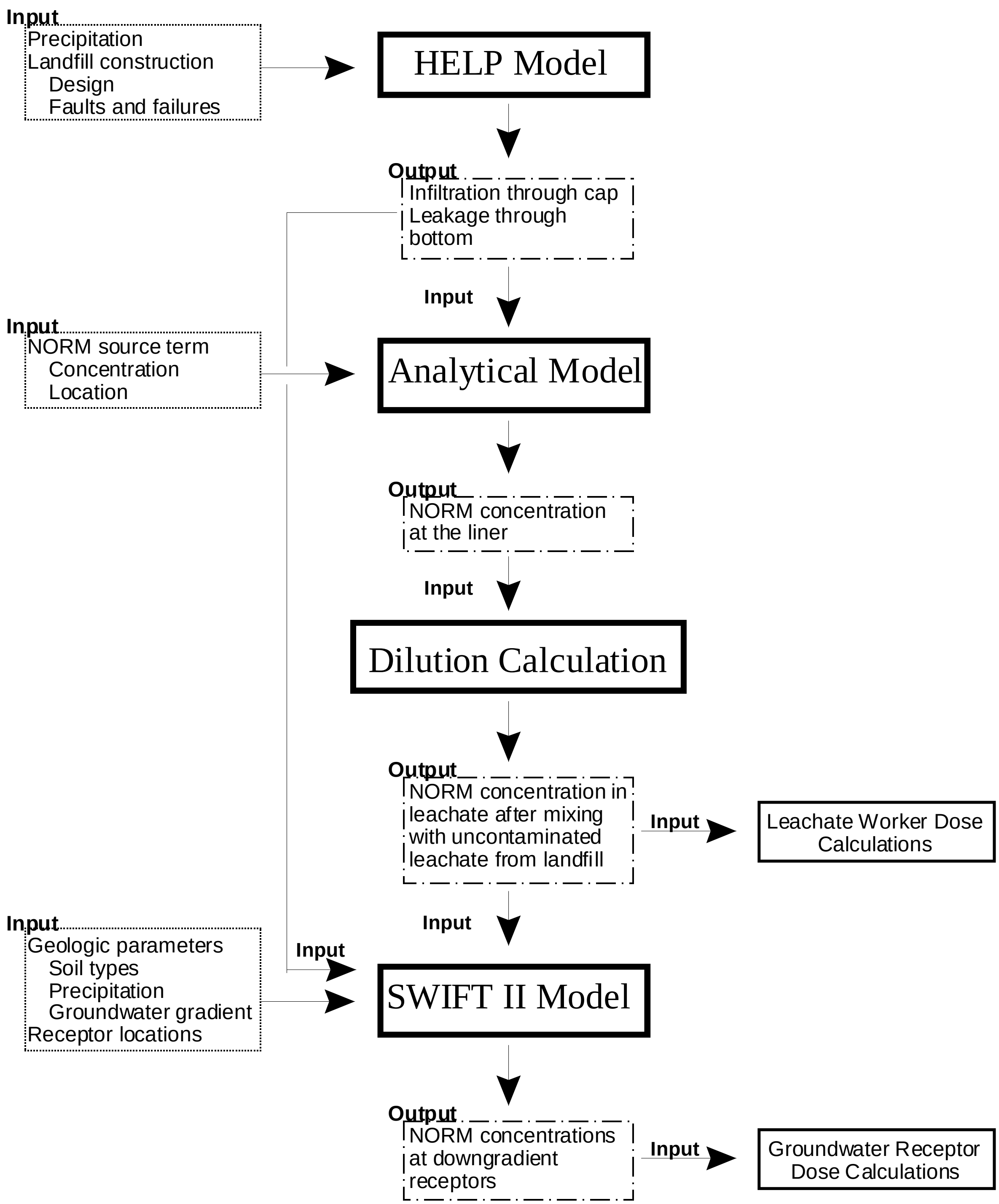

FIGURE 5 Diagram of Models Used to Calculate Leachate and Groundwater Transport

barium and radium dissolve, the solubility of radium is reduced significantly as a result of the common ion effect. However, for the purposes of this study, this effect was neglected. As a result, the radium concentration in solution estimated in this study is much higher than would be expected to occur in the landfill.

Pb-210 is insoluble in its elemental state, which is characteristic of most lead-containing NORM wastes. In addition, the majority of the lead is found on the interior surface of pipes, valves, and other pieces of equipment used in gas production and would not be in direct contact with the leachate in the landfill. Over time, these items would rust and decay, gradually exposing the NORM to the leachate. Because lead is insoluble, even as the NORM was exposed to the leachate, there would be no transport to the collection system or the groundwater beneath the landfill. However, in order to evaluate a worst-case scenario, it was assumed that the lead was in the form of lead sulfate, the most soluble form of lead that might occur in a landfill, with a solubility of 4.25×10^{-3} g/L (Chemical Rubber Company 1968). In addition, it was assumed that leachate passing through the NORM-containing waste instantaneously dissolved the lead to its solubility limit.

The Hydrologic Evaluation of Landfill Performance (HELP) model (Schroeder 1994) was used to conduct a landfill performance study. Specifically, it was used to calculate the amount of fluid that percolates through the surface cover of the landfill and the amount of leachate that could leak through the landfill containment system (e.g., the base liner). Variations in the quality of the landfill liners and various drainage lengths were evaluated to determine how sensitive the results were to these changes.

An analytical model developed by Tomasko (1992) was used to estimate the movement of dissolved Ra-226 and Pb-210 from the original position of the NORM waste in the landfill to the impermeable liner at the bottom of the landfill cell. Because radionuclide transport within the landfill is driven by leachate percolation, this model requires quantification of the amount of fluid that would leak through the landfill surface cover. The HELP model results were used to define this value. Concentrations predicted by the analytical model did not account for the effect of mixing with uncontaminated leachate generated throughout the rest of the landfill. To account for dilution, a calculation was made to estimate the Ra-226 or Pb-210 concentrations at the base of the cell expected to receive the NORM waste and at the base of the entire landfill after mixing.

The SWIFT II model (Reeves et al. 1986) was used to calculate groundwater transport of Ra-226 and its daughter, Rn-222, from beneath the landfill to various groundwater receptor locations. The HELP model was used to estimate the volume of leachate that could be expected to leak from the landfill, while the model developed by Tomasko was used to estimate the NORM concentration in the leachate and the length of time that the leachate would contain NORM. The SWIFT II model was not used for Pb-210 because calculations of Pb-210 movement within the landfill indicated that lead did not reach the liner in appreciable concentrations.

TABLE 2 Methodologies Used to Model NORM Disposal in a Nonhazardous Landfill for Various Scenarios

Receptor	Exposure Pathway	Model
Operational Phase Scenarios		
Driver	External irradiation	TSD-DOSE
Waste-placement operator	External irradiation	TSD-DOSE[a]
	Inhalation of particulates	TSD-DOSE
Leachate worker	External irradiation	TSD-DOSE
Off-site residents	External irradiation	TSD-DOSE
	Inhalation of particulates	TSD-DOSE
	Ingestion of soil and food	TSD-DOSE
Future Use Scenarios		
On-site resident	External irradiation	RESRAD[b]
	Inhalation of particulates and radon	RESRAD
	Ingestion of soil and food	RESRAD
On-site industrial worker	External irradiation	RESRAD
	Inhalation of particulates and radon	RESRAD
Recreational visitor	External irradiation Inhalation of radon	RESRAD
Off-site resident	Groundwater ingestion	HELP model,[c] leachate model,[d] SWIFT II[e]

[a] Pfingston et al. (1998).
[b] Yu et al. (1993).
[c] Schroeder et al. (1994).
[d] Tomasko (1992).
[e] Reeves et al. (1986).

5.5 DOSE ASSESSMENT METHODOLOGIES AND EXPOSURE ASSUMPTIONS

The methodologies used to evaluate doses for each receptor are summarized in Table 2. Doses resulting from waste placement activities (i.e., operational phase scenarios) for the workers, an off-site resident living adjacent to the landfill, and the general public living within a 50-mi radius of the site were evaluated by using the TSD-DOSE computer code, developed jointly by Argonne National Laboratory (ANL) and M.H. Chew and Associates, Inc. for the DOE’s Office of Technical Services (Pfingston et al. 1998). This model can be used to calculate radiological doses to treatment, storage, and disposal (TSD) facility workers and the surrounding public as a result of processing and disposing of waste that is slightly contaminated with radionuclides. It can evaluate potential doses associated with seven different activities that may be undertaken at a TSD facility, including landfill operation.

TSD-DOSE was used to estimate potential doses to (1) the driver at the landfill, resulting from external irradiation; (2) the waste-placement operator, resulting from external irradiation and inhalation of contaminated particulates; (3) the leachate worker, resulting from external irradiation; and (4) the general public, resulting from inhalation of contaminated particulates, external irradiation, incidental ingestion of contaminated particulates, and ingestion of contaminated foodstuff. Summaries of the exposure parameters used to model the worker and off-site resident scenarios are provided in Appendix B, Tables B.1 and B.2, respectively. For the worker scenarios, exposure times were based on the volume of waste to be disposed of. External doses were modeled at a distance of 2 and 3 ft from the source for the driver and leachate worker, respectively. For the waste-placement operator, a distance of 5 ft was assumed for inspection activities and a distance of 10 ft was assumed for disposal activities.

For the waste-placement operator and off-site resident scenarios, radioactive releases from the landfill operations were assumed to occur primarily through the generation of fugitive dust as the NORM-impacted waste was dumped out of the truck. Radioactive material also could become airborne as a result of the mixing, leveling, and rolling of the waste deposited on the landfill; however, the work area within the subject landfill often is sprayed with water to minimize the generation of airborne dust. The fraction of respirable particulates (i.e., particulates that are less than 10 μm in size) released during dumping operations was estimated to be 3×10^{-7} on the basis of EPA methodology (EPA 1989). Releases of radionuclides were calculated by multiplying this fraction by the Ra-226 and Pb-210 concentrations of the waste. The TSD-DOSE code assumes that air contamination disperses as a Gaussian plume (i.e., spatially distributed using a normal distribution*)*.

The RESRAD computer code, Version 5.782 (Yu et al. 1993), was used to calculate potential doses to an on-site resident, industrial worker, and recreational visitor from all applicable pathways. The RESRAD code is a pathway analysis code that implements the methodology prescribed in DOE Order 5400.5 (DOE 1990) for determining residual radioactive soil guidelines. The exposure pathways considered in this analysis included external irradiation; inhalation of resuspended dust and radon; ingestion of crops, milk, and meat grown on the property; incidental ingestion of contaminated soil; and ingestion of fish from a nearby pond. Doses were projected over a period of 1,000 years. The source was adjusted over time to account for radioactive decay and ingrowth, leaching, erosion, and mixing. The various parameters used in the RESRAD code for this analysis are listed in Appendix B, Table B.3.

Potential radiological doses to the off-site groundwater receptor were calculated from the estimated radionuclide concentrations projected by the leachate and groundwater transport models (Section 5.4.2), exposure parameters recommended by the EPA for maximum residential exposures (EPA 1991), and the radionuclide-specific ingestion dose conversion factor (DCF).

TSD-DOSE and RESRAD both use the most conservative DCF as the default in cases where more than one DCF is defined for a specific radionuclide and exposure pathway. Ingestion DCFs are defined on the basis of the fraction of ingested material that will be absorbed from the gastrointestinal tract into the body fluids. Inhalation DCFs are defined on the basis of the radionuclide's lung retention time (i.e., the rate at which deposited material is removed from the region of the respiratory tract). The ingestion and inhalation DCFs are highly dependent on the radionuclide's chemical form. For this analysis, the most conservative DCFs from Federal Guidance Report No. 11 (EPA 1988b) were used. These values may overestimate the doses because the solubility for radium scales is extremely low.

5.6 SENSITIVITY ANALYSES

Sensitivity analyses were conducted for several input parameters that were considered likely to have an effect on potential doses. These analyses were performed only for those scenarios likely to be affected by a given parameter.

For the on-site resident and industrial worker scenarios, sensitivity analyses were conducted to determine the effects of the depth of the NORM waste layer from the surface of the landfill, radon emanation coefficient, thickness of the NORM waste layer, and the source concentration on potential doses. In addition, the effect of breaching the landfill cover during home construction (i.e., building a home with a basement) also was investigated for the on-site resident. For the recreational visitor, a sensitivity analysis on only the depth of the NORM waste layer was conducted.

For the groundwater receptor scenario, sensitivity analyses were conducted within the leachate and groundwater transport models. Parameters examined in these analyses included depth of the NORM waste layer, thickness and areal extent of the waste layer, percolation velocity, distance to the groundwater receptor, and depth of the groundwater receptor below the water table.

6 RESULTS

6.1 LEACHATE AND GROUNDWATER TRANSPORT CALCULATIONS

6.1.1 HELP Model Simulations and Results

Landfill performance was measured in terms of the volume of leachate that potentially could leak from the base of the landfill each year. Table 3 summarizes the results of the nine simulations performed in the HELP modeling study. The base case, Run 1, represents a conservative estimate of landfill performance based on the design of the subject landfill (Figure 4). The output from Run 1 was compared with the outputs of subsequent runs to assess the sensitivity of landfill performance to various input parameters. These subsequent runs used the same basic landfill design criteria used in Run 1 but assumed different values to represent the quality of the geomembrane liners located at the base of the landfill, hydraulic conductivity of the clay cap layer, and length of the gravel drainage layers.

A detailed discussion of the assumptions used in the HELP model and the results of this study are provided in Appendix A. Predictably, the results indicate that potential leakage through the bottom of the landfill increased with decreasing quality or absence of the geomembrane liner and increasing hydraulic conductivity of the clay cap. In addition, leakage increased with increasing length of the gravel drainage layer, which resulted from the increased time that the leachate is in contact with the liner and increased head across the geomembrane liner.

TABLE 3 HELP Model Descriptions and Results

Run Number and Description	Predicted Infiltration through Cap (ft/yr)	Predicted Leakage through Bottom (ft^3/yr/acre)
Run 1: Base case	0.56	61
Run 2: Geomembrane liners of good quality (1 pinhole/acre)	0.56	2.4
Run 3: Geomembrane liners of bad quality (50 pinholes/acre)	0.56	448
Run 4: Drainage length 10 ft	0.53	1.4
Run 5: Drainage length 50 ft	0.51	16
Run 6: Drainage length 150 ft	0.55	126
Run 7: Cap hydraulic conductivity 1.5 ft/d	0.58	61
Run 8: Cap hydraulic conductivity 7.7×10^{-4} ft/d	0.40	31
Run 9: No geomembrane liners	0.56	27,500

6.1.2 Analytic Model Results for Leachate Transport

The vertical transport of radionuclides through the municipal waste layer was modeled by using an analytical model developed by Tomasko (1992) to simulate the movement of the NORM radionuclides through the interior of the landfill. The assumptions used to implement this model were chosen so that the model would predict higher NORM concentrations at the liner than would actually occur. The results of the Ra-226 transport modeling are presented in Table 4 and discussed in Section 6.1.2.1. Results of the Pb-210 calculations are detailed in Section 6.1.2.2. A detailed discussion of the assumptions used in this model and the results for both Ra-226 and Pb-210 are provided in Appendix A.

6.1.2.1 Leachate Transport of Ra-226

The analytical model was used to estimate Ra-226 concentrations in the leachate at three different locations within the landfill: (1) at the liner immediately below the NORM waste layer; (2) at the base of the cell containing the NORM; and (3) within the entire landfill, assuming mixing of leachate from all of the landfill cells is mixed. Table 4 presents the maximum Ra-226 concentrations in the leachate at each of the locations for three different waste thicknesses, assuming a depth of 8 ft to the gravel drainage layer underlaying the municipal wastes. Predictably, the Ra-226 concentration below the NORM waste layer increased with increasing thickness of the waste layer. As the leachate was mixed with leachate generated from larger areas within the landfill, the Ra-226 concentration decreased.

Pb-210 concentrations resulting from the decay of Ra-226 were neglected because of the relatively short half life of Pb-210 (22 years) and high retardation factor. Pb-210 essentially decays to stable Pb-208 before it has moved any appreciable distance. A more detailed discussion is included in Section 6.1.2.2 and Appendix A.

6.1.2.2 Leachate Transport of Pb-210

The analytical model was used to evaluate transport of Pb-210 in the landfill vertically from the location of the NORM waste layer to the liner. The model assumed a source concentration equal to the solubility of lead-sulfate, 4.25×10^{-3} g/L, and a distribution coefficient

TABLE 4 Results of the Leachate Transport Modeling for Ra-226

	Maximum Ra-226 Concentration (pCi/L)		
NORM Waste Layer Thickness	**below NORM Waste Layer**	**in Leachate from NORM Cell**	**in Leachate from Entire Landfill**
1 ft	92	2.1×10^{-1}	2.8×10^{-2}
4 ft	360	2.2×10^{-1}	2.9×10^{-2}
8 ft	740	2.2×10^{-1}	3.0×10^{-2}

(k_d) of 270 mL/g. This results in a retardation factor of approximately 1,135 when a bulk density of 1.8 and an effective porosity of 0.42 are used (Domenico and Schwartz 1990). When an infiltration rate through the landfill cap of 0.55 ft/yr is assumed, movement of Pb-210 through the landfill would be approximately 0.001 ft/yr. Thus, in one half-life (22 years), lead would move only 0.02 ft. By the time the lead moved any appreciable distance from its disposal point, it would have decayed completely to stable Pb-206. The analytical model estimated lead concentrations at the liner on the order of 1×10^{-21} pCi/L. Because of the low calculated concentrations at the liner, the short half-life, and the high retardation of lead, Pb-210 concentrations at the liner were assumed to be zero.

6.1.3 Groundwater Transport Modeling Results

6.1.3.1 Groundwater Transport of Ra-226

The SWIFT II model (Reeves et al. 1986) was used to evaluate groundwater transport of Ra-226 and its progeny, Rn-222, from the point of release below the landfill liner to a receptor located 1,000 ft downgradient of the cell containing the NORM wastes at a depth of 5 ft below the base of the landfill. The results of the SWIFT II modeling runs are shown in Table 5; a detailed discussion of the assumptions used in each run and the results are presented in Appendix A.

The base-case run of the SWIFT II model used input parameters that were chosen to produce estimates of radionuclide concentrations that would be significantly higher than any that realistically would be expected in the subsurface. This approach was taken to produce risk estimates that would be high and bound the problem. This was done due to address the uncertainty in some of the model parameters. The SWIFT II base-case evaluation used input parameters representative of the landfill design shown in Figure 4; however, it was assumed that there were no geomembrane liners present at the base of the landfill. Under this assumption, the leachate leakage rate through the base of the landfill was set at 27,500 ft^3/yr/acre (per HELP

TABLE 5 Maximum NORM Concentrations Predicted by Groundwater Transport Analysis

SWIFT II Run	Maximum Ra-226 Concentration (pCi/L)	Maximum Rn-222 Concentration (pCi/L)	Time to Maximum Concentration (yr)
SWIFT II base case	3.3×10^{-4}	3.9×10^{-8}	109
High conductivity	3.9×10^{-5}	4.5×10^{-9}	100
Low conductivity	2.9×10^{-4}	1.2×10^{-8}	394
High gradient	7.5×10^{-5}	9.0×10^{-9}	100
Low gradient	3.3×10^{-4}	4.2×10^{-8}	427
Increased recharge	1.1×10^{-4}	1.4×10^{-8}	125

model Run 9 [Table 3]). In addition, it was assumed that groundwater was in immediate contact with the landfill liner, when actually the water table at the subject landfill is located at depths ranging from 10 to 30 ft below the liner, and that the movement of Ra-226 and Rn-222 through the subsurface would not be retarded.

The values used in the SWIFT II base case representing the concentration of radium released from the landfill and the duration of the release were derived from the analytical modeling described in Section 6.1.2.1. The Ra-226 concentration in the leachate leaking from the base of the landfill was estimated to be 2.2×10^{-1} pCi/L, on the basis of results generated by the analytical model, which assumed a percolation rate of 0.55 ft/yr, a depth to drainage layer of 8 ft, and a waste layer thickness of 8 ft (Table 4).

As shown in Table 5, under the base-case assumptions, the Ra-226 concentration at the receptor was only 3.3×10^{-4} pCi/L. The estimated Rn-222 concentration also was very low, on the order of only 3.9×10^{-8} pCi/L. Additional simulations were run using SWIFT II to analyze the sensitivity of calculated Ra-226 and Rn-222 concentrations to various input parameters, including hydraulic conductivity, groundwater gradient, aquifer recharge rate, and depth of the groundwater receptor.[8] As shown in Table 5, the Ra-226 and Rn-222 concentrations were affected by changes in hydraulic conductivity, groundwater gradient, and aquifer recharge rates; overall, however, the results of the additional simulations were also very low and within approximately one order of magnitude of the base-case scenario.

Figure 6 presents the results of the sensitivity analyses evaluating the effect of the depth of the well used by the groundwater receptor on the base-case results. This figure plots the Ra-226 concentrations at three depths versus time, as predicted by the base-case run for the downgradient receptor. These three depths correlate to depth below the base of the landfill, which is coincident with depth below the water table. Increasing the depth of the groundwater receptor to 35 and 175 ft below the landfill decreased the maximum calculated concentrations of Ra-266 and Rn-222 by one to two orders of magnitude and had little effect on the timing of the maximum concentrations. Because the water table was assumed to be in contact with the base of the landfill, the decreases in Ra-226 and Rn-222 concentrations were a result of dispersion and decay, not retardation through the vadose zone.

An additional set of SWIFT II runs was made to evaluate the effect of adding a geomembrane liner to the landfill design on estimated Ra-226 concentrations. In these runs, it was assumed that the geomembrane liners were of poor quality, having approximately 10 flaws/acre. Under this scenario, which is analogous to HELP model Run 1, the leachate leakage rate was set at 61 ft^3/yr/acre as opposed to 27,500 ft^3/yr/acre. On the basis of these modeling results (discussed in detail in Appendix A), the presence of the geomembrane liners reduced the estimated Ra-226 concentration by about five orders of magnitude to 5.7×10^{-9} pCi/L. Assuming the geomembrane liners are in place but poorly constructed is more realistic than the extremely conservative assumptions used for the base-case conditions. In addition, if the model also had assumed that the water table was actually 10 to 30 ft below the

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8 Groundwater gradient is a unitless number that measures the pressure drop across a distance. A gradient of 0.01 indicates a 1% change in head over a 1-ft distance. High gradients cause increased flow.

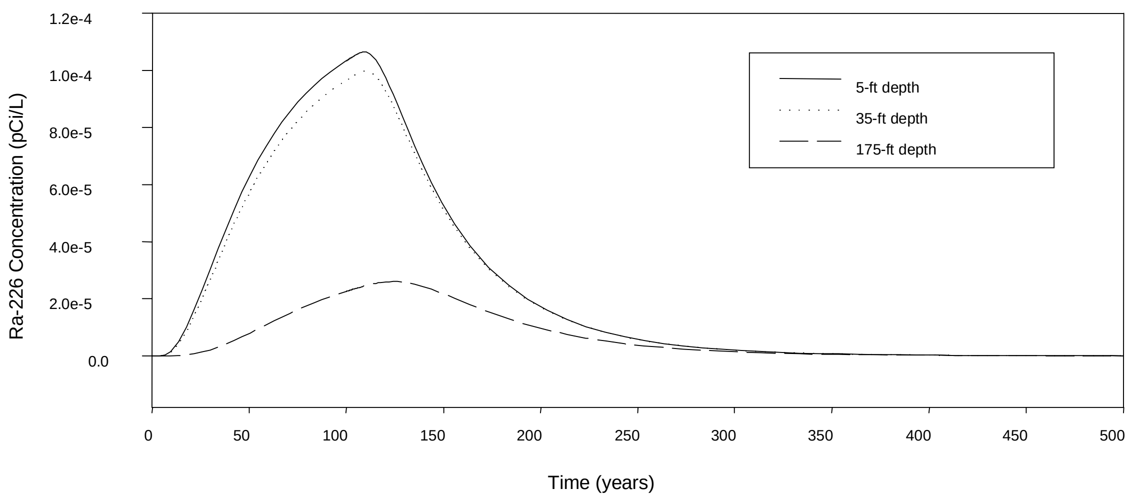

FIGURE 6 Comparison of Ra-226 Concentrations Predicted at Varying Depths for Receptor Located 1,000 Feet Downgradient of the Landfill

base of the landfill, the radionuclide concentrations entering the groundwater could be reduced by an additional three orders of magnitude or more, as a result of the retardation effect of the unsaturated zone.

6.1.3.2 Groundwater Transport of Pb-210

No evaluation of groundwater transport of Pb-210 was made because the estimated Pb-210 concentrations in leachate at the landfill liner were estimated to be zero (Section 6.1.2.2). In addition, because of the relatively short half-life of Pb-210 (approximately 22 years) and its high retardation factor of approximately 1,000, Pb-210 decays to stable Pb-206 before it moves any appreciable distance in subsurface aquifers.

6.2 CALCULATED DOSES AND HEALTH RISKS FOR OPERATIONAL PHASE SCENARIOS

Radiological doses resulting from waste placement activities were calculated for on-site workers (driver, waste-placement operator, and leachate worker) and off-site residents. Corresponding health risks were calculated by using risk factors identified by the ICRP (see Section 5.1). Descriptions of these scenarios are provided in Section 5.2.1. The results for the radium and lead waste streams are discussed separately, since the two waste streams are generated by different processes and most likely would not be co-located in the same landfill.

6.2.1 Radium-Bearing NORM Wastes

6.2.1.1 Landfill Operators

Radiological doses and health risks associated with disposal of NORM-impacted wastes were quantified for a driver and a waste-placement operator. External irradiation was evaluated for both receptors; in addition, inhalation of contaminated particulates also was evaluated for the waste-placement operator. Calculations were based on the disposal of 2,000 m^3 of NORM-impacted waste having a concentration of 50 pCi/g. The total dose estimated for the driver was 0.3 mrem. The total dose for the waste-placement operator was estimated to be 1.7 mrem, more than 99% of which was from external irradiation. The corresponding increased risks of the workers developing a fatal cancer were estimated to be 1×10^{-7} for the driver and 7×10^{-7} for the operator, respectively (i.e., either one or seven excess deaths in a population of ten million similarly exposed persons).

6.2.1.2 Leachate Worker

An annual dose (in mrem per year) was estimated for the leachate worker because this task would be ongoing throughout the life of the landfill. On the basis of calculations made for Ra-226 concentrations in the landfill leachate (Section 6.1.2.1), an upper-bound leachate concentration of 0.22 pCi/L of Ra-226 was estimated on the basis of the assumption that the NORM waste layer was 8 ft thick and was located 8 ft above the gravel drainage layer. To be conservative, it was assumed that the leachate was generated from the NORM cell only and was not diluted with the leachate from the rest of the landfill. The maximum annual dose to the leachate worker was estimated to be 2×10^{-4} mrem/yr. The increased risk of developing a fatal cancer was estimated to be 8×10^{-11}.

6.2.1.3 Off-Site Residents

Radiological doses and estimated health risks were calculated for off-site residents to evaluate potential impacts from any air emissions released during waste placement. Doses were estimated for a resident living adjacent to the landfill (i.e., the maximally exposed individual) and the collective population living within a 50-mi radius around the landfill. The estimated dose to the maximally exposed individual was 6.6×10^{-4} mrem, which corresponds to a risk of 3×10^{-10}. The population dose was estimated to be 2.7×10^{-5} person-rem; the health risk was estimated to be 1×10^{-8}.

6.2.2 Wastes Containing Lead-210

6.2.2.1 Landfill Operators

Doses and health risks associated with the disposal of wastes containing Pb-210 were quantified for the driver and waste-placement receptors. Doses were evaluated for disposal in 55-gal containers, and it was assumed that the containers were not opened at any time during the process. Calculations were based on the disposal of one truckload containing 96 barrels of lead-impacted waste having a concentration of 260 pCi/g. Because the driver would be shielded from the low-energy gamma radiation emanating from the Pb-210 progeny, the dose from external irradiation would be zero. The dose estimated for the waste-placement operator was 1.3×10^{-5} mrem, and the corresponding risk estimate was 5×10^{-12}.

Although the Pb-210 waste would most likely be disposed of in containers, an evaluation also was conducted for bulk disposal in order to evaluate exposures from potential air emissions. Calculations were based on disposal of 20 m^3 (i.e., equivalent to 96 55-gal drums) having a concentration of 260 pCi/g. The estimated dose to the driver was zero (the same as for the containerized disposal). For the waste-placement operator, the dose was estimated to be 2.4×10^{-6} mrem, 99% of which was from external irradiation. The risk of the worker developing a fatal cancer was estimated to be 9×10^{-13}.

6.2.2.2 Leachate Worker

As discussed in Section 6.1.2.2, transport of lead in the landfill leachate would not occur because of the high retardation factor and short half-life of Pb-210. Therefore, the estimated dose to the leachate worker was zero.

6.2.2.3 Off-Site Residents

Radiological doses to the off-site residents were evaluated only for bulk disposal because there would be no air emissions released from containerized waste. Doses were evaluated for a resident living adjacent to the landfill and the collective population. The estimated dose to the resident living adjacent to the landfill was 3.3×10^{-6} mrem, which corresponds to a risk of 2×10^{-12}. The population dose was estimated to be 1.3×10^{-7} person-rem, which corresponds to a risk of 7×10^{-11}.

6.3 CALCULATED DOSES AND HEALTH RISKS FOR FUTURE USE SCENARIOS

Radiological doses resulting from future use of the landfill property or future use of groundwater underlying the landfill were calculated for four receptors: an on-site resident, on-site industrial worker, recreational visitor, and off-site resident consuming groundwater.

Corresponding health risks were calculated by using risk factors identified in ICRP Publication 60 (ICRP 1991). Descriptions of these scenarios are provided in Section 5.2.2.

6.3.1 Radium-Bearing NORM Wastes

6.3.1.1 On-Site Resident

Radiological doses and resultant health risks were evaluated for a hypothetical future resident who constructed a house directly on top of the landfill. The house was assumed to be constructed on a slab. Sensitivity analyses were conducted to determine the effect on potential doses of depth to the NORM waste layer from the surface of the landfill, radon emanation coefficient, thickness of the NORM waste layer, source concentration, and breach of the landfill cap during home construction. The results of the sensitivity analyses are discussed in Section 6.4.

A base-case dose calculation was made for the on-site resident assuming the NORM waste layer was 8 ft thick and that it was located 15 ft below the surface of the closed landfill. The 15 ft of overburden was composed of, from top to bottom, a 6-ft thick layer of clean cover material (the landfill cap), an 8-ft thick layer of municipal waste, and a 1-ft thick layer of clean soil (Figure 7). As discussed in Section 5.3, the 6-ft thick layer of clean cover material was assumed to consist of, from top to bottom, 1.5 ft of top soil, 1.5 ft of gravel, and 3 ft of compacted clay.

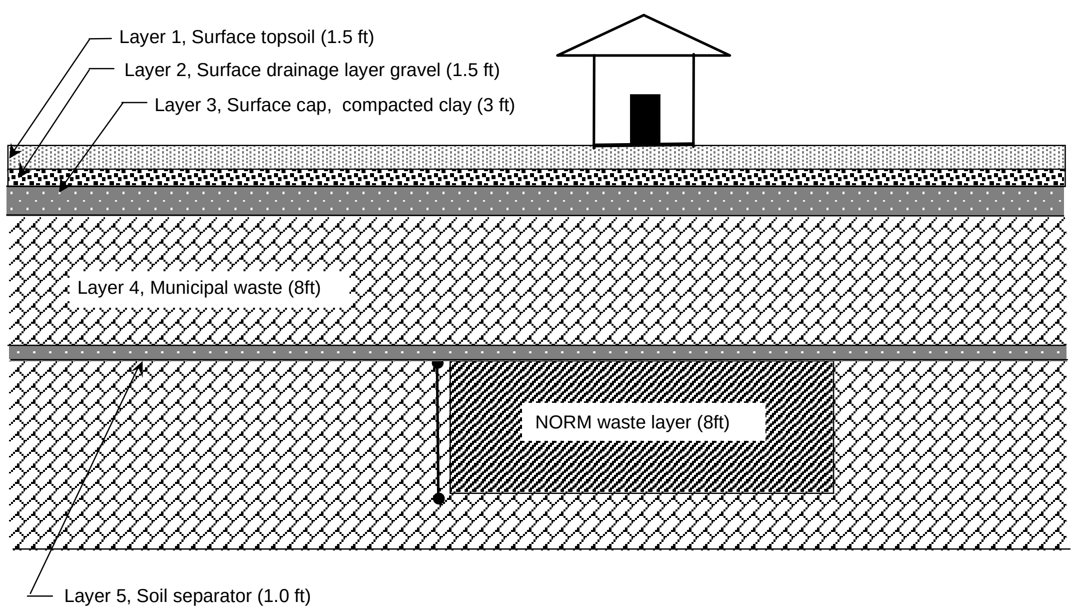

FIGURE 7 Schematic Diagram of Base-Case Assumptions Regarding Placement of NORM Waste within the Landfill

The maximum dose estimated for the base case from all pathways was 7.4 mrem/yr, which corresponds to an increased risk of 4×10^{-6}. This estimated dose resulted exclusively from inhalation of indoor radon. The other pathways do not contribute any dose because the cover layer of soil is so thick. The maximum dose would occur at the time of placement and decrease over time as a result of radioactive decay. The dose from inhalation of radon estimated 1,000 years after placement was 3.1 mrem/yr.

6.3.1.2 On-Site Industrial Worker

Radiological doses and resultant health risks were evaluated for a receptor who worked in a building constructed directly on top of the landfill. The individual was assumed to spend a total of 2,000 hours on site; 75% of that time was assumed to be spent indoors. It was assumed that the building was constructed on a slab. As was done for the on-site residential scenario, a base-case dose calculation was made for the on-site industrial worker assuming the NORM waste layer was 8 ft thick and that it was located 15 ft below the surface of the closed landfill (Figure 7). Sensitivity analyses were conducted to determine the effect of depth to the NORM waste layer, radon emanation coefficient, thickness of the NORM waste layer, and source concentration on potential doses. The results of the sensitivity analyses are discussed in Section 6.4.

The maximum dose estimated for the on-site industrial worker for the base case was 2.2 mrem/yr, which corresponds to an increased risk of 1×10^{-6}. The primary pathway of exposure was inhalation of indoor radon.

6.3.1.3 Recreational Visitor

Potential radiological doses and resultant health risks were evaluated for a future recreational visitor. The visitor was assumed to visit the site 20 times a year. A sensitivity analysis was conducted to determine the effect of depth of the NORM waste layer on potential doses. The results of the sensitivity analyses are discussed in Section 6.4.

Potential doses to a recreational visitor spending time at the landfill after closure were estimated to be very low. A maximum dose of 1.2×10^{-7} mrem/yr was estimated for the base case; all of the dose was from inhalation of radon. This dose corresponds to a health risk of 6×10^{-14}. The thick layer of cover material attenuated almost all of the gamma radiation emanating from the waste layer. The maximum dose occurred at the time of placement and decreased over time due to radioactive decay.

6.3.1.4 Off-Site Resident

The off-site resident scenario was analyzed to evaluate potential doses resulting from future impacts to the underlying aquifer associated with disposal of NORM-impacted waste in the landfill. The only exposure pathway to this receptor was ingestion of groundwater; the receptor was located 1,000 ft downgradient of the landfill at a depth of 5 ft below the base of the landfill. The maximum dose from the groundwater ingestion pathway was 3.2×10^{-4} mrem/yr, which corresponds to a health risk of 2×10^{-10}. This dose was calculated on the basis of a Ra-226 concentration in groundwater of 3.3×10^{-4} pCi/L that was derived from the base-case scenario evaluated in the groundwater transport modeling (Section 6.1.3.1). Under this scenario, it was assumed that the NORM waste layer was 8 ft thick, that it was located 8 ft above the drainage layer, and that there were no geomembrane liners present at the base of the landfill.

6.3.2 Waste Containing Lead-210

Radiological doses resulting from future use of the landfill property were evaluated for disposal of waste containing Pb-210. Evaluations were performed for an on-site resident, on-site industrial worker, recreational worker, and off-site resident consuming groundwater. For all scenarios, the estimated dose was zero.

6.4 SENSITIVITY ANALYSES

Sensitivity analyses for disposal of radium-bearing NORM wastes were conducted on several input parameters for the on-site resident, on-site industrial worker, and recreational visitor scenarios. Only those parameters related to the radon pathway were analyzed because this was the only pathway contributing significantly to dose. These parameters included depth of the NORM waste layer below the landfill surface, radon emanation coefficient, area and thickness of the NORM waste layer, and source concentration. In addition, breach of the landfill cap in home construction was analyzed for the residential scenario. The results of the sensitivity analyses indicated that all of these parameters, except the areal extent of the NORM waste layer, had an impact on estimated doses. Sensitivity analyses for disposal of wastes containing Pb-210 were not performed, with the exception of the parameter defining the depth of the waste.

6.4.1 Depth of Waste Layer

Varying the depth to the NORM waste layer had a significant effect on estimated doses for the radium-bearing NORM wastes. Figure 8 compares the results of the sensitivity analyses on the depth of the NORM waste layer below the landfill surface to the base-case results for the on-site resident scenario. In the base case, it was assumed that the NORM waste layer was located 15 ft below the surface of the closed landfill (Figure 7). In the sensitivity analysis, it was assumed that the NORM waste layer was placed directly on top of the municipal waste layer and was covered only by the 6-ft thick layer of clean cover material. A second case was also analyzed in which it was assumed that the NORM waste layer was placed directly on the bottom of the landfill liner. On the basis of the assumption that the depth to the NORM waste layer was only 6 ft, the estimated peak dose increased from 7.4 to 125 mrem/yr. The corresponding risk increased by one order of magnitude to 6×10^{-5}. For the case where the depth of the NORM layer was increased to 72 ft, the resultant dose was zero. Decreasing the depth of the NORM layer to only 6 ft below the ground surface did not have any effect on the resultant dose

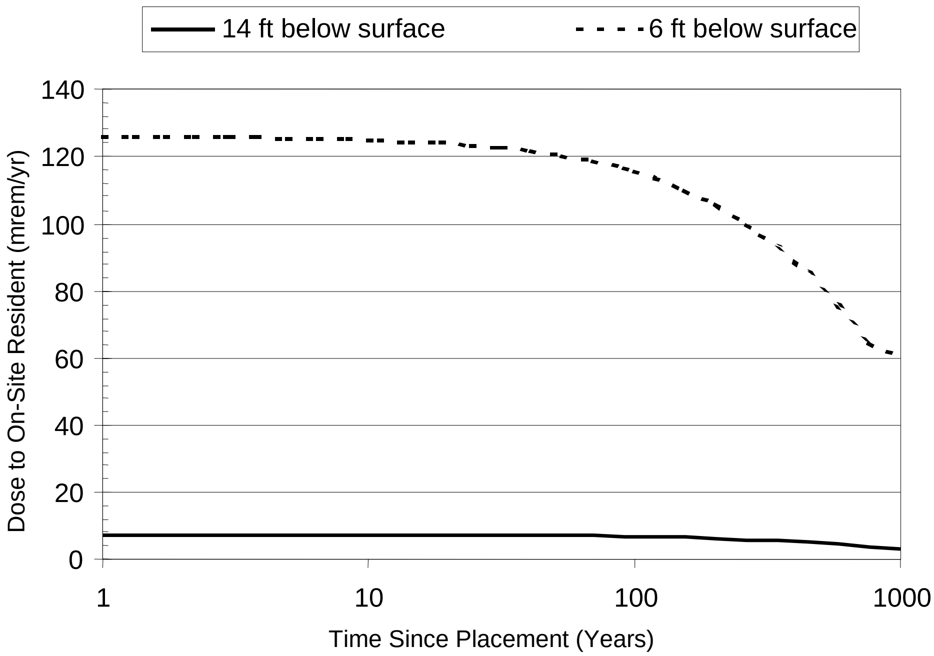

FIGURE 8 Sensitivity of Dose Calculations to Depth of NORM Waste Layer below Landfill Surface

estimated for disposal of the waste containing Pb-210. Because Pb-210 decay progeny produce very weak gamma radiation, the resultant dose was still zero.

Results of the sensitivity analyses on this parameter for the on-site industrial worker and recreational visitor demonstrated a similar relationship between dose and depth of the waste layer. Decreasing the depth of the waste layer to 6 ft increased the dose to the on-site industrial worker and recreational visitor by a factor of 19 and 17, respectively.

6.4.2 Radon Emanation Coefficient

Figure 9 compares the results of the sensitivity analysis of the radon emanation coefficient to the base-case results for the on-site resident. In the base case, a radon emanation coefficient of 0.04 was used. This is equal to the average value measured for petroleum scale by Rood and Kendrick (1996). When the radon emanation coefficient was doubled to 0.08, consistent with the higher average value measured by White and Rood (1998), the resultant peak dose increased from 7.4 mrem/yr to approximately 15 mrem/yr. Decreasing the radon emanation coefficient to 0.02 resulted in a lower peak dose of 3.7 mrem/yr. Results of the sensitivity analyses on this parameter for the on-site industrial worker demonstrated the same relationship between dose and radon emanation coefficient.

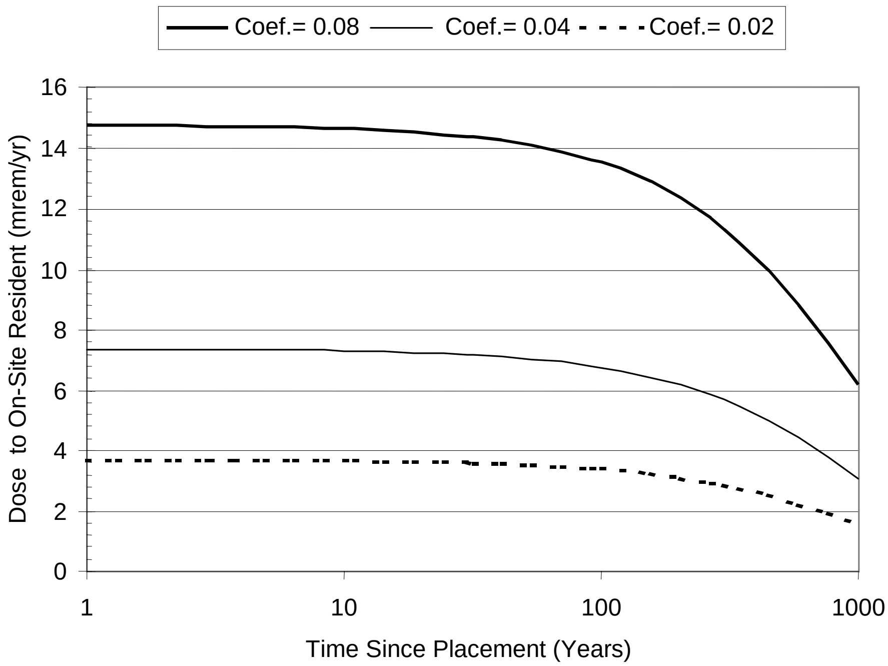

FIGURE 9 Sensitivity of Dose Calculations to Rn-222 Emanation Coefficient

6.4.3 Thickness of Waste Layer

Figure 10 compares the results of the sensitivity analysis of the thickness of the NORM waste layer to the base-case results. In this analysis, the waste thickness was decreased from 8 ft to 4 ft, and necessary adjustments were made to the areal extent to maintain a constant volume of contamination. Halving the waste thickness caused the estimated doses to decrease by approximately 8%. Results of the sensitivity analyses on this parameter for the on-site industrial worker demonstrated the same relationship.

6.4.4 Source Concentration

As illustrated in Figure 11, which presents the base-case dose for the on-site resident as a function of the Ra-226 concentration for the peak dose, dose increases linearly with concentration. For the two depths modeled in the sensitivity analysis on waste layer depth (i.e., 14 and 6 ft), Ra-226 concentrations that would result in a dose of 100 mrem/yr were calculated for each of the future use scenarios. Although this dose is the currently accepted limit for radiological doses to the general public from all sources (Section 2.3.2), it was chosen to define an upper bound on the allowable source concentration. The results of this analysis are presented in Table 6.

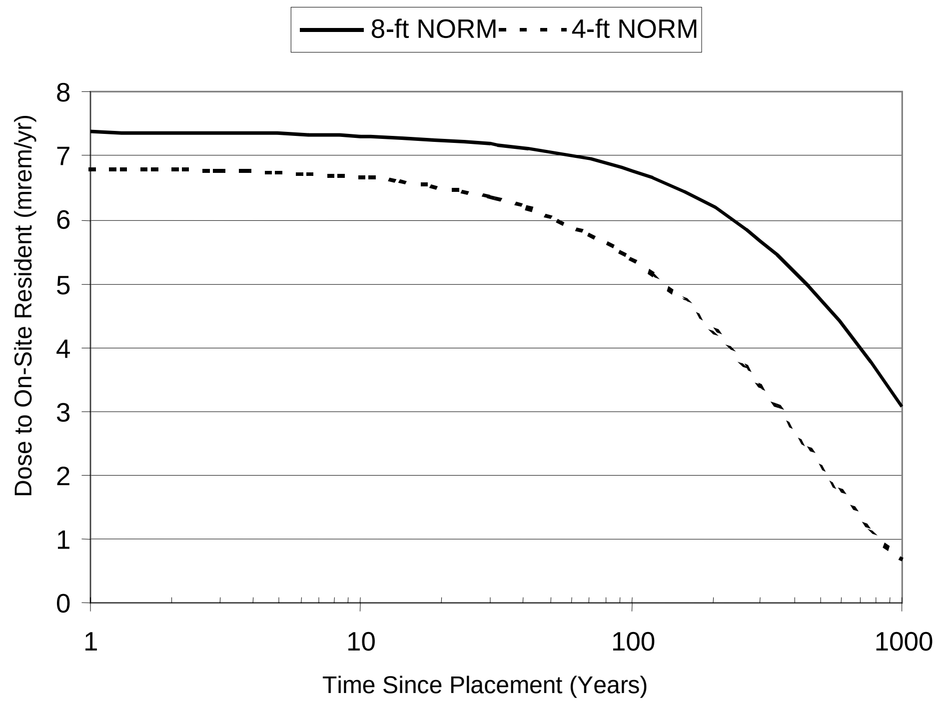

FIGURE 10 Sensitivity of Dose Calculations to Thickness of NORM Waste Layer

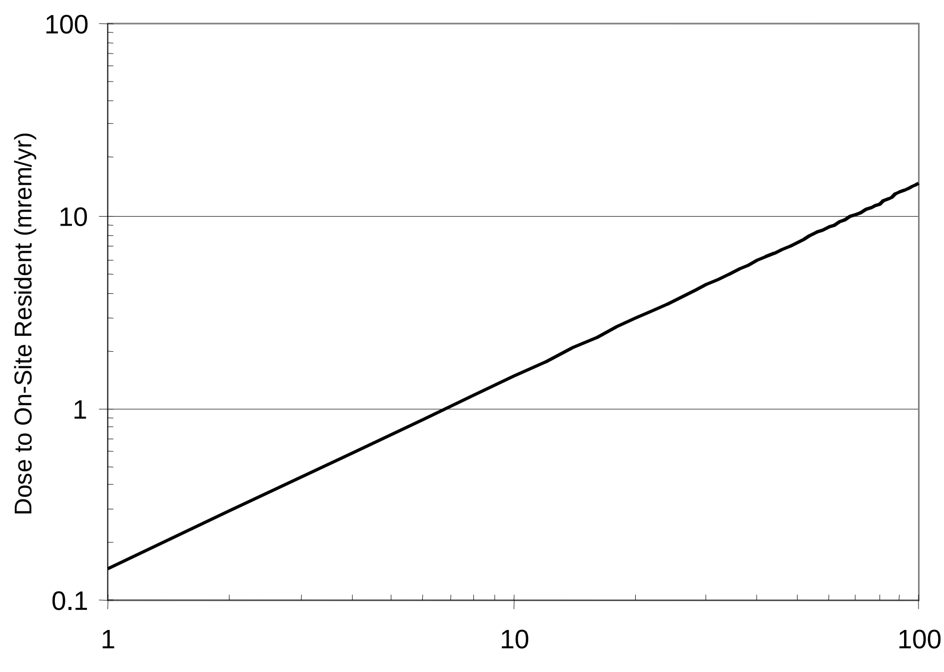

FIGURE 11 Correlation between Calculated Dose and Ra-226 Concentration of Waste

TABLE 6 Equivalent Ra-226 Concentrations Associated with the 100-mrem/yr Dose Limit

Scenario	Equivalent Ra-226 Concentration (pCi/g)	
	Waste Layer 15 ft Deep	Waste Layer 6 ft Deep
On-site resident	680	40
On-site industrial worker	2,300	120
Recreational visitor	4.2×10^{10}	4.7×10^{7}
Off-site resident	4.5×10^{7}	4.5×10^{7}

6.4.5 Breach of Landfill Cap in Home Construction

Breach of the landfill cap was evaluated for the residential scenario assuming the resident lived in a home having a basement that extended 8 ft below the ground surface. This scenario was included in the assessment as a worst-case scenario even though institutional controls should be in place to protect the landfill cover from being breached (Section 2.1). On the basis of the assumption that the home was constructed with a basement, the estimated peak dose increased from 7.4 to 63 mrem/yr. The large increase in dose was a result of the increased radon concentrations inside the home because the distance to the NORM waste layer was effectively decreased.

7 CONCLUSIONS AND RECOMMENDATIONS

7.1 CONCLUSIONS

In this study, existing federal and state laws and regulations were reviewed to determine whether disposing of NORM-impacted wastes in nonhazardous landfills is a feasible disposal option from a regulatory perspective. In addition, this study evaluated potential radiological doses and resultant health risks to workers and the general public resulting from landfill disposal of NORM-impacted wastes. On the basis of the analyses presented in this report, a number of conclusions have been drawn.

7.1.1 Regulatory Feasibility of the Disposal of NORM in Nonhazardous Landfills

Regulatory control over the disposal of NORM-impacted wastes is largely a function of individual states. When these wastes are specifically addressed in the states' regulations, the rules typically focus only on radium isotopes. The disposal of radium-bearing wastes in nonhazardous municipal landfills is explicitly allowed in only one of the states reviewed in this study: Michigan. In several other states, these wastes may be allowed in nonhazardous landfills; however, the conditions under which they are allowed are variable. In other states, there seems to be less latitude both in the state regulations and on the part of individual regulators.

Regulatory feasibility should not be construed to mean widespread acceptance on the part of the solid waste management industry. Even in Michigan, where the policy on disposing of radium-bearing wastes is explicit, some waste management companies are reluctant to accept any radioactive wastes. Their concerns seem to hinge on issues related to public perception and long-term liability, issues which cannot be easily overcome.

7.1.2 Comparison of Disposal Option Costs

Estimating the costs associated with disposing of petroleum industry NORM is difficult. The cost components that must be considered include those associated with the disposal activity, waste analyses, transportation, permitting, and container decontamination. In addition, because the disposal costs depend on a number of factors (e.g., volume, radium content, requirements for waste analyses, competition for market share), they are quite variable. In general, disposal cost comparisons are truly accurate only when they are made on a case-by-case basis. However, on the basis of data compiled for this study describing the range of expenses associated with the various disposal options, it can be concluded that disposal of regulated NORM in nonhazardous landfills could be one of the most cost-effective disposal options available to the petroleum industry if approved on a widespread basis. It can also be concluded that an increase in the number of available disposal options most likely would reduce NORM disposal costs for the industry.

7.1.3 Radiological Risk Associated with Radium-Bearing Wastes

Table 7 presents a summary for each scenario evaluated in this assessment of the estimated doses and carcinogenic risks associated with disposing of 2,000 m^3 of NORM wastes containing 50 pCi/g of radium in a nonhazardous landfill. On the basis of these results and the results of the sensitivity analyses discussed in Section 6.4, the following conclusions can be drawn:

- Potential radiological doses and resultant health risks for workers actively involved in landfill operations would be negligible.
- Potential doses to an individual living adjacent to the landfill during the NORM disposal action and to the general population living within a 50-mi radius would be negligible.
- Potential doses to future industrial and recreational users of the landfill property would be negligible.
- Potential doses to hypothetical future residential users of the landfill property are most sensitive to depth of the NORM waste layer and integrity of the landfill cap. These doses would be negligible on the basis of the assumption that (1) the NORM wastes would be placed at a depth greater than approximately 10 ft below the cap and (2) the landfill cap would not be breached during construction of the home.
- Provided the NORM wastes are placed deeper than approximately 10 ft below the landfill cap, the Michigan policy allowing wastes containing up to 50 pCi/g to be disposed of in Type II landfills is protective of human health.

As noted, this analysis was conducted for a disposal volume of 2,000 m^3. Increasing the total volume would increase the worker doses linearly and could increase the potential doses to the off-site resident via the groundwater pathway. However, it is estimated that doses for these receptors would be negligible, and increasing the volume probably would not change this overall conclusion. Radiological doses to the future-use receptors would not be affected by increasing the total volume; doses to these receptors are primarily affected by changes in the location of the NORM waste within the landfill.

7.1.4 Radiological Risk Associated with Lead-210 Wastes

Table 7 also presents the estimated doses and carcinogenic risks associated with disposing of one truckload of NORM wastes containing approximately 260 pCi/g of Pb-210 in a nonhazardous landfill. On the basis of these results, it can be concluded that the disposal of wastes containing Pb-210 at elevated levels (i.e., several thousand pCi/g) presents a negligible risk to workers or to the general public. Increasing the disposal volume would not significantly change this overall conclusion. Worker doses would increase linearly with volume, but doses to future users of the property would still be zero because once the waste is buried, a complete exposure pathway to a future receptor does not exist.

TABLE 7 Estimated Peak-Year Dose and Carcinogenic Risks for Disposal of NORM-Impacted Wastes in a Nonhazardous Landfill

Receptor	Radium-Bearing NORM[a]		Lead-210 NORM[b]	
	Dose (mrem/yr)	Risk	Dose (mrem/yr)	Risk
Operational phase scenarios				
Driver	0.3	1×10^{-7}	0	0
Waste-placement operator	1.7	7×10^{-7}	2.4×10^{-6}	9×10^{-13}
Leachate worker	2×10^{-4}	8×10^{-11}	0	0
Off-site resident	6.6×10^{-4}	3×10^{-10}	3.3×10^{-6}	2×10^{-12}
General population[c] (50-mi radius)	2.7×10^{-5}	1×10^{-8}	1.3×10^{-7}	7×10^{-11}
Future use scenarios				
On-site resident	7.4	4×10^{-6}	0	0
On-site industrial worker	2.2	1×10^{-6}	0	0
Recreational visitor	1.2×10^{-7}	6×10^{-14}	0	0
Off-site resident	3.2×10^{-4}	2×10^{-10}	0	0

[a] Doses are for bulk disposal of 2,000 m^3 of radium-bearing wastes having an average Ra-226 concentration of 50 pCi/g.

[b] Doses are for bulk disposal of one truckload (20 m^3) of lead-bearing wastes having an average Pb-210 concentration of 260 pCi/g.

[c] Dose for the general population is in person-rem.

7.2 RECOMMENDATIONS

On the basis of the conclusions presented above, the following recommendations are suggested:

- It may be feasible for other states besides Michigan to consider issuing regulations allowing the disposal of NORM wastes containing up to 50 pCi/g of Ra-226 in municipal, nonhazardous landfills. In approving of this type of disposal, regulators should consider the total volume of radium-bearing wastes that are disposed of in a single landfill and cell, as well as the depth of the NORM waste layer within the landfill. Property records denoting that a landfill was in operation at that location should also note that radium-bearing wastes were disposed of therein.
- Regulators should consider allowing the disposal of NORM wastes containing radium in concentrations greater than 50 pCi/g on a case-by-case basis.
- States should also consider regulations governing the disposal of wastes containing Pb-210 in municipal, nonhazardous landfills. As they should for radium-bearing wastes, the regulations should consider the allowable

concentrations of Pb-210 and the total volume that can be disposed of in a single landfill.

- States may want to consider allowing NORM wastes to be disposed of in other categories of nonhazardous landfills, provided the requirements for deed restrictions and protection of the landfill cap are equivalent to those for municipal landfills.

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APPENDIX A: DETAILED DISCUSSION OF LEACHATE AND GROUNDWATER TRANSPORT MODELS AND RESULTS

This appendix provides a more detailed discussion of the leachate and groundwater transport modeling that was presented in Sections 5 and 6. Several of the figures and tables in this appendix are reproduced from the main text.

Three separate models were used to simulate the transport of the NORM radionuclides from the location of the NORM waste layer, through the landfill into the groundwater, and through the groundwater to the receptor locations (Figure A.1). The Hydrologic Evaluation of Landfill Performance (HELP) model was used to determine how much water could leak through the landfill cap into the landfill (infiltration) and how much leachate could leak through the liner system into the groundwater (Schroeder 1994). An analytical model developed by Tomasko (1992), which used the infiltration value calculated by the HELP model as the driving force, was used to calculate the transport of the Ra-226 and Pb-210 in the leachate generated within the landfill. The SWIFT II code (Reeves et al. 1986) was used to calculate transport of Ra-226 and Rn-222 in the groundwater from the location of the landfill leak to the receptors. The SWIFT II model used the leakage calculations from the HELP model and the transport calculations from the analytical model to determine the concentration, volume, and duration of its source term. No SWIFT II calculations were made for Pb-210 because of the extremely low concentrations calculated at the liner.

A.1 HELP Model

The HELP model assumes that a landfill is a layered system. Parameters are defined for each layer on the basis of standard landfill construction. The landfill construction details discussed in Section 5.3 (and shown in Figure 4) were used to set up the HELP model simulations conducted in this study.

Eight model runs were made using the HELP model to simulate various conditions; the results are presented in Table A.1. In Run 1, the base case against which the results of other runs were compared, it was assumed that the geomembrane liners installed at the bottom of the landfill were of poor quality, having approximately 10 flaws or defects per acre. This number was chosen on the basis of recommendations from Giraud and Bonaparte (1989) to represent a landfill where quality assurance checks were limited during installation of the liners. This assumption gives a conservative estimate of the leachate leakage rate. Also in Run 1, the lateral drainage layers were assumed to be 100 ft long, the drainage slope was assumed to be 0.05%, 100% of the leachate was assumed to be recirculated to the waste layer, and the subsurface inflow rate was assumed to be 0.01%. Hydraulic conductivity through the cap was assumed to be 1.8×10^{-1} ft/d (Schroeder et al. 1994). Precipitation, evapotranspiration, and other climate parameters were based on the landfill's location in the upper Midwest. In this area, precipitation is approximately 26.4 in./yr and evapotranspiration is approximately 16 in./yr.

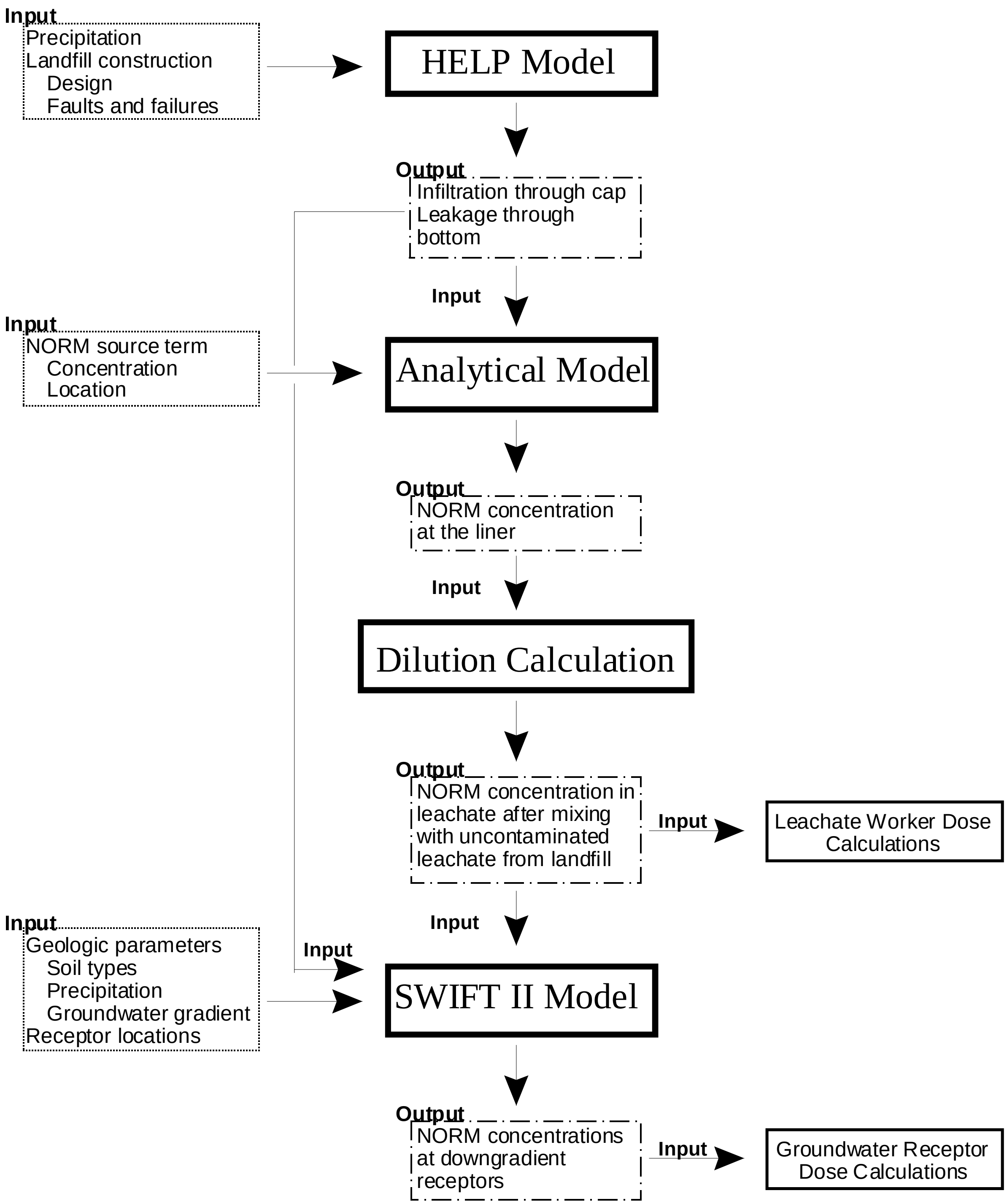

FIGURE A.1 Diagram of Models Used to Calculate Leachate and Groundwater Transport

TABLE A.1 HELP Model Descriptions and Results

Run Number and Description	Predicted Infiltration through Cap (ft/yr)	Predicted Leakage through Bottom (ft^3/yr/acre)
Run 1: Base case	0.56	61
Run 2: Geomembrane liners of good quality (1 pinhole/acre)	0.56	2.4
Run 3: Geomembrane liners of bad quality (50 pinholes/acre)	0.56	448
Run 4: Drainage length 10 ft	0.53	1.4
Run 5: Drainage length 50 ft	0.51	16
Run 6: Drainage length 150 ft	0.55	126
Run 7: Cap hydraulic conductivity 1.5 ft/d	0.58	61
Run 8: Cap hydraulic conductivity 7.7×10^{-4} ft/d	0.40	31
Run 9: No geomembrane liners	0.56	27,500

In Run 1, the leakage rate through the bottom of the landfill was estimated to be 61 ft^3/acre/yr. In Runs 2 and 3, the effect of geomembrane quality on leakage through the bottom of the landfill was evaluated. According to recommendations of Giraud and Bonaparte (1989), in Run 2, a flaw density of 1 flaw/acre was evaluated to represent the geomembrane quality at an intensively monitored landfill. Decreasing the number of defects to 1 flaw/acre decreased the total leakage through the bottom of the landfill to 2.4 ft^3/yr/acre. In Run 3, a flaw density of 50 flaws/acre was evaluated to represent a landfill where quality assurance checks were even less rigorous than in the Run 1 simulation. This increase in defects resulted in a total leakage of 449 ft^3/yr/acre. Run 1 provides a conservative estimate of leakage on the basis of the assumption that the geomembrane liners were not monitored rigorously during installation. In reality, geomembrane liners probably have fewer than 10 flaws/acre, so a more realistic leakage rate would be between 2.4 ft^3/yr/acre (with one pinhole/installation defect) and 61 ft^3/yr/acre (with poor membrane placement).

The HELP model also was used to evaluate the effect of the drainage length on leakage rates. In Run 1, the drainage length was set to 100 ft. In Runs 4, 5, and 6, the drainage lengths were set at 10 ft, 50 ft, and 150 ft, respectively. The results show that a greater drainage length resulted in a greater amount of leakage. This increase occurred because of the increased time that the leachate was in contact with the liner and the increased head across the liner.

Hydraulic conductivity of the cap layer plays an important role in leakage into and out of the landfill. To evaluate the sensitivity of the results to this parameter, two simulations were performed. In Run 7, a hydraulic conductivity of 1.5 ft/d was evaluated. In Run 8, a hydraulic conductivity of 7.7×10^{-4} ft/d was evaluated. These variations had only a small effect on the amount of leachate that leaks through the bottom of the landfill, although there was a slightly greater impact on the amount of infiltration through the cap.

Run 9 modeled the effect of removing the geomembrane liners from the landfill design. This scenario would represent a setting in which NORM waste was buried with only a cap and without any barriers between the waste and the subsurface environment. In Run 9, the geomembrane liners in the design were changed to silty sand layers. Leakage through the base of the landfill increased to 27,500 ft^3/yr/acre.

A.2 Analytical Model

Ra-226 and Pb-210 percolation through the landfill was modeled as a one-dimensional vertical process in which the seepage velocity of the solute was assumed to be equal to the infiltration velocity of the leachate. After closure, seepage velocity would be equal to the infiltration velocity through the cap, i.e., approximately 0.55 ft/yr (Table A.1).

The basic governing equation for contaminant transport of a solute undergoing radioactive decay can be written as:

$$\frac{\partial C}{\partial t} = \frac{D}{R}\frac{\partial^2 C}{\partial Z^2} - \frac{V}{R}\frac{\partial C}{\partial Z} - \lambda C \qquad (1)$$

where the terms on the right hand side of the equation represent dispersion, advection, and decay, respectively. Adsorption is incorporated into the equation through the retardation coefficient, R, in the dispersion and advection terms. R is given by:

$$R = 1 + \frac{\rho_b K_d}{\Phi} \qquad (2)$$

and:

$$K_d = \frac{\partial C_{sorb}}{\partial C} \qquad (3)$$

where Φ is the water content of the porous media, and C_{sorb} is the mass of contaminant adsorbed on the solid part of the porous medium per unit mass of solids. Other parameters of contaminant transport equation include C, the contaminant concentration in the leachate; *t*, the time; D, the dispersion coefficient; Z, the spatial coordinate; and V, the seepage velocity.

This equation was transformed into Laplace space, solved for a step-function contaminant boundary condition, and transformed back into real time and space. This solution was used to simulate NORM transport within the landfill, from the layer in which the NORM was placed, through the landfill materials, to the liner system.

A.2.1 Radium Transport within Landfill

A mass balance calculation was performed to determine how long it would take to dissolve all of the NORM present in the waste layer, thereby defining the source term duration (i.e., the length of the step-function) for the Ra-226 transport calculations. Scenarios were run for three different waste layer thicknesses (1, 4, and 8 ft) assuming a percolation rate of 0.55 ft/yr and an 8-ft depth to the gravel drainage layer located immediately below the municipal wastes (Layer 5 in Figure 4). The results are shown in Table A.2. In calculating the source term durations, it was assumed that the landfill leachate instantly dissolved the Ra-226 to the solubility limit of $RaSO_4$, i.e., 2×10^{-6} g/L (Chemical Rubber Company 1968). Actual radium concentrations would be significantly lower than the solubility limits used in this study because the radium typically co-precipitates with barium, and, as the barium and radium dissolve, the solubility of radium is reduced significantly because of the common ion effect. For purposes of this study, however, these effects were neglected.

The Ra-226 was assumed to leach from the NORM waste layer at solubility limits until all of the Ra-226 present in the waste had been leached out. The Ra-226 was then transported by the leachate down through the municipal waste until it reached the drainage layer. The retardation coefficient used for Ra-226 was 850, which is the value commonly used for Ra-226 transport in silty sands. This value was determined by using a K_d value of 200 mL/g (Sheppard et al. 1984), an effective porosity of 0.42, and a bulk density of 1.8 (Domenico and Schwartz 1990). The actual retardation coefficient for Ra-226 in municipal waste is probably higher because of the high organic content of the wastes. A higher retardation coefficient would produce lower concentrations at the landfill liner and later arrival times than those calculated in this study. When a percolation rate of 0.55 ft/yr is assumed, the retarded velocity for Ra-226 transport would be 6.5×10^{-4} ft/yr (i.e., 0.55 ft/yr divided by 850, the retardation coefficient).

The results of the analytical modeling study are detailed in Table A.3. The concentrations of Ra-226 in the leachate immediately below the NORM waste layer are presented in terms of C/C_0, where C equals the resultant Ra-226 concentration at any given time and C_0 equals the initial Ra-226 concentration in the leachate (i.e., the solubility limit). The last two columns in Table A.3 present the Ra-226 concentrations in the leachate under two mixing scenarios: one in which the leachate is mixed with leachate from the remainder of the cell containing the NORM waste, and the other in which the leachate is mixed with leachate from the entire landfill. Predictably, the Ra-226 increased with increasing thickness of

TABLE A.2 NORM Source Term Descriptions

Waste Layer Thickness (ft)	Time to Dissolve 100% of NORM (yr)	Percentage of Cell Covered by NORM	Percentage of Landfill Covered by NORM
1	0.08	0.23	0.03
4	0.32	0.06	0.008
8	0.65	0.03	0.004

TABLE A.3 Results of the Leachate Transport Modeling for Ra-226

NORM Waste Layer Thickness (ft)	C/C_0 below NORM Waste Layer	Ra-226 Concentration (pCi/L) below NORM Waste Layer	in Leachate from NORM Cell	in Leachate from Entire Landfill
1	4.6×10^{-5}	92	2.1×10^{-1}	2.8×10^{-2}
4	1.8×10^{-4}	360	2.2×10^{-1}	2.9×10^{-2}
8	3.7×10^{-4}	740	2.2×10^{-1}	3.0×10^{-2}

the waste layer, and it decreased as the leachate was diluted by being mixed with leachate from larger areas within the landfill.

Observation indicated that the breakthrough curves for the NORM in the leachate approximated a step-function source term 100 years in duration. This results in an over-prediction of the amount of radium that would be released in leachate leaking from the base of the landfill because (1) removal of contaminated leachate from the landfill would remove part of the NORM before it could be released at the base of the landfill and (2) decay of the NORM radionuclides over the time required to leach through the landfill would reduce the actual amount of NORM released in the leachate. In addition, when it is assumed the source lasted for 100 years, a larger mass of radium would be released into the subsurface beneath the landfill than the amount predicted by the analytical model.

The effect of placing the NORM waste layer at a shallower location within the landfill was examined to determine the impact on Ra-226 concentrations in the leachate. When the NORM waste layer was placed 8 ft above the gravel drainage layer, as assumed in the initial calculations, the time for the center of the NORM plume to move to the liner was approximately 12,000 years, equal to 9 half-lives of Ra-226. When the depth to the drainage layer was increased to 60 ft, the transport time increased to approximately 92,000 years, equal to 72 half-lives of Ra-226. After 72 half-lives, essentially no Ra-226 would be left in the leachate by the time it reached the liner.

A.2.2 Lead Transport within the Landfill

The analytical model also was used to evaluate transport of Pb-210 in the landfill vertically from the location of the NORM waste layer to the liner. The model assumed a source concentration equal to the solubility of lead-sulfate, 4.25×10^{-3} g/L (Chemical Rubber Company 1968) and a k_d of 270 mL/g (Sheppard et al. 1984). These assumptions result in a retardation factor of approximately 1,135, when a bulk density of 1.8 and an effective porosity of 0.42 are used (Domenico and Schwartz 1990). Assuming an infiltration rate through the landfill cap of 0.55 ft/yr, movement of Pb-210 through the landfill would be approximately 0.001 ft/yr. Thus, in one half-life (22 years), lead would move only 0.02 ft. By the time the lead moved any appreciable distance from its disposal point, it would have decayed completely to stable Pb-206. The analytical model estimated lead concentrations at the liner on the order of 1×10^{-21} pCi/L.

Because of the low calculated concentrations at the liner, the short half-life, and the high retardation of lead, Pb-210 concentrations at the liner were assumed to be zero.

A.3 SWIFT II Model

The SWIFT II model (Reeves et al. 1986) was chosen for this study because (1) it is a three-dimensional model that can calculate the mixing that would occur under the landfill, (2) it calculates radionuclide transport with progeny and decay, and (3) it has been widely reviewed and validated. The SWIFT II code was developed to analyze coupled hydrologic, thermal, density, dual-porosity, and solute transport processes in porous media by using a finite difference numerical model. The code has capabilities for simulating continuous and discontinuous layers, time-dependent and constant sources and sinks, and both transient and steady-state groundwater flow.

The SWIFT II model computes fluid concentrations in units of grams of contaminant per gram of fluid, which then can be converted to pCi/L. For this study, SWIFT II was configured to model the transport of Ra-226, and its first decay product, Rn-222. No evaluation of groundwater transport of Pb-210 was made because of the extremely low concentration of Pb-210 calculated to occur at the landfill liner (Section A.2.2). In addition, because of the relatively short half-life of Pb-210 and its high retardation factor of approximately 1,000, Pb-210 would decay to stable Pb-206 before it could move any appreciable distance in a subsurface aquifer. As a result, Pb-210 concentrations in the groundwater were assumed to be zero.

The model developed for this study assumed a general geologic setting with conservative estimates of required parameters. The model was run for 1,000 years; the contaminant source was assumed to be at the maximum value for the first 100 years and zero for the next 900 years. The numerical model was 3,090 ft wide, 12,170 ft long, and 500 ft deep. The site was discretized into a finite difference mesh with 43 divisions in the X direction, 67 divisions in the Y direction, and 10 divisions in the Z direction. Because the model was symmetric about the Y axis and the location of the landfill, a half-model was used.

Both of the horizontal boundaries in the X direction were modeled as constant-head boundaries, which is consistent with the regional aquifer characteristics. The horizontal boundaries in the Y direction and the vertical boundaries in the Z direction were modeled as no-flow boundaries. A steady-state flow regime and transient transport was assumed. A groundwater gradient was induced by tilting the model.

The source location for the NORM injection was located on the X = 1 boundary, Y = 14 point, and Z = 1 boundary. This location achieves symmetry and provides sufficient space above and below the injection point to model transport. The mesh is finer near the source, which aids in numerical stability and yields a higher resolution solution near the injection point and the receptors located closest to the source.

The contaminant was assumed to leak from a single point beneath the landfill, although the volume was calculated on the basis of the entire placement area. The aquifer beneath the landfill was assumed conservatively to be in direct contact with the base of the landfill, and it was assumed that transport of the Ra-226 and Rn-222 in the subsurface was not retarded. The aquifer was assumed to be a homogeneous, clean sand, 500 ft thick. In the base case modeled by using SWIFT II, the regional groundwater gradient was assumed to be 0.0025. This gradient, which is equivalent to a 1-ft change in groundwater elevation over a 400-ft distance, was chosen because it is representative of gradients typically found in aquifers located in the upper Midwest. In the SWIFT II base case, the hydraulic conductivity of the sand layer was assumed to be 100 ft/s, with longitudinal and transverse dispersivities of 100 and 10 ft, respectively. These values were chosen on the basis of the size of the model and recommendations from Freeze and Cherry (1979). The molecular diffusion coefficients for Ra-226 and Rn-222 were assumed to be 1.2×10^{-6} per recommendations from Domenico and Schwartz (1990).

In the SWIFT II base case, the Ra-226 concentration in the leachate was assumed to be 2.2×10^{-1} pCi/L, on the basis of the results of the analytical model simulation that assumed a percolation rate of 0.55 ft/yr, a NORM waste layer thickness of 8 ft, and a depth to the drainage layer of 8 ft (Table A.3). The receptor was located at a depth of 5 ft below the base of the landfill at a distance of 1,000 ft downgradient from the leak. For comparison purposes, additional runs were performed to calculate the Ra-226 concentrations at receptors located 35 and 175 ft below the base of the landfill. In addition, the sensitivity of the model results to the hydraulic conductivity of the sand layers, groundwater gradient, and recharge rate were examined.

Table A.4 lists the maximum concentration of Ra-226 at the downgradient receptor located 5 ft below the base of the landfill for each specific case modeled. For each scenario, the Ra-226 concentration was estimated assuming the absence of the geomembrane liners and a leachate leakage rate of 27,500 ft^3/yr/acre, as calculated in HELP model Run 9 (Table A.1). For comparison purposes, the Ra-226 concentration also was estimated assuming the presence of poorly installed geomembrane liners (i.e., liners having 10 flaws/acre) and a leakage rate of only 61 ft^3/yr/acre, as calculated in HELP model Run 1 (Table A.1).

For the base-case scenario in which the geomembrane liners were not present, the SWIFT II model estimated a Ra-226 concentration at the 5-ft deep receptor of 3.3×10^{-4} pCi/L. In contrast, when the presence of poorly installed liners was assumed, the SWIFT II model estimated a Ra-226 concentration at this receptor of 5.7×10^{-9} pCi/L. Table A.4 also presents the results of the sensitivity analyses of hydraulic conductivity, groundwater gradient, and recharge rate. These results show that while the model is somewhat sensitive to these parameters, the resulting concentrations are within approximately one order of magnitude of each other and are very low.

Figure A.2 presents the results of the sensitivity analyses that evaluated the effect of the depth of the well used by the groundwater receptor on the base-case results. This figure plots the Ra-226 concentrations at three depths versus time. Increasing the depth of the groundwater

TABLE A.4 Radium Concentration at the Groundwater Receptor Located 1,000 ft Downgradient and 5 ft below the Base of the Landfill

	Maximum Ra-226 Concentration (pCi/L)	
Run Description	**with No Liners**	**Maximum Ra-226 Concentration with Liners**
Base case	3.3×10^{-4}	5.7×10^{-9}
High conductivity	3.9×10^{-4}	6.6×10^{-9}
Low conductivity	2.9×10^{-4}	4.8×10^{-9}
High gradient	7.5×10^{-5}	1.3×10^{-8}
Low gradient	3.3×10^{-4}	5.7×10^{-9}
Increased recharge	1.1×10^{-4}	2.0×10^{-9}

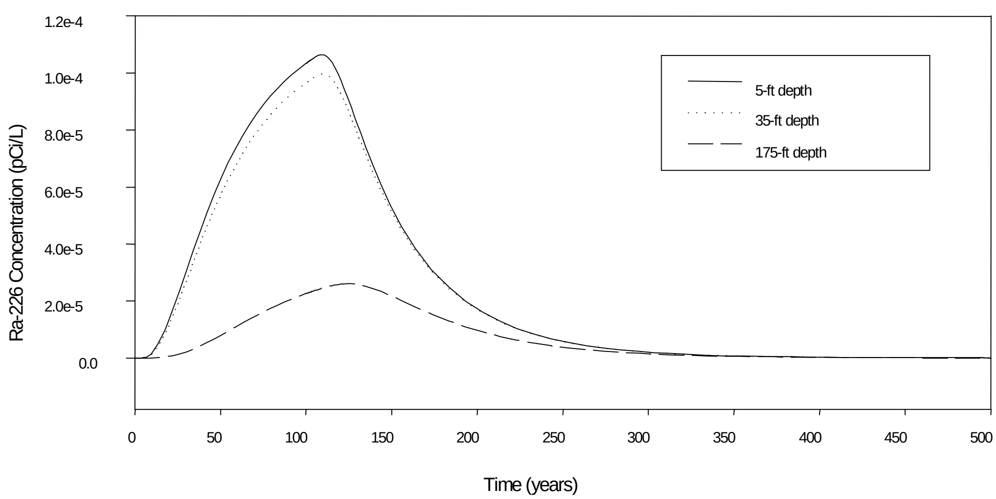

FIGURE A.2 Comparison of Ra-226 Concentrations Predicted at Varying Depths for Receptor Located 1,000 Feet Downgradient of the Landfill

receptor to 35 and 175 ft below the landfill decreased the maximum calculated concentrations of Ra-266 and Rn-222 by one to two orders of magnitude and had little effect on the timing of the maximum concentrations.

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APPENDIX B: TABLES OF ASSUMPTIONS AND EXPOSURE PARAMETERS

TABLE B.1 Exposure Parameters Used to Model Worker Scenarios

Input Parameter[a]	Driver	Waste-Placement Operator	Leachate Worker
Exposure time (h)	0.75[b]	0.75[b]	0.5
Distance to truck (ft)	2	5(10)[c]	3
Shielding thickness (in.)	0	2	0.5
Airborne respirable dust concentration (mg/m^3)	NA[d]	3.3×10^{-7} [e]	NA
Drive time to disposal area (min)	15	NA	NA
Truck bed/tank size (ft)			
Length	25[f]	25[f]	5.5
Width	6[f]	6[f]	2.5
Height	6[f]	6[f]	5.5

[a] Value used for each parameter was based on engineering judgment, unless other reference or rationale is noted.

[b] One half hour assumed for inspection, 15 minutes assumed for disposal.

[c] Five feet assumed for inspection; 10 ft assumed for disposal operations.

[d] NA indicates not applicable.

[e] U.S. Environmental Protection Agency, 1989, *Air/Superfund National Technical Guidance Study Series, Vol. III — Estimation of Air Emissions from Cleanup Activities at a Superfund Site*, EPA-450/1-89-003, Office of Air Quality Planning and Standards, Research Triangle Park, N.C.

[f] Default value from TSD-DOSE code (Pfingston et al., 1998, *TSD-DOSE: A Radiological Dose Assessment Model for Treatment, Storage, and Disposal Facilities*, ANL/EAD/LD-4 (Rev. 1), Argonne National Laboratory, Argonne, Ill.).

TABLE B.2 Exposure Parameters Used to Model Operational Phase Off-Site Resident Scenario

Input Parameter[a]	Maximally Exposed Individual	Population Located within 50 Miles
Exposure time (h)	24	24
Exposure frequency (d/yr)	365	365
Distance to source (mi)	0.18	0–50
Wind speed (ft/s)	13	13
Frequency wind blows in direction	0.5	NA
Population density (persons/mi^2)		
0–20 mi radius	NA[b]	75[c]
0–50 mi radius	NA	217[c]

[a] TSD-DOSE default values were used for all input parameters unless other rationale or reference is specified (Pfingston et al., 1998, *TSD-DOSE: A Radiological Dose Assessment Model for Treatment, Storage, and Disposal Facilities*, ANL/EAD/LD-4 (Rev. 1), Argonne National Laboratory, Argonne, Ill.).

[b] NA indicates not applicable.

[c] Value derived from site-specific data.

TABLE B.3 Exposure Parameters Used to Model Future Use Scenarios

	Scenario[b]				
Input Parameter[a]	**On-Site Resident**	**On-Site Industrial Worker**	**Recreational Visitor**	**Off-Site Resident**	**Reference/Rationale**
Area (ft^2)	147	147	147	147	Site-specific
Cover depth (ft)	14	14	14	14	Site-specific
Waste layer thickness (ft)	8	8	8	8	Site-specific
Density of waste layer (g/cm^3)	2.0	2.0	2.0	NA	Mixture of soil/NORM (EPA 1993)[c]
Density of cover material (g/cm^3)	1.6	1.6	1.6	NA	RESRAD default
Exposure time (h/d)					Engineering judgment
Indoor	12	6	0	NA	
Outdoor	6	2	4	NA	
Exposure frequency (d/yr)	365	250	20	365	
Ingestion rate					RESRAD default
Soil (g/d)	0.1	0.1	0.1	NA	
Meat(kg/yr)	63	NA	NA	NA	
Plant (kg/yr)	160	NA	NA	NA	
Groundwater (L/d)	2	NA	NA	2	
Inhalation rate (m^3/h)	0.96	0.96	0.96	NA	RESRAD default
Rn-222 emanation coefficient	0.04	0.04	0.04	0.04	Rood and Kendrick (1996)[d]
Foundation depth below surface (ft)	1	1	NA	NA	Engineering judgment
Erosion rate (mm/yr)	1.0	1.0	NA	NA	RESRAD default
Plant/soil transfer factor					Auxier & Associates, Inc. (1996)[e]
Radium	6.8×10^{-5}	NA	NA	NA	
Lead	3.3×10^{-5}	NA	NA	NA	
Thorium	1.7×10^{-6}	NA	NA	NA	
Fraction of food from site	0.5	NA	NA	NA	Engineering judgment

[a] RESRAD default values were used for input parameters not listed.

[b] NA indicates not applicable.

[c] U.S. Environmental Protection Agency, 1993, *DRAFT Diffuse NORM Wastes — Waste Characterization and Preliminary Risk Assessment*, Office of Radiation and Indoor Air, Washington, D.C.

[d] Rood, A.S., and D.T. Kendrick, 1996, "Measurement of ^{222}Rn Flux, ^{222}Rn Emanation, and ^{226}Ra Concentration from Injection Well Pipe Scale," *NORM/NARM: Regulation and Risk Assessment*, proceedings of the 29th Midyear Topical Meeting of the Health Physics Society, Scottsdale, Ariz., Jan. 7–10, pp. 139–144.

[e] Auxier & Associates, Inc., 1996, *Leachate Analysis of Martha Oil Field Wastes, Martha, Kentucky*, prepared for Ashland Exploration, Inc., Houston, Tex.