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Radiation

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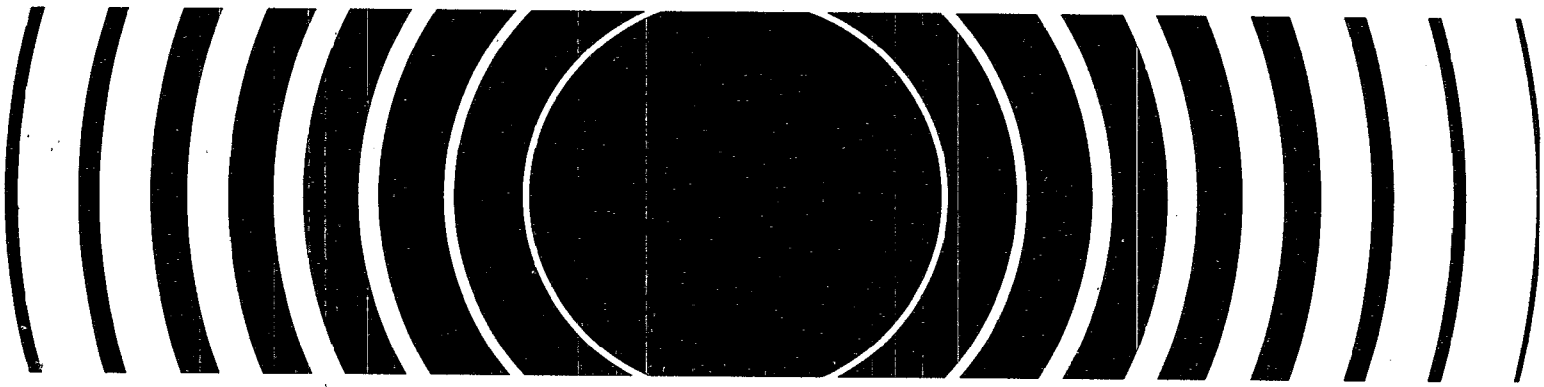
Low-Level and NARM Radioactive Wastes

Draft Environmental Impact Statement for Proposed Rules

Volume 1

Background Information Document



40 CFR Part 193
Environmental Radiation Standards
for Management and Land Disposal
of Low Level Radioactive Wastes

and

40 CFR Part 764
Environmental Radiation Standards
for Land Disposal of Naturally
Occurring and Accelerator-Produced
Radioactive Materials (NARM)

EPA 520/1-87-012-1

DRAFT ENVIRONMENTAL IMPACT STATEMENT

FOR PROPOSED RULES

VOLUME 1

BACKGROUND INFORMATION DOCUMENT

Low-Level and NARM Radioactive Wastes

June 1988
U.S. Environmental Protection Agency
Office of Radiation Programs
Washington, DC 20460

PREFACE

The Environmental Protection Agency is proposing environmental standards for the management and land disposal of low-level radioactive wastes and the land disposal of Naturally Occurring and Accelerator-Produced Radioactive Materials (NARM) waste.

This two-volume Draft Environmental Impact Statement (DEIS) is provided to support EPA's rulemaking for generally applicable environmental standards for the management and land disposal of low-level radioactive wastes and the land disposal of Naturally Occurring and Accelerator-Produced Radioactive Materials (NARM) waste. The first volume of the DEIS, the Background Information Document (BID), presents the technical treatise on the risk assessment. The BID includes the sources of radiation exposures, the routes of exposure, the methodology of the assessment, the individual doses/risk and the population health effects, and model sensitivity and uncertainties in the analysis. Volume 2 of the DEIS, the Economic Impact Assessment (EIA), presents the benefits of the rule, the costs of the controls, and the cost effectiveness of the different regulatory options.

To complete the overall analysis for the DEIS, the Preamble to the rule should be consulted as it discusses how the Agency went about its decision process and why it made such decisions.

Copies of this Draft Environmental Impact Statement (DEIS) and requests for comment have been sent to the following Federal Agencies:

Department of Commerce
Department of Defense
Department of Energy
Department of Health and Human Services
Department of Transportation
Department of The Interior
Nuclear Regulatory Commission

We have also sent copies to those individuals and organizations who have notified us of their interest.

An announcement of the availability of the DEIS has been submitted to the Federal Register.

Comments on this DEIS should be sent (in duplicate if possible) to:

Central Docket Section (LE-131)
Environmental Protection Agency
Attn: Docket No R-82-01
Washington, DC 20460

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Chapter 1: INTRODUCTION

The U.S. Environmental Protection Agency (EPA) is responsible for developing and issuing environmental standards, guidelines, and criteria to ensure that the public and the environment are adequately protected from potential radiation impacts.

Toward this end, EPA is proposing generally applicable environmental standards for the management and disposal of Atomic Energy Act (AEA) low-level radioactive wastes and high-concentration Naturally Occurring and Accelerator-produced Radioactive Materials wastes. These standards provide the basic framework for long-term environmental protection through management and disposal of these types of wastes. When these standards are finalized, they will be Part 193 of Title 40 of the Code of Federal Regulations (40 CFR 193).

In addition, this standard will provide criteria for identifying wastes with sufficiently low levels of radioactivity to qualify as "Below Regulatory Concern" (BRC). Any waste meeting these criteria could be disposed of as a nonradioactive waste.

Low-level radioactive waste (LLW) encompasses basically all radioactive wastes defined by the Atomic Energy Act (AEA) except those that are specifically defined as another class of radioactive waste. Thus, LLW are wastes that are not classified as spent nuclear fuel, high-level radioactive wastes, and transuranic wastes as defined in 40 CFR 191 (EPA85), or uranium and thorium by-product materials (mill tailings) as defined in the Uranium Mill Tailings Radiation Control Act of 1978 (UMT78) and 40 CFR 192 (EPA83b).

Two broad categories of radionuclides not covered under the AEA are naturally occurring radionuclides of insufficient concentration to be considered source material and accelerator-produced radionuclides. Materials containing these nuclides are commonly referred to as naturally occurring and accelerator-produced radioactive materials (NARM). NARM wastes can be further classified as discrete or diffuse. Discrete NARM wastes, such as medical radium sources or radium-dialed instruments, are very similar to most LLW. Diffuse NARM wastes, such as uranium mining overburden, are very different from the waste normally disposed of in LLW facilities.

Sources of LLW in the United States are characterized as being commercially produced and produced from Department of Energy (DOE) research, development, and defense-related activities.

NOTE: Appendix A provides a list of acronyms and a glossary of terms used in this document.

During the 1970's, the level of public concern for health and environmental quality increased rapidly over the management and disposal of radioactive waste. The reasons for this increase included the entrance of radioactivity into the environment from some existing storage and disposal facilities, projections of large increases in the quantity of radioactive waste, realization that radioactive waste would remain hazardous for long times, and recognition that society had not addressed the problem of providing long-term protection from these wastes.

1.1 EPA Authorities for the Rulemaking

These standards are being developed pursuant to the Agency's authorities under the AEA of 1954, as amended, Reorganization Plan No. 3 of 1970, and the Toxic Substances Control Act (TSCA).

Because the AEA excludes NARM radionuclides, the authority for the regulation of NARM waste disposal had to come from some other statute. After a review of the relevant authorities, EPA determined that TSCA was the proper statute for the regulation of NARM wastes. Section 6 of TSCA pertains to the regulation of hazardous chemical substances and mixtures, and specifically gives the Administrator of EPA the authority to regulate the disposal of substances if he finds that there is a reasonable basis to conclude that the unregulated disposal of the substance would present an unreasonable risk of injury to health or the environment (TSC76).

The basic authority for EPA under the AEA, transferred from the Atomic Energy Commission (AEC) to EPA through the Reorganization Plan No. 3 of 1970, is to establish "generally applicable environmental standards for the protection of the general environment from radioactive material. As used herein, standards mean limits on radiation exposures or levels, or concentrations or quantities of radioactive material, in the general environment outside the boundaries of locations under the control of persons possessing or using radioactive material" (Ni70).

1.2 History of the Low-Level Radioactive Waste Program and the EPA Proposed Rulemaking

Since the inception of the nuclear age in the 1940's, LLW has been produced. In the early days of the nuclear age, the civilian sector produced small amounts of wastes compared to the defense programs. At that time most of the civilian wastes were from nonfuel-cycle sources such as hospitals, research laboratories, and certain industries. These wastes were generally low in activity. They were buried at many of the AEC sites or at sea. As the volume of civilian wastes grew, the policy of encouraging the development of regional, commercial burial sites was adopted. The first such sites, licensed in 1962, were near Beatty, Nevada, and at Maxey Flats near Morehead, Kentucky. Over the next decade, sites were licensed near West Valley, New York, Sheffield, Illinois, and Barnwell, South Carolina, to serve their respective regions of the country (Ho78).

Since that time, three of the commercial burial sites have closed, West Valley in 1975 and Maxey Flats in 1977 due to trench leakage and Sheffield in 1978 due to filling of all available trenches (Ho80).

Since its inception, the EPA has participated in many efforts to resolve radioactive waste problems under legislative responsibilities to protect public health and the environment (Me73, Me76a, Me76b, Me77, Pa74, Oc74). In 1972, the EPA Office of Radiation Programs (ORP) began a joint program with the Conference of Radiation Control Program Directors to examine the practice of disposing of LLW in shallow-land burial sites (CRC74, Ho76, EPA77c, EPA78c).

In 1973, the National Academy of Sciences - National Research Council was requested by the AEC to study the conditions, practices, and problems involved in the near-surface ground burial of solid waste contaminated with low levels of radioactive materials. The study was carried out by the Panel on Land Burial of the Committee on Radioactive Waste Management; the report was published in 1976 (NAS76). The Panel's findings included a belief that the Federal Government must exert strong leadership in defining the responsibilities, assigning the authority for setting and implementing standards, and ensuring coordination among Federal, State, and local agencies and private industry for the effective management of radioactive wastes. It was also recognized that EPA was one of the elements of the Federal Government in which concern about various aspects of the problems of radioactive waste is distributed.

Beginning in 1976, the Federal Government intensified its program to develop an interagency effort on waste management. Although the emphasis was on high-level waste processing and disposal, there was usually some mention of the need for standards and research on LLW disposal. The Office of Management and Budget (OMB) established an interagency task force on commercial nuclear wastes in March 1976 (Ly76).

A status report on the management of commercial radioactive nuclear wastes, published in May 1976 by the President's Federal Energy Resources Council (ERC), emphasized the need for coordination of Administration policies and programs relating to energy. The ERC established a nuclear subcommittee to coordinate Federal nuclear policy and programs to assure an integrated government effort. This report called for an accelerated comprehensive government radioactive waste program plan with an interagency task force to coordinate activities among the responsible Federal agencies. The EPA was given the responsibility of establishing general environmental standards governing waste activities, including LLW (FER76).

In October 1976, President Ford issued a major statement on nuclear policy. As part of his comprehensive statement, he announced new steps to assure that the United States has the facilities for management of nuclear wastes from commercial power plants. The President's actions were based on the findings of the OMB interagency task force formed in

March 1976. Among the many steps to be taken was EPA's issuance of general environmental standards governing nuclear facility releases to the biosphere (Fo76).

In 1978, President Carter established the Interagency Review Group (IRG) to develop recommendations for the establishment of an administrative policy with respect to long-term management of nuclear wastes and supporting programs to implement the policy. The IRG report reemphasized EPA's role in developing generally applicable standards for the disposal of all radioactive wastes including LLW (DOE79). The report emphasized that the criteria and standards set by EPA are general rather than site specific and serve as the bases for Nuclear Regulatory Commission (NRC) regulations and DOE operations. In a Message to Congress on February 12, 1980, the President outlined a comprehensive national radioactive waste management program based on the IRG report. The message repeated that the EPA was responsible for creating general criteria and numerical standards applicable to nuclear waste management activities (Ca80).

In November 1978, EPA proposed "Criteria for Radioactive Wastes," which were intended as Federal guidance for storage and disposal of all forms of radioactive wastes (EPA78d). This effort included frequent interaction with the public, which began with a series of public workshops on radioactive waste disposal in 1977 and 1978 (EPA77a,b, EPA78a,b). In March 1981, however, EPA withdrew the proposed criteria. It was decided that the many types of radioactive wastes and different methods necessary to manage and dispose of them made the issuance of generic disposal guidance too complex and that standards based on waste type would be the best approach (EPA81).

EPA efforts continued toward establishing general environmental radiation protection standards for LLW, and on August 31, 1983, EPA published an Advance Notice of Proposed Rulemaking for LLW disposal standards (EPA83a).

In 1980, Congress passed Public Law 96-573, the Low-Level Radioactive Waste Policy Act (LLRWPA), directing that each State would be responsible for providing disposal capacity for all commercial LLW generated within its borders. Regional cooperation through compacts was suggested, and is presently the method by which most States are assuming their responsibility for LLW disposal (LLR80). In 1986, Congress passed the Low-Level Radioactive Waste Policy Amendments Act (Public Law 99-240) to amend the LLRWPA, improve procedures for the implementation of compacts for the establishment and operation of regional disposal facilities for LLW, and allow the States until 1993 to provide disposal capacity (LLR86). Public Law 99-240 also endorsed the BRC concept and required the NRC to establish procedures for acting expeditiously on petitions to exempt specific radioactive waste streams from the NRC's regulations.

In connection with EPA's LLW standards development program, in February 1984 the Agency conducted an independent Peer Review of the PRESTO-EPA computer code risk assessment methodology. The PRESTO-EPA code models the transport of radionuclides through hydrogeologic and atmospheric pathways to the eventual ingestion and inhalation by or direct exposure of humans. (See Chapter 8 for PRESTO-EPA descriptions.) The main purpose of the Peer Review was to discuss the basic assumptions, the applicability of the model, and an evaluation of the PRESTO-EPA code. The Peer Review commented on a number of major issues or problems to which EPA has responded in the form of an enhancement or a modification to the PRESTO-EPA model (and code) and its documentation (PR84).

As part of EPA's standards development, the Agency requested an independent scientific review of the risk assessment for the proposed standards by a special subcommittee of the Agency's Science Advisory Board (SAB). This subcommittee held meetings in 1985 and then prepared a final report dated October 28, 1985 (SAB85). Although the SAB review found the Agency's analyses in support of the proposed standards to be comprehensive and scientifically competent, the report contained several findings and recommendations for improvement. The risk assessment has been revised to incorporate these recommendations. Responses to the SAB report were made on January 13, 1986 (EPA86).

When EPA first started to develop the LLW standard, it did not intend to include NARM wastes since they are outside of the authority of the AEA, which was being used for the LLW standard. In April 1984, EPA conducted public outreach meetings on the development of an LLW standard. At these meetings, State representatives and others indicated that the exclusion of NARM wastes was the most serious deficiency in our program. Similar comments had also been received in response to the Agency's Advanced Notice of Proposed Rulemaking. This was due to the lack of Federal regulation for these wastes, the inconsistent nature of State regulation, and the very hazardous nature of many NARM wastes. Based on the comments EPA received and further studies of the problem, EPA decided to include NARM wastes within the EPA LLW standard development effort.

1.3 Purpose and Scope of the Background Information Document (BID)

The purpose of this document is to provide background information that, when considered together with the proposed generally applicable standards, supports the actions taken by the EPA with regard to the management and disposal of AEA LLW and NARM wastes. It also contains an integrated risk assessment that provides a scientific basis for these actions.

The scope encompasses the conceptual framework for assessing radiation impact, including identification of the sources of possible radionuclide releases, analysis of the movement of the radionuclides from

the source through environmental pathways, estimates of doses received by individuals, and calculations of the potential number of genetic and somatic fatal health effects in future populations.

1.4 Computer Codes Utilized

A family of computer codes has been used as a tool in the Agency's risk analyses. These codes, including PATHRAE-EPA and PRESTO-EPA, are described in Chapter 8.

1.5 Program Technical Support Documents

A number of technical support documents have been prepared and published during the history of the standards development program to help establish the technical basis for the standards. These documents, listed below, should also be considered as part of the technical background for the rulemaking process.

(1) Final Report, Characterization of Health Risks and Disposal Costs Associated with Alternative Methods for Land Disposal of Low-Level Radioactive Waste, Contract No. 68-02-3178, Work Assignment 16, Prepared for EPA/ORP by Envirodyne Engineers Inc., St. Louis, Missouri, 1984.

(2) Appendices A through H, Characterization of Health Risks and Disposal Costs Associated with Alternative Methods for Land Disposal of Low-Level Radioactive Waste, Contract No. 68-02-3178, Work Assignment 16, Prepared for EPA/ORP by Envirodyne Engineers Inc., St. Louis, Missouri, 1984.

(3) Characterization of Land Disposal Alternatives for Low-Level Nuclear Wastes, Prepared for EPA/ORP by TRW Energy Development Group, Lakewood, Colorado, and Rogers and Associates Engineering Corp., Salt Lake City, Utah, September 1983.

(4) Final Report, Radiation Exposures and Health Risks Resulting from Less Restrictive Disposal Alternatives for Very Low-Level Radioactive Wastes, Contract No. 68-02-3178, Work Assignment 20, Prepared for EPA/ORP by Envirodyne Engineers Inc., St. Louis, Missouri, 1984.

(5) Appendices A through K, Radiation Exposures and Health Risks Resulting from Less Restrictive Disposal Alternatives for Very Low-Level Radioactive Wastes, Contract No. 68-02-3178, Work Assignment 20, Prepared for EPA/ORP by Envirodyne Engineers, Inc., St. Louis, Missouri, 1984.

(6) Composite Source Term for Eleven Scenarios Using Candidate BRC Waste Streams, Technical Information Memorandum, TIM-51-1, Rogers and Associates Engineering Corporation, Salt Lake City, Utah, April 6, 1984.

(7) Composite Source Terms of Candidate Below Regulatory Concern Wastes, Technical Information Memorandum TIM-50-1, Rogers and Associates Engineering Corporation, Salt Lake City, Utah, January 24, 1984.

(8) Composite Source Terms for Six Scenarios Using Candidate BRC Waste Streams, Prepared for EPA/ORP by Rogers and Associates Engineering Corporation for Radian Corporation, Research Triangle Park, N.C., March 5, 1984.

(9) J. Neiheisel, Prediction Parameters of Radionuclide Retention at Low-Level Radioactive Waste Sites, USEPA Report No. EPA 520/1-83-025, Washington, D.C., November 1983.

(10) J. M. Gruhlke, J. Neiheisel, and L. Battist, Estimates of the Quantities, Form, and Transport of Carbon-14 in Low-Level Radioactive Waste, U.S. EPA Report No. 520/1-86-019, Washington, D.C., September 1986.

(11) Final Report, PRESTO-EPA-POP: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code, Volume I, Methodology Manual, Prepared for EPA/ORP by Rogers and Associates Engineering Corporation, Salt Lake City, Utah, 1987.

(12) Final Report, PRESTO-EPA-POP: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code, Volume II, User's Manual, Prepared for EPA/ORP by Rogers and Associates Engineering Corporation, Salt Lake City, Utah, 1987.

(13) Final Report, PRESTO-EPA-CPG: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code, Documentation and User's Manual, Prepared for EPA/ORP by Rogers and Associates Engineering Corporation, Salt Lake City, Utah, 1987.

(14) Final Report, PRESTO-EPA-DEEP: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code, Documentation and User's Manual, Prepared for EPA/ORP by Rogers and Associates Engineering Corporation, Salt Lake City, Utah, 1987.

(15) Final Report, PRESTO-EPA-BRC: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code, Documentation and User's Manual, Prepared for EPA/ORP by Rogers and Associates Engineering Corporation, Salt Lake City, Utah, 1987.

(16) Final Report, PATHRAE-EPA: A Performance Assessment Code for the Land Disposal of Radioactive Wastes, Documentation and User's Manual, Prepared for EPA/ORP by Rogers and Associates Engineering Corporation, Salt Lake City, Utah, 1987.

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Ca80 The White House, President J. Carter, The President's Program on Radioactive Waste Management, Fact Sheet, February 12, 1980.

CRC74 Conference of Radiation Control Program Directors, Report of the Task Force on Radioactive Waste Management, 6th Annual National Conference on Radiation Control, San Antonio, Texas, October 1974.

DOE79 U.S. Department of Energy, Report to the President by the Interagency Review Group on Nuclear Waste Management, Report No. TID-29442, March 1979.

EPA77a U.S. Environmental Protection Agency, Proceedings: A Workshop on Issues Pertinent to the Development of Environmental Protection Criteria for Radioactive Wastes, Reston, Virginia, February 3-5, 1977, Office of Radiation Programs, Report No. ORP/CSD-77-1, Washington, D.C., 1977.

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EPA78d U.S. Environmental Protection Agency, Recommendations for Federal Radiation Guidance, Criteria for Radioactive Wastes, Federal Register, 43(221):53262-53268, November 15, 1978.

EPA81 U.S. Environmental Protection Agency, Withdrawal of Proposed Regulations, Federal Register, 46(53):17567, March 19, 1981.

EPA83a U.S. Environmental Protection Agency, Environmental Radiation Protection Standards for Low-Level Radioactive Waste Disposal; Advanced Notice of Proposed Rulemaking, 48(170):39563, August 31, 1983.

EPA83b U.S. Environmental Protection Agency, Environmental Standards for Uranium and Thorium Mill Tailings at Licensed Commercial Processing Sites, Final Rule, Federal Register, 48(196): 45926-45947, October 7, 1983.

EPA85 U.S. Environmental Protection Agency, Environmental Standards for the Management and Disposal of Spent Nuclear Fuel, High-Level and Transuranic Radioactive Wastes, Final Rule, Federal Register, 50(182):38066-38089, September 19, 1985.

EPA86 U.S. Environmental Protection Agency, January 13, 1986 letter from L.M. Thomas, Administrator, to W.J. Schull and N. Nelson, EPA Science Advisory Board, regarding Office of Radiation Programs' Responses to October 28, 1985 Science Advisory Board Report on the March 1985 Draft Background Information Document for Proposed Low-Level Radioactive Waste Standards.

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LLR80 Low-Level Radioactive Waste Policy Act, Public Law 96-573, December 22, 1980.

LLR86 Low-Level Radioactive Waste Policy Amendments Act of 1985, Public Law 99-240, January 15, 1986.

Ly76 Memorandum from J.T. Lynn, OMB, to R. Train, EPA; R. Peterson, CEQ; R. Seamans, ERDA, and W. Anders, NRC; March 25, 1976, concerning Establishment of an Interagency Task Force on Commercial Nuclear Wastes.

Me73 Meyer, G.L., and M.F. O'Connell, Potential Impact of Current Commercial Solid Low-Level Radioactive Waste Disposal Practices on the Hydrogeologic Environment, presented at the International Symposium on Underground Waste Management and Artificial Recharge, AAPG-USGS-IANS, New Orleans, September 1973.

Me76a Meyer, G.L., Preliminary Data on the Occurrence of Transuranium Nuclides in the Environment at the Radioactive Waste Burial Site Maxey Flats, Kentucky, USEPA Report EPA-520/3-75-021, Washington, D.C., February 1976.

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ORP75 Office of Radiation Programs, Environmental Protection Agency, National Radiation Protection Program Strategy, May 1975.

ORP76 Office of Radiation Programs, Environmental Protection Agency, Program Statement, EPA-520/7-76-007, May 1976.

Pa74 Papadopulos, S.S., and I.J. Winograd, Storage of Low-Level Radioactive Wastes in the Ground: Hydrogeologic and Hydrochemical Factors, Prepared for USEPA, Report EPA-520/3-74-009, 1974.

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Chapter 2: CURRENT REGULATORY PROGRAMS AND STRATEGIES

2.1 Introduction

People have always been exposed to natural background radiation from cosmic rays and the naturally occurring radionuclides in the earth. Awareness of radiation and radioactivity dates back only to the end of the last century--to the discoveries of x rays in 1895 and radioactivity in 1896. These discoveries marked the beginning of radiation science and the deliberate use of radiation and radionuclides in science, medicine, and industry.

By the 1920's, the use of x rays in diagnostic medicine and industrial applications was widespread, and radium was being used by industry for luminescent dials and by doctors in therapeutic procedures. By the 1930's, biomedical and genetic researchers were studying the effects of radiation on living organisms, and physicists were beginning to understand the mechanisms of spontaneous fission and radioactive decay. By the 1940's, a self-sustaining fission reaction was demonstrated, which led directly to the construction of the first nuclear reactors and atomic weapons.

Today the use of x rays and radioactive materials is widespread and includes:

- nuclear reactors, and their supporting fuel-cycle facilities, which generate electricity and power ships and submarines, produce radioisotopes for research, space, defense, and medical applications, and are used as research tools by nuclear engineers and physicists;
- particle accelerators, which produce radioisotopes for therapy uses and are also used as research tools for studying the structure of materials and atoms;
- the radiopharmaceutical industry, which provides the radioisotopes needed for biomedical research and nuclear medicine;
- nuclear medicine, which has developed as a recognized medical specialty in which radioisotopes are used in the diagnosis and treatment of numerous diseases;
- x rays, which are widely used as a diagnostic tool in medicine and in such diverse industrial fields as oil exploration and nondestructive testing;
- radionuclides, which are used in such common consumer products as luminous-dial wristwatches and smoke detectors; and
- industrial uses such as thickness gauges and well-logging.

The following is a brief history of the evolution of radiation protection philosophy and a summary of the current regulatory programs and strategies of the government agencies responsible for assuring that radiation and radionuclides are used safely.

2.2 The International Commission on Radiological Protection and the National Council on Radiation Protection and Measurements

Initially, the dangers and risks posed by x rays and radioactivity were poorly understood. By 1896, however, "x-ray burns" were being reported in the medical literature, and by 1910, it was understood that such "burns" could be caused by radioactive materials. By the 1920's, sufficient direct evidence (from the experiences of radium dial painters, medical radiologists, and miners) and indirect evidence (from biomedical and genetic experiments with animals) had been accumulated to persuade the scientific community that a scientific body should be established to make recommendations concerning human protection against exposure to x rays and radium.

In 1928, the first radiation protection commission was created. Reflecting the uses of radiation and radioactive materials at the time, the body was named the International X-Ray and Radium Protection Commission and was charged with developing recommendations concerning protection from radiation. In 1950, the Commission was renamed the International Commission on Radiological Protection (ICRP).

The newly created commission suggested to the nations represented that they appoint national advisory committees to represent their viewpoints before the ICRP, and to act in concert with the Commission in developing and disseminating recommendations on radiation protection. This suggestion led to the formation, in 1929, of the Advisory Committee on X-Ray and Radium Protection as the U.S. advisory group. This Advisory Committee emerged in 1964 in its present form as the Congressionally chartered National Council on Radiation Protection and Measurements (NCRP). The Congressional charter provides for the NCRP to:

- collect, analyze, develop, and disseminate in the public interest information and recommendations about radiation protection and radiation quantities, units, and measurements;
- develop basic concepts about radiation protection and radiation quantities, units, and measurements, and the application of these concepts;
- provide a means by which organizations concerned with radiation protection and radiation quantities, units, and measurements may cooperate to effectively use their combined resources, and to stimulate the work of such organizations; and

- cooperate with the ICRP and other national and international organizations concerned with radiation protection and radiation quantities, units, and measurements.

Throughout their existence, the ICRP and the NCRP have worked together closely to develop radiation protection recommendations that reflect the current understanding of the dangers associated with exposure to ionizing radiation.

The first exposure limits adopted by the ICRP and the NCRP (ICRP34, ICRP38, NCRP36) established 0.2 roentgen/day (R/d)* as the "tolerance dose" for occupational exposure to x rays and gamma radiation from radium. This limit, equivalent to approximately 25 rads/yr as measured in air, was established to guard against the known effects of ionizing radiation on superficial tissue, changes in the blood, and "derangement" of internal organs, especially the reproductive organs. At the time the recommendations were made, high doses of radiation were known to cause observable effects and even to induce cancer. However, no such effects were observed at lower doses, and the epidemiological evidence at the time was inadequate to even imply the carcinogenic induction effects of moderate or low doses. Therefore, the aim of radiation protection was to guard against known effects, and the "tolerance dose" limits that were adopted were believed to represent the level of radiation that a person in normal health could tolerate without suffering observable effects. The concept of a tolerance dose and the recommended occupational exposure limit of 0.2 R/d for x- and gamma radiation remained in effect until the end of the 1940's. The recommendations of the ICRP and the NCRP made no mention of exposure of the general populace.

By the end of World War II, the widespread use of radioactive materials and scientific evidence of genetic and somatic effects at lower doses and dose rates suggested that the radiation protection recommendations of the NCRP and the ICRP would have to be revised downward.

By 1948, the NCRP had formulated its position on appropriate new limits. These limits were largely accepted by the ICRP in its recommendations of 1950 and formally issued by the NCRP in 1954 (ICRP51, NCRP54). The immediate effect was to lower the whole-body occupational dose limit to 0.3 rad/wk (approximately 15 rad/yr); the revised recommendations also embodied several new and important concepts in the formulation of radiation protection criteria.

*The NCRP's recommendation was 0.1 R/d measured in air. This limit is roughly equivalent to the ICRP limit, which was conventionally measured at the point of exposure and included backscatter.

First, the recommendations recognized the differences in the effects of various types and energies of radiation; both ICRP's and NCRP's recommendations included discussions of the biological effectiveness of radiations of differing types and energies. The NCRP advocated the use of the "rem" to express the equivalence in biological effects between radiations of differing types and energy.* Although the ICRP noted the shift toward the acceptance of the rem, it continued to express its recommendations in terms of the rads, with the caveat that neutrons should carry a quality factor of 10.

Second, the recommendations of both organizations introduced the concept of critical organs and tissues. The intent of this concept was to assure that no tissue or organ, with the exception of the skin, would receive a dose in excess of that allowed for the whole body. At the time, scientific evidence was lacking on which to base different recommended limits for the various tissues and organs. Thus, all blood-forming organs were considered critical organs and were limited to the same exposure as the whole body. The skin was allowed a dose of 30 rad/yr and the extremities were allowed 75 rad/yr.

Third, the recommendations of the NCRP included the suggestion that individuals under the age of 18 receive no more than one-tenth the exposure allowed for adults. The reasoning behind this particular recommendation is interesting, as it reflects clearly the limited knowledge of the times. The scientific evidence indicated a clear relationship between accumulated dose and genetic effect. However, this evidence was obtained exclusively from animal studies that had been conducted with doses ranging from 25 to thousands of rad. There was no evidence from exposures less than 25 rad accumulated dose, and the interpretation of the animal data and the implications for humans were unclear and did not support a specific permissible dose. The data did suggest that genetic damage was more dependent on accumulated dose than previously believed, but experience showed that exposure for prolonged periods to the permissible dose (1.0 R/wk) did not result in any observable genetic effects. The NCRP decided that it was not necessary to change the occupational limit to provide additional protection beyond that provided by the reduction in the permissible dose limit to

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* The exact relationship between roentgen, rad, and rem is beyond the scope of this work. In simple terms, the roentgen is a measure of the degree of ionization induced by x- and gamma radiations in air. The rad (radiation absorbed dose) is a measure of the energy imparted to matter by radiation. The rem (roentgen equivalent man) is a measure of equivalence after incorporating the relative biological effect on human tissue of radiations of different types and energies. Over the range of energies typically encountered, the relationship of roentgens to rads to rem for x- and gamma radiation is essentially equality. For beta radiation, rad are approximately equivalent to rem, and for alpha radiation one rad equals 10 to 20 rem.

0.3 R/wk. At the same time, it recommended limiting the exposure of individuals under the age of 18 to assure that they did not accumulate a genetic dose that would later preclude their employment as radiation workers. The factor of 10 was rather arbitrary, but was believed to be sufficient to protect the future employability of all individuals (NCRP54).

Fourth, the concept of a tolerance dose was replaced by the concept of a maximum permissible dose. The change in terminology reflected the increasing awareness that any radiation exposure might involve some risk and that repair mechanisms might be less effective than previously believed. Therefore, the concept of a maximum permissible dose was adopted because it better reflects the uncertainty in our knowledge than does the concept of tolerance dose. The maximum permissible dose was defined as the level of exposure that entailed a small risk compared with those posed by other hazards in life (ICRP51).

Finally, in explicit recognition of the inadequacy of our knowledge regarding the effects of radiation and of the possibility that any exposure might have some potential for harm, the recommendations included an admonition that every effort should be made to reduce exposure to all kinds of ionizing radiation to the lowest possible level. This concept, known originally as ALAP (as low as practicable) and later as ALARA (as low as reasonably achievable), would become a cornerstone of radiation protection philosophy.

During the 1950's, a great deal of scientific evidence on the effects of radiation became available from studies of the radium dial painters, radiologists, and the survivors of the atomic bombs dropped on Japan. This evidence suggested that genetic effects and long-term somatic effects were more important than previously considered. Thus, by the late 1950's, the ICRP and NCRP recommendations were again revised (ICRP59, NCRP59). These revisions included the following major changes: the annual maximum permissible dose for whole-body exposure and the most critical organs (blood-forming organs, gonads, and the lens of the eye) was lowered to 5 rem, with a quarterly limit of 3 rem; the limit for exposure of other organs was set at 30 rem/yr; internal exposures were controlled by a comprehensive set of maximum permissible concentrations of radionuclides in air and water based on the most restrictive case of a young worker; and recommendations were included for some nonoccupational groups and for the general population (for the first time).

The lowering of the annual maximum permissible whole-body dose to 5 rem, with a quarterly limit of 3 rem, reflects both the new evidence and the uncertainties of the time. Although no adverse effects were observed among workers who had received the earlier maximum permissible dose of 0.3 rad in a week, there was concern that the lifetime accumulation of as much as 750 rad (15 rad/yr times 50 yr) was too much. Lowering the maximum permissible dose by a factor of 3 was believed to provide a greater margin of safety. At the same time, operational

experience showed that an annual dose of 5 rem could be met in most instances, particularly with the additional operational flexibility provided by expressing the limit on an annual and quarterly basis.

The recommendations given for nonoccupational exposures were based on concerns of genetic effects. The evidence available suggested that genetic effects were primarily dependent on the total accumulated dose. Thus, having sought the opinions of respected geneticists, the ICRP and the NCRP adopted the recommendation that accumulated gonadal dose to age 30 be limited to 5 rem from sources other than natural background and medical exposure. As an operational guide, the NCRP recommended that the maximum annual dose to any individual be limited to 0.5 rem, with maximum permissible body burdens of radionuclides (to control internal exposures) set at one-tenth that allowed for radiation workers. These values were derived from consideration of the genetically significant dose to the population, and were established "primarily for the purpose of keeping the average dose to the whole population as low as reasonably possible, and not because of the likelihood of specific injury to the individual" (NCRP59).

During the 1960's, the ICRP and NCRP again lowered the maximum permissible dose limits (ICRP65, NCRP71). The considerable scientific data on the effects of exposure to ionizing radiation were still inconclusive with respect to the dose-response relationship at low exposure levels; thus, both organizations continued to stress the need to keep all exposures to the lowest possible level.

The NCRP and the ICRP made the following similar recommendations:

- limit the dose to the whole-body, red bone marrow, and gonads to 5 rem in any year, with a retrospective limit of 10 to 15 rem in any given year as long as total accumulated dose did not exceed 5x(N-18), where N is the individual's age in years;
- limit the annual dose to the skin, hands, and forearms to 15, 75, and 30 rem, respectively;
- limit the annual dose to any other organ or tissue to 15 rem;
- limit the annual dose to any nonoccupationally exposed individual in the population to 0.5 rem; and
- limit the annual average dose to the population to 0.17 rem.

The scientific evidence and the protection philosophy on which the above recommendations were based were set forth in detail in NCRP71. In the case of occupational exposure limits, the goal of protection was to ensure that the risks of genetic and somatic effects were small enough to be comparable to the risks experienced by workers in other industries.

The conservatively derived numerical limits recommended were based on the linear, nonthreshold, dose-response model, and were believed to represent a level of risk that was readily acceptable to an average individual. For nonoccupational exposures, the goal of protection was to ensure that the risks of genetic or somatic effects were small compared with other risks encountered in everyday life. The derivation of specific limits was complicated by the unknown dose-response relationship at low exposure levels and the fact that the risks of radiation exposure did not necessarily accrue to the same individuals who benefited from the activity responsible for the exposure. Therefore, it was necessary to derive limits that gave adequate protection to each member of the public and to the gene pool of the population as a whole, while still allowing the development of beneficial uses of radiation and radionuclides.

In 1977, the ICRP made a fundamental change in its recommendations when it abandoned the critical organ concept in favor of the weighted whole-body dose equivalent concept for limiting occupational exposure (ICRP77). The change, made to reflect our increased understanding of the differing radiosensitivity of the various organs and tissues, did not affect the overall limit of 5 rem/yr and is not intended to be applied to nonoccupational exposures.

Also significant is the fact that ICRP's 1977 recommendations represent the first explicit attempt to relate and justify permissible radiation exposures with quantitative levels of acceptable risk. Thus, the risks of average occupational exposures (approximately 0.5 rem/yr) are equated with risks in safe industries, given as 10^{-4} annually. At the maximum limit of 5 rem/yr, the risk is equated with that experienced by some workers in recognized hazardous occupations. Similarly, the risks implied by the nonoccupational limit of 0.5 rem/yr are equated to levels of risk of less than 10^{-2} in a lifetime; the general populace's average exposure is equivalent to a lifetime risk on the order of 10^{-3} to 10^{-4}. The ICRP believed these levels of risk were in the range that most individuals find acceptable.

The NCRP has not formally changed its recommendations for occupational exposure to correspond to the 1977 recommendations of the ICRP. It has been working diligently, however, to review its recommendations, and has circulated a draft of proposed changes to various interested scientists and regulatory bodies for their comments. The relevant nonoccupational exposure limits are:

- 0.5 rem/yr whole-body dose equivalent, not including background or medical radiation, for individuals in the population when the exposure is not continuous;

- 0.1 rem/yr whole-body dose equivalent, not including background or medical radiation, for individuals in the population when the exposure is continuous; and

- continued use of a total dose limitation system based on justification of every exposure and application of the ALARA philosophy to every exposure.

The NCRP equates continuous exposure at the level of 0.1 rem/yr to a lifetime risk of developing cancer of about one in a thousand. The NCRP has not formulated exposure limits for specific organs, but it notes that the permissible limits will necessarily be higher than the whole-body limit in inverse ratio of the risk for a particular organ to the total risk for whole-body exposure.

2.3 Federal Guidance

The ICRP and the NCRP function as nongovernmental advisory bodies. Their recommendations are not binding on any user of radiation or radioactive materials. The wealth of new scientific information on the effects of radiation that became available in the 1950's prompted President Eisenhower to establish an official government entity with responsibility for formulating radiation protection criteria and coordinating radiation protection activities. Thus, the Federal Radiation Council (FRC) was established in 1959 by Executive Order 10831. The Council included representatives from all of the Federal agencies concerned with radiation protection, and acted as a coordinating body for all of the radiation protection activities conducted by the Federal Government. In addition to its coordinating function, the Council's major responsibility was to "...advise the President with respect to radiation matters, directly or indirectly affecting health, including guidance for all Federal agencies in the formulation of radiation standards and in the establishment and execution of programs of cooperation with States..." (FRC60).

The Council's first recommendations concerning radiation protection standards for Federal agencies were approved by the President in 1960. Based largely on the work and recommendations of the ICRP and the NCRP, the guidance established the following limits for occupational exposures:

- whole body, head and trunk, active blood-forming organs, gonads, or lens of eye--not to exceed 3 rem in 13 weeks and total accumulated dose limited to 5 times the number of years beyond age 18;

- skin of whole body and thyroid--not to exceed 10 rem in 13 weeks or 30 rem/yr;

- hands, forearms, feet, and ankles--not to exceed 25 rem in 13 weeks or 75 rem/yr;

- bone--not to exceed 0.1 microgram of radium-226 or its biological equivalent; and

- any other organ--not to exceed 5 rem per 13 weeks or 15 rem/yr.

Although these levels differ slightly from those recommended by NCRP and ICRP at the time, the differences do not represent any greater or lesser protection. In fact, the FRC not only accepted the levels recommended by the NCRP for occupational exposure, it adopted the NCRP's philosophy of acceptable risk for determining occupational exposure limits. Although quantitative measures of risk were not given in the guidance, the prescribed levels were not expected to cause appreciable bodily injury to an individual during his or her lifetime. Thus, while the possibility of some injury was not zero, it was so low as to be acceptable if there was any significant benefit derived from the exposure.

The guidance also established exposure limits for members of the public. These were set at 0.5 rem/yr (whole body) for an individual, and an average of 5 rem in 30 yr (gonadal) per capita. The guidance also provided for developing a suitable sample of the population as an operational basis for determining compliance with the limit when doses to all individuals are unknown. Exposure to this population sample was not to exceed 0.17 rem/capita/yr. The population limit of 0.5 rem to any individual per year, was derived from many considerations including natural background exposure.

In addition to the formal exposure limits, the guidance also established the Federal policy that there should be no radiation exposure without an expectation of benefit, and that "every effort should be made to encourage the maintenance of radiation doses as far below this guide as practicable." The inclusion of the requirements to consider benefits and keep all exposure to a minimum was based on the possibility that there is no threshold dose for radiation. The linear, nonthreshold dose response was assumed to result in an upper limit on the estimate of radiation risk. However, the FRC explicitly recognized that it might also represent the true level of risk. If so, then any radiation exposure carried some risk, and it was necessary to avoid all unproductive exposures and to keep all productive exposures as "far below this guide as practicable."

In 1967, the Federal Radiation Council issued guidance for the control of radiation hazards in uranium mining (FRC67). The need for such guidance was clearly indicated by the epidemiological evidence that showed a higher incidence of lung cancer in adult males who worked in uranium mines compared with the incidence in adult males from the same locations who had not worked in mines. The guidance established specific exposure limits and recommended that all exposures be kept as far below the guide limits as possible. The limits chosen represented a trade-off between the risks incurred at various exposure levels, the technical feasibility of reducing the exposure, and the benefits of the activity responsible for the exposure. The guidance also applied to nonuranium mines.

In 1970, the functions of the Federal Radiation Council were transferred to the U.S. Environmental Protection Agency (EPA). In 1971, the EPA revised the Federal guidance for the control of radiation hazards in underground uranium mining (EPA71). Based on the risk levels associated with the exposure limits established in 1967, the upper limit of exposure was reduced by a factor of 3. The EPA has also provided Federal guidance for the diagnostic use of X rays (EPA78). This guidance established maximum skin entrance doses for various types of routine x-ray examinations. It also established the requirement that all x-ray exposures be based on clinical indication and diagnostic need, and that all exposure of patients should be kept as low as reasonably achievable consistent with the diagnostic need.

In 1987, the EPA provided new Federal guidance for occupational exposures to supersede the 1960 guidance (EPA87). The 1987 guidance follows the principles set forth by the ICRP in 1977 with respect to combining internal and external doses. The new occupational limit in the guidance is 5 rem/yr, and exposure of the fetus is limited for the first time to avoid possible radiation-induced effects on health (maximum exposure of the unborn is 0.5 rem during the entire gestation period).

2.4 The Environmental Protection Agency

In addition to the statutory responsibility to provide Federal guidance on radiation protection, the EPA has various statutory authorities and responsibilities regarding protection of the general public from exposure to radiation. The standards and the regulations that EPA has promulgated and proposed with respect to controlling radiation exposures are summarized here.

The AEA and Reorganization Plan No. 3 granted EPA the authority to establish generally applicable environmental standards for exposure to radionuclides. The Ash Memorandum of 1973 (OMB73) established the responsibility for setting offsite radiation protection standards for the total amount of radiation entering the general environment from all facilities in the uranium fuel cycle, but not from specific facilities. Pursuant to this, in 1977, the EPA issued standards limiting exposure from operations of the commercial light-water reactor nuclear fuel cycle (EPA77b). These standards cover normal operations of the uranium fuel cycle, excluding mining and waste disposal. The standards limit the annual dose equivalent to any member of the public from all phases of the uranium fuel cycle to 25 millirem (mrem) to the whole body, 75 mrem to the thyroid, and 25 mrem to any other organ. To protect against the buildup of long-lived radionuclides in the environment, the standards also set normalized emission limits for krypton-85, iodine-129, and plutonium-239 combined with other transuranics with a half-life exceeding one year. The dose limits imposed by the standards cover all exposures (excluding radon and its daughters) resulting from radiation and radionuclide releases to air and water from operations of uranium fuel-cycle facilities.

The development of this standard took into account both the maximum risk to an individual and the overall effect of releases from fuel-cycle operations on the population, and balanced these risks against the costs of effluent control in a primarily qualitative way.

Under the authority of the Uranium Mill Tailings Radiation Control Act, the EPA promulgated standards limiting public exposure to radiation and restricting releases of materials from uranium mill tailings piles (EPA83a, EPA83b). Cleanup standards for land and buildings contaminated with residual radioactive materials from inactive uranium processing sites were also established. In these actions, the Agency sought to balance the radiation risks imposed on individuals and the population in the vicinity of the pile against the feasibility and costs of control.

The Agency established regulations and criteria for the disposal of radioactive waste into the oceans in 1973 under the authority of the Marine Protection, Research and Sanctuaries Act of 1972 (MPRSA). These regulations (40 CFR 220-229), which were revised in 1977, prohibit ocean disposal of high-level radioactive wastes and radiological warfare agents and establish requirements for obtaining ocean disposal permits for other radioactive waste (EPA77a).

In 1983, amendments (USC83) to the MPRSA required that: (1) for a 2-yr period after enactment, EPA could issue only research permits relative to LLW disposal; (2) after the 2-yr restriction, all applicants must prepare and submit to EPA a site-specific Radioactive Material Disposal Impact Assessment; and (3) if EPA determines that a permit should be issued to the applicant, the recommendation must be transmitted to both Houses of Congress and approved by a joint resolution within 90 days of receipt. No permits have been issued to date.

In 1982, EPA issued effluent limitations guidelines for the ore mining and dressing point source category under the Clean Water Act. Subpart C - Uranium, Radium and Vanadium Ores Subcategory of 40 CFR 440 limits, among other items, the concentrations of radium and uranium in effluent discharges from such mines and prohibits the discharge of process wastewater from uranium mills in dry climates.

Under the authority of the Safe Drinking Water Act, the EPA issued interim regulations covering the permissible levels of radium-226 and radium-228, gross alpha (excluding uranium and radon), man-made beta, and photon-emitting contaminants in community water systems (EPA76). The limits are expressed in picocuries/liter. The limits chosen for man-made beta- and photon-emitters equate to approximately 4 mrem/yr whole-body or organ dose to the most exposed individual. In the background information for the standard, the 4 mrem/yr exposure through a single pathway that the standard permits is explicitly compared with the overall population standard of 170 mrem/yr, and the conclusion is expressed that the roughly 40-fold decrease is appropriate for a single pathway.

Section 122 of the Clean Air Act amendments of 1977 (Public Law 95-95) directed the Administrator of EPA to review all relevant information and determine if emissions of hazardous pollutants into air will cause or contribute to air pollution that may reasonably be expected to endanger public health. In December 1979, EPA designated radionuclides as hazardous air pollutants under Section 112 of the Act (40 CFR 61). In 1985 and 1986, EPA published National Emission Standards for radionuclides from DOE facilities, NRC-licensed facilities, elemental phosphorus plants, underground uranium mines, and NRC-licensed uranium mill tailings (EPA85a, 85b, 86).

The DOE and NRC facilities' radionuclide emissions to air are limited to that amount which will cause a dose equivalent of 25 mrem/yr to the whole body or 75 mrem/yr to the critical organ of any member of the public. The phosphate plants have an annual release limit of 21 curies of polonium-210, while the mines and mill tailings standards are based on Radon-222 emission control technology requirements.

In 1985, under the authority of the AEA, the EPA issued standards (40 CFR 191) for disposal of spent nuclear fuel, high-level and transuranic radioactive wastes (EPA85c). The standards apply to NRC-licensed facilities and non-NRC-licensed DOE facilities. The standards are divided into two subparts: Under Subpart A, waste management and storage operations must be conducted so that no member of the public receives an annual dose greater than that allowed for other phases of the uranium fuel cycle (i.e., 25 mrem to whole body, 75 mrem to thyroid, and 25 mrem to any other critical organ for NRC-licensed facilities and 25 mrem to whole body and 75 mrem to any critical organ for DOE-operated facilities). Subpart B requires that once the repository is closed, exposure is to be controlled by limiting releases over 10,000 years. The release limits were derived by summing, over long time periods, the estimated risks to all persons exposed to radioactive materials released into the environment. The uncertainties involved in estimating the performance of a theoretical repository led to this unusual approach. In addition to the containment requirements, individual protection requirements provide limits (i.e., 5 mrem to whole body and 75 mrem to any critical organ) for the first 1,000 years after disposal, ground-water protection requirements provide limits for special sources of ground water (the same as 40 CFR 141--the Interim Drinking Water Standards), and 6 assurance requirements are set forth for DOE facilities to provide an extra margin of insurance that the containment requirements will be met despite the large uncertainties that confront the prediction of disposal system performance over 10,000 years. In 1986 several petitions for review of the 40 CFR 191 standards were consolidated in the U.S. Court of Appeals for the First Circuit. The outcome of the action was that Subpart B was remanded to the Agency for further action.

EPA is also proposing to develop guidance and standards for such other areas as land cleanup and residual radioactivity, LLW disposal, and

NARM waste disposal. Besides the 1983 Advanced Notice of Proposed Rulemaking for LLW disposal (see Chapter 1), in June 1986 EPA issued an Advance Notice of Proposed Rulemaking (40 CFR 194) for radiation protection criteria for cleanup of land and facilities contaminated with residual radioactive materials.

The main authority EPA uses to promulgate the various waste disposal standards is the AEA. This authority, which covers radionuclides classified as source, special nuclear, and by-product materials, does not cover two broad classes of radionuclides. These two classes are (1) naturally occurring radionuclides that are not considered source material, such as radium and lower concentration uranium and thorium (less than 0.05 percent by weight), and (2) radionuclides produced by particle accelerators. These two groups are included in the NARM classification discussed in Chapter 1.

Because NARM is not covered by the AEA, EPA has to use another authority to regulate these radioactive materials. The authority EPA is proposing to use is TSCA, which regulates commerce and protects human health and the environment by requiring testing and necessary use restrictions on certain chemical substances. Section 6(a)(6) of TSCA authorizes EPA to prohibit or regulate the disposal of chemical substances or mixtures. This section enables EPA to establish requirements for the proper disposal of NARM wastes.

2.5 Nuclear Regulatory Commission

Under the authority of the AEA, NRC is responsible for licensing and regulating the use of by-product, source, and special nuclear material, and for assuring that all licensed activities are conducted in a manner that protects public health and safety. The NRC has no authority over the licensing or regulation of NARM wastes that are exempt from the AEA. The Federal guidance on radiation protection applies directly to the NRC; therefore, the NRC must assure that none of the operations of its licensees exposes an individual of the public to more than 0.5 rem/yr from all pathways. The dose limits imposed by the EPA's standard for uranium fuel-cycle facilities (40 CFR 190) also apply to the fuel-cycle facilities licensed by the NRC. These facilities are prohibited from releasing radioactive effluents in amounts that would result in doses greater than the 25 mrem/yr limit imposed by that standard.

The NRC exercises its statutory authority by imposing a combination of design criteria, operating parameters, and license conditions at the time of construction and licensing. It assures that the license conditions are fulfilled through inspection and enforcement. The NRC and its Agreement States license more than 20,000 users of radioactivity.

2.5.1 Fuel-Cycle Licensees

The NRC does not use the term "fuel-cycle facilities" to define its classes of licensees. The term is used here to coincide with the EPA use of the term in its standard for uranium fuel-cycle facilities. As a practical matter, this term includes the NRC's large source and special nuclear material, and production and utilization facilities. The NRC's regulations require an analysis of probable radioactive effluents and their effects on the population near fuel-cycle facilities. The NRC also assures that all exposures are as low as reasonably achievable by imposing design criteria and specific equipment requirements on the licensees. After a license has been issued, fuel-cycle licensees must monitor their emissions and take environmental measurements to assure that the design criteria and license conditions have been met. For practical purposes, the NRC adopted the maximum permissible concentrations developed by the NCRP as a basis for relating effluent concentrations to exposure.

In the 1970's, the NRC formalized the implementation of As Low As Reasonably Achievable (ALARA) exposure levels by issuing a regulatory guide for achieving these levels through design criteria. This coincided with a decision to adopt, as a design criterion, a maximum annual permissible dose of 5 mrem from a single nuclear electric generating station. The 5-mrem limit applies to the most exposed individual actually living in the vicinity of the reactor, and refers to whole-body doses from external radiation by the air pathway, plus a 3-mrem limit to the whole body by liquid pathways (NRC77).

2.5.2 By-product Material Licensees

The NRC's licensing and inspection procedure for by-product material users is less uniform than that imposed on major fuel-cycle licensees for two reasons: (1) the much larger number of such licensees, and (2) the much smaller potential for releasing significant quantities of radioactive materials into the environment. The prelicensing assurance procedures of imposing design reviews, operating practices, and license conditions prior to construction and operation are similar. The amount of protection that is afforded the public from releases of radioactive materials from these facilities can vary considerably because of three factors. First, the requirements that the NRC imposes for monitoring effluents and environmental radioactivity are much less stringent for these licensees. If the quantity of materials handled is small enough, the NRC might not impose any monitoring requirements. Second, and more important, the level of protection can vary considerably because the point where the licensee must meet the effluent concentrations for an area of unrestricted access is not consistently defined. Depending on the particular licensee, this area has been defined as the nearest inhabited structure, as the boundary of the user's property line, as the roof of the building where the effluents are vented, or as the mouth of the stack or vent. Finally, not all users are allowed to reach

100 percent of the permissible concentrations in their effluents. In fact, the NRC has implemented as low as reasonably achievable considerations on many of these licensees by limiting them to 10 percent of the maximum permissible concentration in their effluents.

2.5.3 Radioactive Waste Disposal Licensees

The NRC's requirements for radioactive waste disposal are contained in 10 CFR 60, Disposal of High-Level Radioactive Wastes in Geologic Repositories: Technical Criteria (NRC83). The NRC has also issued a package of amendments to 10 CFR Parts 2, 19, 20, 21, 30, 40, 51, 60, and 70, entitled "Disposal of High-Level Radioactive Wastes in Geologic Repositories: Licensing Procedures" (NRC81a); and another to amend 10 CFR Parts 2, 19, 20, 21, 30, 40, 51, 61, 70, 73, and 170, entitled "Licensing Requirements for Land Disposal of Radioactive Waste" (NRC82); 10 CFR 40, Uranium Mill Licensing Requirements (NRC80); and 10 CFR 20.301, Biomedical Waste Disposal (NRC81b). The NRC has also issued policy statements on radioactive waste that is below regulatory concern (NRC86) and on LLW volume reduction (NRC81c).

Specifically, the 10 CFR 61 regulations establish performance objectives for: land disposal of LLW; technical requirements for the siting, design, operations, and closure activities for a near-surface disposal facility; technical requirements concerning the waste form that waste generators must meet for the land disposal of waste; classification of waste; institutional requirements; and administrative and procedural requirements for licensing a disposal facility.

The performance objectives provide for the protection of the general population from releases of radioactivity, so that no release results in an annual dose exceeding 25 mrem to the whole body, 75 mrem to the thyroid, and 25 mrem to any other organ.

The biomedical waste disposal rule provides for the disposal of liquid scintillation media and animal carcasses containing tracer levels (0.05 microcurie or less per gram) of tritium or carbon-14 without regard to their radioactivity (NRC81b).

2.6 Department of Energy

The U.S. Department of Energy (DOE) operates a complex of national laboratories and weapons facilities. These facilities are not licensed by the NRC. Under the AEA, the DOE is responsible for keeping radionuclide emissions at these facilities as low as reasonably achievable. The EPA has promulgated a final standard, consistent with the requirements of the Clean Air Act, that sets the maximum radionuclide air emissions from DOE facilities to that amount which will cause a dose equivalent of 25 mrem/year to the whole body or 75 mrem/year to the critical organ of any member of the public. These limits generally reflect current emission levels achieved by existing control technology and operating practices at DOE facilities (EPA85a).

For practical purposes, the DOE has adopted the NCRP's maximum permissible concentrations in air and water as a workable way to assure that the Federal guidance annual dose limits of 0.5 rem whole body and 1.5 rem to any organ are being observed. The DOE also has a requirement that all doses be kept ALARA; however, latitude is provided to DOE's Operations Offices in determining when policies and procedures are either appropriate or inappropriate for assuring that all doses are kept to the lowest reasonably achievable level.

The DOE assures that its operations are within its operating guidelines by requiring its contractors to maintain radiation monitoring systems around each of its sites and to report the results in an annual summary report. New facilities and modifications to existing facilities are subject to extensive design criteria reviews, and each requires the preparation of an Environmental Impact Statement pursuant to the National Environmental Policy Act of 1970 (NEPA70). In the mid-1970's, the DOE initiated an effluent-reduction program that resulted in the upgrading of many facilities and effected a corresponding reduction in the effluents (including airborne and liquid radioactive materials) released to the environment.

2.7 Department of Transportation

The U.S. Department of Transportation (DOT) has statutory responsibility for regulating the shipment and transportation of radioactive materials. This authority includes the responsibility to protect the public from exposure to radioactive materials while they are in transit. For practical purposes, the DOT has implemented its authority through the specification of performance standards for shipment containers, and by setting maximum exposure rates from any package containing radioactive materials. These limits were set to assure compliance with the Federal guidance for occupational exposure, and they are believed to be sufficient to protect the public from exposure. The DOT also controls potential public exposure by managing the routing of some radioactive shipments to avoid densely populated areas.

DOT regulations include a statutory definition (49 CFR 173.403(y)) of radioactive materials that requires a material to contain greater than 0.002 microcurie per gram of radioactivity to be considered radioactive (CFR85). However, in 1985, the DOT amended its regulations to exempt materials covered by the NRC biomedical waste disposal rule (10 CFR 20.306) from DOT requirements pertaining to radioactive materials when transported for disposal or recovery (DOT85).

2.8 State Agencies

States have authority for protecting the public from the hazards associated with ionizing radiation. Twenty-nine States have assumed NRC's inspection, enforcement, and licensing responsibilities for users of source and by-product materials and users of small quantities of

special nuclear material. These "NRC-agreement States," which license and regulate more than 11,500 users of radiation and radioactive materials, are bound by formal agreements to adopt requirements consistent with those imposed by the NRC. The NRC continues to perform this function for all licensable uses of source, by-product, and special nuclear material in the 24 States that are not agreement States.

Nonagreement States, as well as NRC-agreement States, retain the authority to regulate the use of NARM (i.e., radium). Many States regulate NARM in the same manner as AEA-regulated materials, although other States do not regulate it at all. Because of this, the level of NARM regulation varies considerably from State to State, leading to uncertainty in protecting the public during its use and for disposal.

The passage of the LLRWPA and the subsequent Amendments Act (see Chapter 1) by Congress directed each State by 1993 to provide disposal capacity for all commercial LLW generated within its borders either individually or through regional compacts.

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Chapter 3: QUANTITIES, SOURCES, CHARACTERISTICS, AND DISPOSAL OF LOW-LEVEL RADIOACTIVE WASTE

3.1 Description of Low-Level Radioactive Waste

Low-level radioactive waste encompasses basically all radioactive wastes except those that are specifically defined as another class of radioactive waste. Thus, radioactive wastes that are not LLW include spent nuclear fuel, high-level radioactive wastes, and transuranic wastes as defined in EPA 40 CFR 191 (EPA85) and uranium and thorium by-product materials (mill tailings) as defined by Congress in the Uranium Mill Tailings Radiation Control Act of 1978 (P.L. 95-604), which amends the AEA. All other radioactive wastes are low level.

For regulatory purposes, LLW is divided into two classes: that controlled under the AEA and that which cannot be controlled under the AEA. However, for all practical purposes, most radioactive waste is controlled under the AEA. Radioactive wastes that are not controlled are limited to a few NARM radionuclides which are not source or special nuclear material (as defined by the AEA).

3.2 Quantities and Sources of Low-Level Radioactive Waste

Low-level waste is generated by government, utilities, industries, and institutional facilities. Virtually all AEA LLW may be partitioned into two regulatory categories, commercial LLW and DOE/defense LLW. The waste regulated by the NRC is referred to as commercial LLW and includes nuclear fuel-cycle, industrial, and institutional LLW. The DOE, of course, regulates the disposal of its DOE/defense LLW. Both the NRC and DOE regulate LLW under the AEA.

DOE has classified all of its LLW as falling into one of six general categories: (1) uranium/thorium; (2) fission products; (3) induced activity; (4) tritium; (5) alpha or transuranic (< 100 nCi/g); and (6) other. Many of the DOE/defense facilities are one-of-a-kind. However, DOE has indicated that its LLW is similar to LLW produced in the commercial sector (DOE84).

The NRC has developed an extraordinary amount of data describing the numerous commercial LLW categories falling under the purview of its 10 CFR 61 rulemaking (NRC81b, NRC82, NRC86). Each waste category, which is called a waste stream, consists of a consolidation of several groups of wastes having similar sources and physical, chemical, and radiological characteristics. The assessment for the NRC's final environmental impact statement examined 37 waste streams (NRC82). Routine commercial LLW was analyzed in these assessments. A more recent assessment by NRC has built upon the previous assessments to create a much more detailed and comprehensive LLW source term (NRC86). A total of 148 waste streams are described, with greater emphasis on higher activity wastes and

nonroutine sources of LLW generation. So-called nonroutine wastes from such sources as uranium fuel reprocessing activities, the DOE West Valley Demonstration Project, and decontamination of the Three Mile Island Unit 2 nuclear power plant are also characterized in this LLW assessment (NRC86).

Other sources of routine LLW include Light-Water Reactor (LWR) decommissioning, the Formerly Utilized Sites Remedial Action Program (FUSRAP), and the Surplus Facilities Management Program (SFMP). The FUSRAP has been developed to provide for decontamination and decommissioning (D&D) of facilities used many years ago in the Manhattan Engineering District and AEC operations. The SFMP includes approximately 500 DOE facilities, covering a wide range of facilities and disposal areas. These facilities are to be decontaminated to minimize hazards to public health and allow reuse of certain facilities. The LWR decommissioning is another routine source of LLW that will be produced in the future (Ro82, DOE86). Table 3-1 provides an overall picture of current and projected LLW generation. Volumes of LLW anticipated until the year 2020 are dominated by routine LLW arising from commercial and DOE sources. Nonroutine LLW contribute very small volumes to LLW as a whole. It should be noted that the volumes shown for LWR D&D presume that such activities begin within a few years of reactor shutdown. The start of actual LWR D&D activities may be much later to allow plant radiation levels to decrease. Both the volume and activity associated with such activities decrease significantly after a 50-yr period of storage (DOE86). FUSRAP and SFMP primarily contribute radionuclides associated with the processing of uranium and thorium. Commercial LWR decommissionings are projected to contribute mainly low specific activity wastes. Approximately 98 percent of the reactor D&D wastes are expected to be Class A, 1.5 percent Class B, and the remainder Class C or greater-than-Class C LLW, where 10 CFR 61 defines the waste classifications given above (DOE86).

Though the discovery and application of radioactivity originated with NARM radionuclides, the present regulatory framework of NARM is inconsistent. NARM wastes are not covered under the AEA, which was created to assure controls on radionuclides associated with the various aspects of nuclear fission (NRC77). In order to provide a basis on which to consider the regulation of NARM, EPA commissioned a study of NARM wastes (PEI85). Special emphasis was placed on higher specific activity wastes and those exhibiting characteristics analogous to LLW regulated under the AEA. The NARM waste streams included in EPA's LLW radiological source term are based on this study.

To characterize waste streams appropriate for an analysis of the applicability of the BRC concept to LLW disposal, a select group of LLW waste streams was constructed. These waste streams have been designated as surrogate BRC wastes, since such wastes do not yet exist, at least in

Table 3-1. Current and projected cumulative quantities of LLW[a]

Source of material	Year 1985	2000	2010	2020
		ROUTINE LLW		
DOE/Defense, 10^3 m^3				
LLW-Routine	2,181	4,043	5,159	6,256
FUSRAP LLW	119	870	870	870
SFMP LLW	24	791	800	800
Commercial, 10^3 m^3				
LLW (No Reprocessing)	1,160	2,441	3,545	4,972
D&D LLW	—	0.6	91.3	795.9
		NONROUTINE LLW		
Facility, 10^3 m^3				
West Valley D&D	3.0	12.9	12.9	12.9
Three Mile Is. II D&D	3.5	9.3[b]	9.3	9.3
Fuel Reprocessing	—	6.7[c]		
Mixed Oxide Fuel Fabrication	—	0.5[c]		

[a]All figures from DOE86, except as noted.
[b]From NRC81b, Appendix D, Worst-Case Conditions.
[c]Volumes are for one year of operation. The fuel reprocessing plant has a capacity of 2,000 MTHM/yr. The mixed oxide fuel fabrication plant handles 400 MTHM/yr. Volumes are from NRC86.

the context of individual waste streams deregulated within the framework of a generic BRC level for LLW disposal. In general, the surrogate BRC waste streams are lower activity LLW waste streams, or where enough information is available, substreams of previously defined LLW waste streams. Various waste generators are represented by these surrogate BRC wastes, including nuclear fuel-cycle, institutional, and industrial LLW generators.

The following sections describe in more detail the basis and form of the source term used in the EPA risk assessment for the land disposal of both LLW and BRC wastes.

3.3 EPA Low-Level Radioactive Waste Source Term

EPA has developed a LLW source term tailored to the requirements of a radiological risk assessment supporting an environmental radiation protection standard for all LLW. This source term consists of three separate but complementary LLW radiological sources: (1) LLW defined under the AEA, (2) NARM, and (3) BRC.

3.3.1 Low-Level Radioactive Waste Regulated Under the AEA

Over the past few years, the NRC has developed an extensive, systematic characterization of LLW from commercial nuclear fuel-cycle facilities (NRC81b, Wi81, NRC82, NRC86). The Agency has relied upon this massive LLW data base to derive a LLW source term more suitable for EPA's risk assessment models. The inherent complexity of the EPA risk models necessitated the creation of a condensed version of the NRC LLW source term. Table 3-2 illustrates a comparison of the waste streams defined by NRC in the draft EIS for its 10 CFR 61 rulemaking and the resulting condensed waste streams defined by EPA. The NRC's 1982 final EIS for 10 CFR 61 (NRC82) and a later issued update report (NRC86) further supplemented the 1981 draft EIS LLW source term (NRC81b). In particular, the update report expands the number of waste streams to 148. Where the NRC waste streams have been broken down into substreams, EPA has used this more detailed data to properly weight each substream and condense them back to the original EPA waste streams in Table 3-2. As explained in the previous section, because of the great uncertainty and relatively small contributions of nonroutine commercial LLW streams (e.g., fuel reprocessing, power reactor decommissioning), such wastes are not included in the EPA LLW source term. For similar reasons, LLW from FUSRAP and SFMP are not included in EPA's LLW source terms. Table 3-3 lists the shorthand symbols and corresponding waste form descriptions for the waste streams regulated under the AEA and included in the EPA LLW source term.

For the most part, radionuclide concentrations are based on the NRC update report of the 10 CFR 61 analysis methodology (NRC86). This analysis includes numerous short-lived radionuclides for some waste

Table 3-2. Waste groups and streams

Waste streams (NRC81b)	EPA condensed waste streams
Group I: LWR Process Wastes	**Group I:**
PWR Ion Exchange Resins	LWR Ion Exchange Resins
PWR Concentrated Liquids	LWR Concentrated Liquids
PWR Filter Sludges	LWR Filter Sludges
PWR Filter Cartridges	PWR Filter Cartridges
BWR Ion Exchange Resins	
BWR Concentrated Liquids	
BWR Filter Sludges	
Group II: Trash	**Group II:**
PWR Compactible Trash	LWR Compactible Trash
PWR Noncompactible Trash	LWR Noncompactible Trash
BWR Compactible Trash	Fuel Fabrication Compactible Trash
BWR Noncompactible Trash	Fuel Fabrication Noncompactible Trash
Fuel Fabrication Compactible Trash	Institutional Trash
Fuel Fabrication Noncompactible Trash	Industrial SS Trash
Institutional Trash (large facilities)	Industrial Low Trash
Institutional Trash (small facilities)	
Industrial SS Trash (large facilities)	
Industrial SS Trash (small facilities)	
Industrial Low Trash (large facilities)	
Industrial Low Trash (small facilities)	
Group III: Low Specific Activities Wastes	**Group III:**
Fuel Fabrication Process Wastes	Fuel Fabrication Process Wastes
UF_6 Process Wastes	UF_6 Process Wastes
Institutional LSV Waste (large facilities)	Institutional LSV Waste
Institutional LSV Waste (small facilities)	Institutional Liquid Waste
Institutional Liquid Waste (large facilities)	Institutional Biowaste
Institutional Liquid Waste (small facilities)	Industrial SS Waste
Institutional Biowaste (large facilities)	Industrial Low Activity Waste
Institutional Biowaste (small facilities)	
Industrial SS Waste	
Industrial Low Activity Waste	

Table 3-2. Waste groups and streams (continued)

Waste stream (NRC81b)	EPA condensed waste streams
Group IV: Special Wastes	Group IV:
LWR Decontamination Resins	LWR Decontamination Resins
Waste from Isotope Production Facilities	Waste from Isotope Production Facilities
Tritium Production Waste	Tritium Production Waste
Accelerator Targets	Accelerator Targets
Sealed Sources	Sealed Sources
High Activity Waste	LWR Nonfuel Reactor Components (incl. High Activity Waste)
LWR Nonfuel Reactor Components	

BWR: Boiling water reactor
LSV: Liquid scintillation vial
LWR: Light water reactor
NFRCOMP: LWR nonfuel reactor components
PWR: Pressurized water reactor
SS: Source and special nuclear material

Table 3-3. Symbols and descriptions of EPA's AEA low-level radioactive waste streams

Symbol	Waste stream description
Nuclear Fuel-Cycle	
L-IXRESIN	LWR Ion Exchange Resins
L-CONCLIQ	LWR Concentrated Liquids
L-FSLUDGE	LWR Filter Sludges
P-FCARTRG	PWR Filter Cartridges
L-DECONRS	LWR Decontamination Resins
L-NFRCOMP	LWR Nonfuel Reactor Components
L-COTRASH	LWR Compactible Trash
L-NCTRASH	LWR Noncompactible Trash
F-PROCESS	Fuel Fabrication Process Waste
U-PROCESS	Uranium Conversion Process Waste
F-COTRASH	Fuel Fabrication Compactible Trash
F-NCTRASH	Fuel Fabrication Noncompactible Trash
Institutional Wastes	
I-LQSCNVL	Liquid Scintillation Vials
I-ABSLIQD	Various Absorbed Liquids
I-BIOWAST	Biological Wastes
I-COTRASH	Mostly Compactible/Combustible Trash
Industrial Wastes	
N-LOWASTE	Low-Activity Waste
N-LOTRASH	Low-Activity Trash
N-SSTRASH	Source and Special Nuclear Material Trash
N-SSWASTE	Source and Special Nuclear Material Waste
N-ISOPROD	Isotope Production Wastes (Medical)
N-SOURCES	Sealed Sources
N-TRITIUM	Production of H-3, C-14 Labeled Products
N-TARGETS	Accelerator Targets

LWR: Light water reactor
PWR: Pressurized water reactor
Note: L-NFRCOMP includes the much smaller industrial high activity waste stream.

streams. Since the EPA risk analysis is principally directed at estimating and comparing impacts over the long term, the EPA LLW source term considers only isotopes with half-lives of a year or more. Other factors used to qualify radionuclides for inclusion in the EPA source term were relatively high radiological toxicity and being present in significant amounts in LLW. Table 3-4 lists the radionuclides, along with their decay constant and half-life, considered in the EPA risk analysis for LLW regulated under the AEA (Gr86).

Tables 3-5 and 3-6 provide the radionuclide concentrations in curies per cubic meter (Ci/m^3) and waste volumes (m^3) projected for 1985-2004, respectively, for LLW regulated under the AEA (Gr86). Later sections in this chapter describe in more detail the rationale for selection of the radionuclide concentrations and volumes representing each waste stream.

Table 3-7 lists, on a waste stream-by-waste stream basis: (1) the waste class (A, B, or C) as per 10 CFR 61.55; (2) the "as-generated" waste form; (3) the optional waste forms afforded by various treatment options; and (4) a brief description of the waste stream. The PRESTO analysis covers four forms of LLW: trash, absorbing waste, activated metal, and solidified waste. Liquid waste disposal via shallow-land disposal is not practiced anymore. Use of a high integrity container (HIC) involves the placement of the as-generated waste form in an HIC, which is postulated to have a container lifetime of 300 yr.

(A) Nuclear Fuel-Cycle

Nuclear fuel-cycle wastes are generated from various facilities associated with the commercial generation of electricity. The present nuclear fuel-cycle is termed a once-through uranium fuel-cycle because once the nuclear fuel is used to produce electricity, the resulting spent fuel becomes a waste product.

In the once-through uranium fuel-cycle, uranium ore is extracted from the earth at uranium mines and refined to yellowcake at uranium mills. This yellowcake is shipped to conversion plants where the yellowcake (U_3O_8) is converted to uranium hexafluoride (UF_6). Up to this point, the uranium contains its naturally occurring ratios of the various uranium isotopes. In the next step, the UF_6 is sent to an enrichment facility, where the content of fissile U-235 is raised from about 0.7 weight percent to about 2 to 4 weight percent. This amount of U-235 is required to allow for a continuous fissioning process in the cores of today's light-water-cooled nuclear power reactors.

The enriched UF_6 is next shipped to a fuel fabrication plant that processes UF_6 into uranium dioxide (UO_2) pellets for insertion into long, cylindrical metal rods, called fuel rods. Fuel rods are grouped into fuel assemblies and shipped to nuclear power plants. Depending upon the design of the nuclear power plant, around 200 to 700 fuel assemblies

Table 3-4. Radionuclides considered in the EPA source term for LLW regulated under the AEA (Gr86)

Nuclide	Decay constant (yr^{-1})	Half-life (yr)
Hydrogen-3	5.64E-02*	1.23E+01
Carbon-14	1.21E-04	5.73E+03
Iron-55	2.57E-01	2.60E+00
Nickel-59	8.66E-06	8.00E+04
Cobalt-60	1.32E-01	5.26E+00
Nickel-63	7.53E-03	9.20E+01
Strontium-90	2.42E-02	2.81E+01
Niobium-94	3.47E-05	9.59E-02
Technetium-99	3.25E-06	2.12E+05
Ruthenium-106	6.89E-01	1.00E+00
Antimony-125	2.50E-01	2.70E+00
Iodine-129	4.08E-08	1.70E+07
Cesium-134	3.36E-01	2.05E+00
Cesium-135	2.30E-07	3.00E+06
Cesium-137	2.31E-02	3.00E+01
Barium-137m	1.43E+05	4.80E-06
Europium-154	8.15E-02	1.60E+01
Uranium-234	2.83E-06	2.47E+05
Uranium-235	9.85E-10	7.1E+08
Neptunium-237	3.30E-07	2.14E+09
Uranium-238	1.55E-10	4.51E+09
Plutonium-238	7.90E-03	8.6E+01
Plutonium-239	2.87E-05	2.44E+04
Plutonium-241	5.25E-02	1.32E+01
Americium-241	1.51E-03	4.58E+02
Plutonium-242	1.83E-06	3.79E+05
Americium-243	9.40E-05	7.95E+03
Curium-243	2.17E-02	3.2E+01
Curium-244	3.94E-02	1.76E+01

*See Appendix A scientific notation section for an explanation of the E notation system.

Table 3-5. Radionuclide concentrations of AEA* waste streams (Gr86)
(Ci/m^3)

NUCLIDE	Waste stream							
	L-IXRESIN	L-CONCLIQ	L-FSLUDGE	P-FCARTRG	L-DECONRS	L-NFRCOMP	F-PROCESS	U-PROCESS
H-3	3.42E-01	1.89E-02	1.36E-02	2.77E-03				
C-14	1.28E-02	7.10E-04	8.29E-04	1.02E-04		6.43E-03		
Fe-55	8.19E-01	1.95E-01	1.56E+00	1.34E+00	2.63E+00	5.54E+01		
Ni-59	8.89E-04	2.20E-04	1.62E-03	1.59E-03		3.45E-02		
Co-60	1.44E+00	3.58E-01	2.62E+00	2.58E+00	1.89E+01	3.98E+01		
Ni-63	1.19E-01	4.59E-02	5.32E-02	4.91E-01	9.96E-01	4.76E+00		
Sr-90	2.62E-02	1.45E-03	2.50E-03	2.02E-04				
Nb-94	2.82E-05	6.98E-06	5.10E-05	5.03E-05		2.04E-04		
Tc-99	1.45E-04	8.12E-06	5.36E-05	8.62E-07				
Ru-106	3.87E-03	2.16E-04	1.39E-03	2.30E-05	8.46E-01			
Sb-125	1.16E-02	2.86E-03	2.09E-02	2.06E-02	1.88E-03			
I-129	4.18E-04	2.33E-05	1.39E-04	2.55E-06				
Cs-134	3.87E+00	2.16E-01	1.39E+00	2.30E-02				
Cs-135	1.45E-04	8.12E-06	5.24E-05	8.62E-07				
Cs-137	3.87E+00	2.16E-01	1.39E+00	2.30E-02				
Ba-137m	3.87E+00	2.16E-01	1.39E+00	2.30E-02				
Eu-154	1.16E-03	2.87E-04	2.10E-03	2.07E-03	3.76E-05			
U-234	1.59E-04	9.62E-06	9.95E-06	2.36E-05			5.20E-04	3.64E-04
U-235	2.55E-06	1.54E-07	1.60E-07	3.79E-07			2.30E-05	1.65E-05
Np-237	1.14E-09	6.89E-11	7.14E-11	1.69E-10				
U-238	4.65E-05	2.82E-06	2.92E-06	6.91E-06			8.54E-05	3.64E-04
Pu-238	3.29E-03	4.66E-04	4.95E-04	6.05E-04	1.13E-02			
Pu-239	2.30E-03	2.68E-04	2.72E-04	9.15E-04	7.52E-03			
Pu-241	1.01E-01	1.21E-02	1.32E-02	4.00E-02				
Am-241	2.35E-03	2.76E-04	2.08E-04	3.95E-04				
Pu-242	5.04E-06	5.76E-07	5.41E-07	2.01E-06				
Am-243	1.58E-04	1.86E-05	1.40E-05	2.65E-05				
Cm-243	1.25E-06	3.16E-07	3.62E-07	4.65E-07	1.13E-02			
Cm-244	1.73E-03	3.03E-04	2.63E-04	2.65E-04	3.76E-03			
TOTAL	1.45E+01	1.29E+00	8.46E+00	4.54E+00	2.34E+01	1.00E+02	6.28E-04	7.45E-04

*Wastes regulated under the Atomic Energy Act.

Table 3-5. Radionuclide concentrations of AEA waste streams (continued)
(Ci/m^3)

NUCLIDE	Waste stream							
	I-LQSCNVL	I-ABSLIQD	I-BIOWAST	N-LOWASTE	N-ISOPROD	N-SOURCES	N-TRITIUM	N-TARGETS
H-3	5.01E-03	1.42E-01	1.75E-01	1.63E-02	5.52E-02	2.88E+01	2.21E+02	7.80E+02
C-14	2.51E-04	8.16E-03	1.01E-02	9.36E-04	7.79E-05	4.57E-03	2.76E-01	
Fe-55					9.64E-01			
Ni-59								
Co-60		3.12E-02	3.99E-03	1.47E-03	1.48E+00	2.24E+01		
Ni-63					1.48E-02	1.56E-02		
Sr-90	4.34E-03	4.34E-03	8.33E-03	1.31E-03	7.09E+01	3.77E+01		
Nb-94								
Tc-99		1.02E-08	6.51E-09	7.76E-10	5.10E-06			
Ru-106					1.46E-01			
Sb-125								
I-129					4.24E-08			
Cs-134					4.70E-01			
Cs-135					5.10E-06			
Cs-137		1.37E-02	8.76E-03	1.04E-03	4.78E+00	4.45E+02		
Ba-137m		1.37E-02	8.76E-03	1.04E-03	4.78E+00	4.45E+02		
Eu-154								
U-234					1.20E-03			
U-235					3.15E-05			
Np-237					6.20E-15			
U-238					3.47E-07			
Pu-238					2.29E-06	8.89E-01		
Pu-239					6.45E-07			
Pu-241					8.25E-05			
Am-241					4.50E-02	1.47E+00		
Pu-242					1.11E-09			
Am-243					1.46E-08			
Cm-243					3.35E-09			
Cm-244					1.93E-06			
TOTAL	9.60E-03	2.13E-01	2.15E-01	2.21E-02	8.37E+01	9.81E+02	2.21E+02	7.80E+02

Table 3-5. Radionuclide concentrations of AEA waste streams (continued)
(Ci/m^3)

NUCLIDE	Waste stream							
	L-COTRASH	L-NCTRASH	F-COTRASH	F-NCTRASH	I-COTRASH	N-LOTRASH	N-SSTRASH	N-SSWASTE
H-3	3.56E-04	3.17E-03			9.13E-02	2.85E-02		
C-14	1.39E-05	1.19E-04			5.26E-03	1.64E-03		
Fe-55	9.19E-03	6.87E-02						
Ni-59	1.05E-05	8.09E-05						
Co-60	1.71E-02	1.31E-01			1.04E-02	3.25E-03		
Ni-63	2.41E-03	2.24E-02						
Sr-90	2.96E-05	2.43E-04			1.45E-03	4.53E-04		
Nb-94	3.33E-07	2.56E-06						
Tc-99	2.26E-07	1.32E-06			3.39E-09	1.06E-09		
Ru-106	6.01E-06	3.54E-05						
Sb-125	1.36E-04	1.05E-03						
I-129	6.32E-07	3.82E-06						
Cs-134	6.01E-03	3.54E-02						
Cs-135	2.26E-07	1.33E-06						
Cs-137	6.01E-03	3.54E-02			4.56E-03	1.42E-03		
Ba-137m	6.01E-03	3.54E-02			4.56E-03	1.42E-03		
Eu-154	1.37E-05	1.05E-04						
U-234	2.43E-07	2.19E-06	2.68E-05	2.56E-05			2.56E-06	4.97E-05
U-235	3.89E-09	3.52E-08	1.18E-06	1.13E-06			1.42E-07	2.77E-06
Np-237	1.74E-12	1.57E-11						
U-238	7.11E-08	6.43E-07	4.40E-06	4.20E-06			8.80E-06	1.71E-04
Pu-238	7.46E-06	6.39E-05						
Pu-239	6.49E-06	5.75E-05						
Pu-241	2.85E-04	2.52E-03						
Am-241	4.69E-06	4.14E-05			4.82E-06	1.51E-06		
Pu-242	1.41E-08	1.26E-07						
Am-243	3.33E-08	2.80E-06						
Cm-243	3.84E-09	3.04E-08						
Cm-244	3.50E-06	2.84E-05						
TOTAL	4.76E-02	3.35E-01	3.24E-05	3.09E-05	1.18E-01	3.67E-02	1.15E-05	2.23E-04

Table 3-6. Waste volumes projected for 1985-2004 (Gr86)

Waste stream	Volume (m^3)
AEA LLW	
L-IXRESIN	9.91E+04
L-CONCLIQ	3.31E+05
L-FSLUDGE	1.31E+05
P-FCARTRG	1.28E+04
L-DECONRS	2.24E+03
L-NFRCOMP	6.45E+04
F-PROCESS*	5.95E+04
U-PROCESS*	2.14E+04
L-COTRASH	5.98E+05
L-NCTRASH	4.78E+05
F-COTRASH*	1.79E+05
F-NCTRASH*	3.17E+04
I-COTRASH*	2.82E+05
N-LOTRASH*	1.01E+05
N-SSTRASH*	3.59E+05
N-SSWASTE*	6.34E+04
I-LQSCNVL*	1.50E+04
I-ABSLIQD*	1.11E+04
I-BIOWAST*	7.52E+03
N-LOWASTE*	6.03E+04
N-ISOPROD	9.97E+03
N-SOURCES	5.82E+02
N-TRITIUM	6.94E+03
N-TARGETS	2.23E+02
NARM LLW	
R-RASOURC	4.45E-01
R-RAIXRSN	6.60E+03
SURROGATE BRC WASTES	
B-COTRASH	3.32E+05
P-COTRASH	2.65E+05
P-CONDRSN	7.39E+03
L-WASTOIL	2.12E+04
REFERENCE BRC WASTES	
C-SMOKDET	9.20E+04
C-TIMEPCS	1.04E+04

*Note: These wastes may also serve as surrogate BRC waste streams in various analyses (see Section 3.3.3).

Table 3-7. Characterization of the AEA and selected NARM waste streams

Waste stream	10 CFR 61 waste class	As-generated waste form[a]	Optional waste forms[b]	Description of the waste stream
L-IXRESIN	B	AW	SW,IS,HIC	Dewatered Resin in Drum
L-CONCLIQ	A	AW	SW,-,HIC	Absorbed Liquid in Drum
L-FSLUDGE	B	AW	SW,IS,HIC	Dewatered Sludge in Drum
P-FCARTRG	A	TR	SW,-,HIC	Filter in Drum
L-DECONRS	C	AW	SW,IS,HIC	Dewatered Resin in Drum
L-NFRCOMP	A	AM	SW,-,HIC	Activated Steel in Drum or Cask
F-PROCESS	A	AW	SW,-,HIC	Limestone, Oxides in Drum
U-PROCESS	A	AW	SW,-,HIC	Bed Materials etc. in Drum
L-COTRASH	A	TR	SW,IS,HIC	Compactible Trash (Paper, Plastics, etc.)
L-NCTRASH	A	TR	SW,-,HIC	Non-Compactible Trash (Equipment, Used Tools, Parts)
F-COTRASH	A	TR	SW,IS,HIC	Compactible Trash
F-NCTRASH	A	TR	SW,-,HIC	Non-Compactible Trash
I-COTRASH	A	TR	SW,IS,HIC	Compactible Trash
N-LOTRASH	A	TR	SW,IS,HIC	Mostly Compactible Trash
N-SSTRASH	A	TR	SW,IS,HIC	Compactible Trash Plus Miscellaneous Non-Compactible
N-SSWASTE	A	TR	SW,-,HIC	MgF_2 Slag, U Cuttings, Scraps, Used Equipment
I-LQSCNVL	A	AW	SW,IS,HIC	Absorbed Scintillation Fluids in Drum
I-ABSLIQD	A	AW	SW,-,HIC	Absorbed Aqueous Fluids in Drum
I-BIOWAST	A	AW	SW,IS,HIC	Biological Wastes With Absorbent in Drum
N-LOWASTE	A	AW	SW,IS,HIC	Fluids, Biowastes With Absorbent in Drum
N-ISOPROD	C	TR	SW,-,HIC	Compactible Trash Plus Miscellaneous Solid Items
N-SOURCES	C	AM	SW,-,HIC	Encapsulated Sources in Drum
N-TRITIUM	B	TR	SW,-,HIC	Trash, Metal Items in Drum
N-TARGETS	B	AM	SW,-,HIC	Spent Targets (Foils) in Drum
R-RAIXRSN	(C)	AW	SW,IS,HIC	Dewatered Resin in Drum
R-RASOURC	(C)	AM	SW,-,HIC	Encapsulated Ra Sources in Drum

[a]AW=Absorbing Waste, AM=Activated Metal, TR=Trash
[b]SW=Solidified Waste, IS=Incinerated/Solidified Waste, HIC=High Integrity Container

are inserted into the reactor core structure. After generating heat in the reactor core for a predetermined time, the nuclear fuel is termed spent, i.e., the content of the fissile U-235 has been lowered by the numerous fissions of U-235 atoms. Spent fuel becomes a high-level nuclear waste in the once-through uranium fuel cycle. This fuel is stored for final disposition in a Federal high-level radioactive waste repository to be constructed by DOE.

As indicated previously, the EPA LLW source term is a condensation of the detailed NRC LLW source term developed to support the 10 CFR 61 rulemaking. The NRC has characterized LLW from commercial nuclear power reactors, fuel fabrication plants, and uranium conversion plants to represent LLW from the nuclear fuel cycle. LLW from the three Federally owned and operated gaseous diffusion plants were not characterized in the NRC analysis.

(B) Nuclear Power Reactors

LLW is produced from the operation of various liquid and solid waste handling systems at nuclear reactors. Wet wastes, such as resins and sludges, result from operation of the liquid treatment components (evaporators, filters, ion exchange resins) in the liquid radioactive waste treatment system. Dry wastes, such as trash and miscellaneous discarded equipment, are produced by the solid radioactive waste handling system. On occasion, highly activated metals from equipment in or near the reactor core may be discarded. At certain intervals, the reactor coolant piping may be decontaminated with certain chemicals. Waste liquids from this process are treated with ion exchange resins, which become another form of LLW. Table 3-3 identifies the kinds of power reactor wastes considered. The LWR waste category represents LLW from both boiling water reactors (BWRs) and pressurized water reactors (PWRs). Since filter cartridges are used only at PWRs, they are indicated as a PWR waste only.

Table 3-5 presents the radionuclide concentrations of the as-generated LLW streams for nuclear power reactors. For the most part, these concentrations are based on NRC's update report (NRC86). EPA has added a few radionuclides that possess moderately long half-lives (a year to a few years) and contribute somewhat to the radionuclide inventory (Gr86): Ru-106, Sb-125, Cs-134, and Eu-154. In addition, U-234 has been added because of its long half-life, presence to varying degrees in nuclear fuel-cycle wastes, and dose conversion factors comparable to U-238 (Pe75, Se77). Inclusion of the isotopes Ru-106, Sb-125, Cs-134, and Eu-154 is based on scaling factors (a standard industry practice) to standard isotopes presented in a previous EPA characterization of LLW (TRW83). Accordingly, Ru-106 and Cs-134 were compared to Cs-137, while Sb-125 and Eu-154 were compared to Co-60.

Table 3-2 illustrates the manner in which EPA has condensed the NRC waste streams representative of nuclear power reactor LLW. Except for PWR filter cartridges, which are used only at PWRs, EPA does not distinguish, for purposes of LLW analysis, between LWR types. In order to derive radionuclide concentrations in waste streams representative of LWRs (i.e., both PWRs and BWRs), EPA first grouped corresponding waste forms together. For example, ion exchange resins from BWRs and PWRs were grouped together. Similar groupings were made for concentrated liquids, filter sludges, compactible trash, and noncompactible trash. To arrive at a radionuclide concentration representative of LWRs, the radionuclide concentrations for similar BWR and PWR waste streams, as given in the NRC update report (NRC86), were weighted by the volume of each waste. Radionuclide concentrations for the two remaining power reactor waste streams, LWR decontamination resins (L-DECONRS) and LWR nonfuel reactor core components (L-NFRCOMP), are also based on the NRC update report. The radionuclide concentrations for L-DECONRS are taken directly from the NRC update. For the L-NFRCOMP waste stream, EPA has combined two similar NRC streams, L-NFRCOMP and N-HIGHACT. The latter waste is composed of highly activated metals and equipment, whereas the former waste is highly activated metallic components (except fuel rods). The resulting radionuclide concentration for L-NFRCOMP in the EPA LLW source term is calculated by volume-weighting the two NRC activated metal waste streams, L-NFRCOMP and N-HIGHACT.

EPA's projected volumes for 1985-2004 are based on the volume generation rates in NRC's update report (NRC86). Volume generation rates of power reactor waste streams are provided for major categories of power reactor designs. Using these generation rates and the projected annual additions of specific power reactors, EPA projected power reactor waste stream volumes from 1985 through 2004 (PHB85). These projections were first performed for the breakdown of reactor waste streams as given in the NRC update report. Then, as described above, the waste volumes of corresponding waste forms in BWRs and PWRs were simply combined to calculate a projected volume for the consolidated LWR waste stream. Thus, the projected volumes for BWR and PWR ion exchange resins were added to arrive at the volume for the EPA waste stream, LWR ion exchange resins. A similar procedure was used for concentrated liquids, filter sludges, compactible trash, and noncompactible trash. The PWR filter cartridge volumes were projected using the volume generation rates and projections of PWR generating capacity given by NRC (NRC86). The LWR decontamination resin volumes were also calculated according to the NRC update report (NRC86). The EPA waste stream volume for L-NFRCOMP represents the summation of the NRC waste stream volumes for L-NFRCOMP and N-HIGHACT, as given in the update report. Table 3-6 provides the projected waste volumes for 1985-2004 for LLW streams from nuclear power reactors.

(C) Fuel Supply

In the nuclear fuel-cycle, the fuel supply portion includes facilities for uranium conversion, enrichment, and fuel fabrication. Although the LLW resulting from enrichment plants is not characterized in the NRC or EPA analyses of LLW disposal, estimated quantities of such LLW are very small in comparison to quantities coming from other nuclear fuel-cycle LLW generators. In 1980, DOE reported the following solid LLW generated by the three uranium enrichment facilities (DOE81):

Facility	1980 solid LLW generated (m^3)
Portsmouth	1.045 E+01
Oak Ridge	2.943 E+02
Paducah	1.325 E+01

Wastes from uranium conversion plants and fuel fabrication plants have been characterized by NRC for the 10 CFR 61 rulemaking. EPA has used the NRC update report (NRC86) to construct the EPA LLW source term for these facilities. The radioactive waste streams from such facilities are identified in Tables 3-2 and 3-3. Tables 3-5 and 3-6 provide the radionuclide concentrations and the projected waste volumes (1985-2004), respectively, for the LLW streams from fuel fabrication plants and uranium conversion facilities. Note that EPA has included U-234 in these source terms. Studies of the environmental impacts of model fuel fabrication and uranium conversion facilities identified U-234 as a major contributor of the radiation dose to individuals living in the vicinity of such plants (Pe75, Se77).

For the EPA uranium conversion waste stream, U-PROCESS, the U-235 and U-238 activity concentrations are provided by NRC in its update report. U-234 is the daughter of U-238 and is presumed in equilibrium with U-238. Therefore, the U-234 concentration in U-PROCESS is taken to be equal to that of U-238, as shown in Table 3-5.

The relative concentrations of uranium isotopes in fresh fuel depend upon the degree of enrichment of the uranium isotopes during the enrichment process. For U-235 and U-238, EPA has used the waste stream concentrations for fuel fabrication wastes as provided in NRC's update report (NRC86). EPA has added U-234 by using the results of analyses supporting EPA's high-level waste standards (EPA77). The results for the throwaway or once-through uranium fuel-cycle are used. Accordingly, the activity concentration of U-234 in fresh fuel is about a factor of 6 higher than the U-238 concentration. Therefore, EPA has included U-234 in the fuel fabrication waste streams at a concentration that reflects the above U-234/U-238 ratio.

(D) Institutional Generators

Institutional generators of LLW include academic and medical facilities. Academic sources include university hospitals, research facilities, colleges, and universities. Medical LLW producers include hospitals and clinics, medical research facilities and laboratories, and private medical offices (An78, CRC84, NRC86). The 1983 assessment by the CRCPD of LLW shipped to commercial disposal sites indicates a total of 8,485 NRC licensees in the institutional sector (CRC84). Between 80 and 90 percent of these licensees fall in the medical category (An78, EPA80).

EPA has used the NRC characterization of institutional wastes (NRC86), though no distinction is made by EPA between large and small generators. The EPA source term combines volumes of large and small generators into one overall volume for each waste stream. The radionuclide concentrations used by NRC (and EPA) are based on surveys of institutional generators and unpublished disposal site radioactive waste shipment records (NRC81b). Accordingly, institutional LLW has been classified as trash, liquid scintillation vials, absorbed aqueous and organic liquids, and biological wastes. These are identified in Tables 3-2 and 3-3. Radionuclide concentrations are shown in Table 3-5 and are the same as provided in the NRC update report (NRC86). EPA has used the same assumptions as NRC for institutional waste volume projections. Table 3-6 shows the resulting institutional waste volumes projected for 1985 through 2004 (PHB85).

(E) Industrial Generators

Industrial generators of LLW are involved in a wide variety of activities. Such generators produce and distribute isotopes to other industrial and institutional facilities, which incorporate these isotopes in various products, procedures, and analyses. The 1983 assessment of LLW (CRC84) indicates a total of 10,150 industrial licensees. Due to the large number of licensees and the extreme diversity of processes and uses of radionuclides in the industrial sector, industrial LLW is the most difficult category of routine LLW generators to characterize. Many industrial licensees use one or a few radionuclides for applications specific to the licensee's process and facility. Generation of the waste may occur at irregular intervals.

EPA has relied on the NRC characterization of industrial wastes (NRC86) in developing a condensed version of the NRC industrial LLW source term. Table 3-2 illustrates the corresponding NRC and EPA industrial waste streams. It should be noted that Table 3-2 reflects the waste streams characterized by NRC in its 1981 draft EIS (NRC81b). With the update report (NRC86), NRC has greatly expanded the number of higher specific activity waste streams characterized. In particular, waste from isotope production facilities (N-ISOPROD) and tritium production waste (N-TRITIUM) are broken down into numerous substreams. EPA has derived an independent source term for sealed sources (N-SOURCES). Table 3-8

Table 3-8. NRC and EPA LLW source terms: industrial waste groups and streams

NRC symbol (NRC86)	Waste stream description	EPA symbol
	Source and SNM[a] Wastes	
N-SSTRASH	Source and SNM Trash (LF)[b]	N-SSTRASH
N+SSTRASH	Source and SNM Trash (SF)[c]	
N-SSWASTE	Source and SNM Waste	N-SSWASTE
	Low Activity Wastes	
N-LOTRASH	Low Activity Trash (LF)	N-LOTRASH
N+LOTRASH	Low Activity Trash (SF)	
N-LOWASTE	Low Activity Wastes	N-LOWASTE
	Isotope Production Wastes	
N-ISOPROD	High Activity Production Trash	N-ISOPROD
N-ISOTRSH	Low Activity Production Trash	
N-SORMFG1	Sealed Source Manufacturing Wastes:	
N-SORMFG2	Various Facilities	
N-SORMFG3		
N-SORMFG4		
	Large and Small Tritium, C-14 Manufacturers	
N-NECOTRA	Compactible Trash	N-TRITIUM
N-NEABLIQ	Absorbed Organic Liquid	
N-NESOLIQ	Solidified Aqueous Liquid	
N-NEVIALS	Reject Product Vials	
N-NENCGLS	Noncompactible Glass	
N-NEWOTAL	Noncompactible Wood/Metal	
N-NETRGAS	Tritium Gas	
N-NETRILI	Absorbed Tritiated Liquid	
N-NECARLI	Absorbed C-14 Liquid	
N-MWTRASH	Laboratory Trash	
N-MWABLIQ	Absorbed Organic Liquid	
N-MWSOLIQ	Solidified Aqueous Liquid	
N-MWWASTE	Miscellaneous Waste	
N-TRIPLAT	Tritium in Paint or as Plating	
N-TRITGAS	Gaseous Tritium	
N-TRISCNT	High Activity Scintillation Liquids	
N-TRILIQD	Tritium in Aqueous Liquid	
N-TRITRSH	Miscellaneous Trash	

Table 3-8. NRC and EPA LLW source terms: industrial waste groups and streams (continued)

NRC symbol (NRC86)	Waste stream description	EPA symbol
	Accelerator Waste	
N-TRIFOIL	Accelerator Targets	N-TARGETS
	Sealed Sources and Devices	
N-TRITSOR	Tritium Sources	N-SOURCES[d]
N-CARBSOR	Carbon-14 Sources	
N-COBLSOR	Cobalt-60 Sources	
N-NICKSOR	Nickel-63 Sources	
N-STROSOR	Strontium-90 Sources	
N-CEISOR	Cs-137 Sources	
N-PLU8SOR	Plutonium-238 Sources	
N-PLU9SOR	Plutonium-239 Sources	
N-AMERSOR	Americium-241 Sources	
N-PUBESOR	Plutonium-238 Neutron Sources	
N-AMBESOR	Americium-241 Neutron Sources	

[a]Special Nuclear Material
[b]Large Facility
[c]Small Facility
[d]EPA has derived an independent source term for sealed sources.

illustrates the breakdown of industrial LLW streams provided by NRC in its update report and the corresponding EPA LLW streams. For source and special nuclear material trash (N-SSTRASH) and low activity trash (N-LOTRASH), EPA does not distinguish between large and small facilities. With respect to volume generation rates, EPA combines the volume generation rates for NRC's large and small facilities to arrive at the total volume generation rates used in EPA's N-SSTRASH and N-LOTRASH waste streams. Volumes for the EPA N-ISOPROD and N-TRITIUM waste streams represent the combined volume generation rates of the corresponding NRC waste streams. (See Table 3-8.)

The radionuclide concentrations in the EPA industrial LLW streams are reflective of the most recent NRC data for the NRC LLW streams shown in Table 3-8 (NRC86). Industrial waste streams are identified in Tables 3-2 and 3-3. Table 3-4 provides the radionuclide concentrations for the waste streams identified in Table 3-3. Radionuclide concentrations N-LOTRASH and N-LOWASTE are identical in both the NRC update report (NRC86) and the EPA LLW source term. The NRC N-TRIFOIL waste stream concentration is identical to the EPA N-TARGETS waste stream. Both represent tritium contained in, or on the surface of, metal foils. For isotope production waste and wastes from large and small tritium/carbon-14 manufacturers, EPA grouped the NRC waste streams as shown in Table 3-8. EPA then weighted the radionuclide concentration of each of the corresponding NRC waste streams by its volume contribution to the volume represented by the EPA waste stream. For example, Table 3-8 shows that six NRC waste streams are combined to form the EPA N-ISOPROD waste stream. EPA's N-ISOPROD waste stream radionuclide concentrations are derived by weighting the radionuclide concentrations of each of the six NRC waste streams by their respective volumetric contribution to the total volume for the EPA N-ISOPROD waste stream. A similar procedure was used to condense the 18 NRC waste streams representing large and small tritium/carbon-14 manufacturers into the single EPA waste stream labeled N-TRITIUM.

Deviations from the radionuclide concentrations of the NRC update report (NRC86) for industrial LLW streams include the adjustment of concentrations for uranium isotopes and the development of an independent source term for sealed sources (N-SOURCES). These adjustments are discussed in the detailed EPA report on LLW source terms (Gr86).

3.3.2 Naturally Occurring and Accelerator-Produced Radioactive Material (NARM) Wastes

Naturally occurring and Accelerator-Produced Radioactive Materials (NARM) are not regulated under the AEA or any other Federal regulation. At the State level, regulation is non-uniform (NRC84). In order to provide a basis on which to consider regulatory options for NARM, EPA commissioned a study of NARM wastes (PEI85). Over 70 specific waste types were catalogued. Information in the open literature and numerous telephone conversations with knowledgeable people in Federal and State

agencies, private industry, and medical/academic institutions formed the data base for the EPA study. The study showed that the radionuclides which make up NARM are of two types: (1) those produced by a particle accelerator and (2) those occurring naturally. Most of the accelerator-produced NARM radionuclides are used in medicine or for research and have very short half-lives. A few accelerator-produced radionuclides are longer-lived. These nuclides, however, are indistinguishable from those that are AEA-related, and the facilities that use them are usually AEA licensed. Therefore, these facilities usually dispose of the NARM radionuclides with their AEA LLW.

An additional component of accelerator-produced radioactive waste is long-lived nuclides associated with activated components of the accelerator or its shielding. There is little information on this waste stream, but what is available suggests that only small amounts would require disposal.

The other source of NARM is the naturally occurring radioactive materials, principally uranium, thorium, and radium. The wastes containing naturally occurring radioactive materials are of two very different types: (a) discrete sources or waste streams of higher radioactive concentration, such as radium needles used in medical practice or radium-contaminated drinking water cleanup resins, and (b) lower activity diffuse sources such as residuals from mining and extraction industries. In terms of their radiation characteristics and physical form, the higher concentration NARM wastes, such as medical irradiation sources and radium-contaminated ion exchange resins, are similar to much of the AEA LLW. When the concentration of uranium or thorium materials exceeds 0.05 percent by weight, they are classified as source material under the AEA, and their disposal is subject to AEA regulation.

The lower activity NARM wastes, such as uranium mine overburden or phosphogypsum waste piles, are very different from the AEA LLW covered under this Standard, both in terms of concentration and volume. These wastes have very low concentrations of radionuclides and are produced in large volumes. The sheer volume of these materials would make disposal in a LLW facility impractical.

A total of 10 waste streams evolved from the initial data base on 70 specific waste types. In forming the 10 NARM waste streams, only those NARM wastes were included for which generation rates and radionuclide data could be found or reasonably estimated. Grouping of the specific NARM waste types into waste streams was based on similarities in source type, waste form, and/or waste processing.

An initial analysis was done on 10 waste streams to determine which streams should be included in our final analysis of impacts from the disposal of LLW. Because only NARM waste streams that were similar to AEA-regulated LLW were to be included in the analysis, only two waste

streams, characterized as higher activity NARM wastes, were included in the final LLW analysis.

As pointed out, there are many types of NARM, including materials containing uranium and thorium in concentrations of less than 0.05 percent by weight, lower activity radium-contaminated items such as radium dial watches and instruments, and accelerator decommissioning wastes. The majority of NARM wastes with higher activity, however, can be characterized by two waste streams: radium sources and radium-contaminated ion exchange resins. EPA has used these two waste streams as surrogates for all higher activity NARM waste in performing analyses of disposal impact. This assumption does not affect the overall analysis, as the few additional NARM wastes would provide little additional activity or volume.

The radium sources waste stream is made up of high activity, discrete sources such as radium needles used for radiation therapy in hospitals, radium-beryllium sources used to generate neutrons for research, or radium sources used in industrial measurement devices. The average activity of these items ranges between 1.0 and 500 mCi for the radium needles and thickness gauges, while the radium-beryllium sources range as high as 1 Ci.

While radium sources are no longer produced in large quantities, many of these activity sources are still in use or are being stored, awaiting proper disposal. According to the CRCPD, between 1912 and 1961, nearly 2,000 Ci of radium were processed or imported into the United States. Less than 200 Ci have been disposed of in licensed disposal facilities, implying a large quantity of radium requiring proper disposal. A recent survey of the States by the CPCPD shows that State regulatory agencies know of at least 400 sources requiring disposal (CRC85), while a preliminary survey for the DOE shows over 500 high activity commercial sources requiring disposal (INEL87).

While a large amount of activity is associated with radium sources, the volume is very small. It is estimated that less than 1 m^3 of radium sources will require disposal over the next 20 yr. This volume does not take into account the disposal containers and their shielding. When the volume of the disposal containers is included, the total volume will increase to approximately 1,200 m^3.

Radium-contaminated ion exchange resins result from the removal of radium from water by municipal drinking water facilities or by uranium recovery operations. The ion exchange resins can become highly concentrated with radium and require special methods to ensure their safe disposal. Relatively few water treatment facilities produce contaminated resins at this time. Their use is expected to increase in the next few years, however, as efforts grow to reduce the level of radium in drinking water. It is estimated that approximately 6,600 m^3 will be produced in the next 20 yr, with activities averaging approximately 40 nCi/gm.

Table 3-6 lists the estimated 20-yr volume generation rates and Table 3-9 provides the radionuclide concentrations for these two higher activity NARM waste streams. As can be seen in Table 3-6, the anticipated volumes of NARM wastes are extremely small (less than 0.01 percent) when compared with LLW regulated under the AEA.

No accelerator-produced wastes are included in the waste streams defining NARM. Contacts with several hospitals and medical associations revealed that the radiopharmaceuticals and other wastes associated with such accelerator operations have short half-lives (on the order of hours). Wastes are generally stored until they are no longer considered radioactive. Research accelerator and accelerator decommissioning wastes are also not included. Research accelerators produce small volumes of wastes annually (approximately one 55-gal drum per year for a large accelerator). Wastes from both research accelerators and accelerator decommissioning are poorly characterized and are currently either recycled or disposed of onsite or in regulated LLW disposal facilities (PEI85, Ba86).

3.3.3 Surrogate Below Regulatory Concern (BRC) Wastes

EPA is investigating the concept of BRC with respect to the disposal of LLW. The BRC concept defines radiation exposures associated with radioactive waste disposal so low that regulation with respect to radiation hazard is not warranted. This concept derives from the reality that all waste, including household garbage, has some level of radioactivity associated with it. Since many LLW streams are treated as such because of trace or even suspected levels of man-made radioactivity, the establishment of a BRC level would allow such wastes to be disposed of in a less restrictive manner, at substantial cost savings, and with minimal risk to the public.

In order to examine the viability of the concept of BRC, EPA has derived a source term that approximates the kinds of LLW, with very low levels of radioactivity, that might be candidates for less restrictive disposal. These waste streams are called surrogate waste streams since they are representative of many potential LLW streams that could be determined to be BRC. As such, they serve as an analytical tool to examine the many aspects of implementing a BRC level.

Models of existing, rather than hypothetical, waste streams are used. Such models provide more direct answers to some of the practical questions surrounding the application of a BRC level than would hypothetical waste stream models. For example, questions about the BRC concept's generic applicability, practical implementation, and the magnitude of cumulative impacts from numerous deregulations are much better answered using real waste streams as candidates.

Table 3-9. Radionuclide concentrations of NARM waste streams (Gr86) (Ci/m^3)

Nuclide	Waste streams	
	R-RAIXRSN	R-RASOURC
Ra-226	1.8E-02	1.4E+03
Rn-222	9.0E-03	1.4E+03
Pb-214	9.0E-03	1.4E+03
Bi-214	9.0E-03	1.4E+03
Pb-210	9.0E-03	1.4E+03
Po-210	9.0E-03	1.4E+03
TOTAL	6.3E-02	8.4E+03

Selection of LLW streams for inclusion as surrogate BRC waste streams was therefore based on numerous criteria. First, the waste stream should be reasonably well characterized as to quantity, radioactivity level, and radionuclides involved. The large data base on LLW developed over the last few years by the NRC, EPA, the Atomic Industrial Forum (AIF), and the Edison Electric Institute, provides enough data to characterize several waste streams having a potential to be classified as BRC. Second, the waste stream should have a low radioactivity concentration. Third, the waste streams selected should represent a wide variety of LLW generators. Finally, most of the waste streams were of a general trash-type of waste, i.e., high volume and few radionuclides.

Two consumer products, ionization chamber smoke detectors (with Am-241) and timepieces (with tritium), were included in the source term as consumer waste streams. These items are manufactured under the NRC licensing process, but are used and disposed of by the public virtually without restriction. Another waste stream used as a special source term is the liquid scintillation media and animal carcasses containing tracer levels of tritium (H-3) or C-14 that was deregulated by the NRC (NRC81a).

(A) Surrogate BRC Waste Streams

Table 3-10 identifies the 18 LLW streams chosen as surrogate BRC waste streams plus the special BIOMED reference waste stream. A variety of waste generators are represented: nuclear fuel-cycle, industrial, institutional, and consumer product sources. The industrial and institutional waste streams are identical to those listed in Table 3-3 and described in Table 3-2. (See Section 3.3.1 for more details on the various LLW streams.) Four of the nuclear fuel-cycle waste streams are identical to those listed in Table 3-3 and described in Table 3-2: F-PROCESS, F-COTRASH, F-NCTRASH, and U-PROCESS. These wastes generally contain low concentrations of uranium isotopes. The other four nuclear fuel-cycle wastes originate from power reactors. These waste streams, however, are actually substreams of the power reactor waste streams listed in Table 3-3 and described in Table 3-2:

AEA waste stream (Table 3-3)	Substream for BRC analysis
L-COTRASH	P-COTRASH and B-COTRASH
L-IXRESIN	P-CONDRSN
L-CONCLIQ	L-WASTOIL

Characterizations of P-COTRASH and B-COTRASH are based upon the NRC's update report (NRC86). The two other surrogate BRC waste streams (P-CONDRSN and L-WASTOIL) are based on independent studies of specific power reactor waste types that were considered good candidates for less restrictive disposal practices (AIF78, B183). Table 3-11 lists the radionuclide concentrations for the surrogate BRC waste streams to be

Table 3-10. Surrogate waste streams for BRC analysis

Waste stream	Identification
	Nuclear Fuel-Cycle Sources
B-COTRASH	BWR Compactible Trash
P-COTRASH	PWR Compactible Trash
P-CONDRSN	PWR Condensate Resins
L-WASTOIL	LWR Waste Oils
F-NCTRASH	Fuel Fabrication Noncompactible Trash
F-COTRASH	Fuel Fabrication Compactible Trash
F-PROCESS	Fuel Fabrication Process Waste
U-PROCESS	Uranium Hexafluoride Process Waste
	Industrial Sources
N-SSWASTE	Source and Special Nuclear Material Waste
N-SSTRASH	Source and Special Nuclear Material Trash
N-LOWASTE	Low Activity Waste
N-LOTRASH	Low Activity Trash
	Institutional Sources
I-LQSCNVL	Liquid Scintillation Vial Wastes
I-BIOWAST	Animal Carcasses, Tissues, and Excreta
I-ABSLIQD	Absorbed Liquid Wastes
I-COTRASH	Compactible Trash
	Consumer Products Sources
C-SMOKDET	Residential Smoke Detectors (Using Am-241)
C-TIMEPCS	Radioluminous Timepieces (Using Tritium)
	Special Reference Waste Source
BIOMED	NRC deregulated liquid scintillation media and animal carcasses

BWR - Boiling Water Reactor
PWR - Pressurized Water Reactor
LWR - Light Water Reactor (i.e., representative of both BWRs and PWRs)

Table 3-11. Radionuclide concentrations of surrogate BRC waste streams (Gr86)
(Ci/m^3)

NUCLIDE	Waste stream								
	B-COTRASH	P-COTRASH	P-CONDRSN	L-WASTOIL	F-PROCESS	F-COTRASH	F-NCTRASH	U-PROCESS	N-SSWASTE
H-3	5.67E-05	7.32E-04	1.60E-06						
C-14	3.50E-06	2.70E-05	5.84E-08						
Fe-55	5.03E-03	1.44E-02	1.92E-05						
Ni-59	5.21E-06	1.72E-05	1.68E-08						
Co-60	8.47E-03	2.78E-02	2.76E-05	5.50E-05					
Ni-63	1.14E-04	5.29E-03	5.17E-06						
Sr-90	1.06E-05	5.34E-05	1.56E-07						
Nb-94	1.64E-07	5.45E-07	4.25E-10						
Tc-99	2.25E-07	2.28E-07	5.16E-11						
Ru-106	5.99E-06	6.04E-06	8.16E-08						
Sb-125	6.78E-05	2.22E-04	2.17E-07						
I-129	5.99E-07	6.73E-07	1.52E-10						
Cs-134	5.99E-03	6.04E-03	2.88E-04						
Cs-135	2.25E-07	2.28E-07	1.06E-08						
Cs-137	5.99E-03	6.04E-03	2.76E-04	5.30E-06					
Ba-137m	5.99E-03	6.04E-03	2.76E-04	5.30E-06					
Eu-154	6.79E-06	2.23E-05	2.92E-09						
U-234	2.74E-08	5.12E-07	7.86E-09		5.20E-04	2.68E-05	2.56E-05	3.64E-04	4.97E-05
U-235	4.41E-10	8.22E-09	3.79E-11		2.30E-05	1.18E-06	1.13E-06	1.65E-05	2.77E-06
Np-237	1.97E-13	3.67E-12	7.28E-15						
U-238	8.04E-09	1.50E-07	2.99E-10		8.54E-05	4.40E-06	4.20E-06	3.64E-04	1.71E-04
Pu-238	1.92E-06	1.44E-05	2.09E-08						
Pu-239	9.72E-07	1.34E-05	7.32E-09						
Pu-241	4.72E-05	5.82E-04	6.38E-07						
Am-241	8.11E-07	9.56E-06	1.50E-08						
Pu-242	2.12E-09	2.92E-08	3.20E-11						
Am-243	5.47E-08	6.47E-09	1.01E-09						
Cm-243	1.62E-09	6.63E-09	7.98E-12						
Cm-244	1.26E-06	6.31E-06	1.11E-08						
TOTAL	3.18E-02	6.73E-02	8.95E-04	6.56E-05	6.28E-04	3.24E-05	3.09E-05	7.45E-04	2.23E-04

Table 3-11. Radionuclide concentrations of surrogate BRC waste streams (continued)
(Ci/m^3)

NUCLIDE	Waste stream									
	N-SSTRASH	N-LOWASTE	N-LOTRASH	I-COTRASH	I-LQSCNVL	I-ABSLIQD	I-BIOWAST	C-SMOKDET	C-TIMEPCS	BIOMED
H-3		1.63E-02	2.85E-02	9.13E-02	5.01E-03	1.42E-01	1.75E-01		3.62E+01	4.45E-02
C-14		9.36E-04	1.64E-03	5.26E-03	2.51E-04	8.16E-03	1.01E-02			4.45E-02
Fe-55										
Ni-59										
Co-60		1.47E-03	3.25E-03	1.04E-02		3.12E-02	3.99E-03			
Ni-63										
Sr-90		1.31E-03	4.53E-04	1.45E-03	4.34E-03	4.34E-03	8.33E-03			
Nb-94										
Tc-99		7.76E-10	1.06E-09	3.39E-09		1.02E-08	6.51E-09			
Ru-106										
Sb-125										
I-129										
Cs-134										
Cs-135										
Cs-137		1.04E-03	1.42E-03	4.56E-03		1.37E-02	8.76E-03			
Ba-137m		1.04E-03	1.42E-03	4.56E-03		1.37E-02	8.76E-03			
Eu-154										
U-234	2.56E-06									
U-235	1.42E-07									
Np-237										
U-238	8.80E-06									
Pu-238										
Pu-239										
Pu-241										
Am-241			1.51E-06	4.82E-06				2.17E-03		
Pu-242										
Am-243										
Cm-243										
Cm-244										
TOTAL	1.15E-05	2.21E-02	3.67E-02	1.18E-01	9.60E-03	2.13E-01	2.15E-01	2.17E-03	3.62E+01	

used by EPA in its analysis of the BRC concept. Table 3-6 shows the projected 20-yr waste volumes for these wastes (PHB85).

(1) Fuel-Cycle Waste Substreams

Many PWRs use ion exchange resins to purify water in the secondary, or condensate, coolant system. About half of all PWRs are reported to use condensate ion exchange resins (Oz84). PWR condensate resins should have very low concentrations of radionuclides unless there is leakage of the primary coolant into the secondary coolant system. Even then, radionuclide concentrations in the secondary system condensate resins may still be low unless the primary coolant has high activity levels from failed fuel rods. Fuel performance has been excellent in recent years, with many vendors reporting 0.02 percent failed fuel or less, so that PWR condensate resins should typically have very low radionuclide concentrations (Ba85). The radionuclide concentrations shown in Table 3-11 for P-CONDRSN are based on the AIF study (AIF78) and an independent EPA evaluation of this waste stream (Ra84).

During the operation of PWRs and BWRs, lubricating oils for many equipment items become slightly contaminated with radioactivity. Oil changing, subsequent handling, or off-normal operating conditions may be responsible for introducing radioactive contamination into the oil. At BWRs, the principal source of waste oil is the turbine seal where reactor coolant may directly contact the turbine oil. The primary coolant system pump motors are the principal waste oil sources at PWRs. The pump oil is contaminated by the containment building atmosphere (Bl83). Numerous other sources of waste oil exist at both BWRs and PWRs, but contribute much smaller and more varied oils. The radionuclide concentrations shown in Table 3-11 for L-WASTOIL are based on a study of these wastes by Bland (Bl83) and an independent EPA evaluation of this waste stream (Ra84).

(2) Consumer Product Wastes

Two consumer products were included in the EPA set of surrogate BRC wastes to serve as reference wastes: residential smoke detectors using Am-241 and timepieces using tritium. Both are in widespread use, familiar to most people, contain small amounts of radioactivity, and can serve as benchmarks for comparison of calculated risks with the other surrogate BRC wastes.

In simplest terms, the smoke detector used in homes consists of electronics and an ionization chamber. During normal operation, a small radioactive source ionizes the air between two electrodes, allowing a current to flow. During a fire, particles from combustion also pass between the electrodes, attaching themselves to the ions, and thus reducing current flow between electrodes and triggering the alarm circuit. The most widely used radionuclide by far is Am-241, an alpha emitter with a 458-yr half-life. The radioactive source is in the form of a small foil fixed onto a metallic or plastic source holder located in

an appropriate position for the electronic circuitry associated with the alarm system. This circuitry is generally encased in its own metal or plastic container, which is located typically at the center of a plastic housing approximately 2.5-cm thick and 15.2 cm in diameter. About one microcurie of Am-241 is used in each detector, resulting in a radionuclide concentration of about 2.17 E-03 Ci/m^3 (Bu80, NRC78). A volume of about 92,000 m^3 is estimated as the disposal volume for discarded smoke detectors between 1985 and 2004 (PHB85). Timepieces containing radioluminous paint have been distributed for over 70 yr in the United States. For many years tritium and Pm-147 have been the principal radionuclides used in timepieces. Since tritium is by far the predominantly used radionuclide, this analysis will focus on the use of tritium watches and clocks (NRC78). A previous study of radioluminous timepieces estimated that each watch contained 2 mCi and each clock contained 0.5 mCi of tritium (McD78). Watches were assumed to be 1-cm thick and 3.3 cm in diameter, while clocks were assumed to be 4-cm thick and 10 cm in diameter. Thus, each watch had a volume of 30 cm^3 at disposal and each clock was assumed to have a disposal volume of 630 cm^3 (allowing a factor of 2 reduction for crushing). This results in a weighted average concentration of 36.2 Ci/m^3 for tritium in timepieces, assuming 6.6 million watches and 0.5 million clocks are disposed of annually (McD78).

(3) BIOMED Wastes

The special deregulated waste source stream called BIOMED was included to serve as an additional reference point for comparison to the other BRC surrogate waste streams. This waste stream was based on the NRC deregulation ruling for trace levels of C-14 and H-3 in scintillation media and animal carcasses. It is made up of the I-BIOWAST and the I-LQSCNVL waste streams which contain only C-14 and H-3 at the maximum concentrations of 0.05 μCi/g of media or animal tissue. The NRC analysis for the deregulation ruling was based on some 11,000 m^3/yr of liquid scintillation vials and 2,200 m^3/yr of animal carcasses fitting this class of BRC waste (NRC81a).

(B) Transportation of BRC Wastes

External gamma doses for transportation workers handling certain BRC waste streams were examined (Ro86). Additional short-lived radionuclides were included that are capable of giving gamma exposure. Table 3-12 presents these nuclides and their concentrations and waste volumes considered for the analysis.

3.4 Status of Low-Level Radioactive Waste Disposal Sites

The goal for the disposal of LLW is the continuing protection of public health and the environment from these hazardous materials. The State of Texas recently summarized several performance objectives for land disposal of LLW: protection of the general population; protection

Table 3-12. Concentrations and annual waste volumes for BRC short-lived radionuclides used in transportation analysis (Ro86) (Ci/m^3)

Short-lived nuclides	Volume (m^3) waste stream	625 P-COTRASH	17.4 P-CONDRSN	22.3 L-WASTOIL	1620.7 B-COTRASH	57.7 L-WASTOIL	187.7 I-COTRASH	7.1 I-ABSLIQD	4.8 I-BIOWAST	89.5 N-LOTRASH	53.2 N-LOWASTE
Na-22							6.44E-04	1.93E-03	2.47E-04		
Co-58		1.37E-02	2.80E-05	5.32E-05	4.17E-03	1.84E.-05				4.41E-05	3.23E-05
Fe-59		2.65E-04	1.20E-06		2.51E-06		8.34E-04	2.50E-03	3.20E-04		
Mo-99		1.84E-02	1.56E-05		1.68E-04		4.69E-02				
I-131		7.26E-02	4.03E-05		4.41E-04		2.06E-02	6.18E-02	7.91E-03		
Cs-136		3.37E-03	1.71E-06		2.38E-06						

of individuals; and stability of the disposal site after closure (Al83). The NRC has developed broad guidelines for the selection of disposal sites (Si82). These criteria and guidelines are based on the premise that the disposal site itself provides the greatest protection from the hazards of LLW. This premise is illustrated in Figure 3-1 (Bo82). It is estimated that a site location provides about 50 percent of the total isolation achievable by an advanced land disposal design.

3.4.1 Existing Low-Level Waste Disposal Sites

There are 23 sites where shallow-land disposal of LLW has taken place (DOE86). The disposal of LLW at commercially operated burial sites began in 1962 at Beatty, Nevada. Since that time the industry expanded to six commercial sites, but only three are currently in operation. Commercial operations at Maxey Flats, Kentucky; West Valley, New York; and Sheffield, Illinois, have been halted. However, a second NRC-licensed burial ground at West Valley continues to receive LLW generated onsite from cleanup and water treatment operations. The three sites still in operation are at Barnwell, South Carolina; Richland, Washington; and Beatty (HO78, DOE86). DOE considers only 6 of its 17 sites as major disposal sites. The other 11 sites are used primarily for the disposal of uranium and thorium wastes and very limited quantities of other radioactive waste, most of which is generated locally. Some DOE disposal sites are no longer being used (e.g., BNL and ANL). See Table 3-13 for a listing of the disposal sites.

The status of the LLW disposal sites (six major DOE/defense and six commercial) is summarized in Table 3-14 (DOE86, Ho80a). The total land area usable for burial is estimated, along with the land area used through 1982. Usable land area is not a severe restriction at some of the DOE sites. The three commercial sites that are closed are indicated, along with their closure dates.

3.4.2 Quantities of Low-Level Waste at Disposal Sites

The total volume of LLW buried through 1985 is illustrated in Figure 3-2 for each of the major sites. About two-thirds of the total volume (65 percent) is buried at the DOE sites. Two of the DOE sites having significant waste volumes (NLO and WSPG) are listed as containing only uranium and thorium wastes and are not considered principal DOE sites.

Historical annual additions and total volumes of LLW disposed of at the principal DOE sites are listed in Table 3-15 (DOE86). Volumes and radionuclide characteristics of LLW disposed of at all DOE sites are shown in Table 3-16.

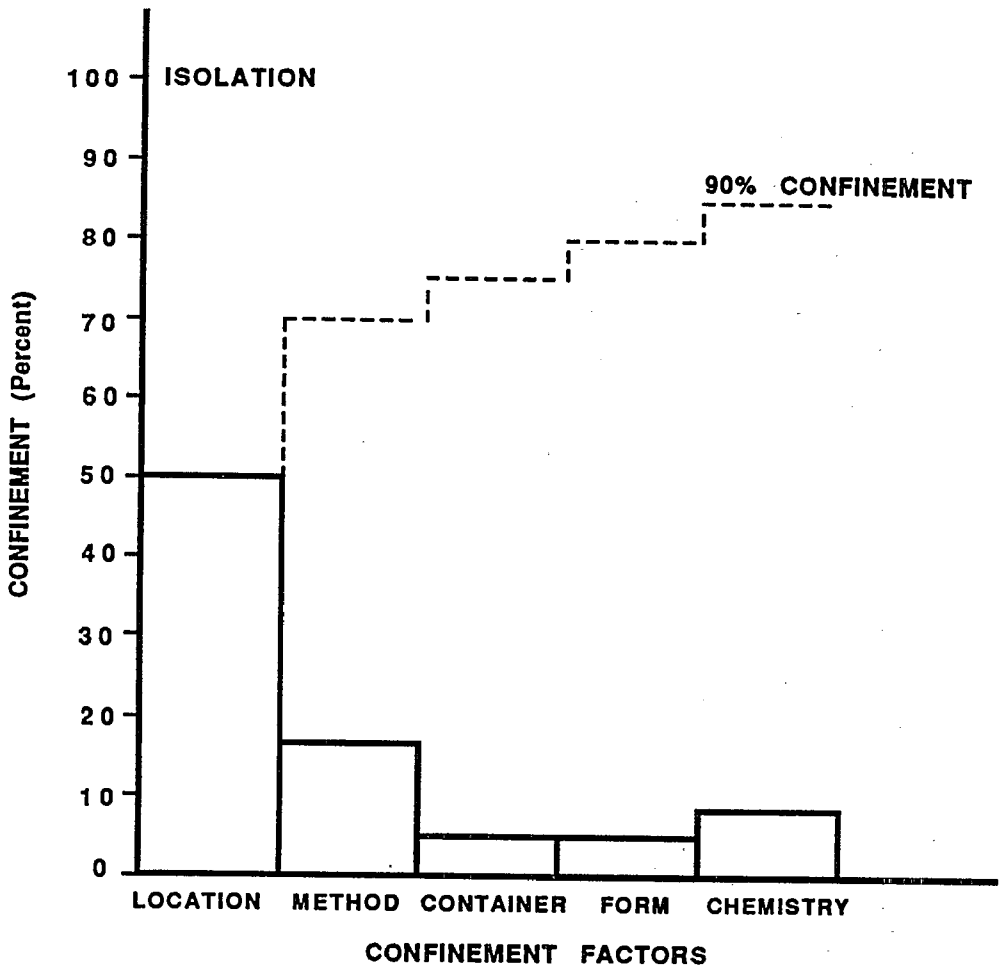


Figure 3-1. Hypothetical Confinement Chart for Land Disposal of LLW (Bo82)

Table 3-13. Existing shallow-land LLW disposal sites

Name	Abbreviation	Location
DOE/defense		
Brookhaven National Laboratory**	BNL	Upton, New York
Hanford Reservation*	HANF	Richland, Washington
Idaho National Engineering Laboratory*	INEL	Idaho Falls, Idaho
Lawrence Livermore National Laboratory	LLNL	Livermore, California
Los Alamos National Laboratory*	LASL	Los Alamos, New Mexico
National Lead of Ohio	NLO	Fernauld, Ohio
Nevada Test Site*	NTS	Mercury, Nevada
Niagara Falls	NIGF	Niagara Falls, New York
Oak Ridge Gaseous Diffusion Plant	ORGDP	Oak Ridge, Tennessee
Oak Ridge National Laboratory*	ORNL	Oak Ridge, Tennessee
Oak Ridge Y12 Plant	Y12	Oak Ridge, Tennessee
Paducah Gaseous Diffusion Plant	PAD	Paducah, Kentucky
Pantex Plant	PANT	Amarillo, Texas
Portsmouth Gaseous Diffusion Plant	PORT	Portsmouth, Ohio
Sandia National Laboratories	SAND	Albuquerque, New Mexico
Savannah River Plant*	SRP	Aiken, South Carolina
Weldon Springs Quarry	WSPG	Weldon Springs, Missouri
Commercial		
Barnwell	BARN	Barnwell, South Carolina
Beatty	BETY	Beatty, Nevada
Maxey Flats**	MFKY	Maxey Flats, Kentucky
Richland	RICH	Richland, Washington
Sheffield**	SHEF	Sheffield, Illinois
West Valley**	WVNY	West Valley, New York

* Major DOE disposal facilities.

** Facility closed.

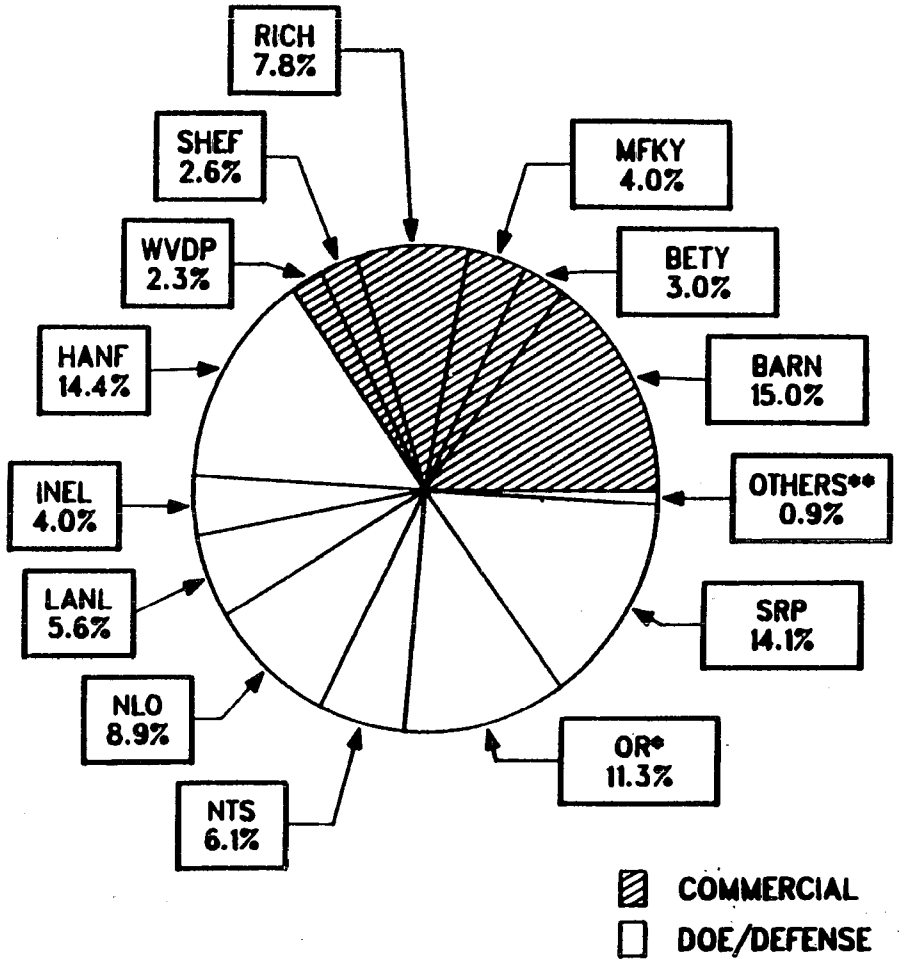
Table 3-14. Status of major LLW disposal sites

Site	Site size[a] (ha)	Estimated total usable burial area[a] (ha)	Calculated land usage factor[a] (m^3/ha)	Estimated burial area utilized through 1982 (ha)[b]
DOE/defense				
Los Alamos National Laboratory, New Mexico	36.4	24.7	29,490	15.5
Idaho National Engineering Laboratory, Idaho	35.6	31.6	20,000	26.4[c]
Nevada Test Site, Nevada	42.5	d	d	d
Oak Ridge National Laboratory, Tennessee	27.5	5.8	6,580	3.4
Hanford Reservation, Washington	>405.	e	4,767	25.9[e]
Savannah River Plant, South Carolina	78.8	72.7	11,150	50.3
Total	>626	>135		121.5
Commercial				
West Valley, New York (Closed March 11, 1975)	8.9[f]	<8.9[g]	22,470	3.0
Maxey Flats, Kentucky (Closed December 27, 1977)	102	<51[h]	8,820[h]	10.0[h]
Sheffield, Illinois (Closed April 8, 1978)	8.9	3.9[g]	22,650	3.9[i]
Barnwell, South Carolina[j]	97.1[k]	46.7[k]	19,550[k]	20.3[k]
Beatty, Nevada	32	18.6[l]	8,132[m,n]	11.6
Richland, Washington	40	35.4[m]	8,714[m]	16.2
Total	289	182		65.0
Grand Total	>915	>317		186.5

[a]Calculated from data given in Ga79 (for DOE) and Ho80b (for commercial), except for Barnwell (La80).
[b]Calculated by dividing volume added in 1982 by the average land usage factor to obtain area used in 1982. This value was added to the 1981 value reported in DOE82a.
[c]In addition, prior to 1970, about 2 ha was used for TRU waste, which was considered to be LLW at the time of burial.
[d]This pertains to the radioactive waste management site in Area 5 of the Nevada Test Site. The availability of land that could be used for shallow-land burial is not clearly defined because of the classified nature of the site and the abundance of land. A land usage factor is not applicable at NTS.

Table 3-14. Status of major LLW disposal sites (continued)

[e]There appears to be no problem in designating additional land for shallow-land burial as it is needed. Land utilized value is for 200-Area only; if the closed 100- and 300-Area burial grounds are considered, ~16.8 ha must be added to this total.
[f]Including fuel reprocessing area, total site is 1,350 ha (Ja80).
[g]Data from Ho80.
[h]Data from Ho78. Area utilized value provided by Site Manager, John E. Razor, letter to A.H. Kibbey, ORNL, July 20, 1983.
[i]Full when closed.
[j]Limit on volume of waste accepted began in November 1978; with present schedule, available land could last until 1997; however, current compact legislation requires closure in 1992.
[k]Calculated from data given in La80. Remaining usable area (26.4 ha) provided by V.R. Autry, S.C. Bureau of Radiological Health, letter to A.H. Kibbey, ORNL, June 21, 1983.
[l]Calculated from data in Ja80 and Ph79.
[m]Based on data given in DOE81.
[n]This is an average value. In the last few years, deeper trenches have been used; therefore, the current land usage factor is somewhat higher.



SITE	CUBIC METERS
BARN	5.02E+05
BETY	9.86E+04
MFKY	1.35E+05
RICH	2.61E+05
SHEF	8.83E+04
WVDP	7.53E+04
HANF	4.80E+05
INEL	1.33E+05
LANL	1.87E+05
NLO	2.99E+05
NTS	2.03E+05
OR*	3.79E+05
SRP	4.71E+05
OTHERS**	2.95E+04
TOTAL	3.34E+06

*Includes contributions from ORNL, Y-12, and ORGDP.

**Includes contributions from PANT, SNL, LLNL, BNL, PAD, and PORT.

Figure 3-2. Total Volume of Buried LLW Through 1985 (DOE86)

Table 3-15. Historical annual additions and total volume of LLW disposed of at DOE/defense sites[a]

Year	Volume of waste buried annually (10^3 m^3)								
	Los Alamos National Laboratory	Idaho National Engineering Laboratory	Nevada Test Site	Oak Ridge National Laboratory	Hanford Reservation[b]	Savannah River Plant	All other	Total annual addition	Total volume accumulated
Through 1975	131.6	85.2	8.3	181.5	349.9[c]	269.1	405.2[d]	1,430.8	1,431
1976	8.8	6.2	2.9	3.8	4.7	8.1	18.0	52.6	1,483
1977	3.6	6.5	0.9	2.4	10.8	14.7	5.4	44.3	1,528
1978	7.5	6.7	13.0	2.0	9.9	15.5	6.5	61.1	1,589
1979	4.9	5.3	34.0	2.1	15.8	18.2	3.8	84.1	1,673
1980	4.8	5.1	12.4	2.0	10.6	19.6	3.4	57.8	1,731
1981	5.5	3.1	14.6	1.4	12.9	20.1	4.2	61.9	1,793
1982	4.5	3.0	39.2	1.3	11.7	22.4	7.0	89.1	1,882
1983	3.2	5.4	26.6	1.8	18.0	26.7	8.2	90.0	1,972
1984	5.4	3.8	12.1	2.2	18.7	26.1	21.4	89.6	2,061
1985	6.7	3.1	39.4	2.2	16.4	30.5	21.5	119.8	2,181
Total	186.5	133.4	203.4	202.8	479.5	471.0	504.6	2,181.0	

[a]No TRU waste included.

[b]Numerous changes were made to the historic LLW data at Hanford. Every year shown was affected; there were reclassifications of old data that caused a major increase in the "through 1975" entry.

[c]Includes 116.7 x 10^3 m^3 of 300-Area LLW that until 1982 was classified as contact-handled TRU waste. The 300-Area burial ground was closed in 1972 and this waste was incorporated into the buried LLW data base as a single entry in that year. Also includes 86.54 x 10^3 m^3 in the 100 Area that previously was reported under "All other."

[d]To avoid double counting, in 1984, African Metals low-level waste at Niagara Falls, NY, was removed from the Inventory for buried low-level waste. Likewise, the low-level wastes in the Weldon Springs, MO, Raffinate Pits and Quarry were removed in 1985. These are inactive sites.

Table 3-16. Volumes and radionuclide characteristics of LLW disposed of at DOE/defense sites[a]

Site	Accumulated amount buried		Summation of activities at time of burial (Ci)[b]					
	Volume (10^3 m^3)	Uranium thorium (kg)	Fission products	Induced activity	Tritium	Alpha (<10 nCi/g)	Other activity	Total gross activity
Los Alamos National Lab	186.5	8.16E+04	1.46E+04	1.67E+04	7.20E+05	4.02E+03	0	7.55E+05
Idaho National Engineering Lab	133.4	6.87E+04	2.93E+06	6.54E+06	0	1.24E+03	0	9.47E+06
Nevada Test Site	203.4	1.56E+06	9.31E+04	9.83E+01	6.93E+06	5.13E+04	2.65E+05	7.34E+06
Oak Ridge National Lab	202.8	9.97E+04	1.73E+05	4.84E+04	7.54E+03	5.00E+02	4.22E+05	6.52E+05
Hanford Reservation[c]	479.5	5.33E+04	5.49E+06	1.45E+06	0	0	0	6.94E+06
Savannah River Plant	471.0	6.74E+02	7.15E+05	4.52E+06	4.30E+06	5.16E+03	2.13E+05	9.76E+06
Principal site total	1676.6	1.86E+06	9.42E+06	1.26E+07	1.20E+07	6.22E+04	9.00E+05	3.49E+07
All other sites[d]:								
National Lead of Ohio	298.5	2.49E+06	0	0	0	0	0	0
Paducah Gaseous Diffusion Plant	7.6	3.30E+06	2.02E+00	0	6.00E-01	0	0	2.62E+00
Oak Ridge Gaseous Diffusion Plant	76.9	9.27E+01	0	0	0	0	0	0
Oak Ridge Y12 Plant	99.1	1.81E+07	0	0	0	0	0	0
Pantex Plant	0.1	2.28E+04	0	1.81E-02	0	1.10E-07	0	1.81E-02
Sandia National Lab	1.9	9.28E+03	6.07E+02	5.16E+03	2.67E+03	2.88E+00	3.14E+00	8.44E+03
Lawrence Livermore National Lab	9.1	3.69E+04	4.35E-03	1.43E-02	0	0	0	1.87E-02
Brookhaven National Lab	0.8	0	0	1.98E+00	2.79E+00	0	7.17E-01	5.49E+00
Portsmouth Gaseous Diffusion Plant	10.0	3.71E+03	0	0	0	0	0	0
Total	504.0	2.40E+07	6.09E+02	5.16E+03	2.67E+03	2.88E+00	3.85E+00	8.43E+03
Grand Total	2180.6	2.58E+07	9.42E+06	1.26E+07	1.20E+07	6.22E+04	9.00E+05	3.49E+07

Table 3-16. Volumes and radionuclide characteristics of LLW disposed of at DOE/defense sites (continued)

[a]From DOE83a. No TRU waste is included. As of December 31, 1982.
[b]Decay has not been allowed for. Present activities are less than the sum of what was buried.
[c]During 1982 numerous changes were incorporated into the data file for Hanford. The greatest change was due to inclusion of LLW buried in the 300-Area that had formerly been classified as contact-handled TRU waste. Also, 100-Area waste was formerly included with "All other."
[d]To avoid double counting, in 1984, African metals low-level waste at Niagara Falls, NY, was removed from the inventory for buried low-level waste. Likewise, the low-level wastes in the Weldon Springs, MO, Raffinate Pits and Quarry were removed in 1985.

Historical annual additions and total volumes of LLW disposed of at the commercial sites are listed in Table 3-17 (DOE86). Historical annual additions of total radioactivity of by-product material, kilograms of source material, and grams of special nuclear material disposed of at commercial sites are provided in Table 3-18 (DOE86).

3.4.3 Experience at Low-Level Radioactive Waste Disposal Sites

The primary method of disposing of LLW since the early 1940's has been by shallow-land disposal. Relatively simple and inexpensive, it is a potentially effective method of disposing of the large volumes of wastes resulting from nuclear activities. However, it appears that some disposal sites have not performed up to their original expectations of confining the radioactive materials on the site during the period the wastes remain a hazard (Me76).

Two commercial disposal sites at West Valley, New York, and Maxey Flats, Kentucky, have not performed as planned. The burial site at West Valley discontinued operations in 1975 because water containing H-3 and Sr-90 was seeping from two of the trench caps (Ho80a). In 1977, the disposal site at Maxey Flats became the second facility to discontinue operations after it was found that leakage in trenches had resulted in some onsite radionuclide migration of LLW (Ho80a).

The West Valley and Maxey Flats sites, having relatively high precipitation and low soil permeability, experienced the bathtub effect. This effect occurs at low permeability sites during periods of high precipitation (storms) when the waste trenches fill with water, which then overflows on the surface. This can lead to significant spreading of radionuclides if there has been leaching of radioactive waste by trench waters (Me76, Gi79, Me79, Ha79, Ke79).

Radionuclides have moved into ground water at some sites (Fi83, We79, Oa79, Cob79, Eb79). At most of these sites the radionuclides have traveled only short distances. However, at least at one site, Oak Ridge, radionuclides have been transported into the Clinch River by overflow to surface water and by ground-water flow (Oa79). Also, Cahill (Ca82) has estimated that tritium could travel from the trenches at Barnwell to the nearest stream in a period as short as 50 yr. Contamination of ground water from the numerous waste disposal methods used at Hanford has been experienced (ERDA75).

A third pathway for uncontrolled releases of radioactivity to the environment appears to be emanation of gaseous tritium and carbon-14 upward through the trench cap by gas drive and capillary action, as reported at West Valley (Ma79, Gi79). Radon could also escape through the cap from radium sources disposed of by shallow-land burial methods.

Table 3-17. Historical annual additions and total volume of LLW disposed of at commercial sites

	Volume (m^3)[a]							
Year	Beatty	West Valley[b]	Maxey Flats	Richland	Sheffield	Barnwell	Annual total	Total accumulation
1962	1,861						1,861	1,861
1963	3,512	127	2,206				6,240	8,101
1964	2,836	5,940	3,872				13,096	21,197
1965	1,988	5,192	5,751	668			13,124	34,321
1966	3,533	3,951	5,556	2,402			16,188	50,509
1967	3,206	7,475	7,820	773[c]	2,527		19,272	69,781
1968	3,576	3,490	8,177	1,359	2,713		20,330	90,111
1969	4,526	4,099	10,353	438	2,012		21,603	111,714
1970	5,152	4,906	12,520	423	2,825		26,016	137,730
1971	4,916	7,002	13,171	584	4,430	1,171	30,634	168,364
1972	4,301	9,045	15,577	654	5,956	3,757	37,299	205,663
1973	4,076	7,535	10,072	1,033	8,524	15,839	47,041	252,704
1974	4,103	8,866	8,897	1,411	12,373	18,244	53,602	306,306
1975	4,943	2,243	17,109	1,500	14,116	18,072	57,629	363,935
1976	3,864	427	13,783	2,867	13,480	40,227	74,221	438,156
1977	4,742	351	423[d]	2,718	17,643	45,663[e]	71,189	509,345
1978	8,874[f]	144		7,422	1,735[g]	61,554[h]	79,585	588,930
1979	6,491	138		12,185		63,861	82,537	671,467
1980	12,717[i]	141		24,819		54,723[j]	106,765	778,232
1981	3,351	216	(~1,133)[k]	40,732		39,427[j]	87,789	866,021
1982	1,505	627[l]	0	39,606		34,779	75,890	941,911
1983	1,111	1,765	(~850)[k]	40,458		35,132	78,466	1,006,979
1984	2,067	822	0	38,481		34,879	76,249	1,083,228
1985	1,388	808	(946)[k]	40,135		34,389	76,720	1,159,948
Total	98,639	75,310	135,280	260,668	88,334	501,717		1,159,948

[a]Except where noted, data were taken from the following sources: 1962–1978, Ho80a; 1979, Ho80b; 1980, EGG82a; 1981, EGG82b; 1982, No83; and 1983–1985, DOE86.

[b]Includes commercial State-licensed facility that opened November 18, 1963, and closed March 11, 1975; and NRC-licensed facility (for onsite fuel reprocessing wastes) that opened in 1966 and continues to receive only onsite-generated LLW associated with water treatment and site cleanup. Also includes revised data for 1963–1975.

[c]Calculated from data given in C181.

[d]Closed December 27, 1977.

[e]Assumed the value (46,563 m^3) given in Ho80a had transposed digits in order to make the total correct.

Table 3-17. Historical annual additions and total volume of LLW disposed of at commercial sites (continued)

[f]Adjusted (+47 m^3) data given in Ho80a to bring total accumulation into agreement with the 1979 State running total.
[g]Closed April 8, 1978; value adjusted (+1,633 m^3) to bring into agreement with total accumulation reported in Ka81 on a trench-by-trench basis.
[h]Adjusted (-12 m^3) data given in Ho80a to bring total accumulation into agreement with the State running total.
[i]A corrected value provided by the DOE Low-Level Waste Management Program.
[j]These values exclude almost 19,000 m^3 (~14,506 in 1980 and ~4,279 in 1981) of very low-level activity settling pond sludge that was not counted against the annual quota.
[k]These wastes, which are generated on site, are not included in the total volume.
[l]The West Valley Demonstration Project (WVDP) began in 1982. The LLW volumes reported for 1982 and subsequent years are for the WFDP only.

Table 3-18. Historical annual additions and total radioactivity of LLW disposed of at commercial sites[a]

Year	Beatty	West Valley[b]	Maxey Flats[c]	Richland	Sheffield[d]	Barnwell	Annual total	Total accumulation
				A. By-product material (Ci)[e]				
1962	Not reported							
1963	5,690	100	22,556				29,618	29,618
1964	6,477	10,400	147,218				165,050	194,668
1965	6,377	22,600	63,828	144			91,864	286,532
1966	11,974	35,400	52,737	1,606			107,373	393,905
1967	10,894	123,100	23,272	5,378	3,850		94,624	488,529
1968	6,808	10,600	45,578	64,432[f]	2,381		170,874	659,403
1969	9,761	36,000	31,028	55,964	2,192		122,209	781,612
1970	12,304	91,900	56,969	52,820	5,427		163,811	945,423
1971	4,316	436,700	710,147	23,916	7,895	4,151	792,883	1,738,306
1972	5,228	131,300	217,350	31,809	4,857	13,575[g]	334,027	2,072,333
1973	5,704	346,000	123,779	57,037	2,834	48,212[g]	408,118	2,480,451
1974	23,904	6,600	143,656	12,773	3,229	13,557[g]	252,648	2,733,099
1975	18,388	11,600	289,751	113,341	6,103	17,428	455,284	3,188,383
1976	4,493	1,200	211,356	104,306	7,744	90,205[g]	418,104	3,606,487
1977	23,811[h]	900	267,063	7,465	11,147	390,121[g]	699,607	4,306,094
1978	5,685	700		235,548	2,547	652,061	895,841	5,201,935
1979	8,897	400		164,787[i]		314,938	488,622	5,690,557
1980[j]	148,312	300		41,031		143,502	332,845	6,023,402
1981[j]	52,214	229		43,905		183,744	279,863	6,303,265
1982[j]	80,929	293		59,007		273,962	413,898	6,717,163
1983[o]	1,356	255		120,534		383,450	505,595	7,905,704
1984[o]	544	25		215,286		385,079	600,934	8,506,638
1985[p]	453	[q]		287,849		385,078	673,380	9,180,018
Total	454,519	1,266,602	2,400,690	1,698,938	60,206	3,299,063		9,180,018

Table 3-18. Historical annual additions and total radioactivity of LLW disposed of at commercial sites[a] (continued)

Year	Beatty	West Valley[b]	Maxey Flats[c]	Richland	Sheffield[d]	Barnwell	Annual total	Total accumulation
				B. Source material (kg)				
1962	296						296	296
1963	472	7,582	5,221				13,264	13,571
1964	331	10,068	5,599				15,993	29,569
1965	236	22,220	568	1			23,025	52,594
1966	91	38,325	690	2,530[k]			41,636	94,230
1967	346	20,275	5,682	1	3,930		30,229	124,464
1968	1,043	6,461	6,252	3	8,705		22,459	146,928
1969	290	80,014	2,556	89	6,334		89,281	243,121
1970	323	31,720	7,224	31	2,004		41,296	284,423
1971	428	51,455	5,740	607	212	12,549	70,986	355,414
1972	9,342	72,543	8,265	3,110	3,596	15,897	112,746	468,167
1973	11,460	44,107	10,998	2,245	2,409	38,460	108,021	577,846
1974	9,717	61,703	13,117	20	13,914	20,814	119,285	697,131
1975	1,438	16,291[r]	82,509	215	35,950	40,339	176,716	873,873
1976	5,000		87,268	5,011	3,854	24,372	114,181	999,378
1977	10,634		297	1,249[l]	184,814	166,818	363,762	1,363,190
1978	77,647			5,264	226,548[m]	803,956	1,113,415	2,476,605
1979	131,253			12,922		1,221,724	1,365,899	3,842,504
1980	194,921			125,419		444,175	765,515	4,607,019
1981	43,136			1,156,661		341,973	1,541,770	6,148,789
1982	708			1,325,484		543,471	1,869,663	8,018,452
1983[o]	139,300			1,170,300		1,017,053	2,326,653	10,345,105
1984[o]	123,284			565,672		993,308	1,682,264	12,027,369
1985	q			q		q		
Total	761,696	469,674	241,986	4,376,834	492,270[m]	5,684,909		12,027,369

Table 3-18. Historical annual additions and total radioactivity of LLW disposed of at commercial sites[a] (continued)

Year	Beatty[n]	West Valley[b]	Maxey Flats[c]	Richland	Sheffield[d]	Barnwell	Annual total	Total accumulation
				C. Special nuclear material (g)				
1962	0						0	0
1963	3,590	952	959				5,501	5,501
1964	7,000	3,273	11,770				22,043	27,544
1965	11,980	2,433	4,261	3			18,677	46,221
1966	10,150	4,999	7,461	1,418			24,029	70,249
1967	25,290	3,446	14,842	<1	1,238		44,817	115,066
1968	8,800	2,045	17,771	<1	1,754		30,371	145,437
1969	6,220	7,301	31,504	32	3,843		48,902	194,337
1970	9,310	8,273	47,562	200	5,649		70,994	265,332
1971	20,060	4,816	72,771	15	9,934	20,361	127,956	393,289
1972	20,930	7,321	71,443	832	5,898	65,294	171,718	565,007
1973	6,520	7,710	46,235	6,558	6,126	85,815	158,978	723,971
1974	16,950	2,986	23,850	5,284	8,144	98,745	155,959	879,930
1975	31,280	1,240	25,690	18,978	5,285	76,983	159,456	1,039,386
1976	2,100		27,767	24,378	1,738	122,261	177,951	1,217,630
1977	11,290		27,878	25,937	5,310	183,256	253,671	1,471,301
1978	7,670			18,312	2,134	220,866	248,982	1,720,283
1979	4,770			7,888		180,275	192,933	1,912,216
1980j	13,600			0		239,315	252,915	2,166,131
1981j	5,119			0		171,506	176,625	2,342,756
1982j	2,860			0		195,074	197,934	2,540,690
1983o	1,030			4,948,300		195,104	5,144,434	7,685,124
1984o	0			9,548		159,479	169,027	7,854,151
1985p	q			q		q		
Total	226,519	56,795	431,765	5,067,685	57,053	2,014,334		7,854,151

Table 3-18. Historical annual additions and total radioactivity of LLW disposed of at commercial sites[a] (continued)

[a]Taken from Ho80a except where noted. By-product, source, and special nuclear materials are as defined in Title 10, Code of Federal Regulations, Parts 30, 40, and 70.
[b]Includes both burial grounds: commercial State-licensed LLW facility (November 18, 1936 to March 11, 1975); NRC-licensed facility for reprocessing waste 1966-1981. Beginning in 1982, the values given are for the West Valley Demonstration Project only. Also includes revised data for 1963-1975.
[c]Closed December 27, 1977. Upgraded data were received from Site Manager, John E. Razor, too late for incorporation in this edition.
[d]Closed April 8, 1978.
[e]Radioactivity at time of burial; decay has not been allowed for. Present activities are less than the sum of what was buried.
[f]Made adjustment (+54, 102 Ci) to bring into agreement with State total and also into rough agreement with 1968 value in C181.
[g]Corrected values provided by V.R. Autrey, S.C. Bureau of Radiological Health, letter to A.H. Kibbey, ORNL, June 15, 1983.
[h]Made adjustment (+925 Ci) to bring into agreement with later State running total.
[i]Washington State correction.
[j]EGG82a, 1980; EGG82b, 1981; and No83, 1982.
[k]Corrected decimal error (253 to 2,530).
[l]Corrected conversion from lb to kg.
[m]Corrected to agree with trench-by-trench study in Ka81; a conversion error (kg vs. lb) in C181 was corrected here.
[n]Beatty data from C181 except for 1981 and 1982.
[o]DOE 85.
[p]EGG86.
[q]Data not available.
[r]Corrected (+38 kg) total u/Th at commercial site.

Other pathways along which radioactive contaminants have or could move from the trenches have been identified as the following: surface runoff and erosion; lateral movement through the soil and permeable weathered zones; sub-surface movement through sand lenses, joints, fractures, and normal intergranular movement; and movement between trenches through failure joints and sand lenses (Me79).

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Chapter 4: DISPOSAL METHODS FOR LOW-LEVEL RADIOACTIVE WASTE

4.1 General Considerations

Land disposal is the placement of waste in a manner not intended for future recovery. If future recovery or movement of the waste is intended, it is more appropriately called storage. Disposal methods are designed to provide public health and environmental protection, to assure that protection is achieved over a reasonable time (e.g., that time over which the majority of health impacts are expected to occur), and to protect workers who handle the waste.

Factors influencing the design of disposal methods for LLW include the quantity and concentration of the radionuclides in the waste; the lifetimes of these materials; their physical and chemical forms; and site-specific factors such as meteorology, geology, hydrology, topography, and geochemistry. Additional factors important for long-term protection include design against intrusion and designs to prevent or inhibit subsidence.

In many cases, disposal methods must be tailored to specific disposal locations. In fact, location characteristics may dictate which disposal methods can be considered. For instance, hydrofracture could only be considered if the geology was appropriate. This close relationship should be kept in mind in the following discussions of disposal methods.

Disposal method design must then address two kinds of failures: those caused by long-term processes, such as weathering, and those caused by more or less discrete processes, such as intrusion by man or biological activity or the sudden subsidence of a trench cap. In general, discrete events pose more difficult design problems because of their random or unpredictable nature.

Disposal methods should be designed to circumvent or solve problems encountered in the past. Figure 4-1 illustrates common problems faced with shallow-land burial of waste (Sp82). This figure shows several problems, or potential failure mechanisms, that must be addressed in the design of disposal methods, including precipitation, runoff, perched water, cover collapse, fractures, etc.

4.2 Methods Considered in the EPA Risk Analysis

The Agency has selected a wide range of disposal methods for inclusion in its radiological risk assessment. Each method is described briefly in the following sections. Detailed information is available on each method and its important characteristics for retaining waste (A182).

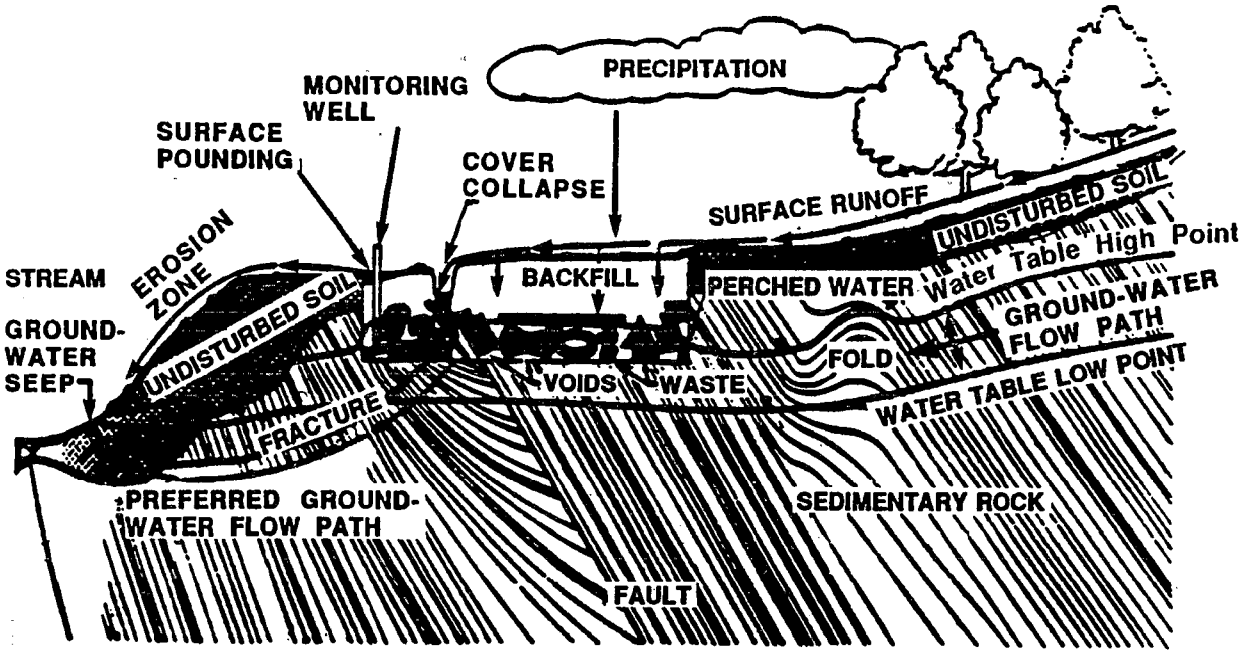

Figure 4-1. Problems Encountered in the Shallow-Land Disposal of Low-Level Radioactive Waste To Be Addressed by Corrective Measures Technology (Sp82)

Seven major reference near-surface disposal methods were considered in the EPA risk analysis: (1) disposal at an LLW regulated sanitary landfill; (2) conventional shallow-land disposal; (3) improved shallow-land disposal; (4) 10 CFR 61 disposal technology; (5) intermediate depth disposal; (6) earth-mounded concrete bunker disposal; and (7) concrete canister disposal.

In addition, three other special waste disposal methods were considered in the risk analysis: (1) deep geological disposal; (2) hydrofracturing; and (3) deep-well injection. However, these methods are suitable only for certain wastes.

Facilities using the seven major reference disposal methods have an assumed period of active operation of 20 years. The capacity of a facility, 250,000 cubic centimeters, was estimated by dividing EPA's projected volume of LLW to be disposed of in the U.S. over 20 years by an assumed 10 to 12 disposal sites. The 20-year period was chosen as it represents a period that a particular waste disposal facility and practice might be expected to exist and also a period for which we have some confidence in waste projection estimates.

For the risk assessment dealing with near-surface disposal, several base case method scenarios were analyzed: (1) LLW regulated sanitary landfill; (2) conventional shallow-land disposal; (3) improved shallow-land disposal; (4) 10 CFR 61 disposal technology; (5) intermediate depth disposal; and (6) concrete canister disposal. These analyses are discussed in Chapter 9. Analyses covering the other disposal methods are discussed with the sensitivity analyses in Chapter 11.

Generally, the various methods are all analyzed for the three LLW classes (A, B, and C). Appendix B lists the NRC waste classification criteria for Classes A, B, and C. While disposal at the greater depths should provide better health protection, this may vary with hydrogeology. The two disposal methods, intermediate depth and deep geological, may be assumed to consider not only Class C but also greater-than-Class C wastes.

4.2.1 LLW Regulated Sanitary Landfill

Sanitary landfills (SLF) are currently used for the disposal of nonhazardous solid wastes (see Figure 4-2). This disposal method involves daily placement of a dirt cover over the disposed refuse material in a manner designed to minimize environmental pollution. The SLF method should not be confused with open dumping and open incineration methods, which are no longer permitted under the Resource Conservation and Recovery Act of 1976 (although in some instances they may still be used). Neither should they be confused with hazardous material landfills and surface impoundments. The standards for the design and operation of an SLF have been established by the EPA in 40 CFR 241 to 257. The actual

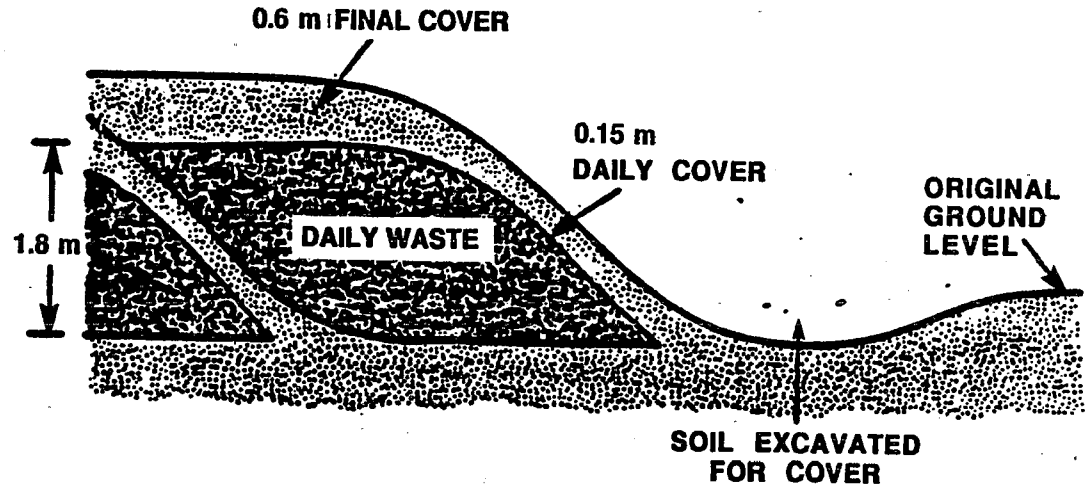

Figure 4-2. Cross-Section of a Typical Sanitary Landfill

licensing of a municipal SLF is normally conducted by municipal or county agencies.

The SLF considered here is a designated and regulated facility for LLW disposal. It would be regulated by DOE, NRC, or Agreement States. The basic disposal method, however, is the same as an EPA-approved municipal SLF. It is assumed that up to 100 percent of the LLW would be suitable for disposal at an SLF. Land requirements are then about 52.6 hectares (ha), including a 100-meter (m) buffer zone surrounding the SLF facility.

The conceptual SLF is operated in much the same manner as a municipal SLF. These wastes are dumped at the base of the landfill. Once or twice a day, a bulldozer spreads and compacts this new waste on the slope left from the previous day's cover. These slopes may be on the order of 2 meters high. At the end of each day, this waste is covered with 0.15 meter of soil. When a large enough area has been filled, an additional 0.6 meter of soil is placed over the waste cells. Typical volumetric ratios of wastes to cover range from 3:1 up to 4:1.

If the cover material has very low permeability, then a gas vent system may have to be added prior to the final cover. This gas control system may consist of periodic layers of gravel over the daily cover or a series of standpipes with perforated laterals.

Because of the spreading and compaction of wastes with a bulldozer (as described above), it is assumed that all of the packages containing the radioactive LLW in the forms of trash and absorbed waste will be breached during disposal. It is also probable that some of these wastes will be mixed with the daily cover. It is assumed that approximately 0.1 percent of the trash and absorbed radwaste will be mingled with the exposed cover material at the surface of the SLF. It is assumed that no spillage of activated metals or solidified waste occurs because of the solid form, whereas for trash or absorbing waste the lack of a solid form would allow some spillage.

Prior to closure and license termination, the landfill will be inspected by the regulating agency. Because we are assuming a regulated SLF, site closure procedures will also be assumed to be similar to those described in Section 4.2.2 for shallow-land burial.

Settlement after disposal compaction is primarily caused by waste decomposition whose rate is controlled by many factors including temperature and local water conditions. Landfill settlement can be as much as 20 percent of the initial waste height in the first year, decreasing logarithmically for several years thereafter. Some landfills settle by as much as 33 percent over several years. In general, the bulk of the settlement occurs in the first five years.

For the purpose of risk assessment, it is assumed that the trench cap area will begin to fail during the first year after closure, directly exposing that fraction of the surface area of the waste contents of the trench. For this assessment, EPA has assumed that the trench cap failure increases at a constant rate until it reaches 30 percent of the area of the cover 40 years after closure. Trench cap failure is then assumed to remain constant at 30 percent for the remainder of the period of analysis.

4.2.2 Conventional Shallow-Land Disposal

Conventional shallow-land disposal (SLD) is best described as the land disposal practices that were used at licensed LLW disposal sites during the 1970's and 1980's. In particular, the operations at Barnwell, South Carolina, Beatty, Nevada, and West Valley, New York, during this period were combined to form the design of a generic SLD facility for this assessment. Certain design features pertaining to the disposal of Class A wastes of the NRC shallow-land disposal reference facility (NRC81) were also included.

The trench design is shown in Figures 4-3 and 4-4. The life cycle of a conventional SLD facility consists of a 3-year construction period, 20-year operating life, 2-year closure period, and 100-year long-term care period.

The conceptual, conventional SLD facility is located on a 52.6-hectare site, of which only 10.9 hectares will be directly used for waste disposal. The disposal area is surrounded by a buffer zone, 100 meters wide, which permits additional monitoring and allows corrective measures to be taken, if required. An all-weather gravel road provides access to both the administrative and disposal areas on the site. Site access control is maintained by a fence surrounding the administrative and disposal areas, as well as a set of inspection points at the entrances to the disposal area.

To better characterize the activities going on at a disposal facility and to allow cost estimates to be made, it is assumed that the following onsite buildings will be: an administrative building, health physics/security building, warehouse, garage, and waste activities building. The administrative building contains office space for the site management and operations support personnel (accounting, shipping and receiving, records, etc.). The health physics/security building houses both the health physics and security personnel and provides locker and luncheon facilities for the operational crews. The frisker station, where personnel leaving the site are monitored for contamination, is also located in this building. The warehouse is used to store the supplies needed for site operation and maintenance. The garage provides onsite maintenance capability for heavy equipment. The waste activities building contains a decontamination bay, a liquid treatment area, a waste solidification/packaging area, a supply room, and a small waste storage area. Temporary waste storage is usually obtained by leaving the wastes

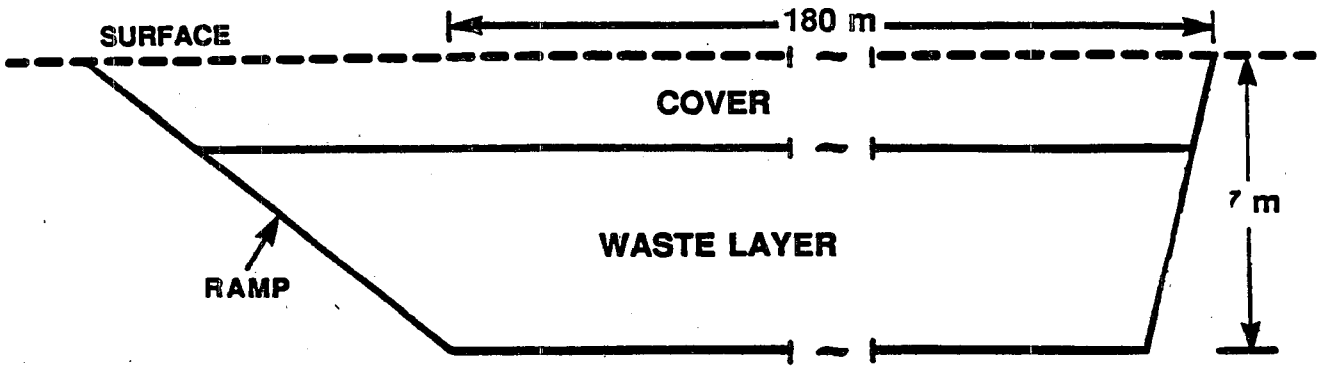

Figure 4-3. Profile of a Shallow-Land Disposal Trench Along the Long Axis

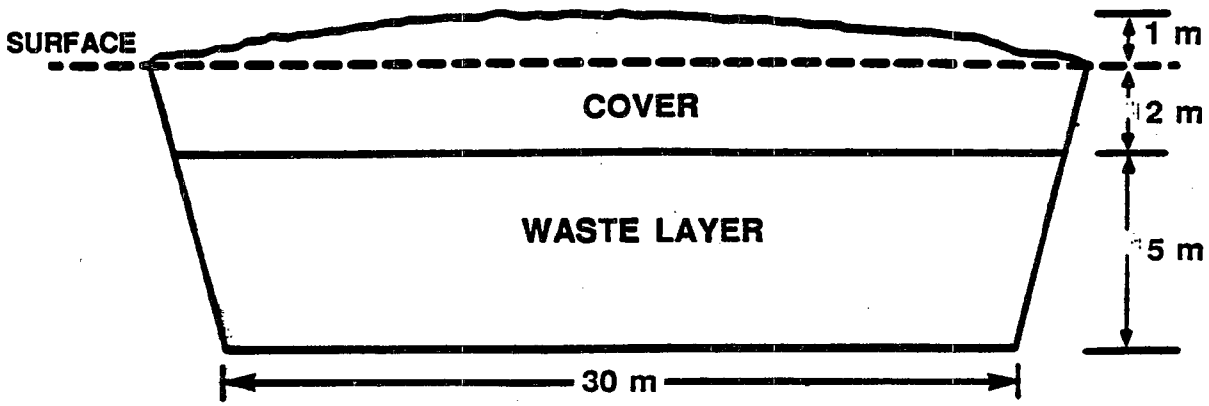

Figure 4-4. Cross-Section of a Shallow-Land Disposal Trench Perpendicular to the Long Axis

on the transport vehicle. The waste storage area is used primarily for common carrier packages or under special circumstances when the disposal operations cannot keep pace with the arrival of wastes. When a waste shipment arrives onsite, its shipment documents (manifests) are processed, while the waste packages are inspected by the health physics personnel to ensure compliance with Federal and State regulations. If the packages meet the appropriate regulations, the transport vehicle is directed to the current disposal area where the packages are unloaded and buried. If the packages do not meet the regulations, corrective action (e.g., repackaging of failed containers) is taken using the facilities in the waste activity building.

The SLD trenches are 180 m long by 30 m wide (at the bottom) by 7 m deep with an average spacing of 3 m between trenches. The trench walls are assumed to have a fairly steep slope of 1:4 (horizontal to vertical) for typical cohesive soils. The bottom of the trench slopes gently toward a French drain that extends the complete length of the trench. Water is pumped from this drain to avoid standing water in the trench.

Wastes are emplaced in these trenches in a random manner with approximately 50 percent utilization of the trench volume. Large boxes are typically lowered into place using cranes, with barrels and drums rolled into the trenches to fill in the void space. Wastes occupy the bottom 5 meters of the trench. Each trench requires that 40,320 m^3 of dirt be excavated and has a 17,500 m^3 waste capacity.

In the conventional SLD facilities, the only wastes that must be segregated are special nuclear materials (SNM) (NRC81). Packages containing SNM must be stored at least 3.7 meters from other SNM packages and must be buried with at least 0.2 meter of earth (or other wastes) between individual SNM packages. They may be buried in the same trenches as the other wastes.

As the wastes are placed in the trench, the trench is backfilled with dirt that was removed during trench excavation. The backfill dirt fills in the voids between the waste packages and is piled to a height of approximately 1 meter above the local grade at the center of the trench and slopes gently to the sides for drainage. This provides approximately 3 meters to the top of the waste at the center and 2 meters at the edges of the trench. No special compaction of the backfill and cap is performed except for the movement of heavy equipment over the cap.

When a trench is filled, the cap is covered with topsoil and seeded with short-rooted grass. The corners of the trench are marked and a monument erected upon which is inscribed a trench identification number, the volume, and the total activity of wastes in the trench, as well as the date of completion. After 20 years, the site will have 24 trenches filled with wastes. The caps on these trenches may have to be regraded during the site operating period to account for the settlement of the waste and the trench cover. The major portion of the settlement normally

occurs within the first year after backfilling the trenches and decreases gradually in the time period of seven to ten years (NRC81).

During the closure period, all the site buildings except the health physics building are dismantled. The health physics building is used as the base of operations for the closure and long-term care periods. Contaminated debris and equipment are buried in the last trench. This trench is then backfilled and capped. The complete site is reseeded with a grass cover as necessary.

The closure operations are estimated to take one to two years. Following closure and a period of up to five years for post-closure observation and maintenance, the licensee may then apply to terminate the operator's license and transfer site control to a Federal or State government agency (NRC82). This begins the long-term institutional care period.

Since trench settlement will continue for several years after closure, some sort of active site maintenance must be available during this period. Site activities should decrease substantially after ten years and be reduced to just monitoring operations after 25 years.

4.2.3 Improved Shallow-Land Disposal

Improved shallow-land disposal (ISD) incorporates all of the design and operating requirements specified in the NRC regulation 10 CFR 61 (NRC82). One of the primary differences between ISD and conventional SLD is the requirement to segregate the wastes into three classes (A, B, and C). Class A LLW requires improved packaging in the as-generated waste form. Class B and Class C wastes require a more stable waste form than does Class A. The Class C LLW disposal method also requires a minimum 5-meter cover between waste and top surface or an appropriate intruder barrier suitable for 500 years. Wastes exceeding Class C isotope concentrations are not considered suitable for near-surface disposal.

EPA has estimated that at any ISD site, 88 percent of the wastes will be Class A, 11 percent will be Class B, and 1 percent Class C after BRC wastes are excluded. The life cycle of the facility includes a 3-year construction period, a 20-year operating life, up to 5 years for closure (NRC82), and a 100-year institutional care period.

For the ISD method, all LLW are buried in Class C type slit trenches, 20 m long, 3 m wide, and 8 m deep. These trenches have nearly vertical sides and are spaced 3 meters apart. Using the slit trenches and these dimensions is appropriate for reducing ground subsidence and external radiation to the workers. Wastes are stacked to a height of 7 meters in the trench, backfilled to the surface, and initially covered with an additional 1-meter cap. Above the cap of the trenches, an additional 5.5-meter intruder barrier cover is installed. This cover

includes layers of sand, clay, gravel, cobbles, boulders, and topsoil (NRC81). A cross-section of the conceptual ISD Class C type slit trench for the LLW is shown in Figure 4-5 and the Class C type slit trench intruder barrier concept is depicted in Figure 4-6.

Because of the buffer zone requirements between the trenches, the total land requirements are 54.6 hectares versus 52.6 hectares for the conventional SLD. The site operations are similar to conventional SLD except for the compaction of the backfill and cap, and the thicker cover and intruder barrier over the trenches. Segregation of the three classes is still part of the acceptance procedures at the site, although only one trench design is used.

4.2.4 10 CFR 61 Disposal Technology

This is the method for disposal of commercial LLW required by the NRC under its 10 CFR 61 regulations (NRC82). One of the major requirements is to segregate the LLW into three classes, A, B, and C. The Class A LLW with improved packaging in the as-generated form is disposed of by the conventional SLD method, except that the waste is stacked in the trenches. The Class B LLW is solidified and disposed of by the conventional SLD method in trenches separate from Class A waste. Class C LLW, after solidification, is disposed of using the slit trench method described for ISD disposal (Section 4.2.3).

The Class A and B trenches are identical to those used for the SLD. The Class A and B trenches, 180 m long, 30 m wide, and 8 m deep, are identical to those used for conventional SLD. The distance between trenches of a single waste class is 3 meters, with a 30-meter buffer zone separating the Class A and Class B trenches. Wastes are stacked to a height of 5 meters, followed by backfilling the trench up to the original grade and placing an additional 1-meter thick cap. The backfill and cap are compacted using a vibratory compactor. Because of the reduced land utilization efficiency with slit trenches, as well as the buffer zone requirements between the Class A and the other trenches, the total land requirements will be larger than those for the conventional SLD. The 10 CFR 61 facility will occupy 54.6 hectares and the site operations will be identical to those for ISD.

4.2.5 Intermediate Depth Disposal

Intermediate depth disposal (IDD) is also designed to meet the requirements of 10 CFR 61. Instead of using an 8-meter deep slit trench for Class C wastes, however, the IDD facility uses a wider trench, 15 meters deep.

The IDD facility includes conventional SLD trenches and IDD trenches. The facility life cycle is 3 years for design and construction and 20 years for disposal operations, followed by a closure period of up to 5 years and a 100-year institutional care period.

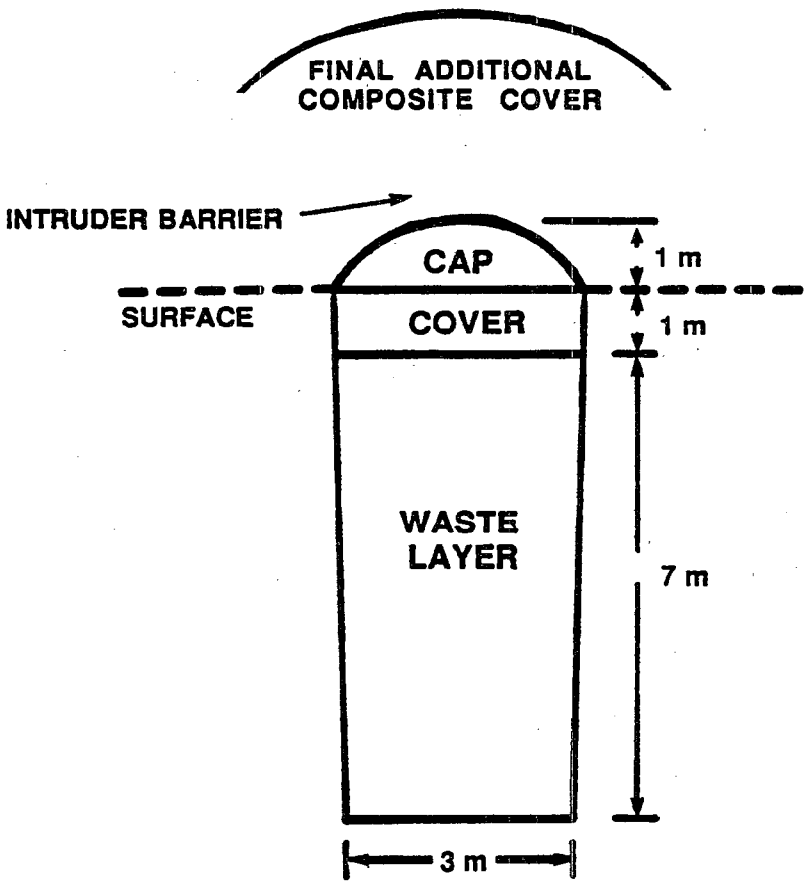

Figure 4-5. Cross-Section of an Improved Shallow-Land Disposal Class C Trench

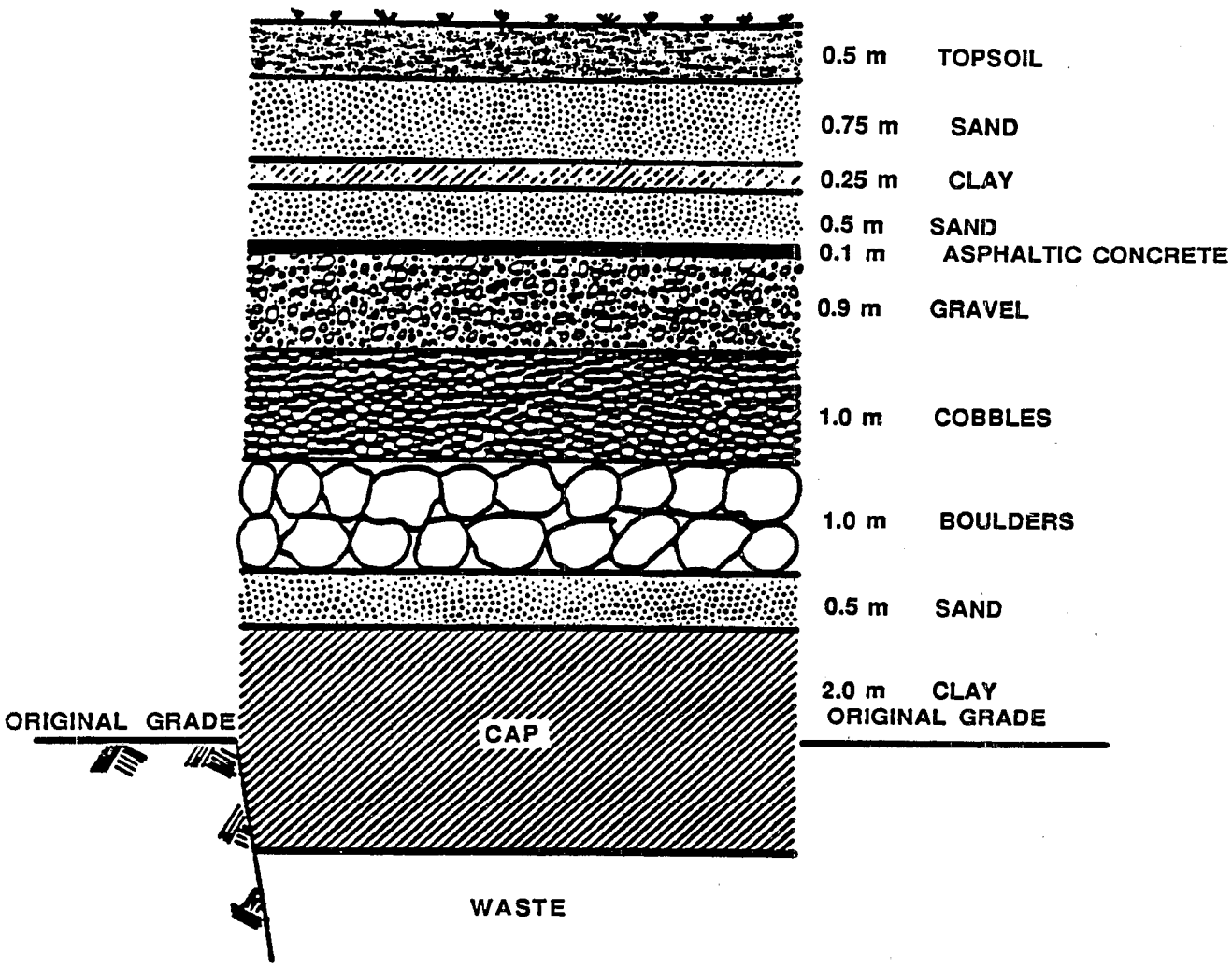

Figure 4-6. Engineered Intruder Barrier Form for Class C LLW Trench (NRC81)

The Class A and B trenches are identical to those used for the ISD, measuring 180 m long, 30 m wide, and 8 m deep. The distance between trenches containing the same waste class is 3 meters, with a 30-meter buffer between the Class A and B trenches. Wastes are stacked to a height of 5 meters, followed by backfilling the trench up to the original grade and then placing an additional 1-meter thick cap. The backfill and cap are compacted using a vibratory compactor.

Class C wastes are buried in deep trenches measuring 180 m long, 30 m wide at the base, and 15 m deep. A cross-section of such a trench is shown in Figure 4-7. The bottoms of these trenches are gently sloped to a French drain that extends the complete length of the trench. The trenches have steep walls which are shored to ensure safe operation. Wastes are stacked 6 meters deep in these trenches, using cranes from the top of the trenches and forklifts, when required, inside the trench. The trenches are then backfilled to the original grade, and a 1-meter thick cap is added to ensure proper surface runoff. The backfill and cap are compacted with a vibratory compactor.

The IDD facility will occupy 54.6 hectares and contain 13 Class A trenches, 2 Class B trenches, and 1 Class C trench. Site operations are identical to those for ISD.

4.2.6 Earth-Mounded Concrete Bunkers

The earth-mounded concrete bunker (EMCB) is the method used in France for the disposal of LLW (NRC84). The EMCB disposal method involves the excavation of trenches, construction of below-ground vaults and aboveground earth mounds, segregation of wastes according to their levels of radioactivity, and surveillance of the disposal site. A typical EMCB trench is shown in Figure 4-8.

It is assumed that trenches at the designed EMCB facility are 180 m x 30 m x 8 m. The sides of the trenches are sloped in a manner to provide temporary stability. The bottom of the trench is covered with a layer of concrete. A drainage system is provided on and around the concrete pad to collect any runoff or infiltration that may occur during the construction and initial operation stages.

The concrete bunker (CB) disposal units are composed of numerous compartments whose outside dimensions are approximately 6 m x 6 m x 6 m. Each wall is composed of 0.76-meter thick, steel-reinforced, cast-in-place concrete. Class B and C wastes are lowered by crane into the compartments in successive layers. After each layer within a compartment is completed it is backfilled with concrete. When the last layer of waste has been placed in a compartment, reinforcing steel is placed on top of the layer and the compartment is completely backfilled with concrete. This covers the top waste layer with 0.76 meter of concrete, thereby embedding the waste in one large concrete monolith. The monolith or combined concrete bunker compartment is generically described as a buried structure with fill.

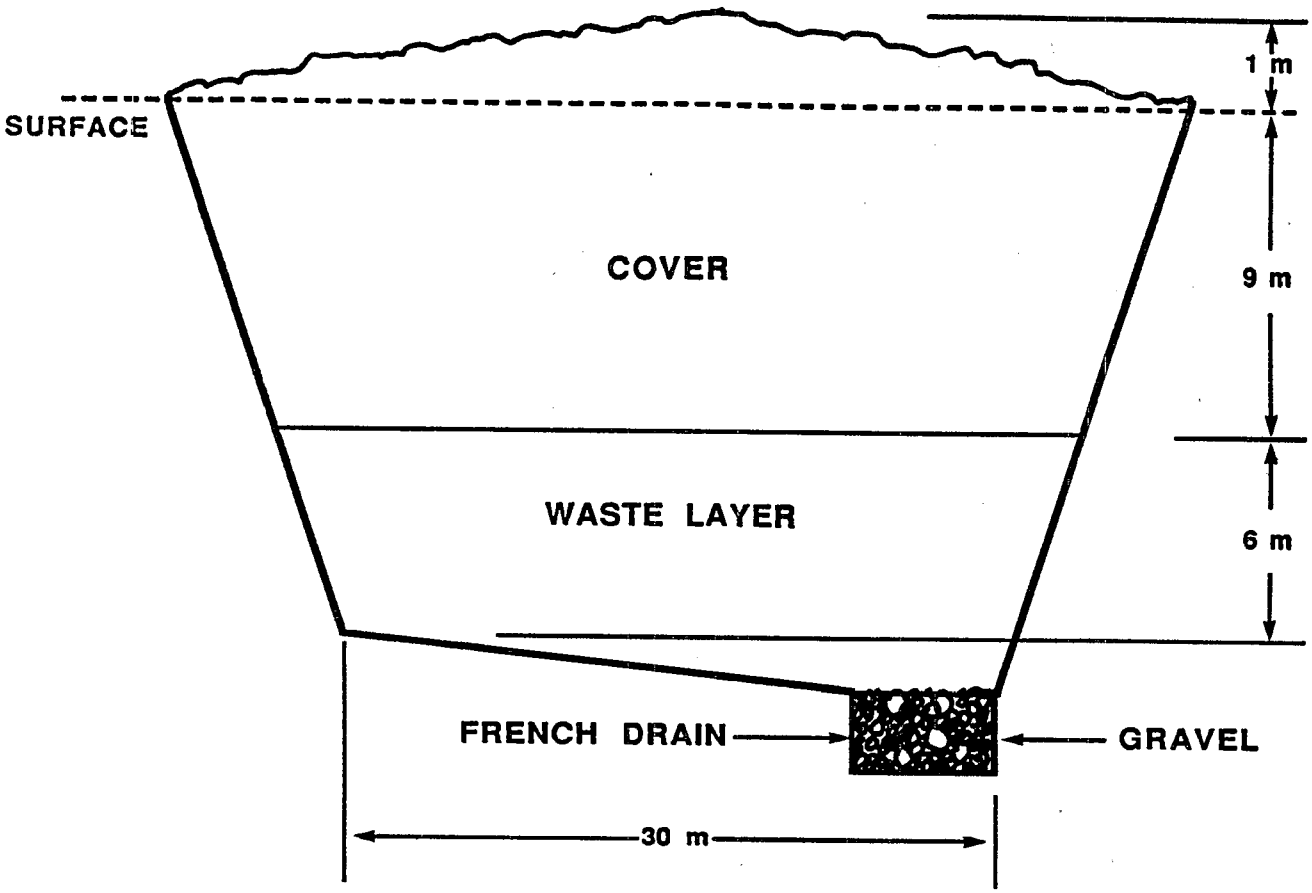

Figure 4-7. Cross-Section of an Intermediate Depth Disposal Trench

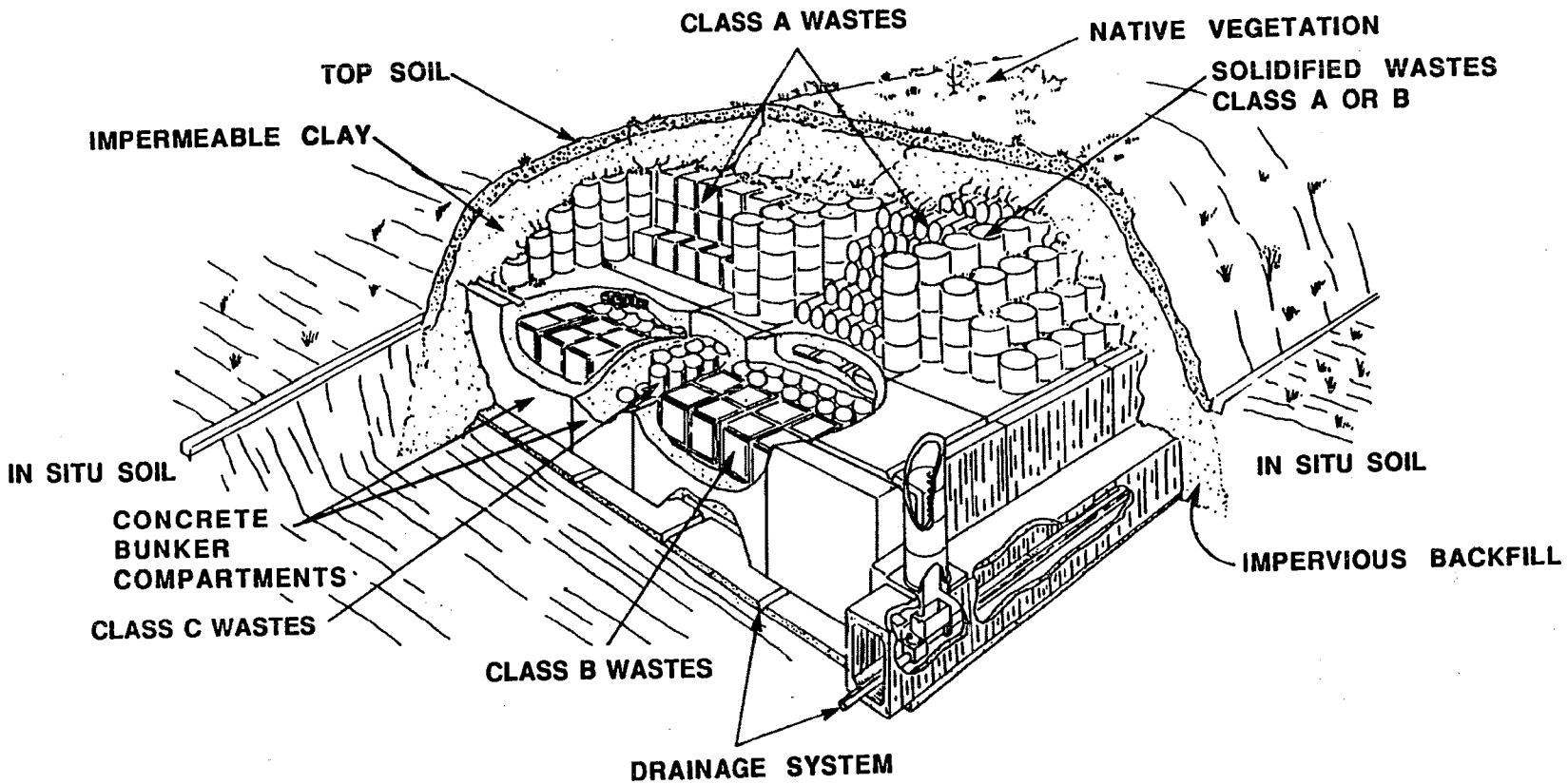

Figure 4-8. Schematic Diagram of the Earth-Mounded Concrete Bunker (NRC84)

The concrete bunker compartments are constructed in sequence. The construction operation is continuous, creating monoliths side-by-side until the trench is filled. Once the last concrete bunker in a trench is completed, the large concrete "platform" of monoliths is waterproofed with a layer of asphalt. Another drainage system is then installed to catch runoff accumulated during further construction.

Each 6-meter cubic cell will accommodate about 25 cubic meters of waste and each CB disposal unit, composed of 150 cells, is capable of disposing of about 3,700 cubic meters of waste. Six concrete bunkers are required for the disposal of all Class B and C wastes at the reference EMCB facility.

After a CB disposal unit is completed, an earth mound (EM) is formed from Class A containerized waste on top of the monolith, as shown in Figure 4-8. Some Class A and some low-activity Class B wastes are placed in cylindrical steel-reinforced concrete overpack canisters and stacked across the middle as well as around the perimeter to provide a structural framework for the earth mounds. These canisters are stacked by crane or forklift to a maximum height of about 7.5 meters. This stepped arrangement, together with the earth cover system, forms a sloping mound.

The voids between drums and canisters are backfilled with cohesionless earthen materials. This reduces the possibility of future settlement and promotes mound stability. When all concrete canisters and metal drums have been emplaced, the entire area is backfilled to increase the stability of the completed earthen mound.

Each concrete canister will hold an average of about 12 cubic meters of waste. The EM is composed of about 3,000 canisters plus about 1,200 drums containing solidified Class A or B waste. A total of 7 EM's will accommodate all the Class A waste to be disposed of in the reference EMCB facility. The EM is described generically as covered modules with fill.

Each EM is covered with a 2-meter cover system. The side slopes of the cover system must not exceed 1:4 (rise to run) in order to hold surface erosion to a minimum. The facility is surrounded by a final drainage system designed to collect rainwater flowing from the earthen cover system. The EMCB is completed by planting the covered EM with native vegetation to stabilize the surface soil and encourage drying.

The EMCB facility uses disposal site areas somewhat more efficiently than do other disposal technologies, largely because the EM's are located above grade and are placed on top of the concrete monoliths. The disposal site consists of about 6.1 hectares and, allowing for a 100-meter buffer zone around the disposal area, the total site area is about 24.3 hectares.

In addition to the disposal area, the site includes the following facilities:

1. A temporary storage area for Class A wastes, which will eventually be placed in the EM's.

2. A plant for treatment and conditioning of raw wastes in order to immobilize them prior to disposal. This plant is equipped with a press for compacting containerized wastes. The press is able to compact ten 200-liter drums into slabs that are placed in a single concrete container. The concrete container is subsequently filled with concrete.

3. Buildings for other technical and administrative functions, such as health physics, storage, and onsite maintenance.

4.2.7 Concrete Canister Disposal Method

The concrete canister (CC) disposal method uses concrete as a barrier to limit the release of radioactive material to the environment. One of the most important features of this method is its retrievability option. The system allows the canister to be removed so that waste can be retrieved or remedial action taken. Wastes received at the disposal site are expected to be generated by hospitals, universities, research laboratories, utilities, and nuclear fuel-cycle industries.

This method involves a system of natural barriers complemented by engineered barriers. All three classes of wastes are received at the facility, where they are re-packaged in precast concrete overpacks called Subsurface Recoverable Packaging Systems (SUREPAKs).* These SUREPAKS are illustrated schematically in Figure 4-9. The contents are inventoried and the inventory list saved for use in identifying the location of specific wastes in the disposal trench. Wastes are disposed of in a shallow-land disposal trench excavated in accordance with the criteria in 10 CFR 61.

The site design is based on the requirements of 10 CFR 61. In many respects, it is identical to a conventional SLD facility design. The waste site would have an area of about 259 hectares. The disposal area is surrounded by a buffer zone about 200 meters wide, which permits monitoring and allows corrective measures to be taken, if required. To better characterize the type of facility required and to allow cost estimates to be made, it is assumed that the following buildings onsite will be: (1) a guard station and driver's day room, (2) an office building, (3) change rooms and a laboratory, (4) a decontamination building, (5) a maintenance building and warehouse, (6) a waste compaction facility, and (7) a waste packaging and documentation building. The access road and site access descriptions are the same as for SLD (Section 4.2.2).

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*A proprietary system of Westinghouse Electric Corporation.

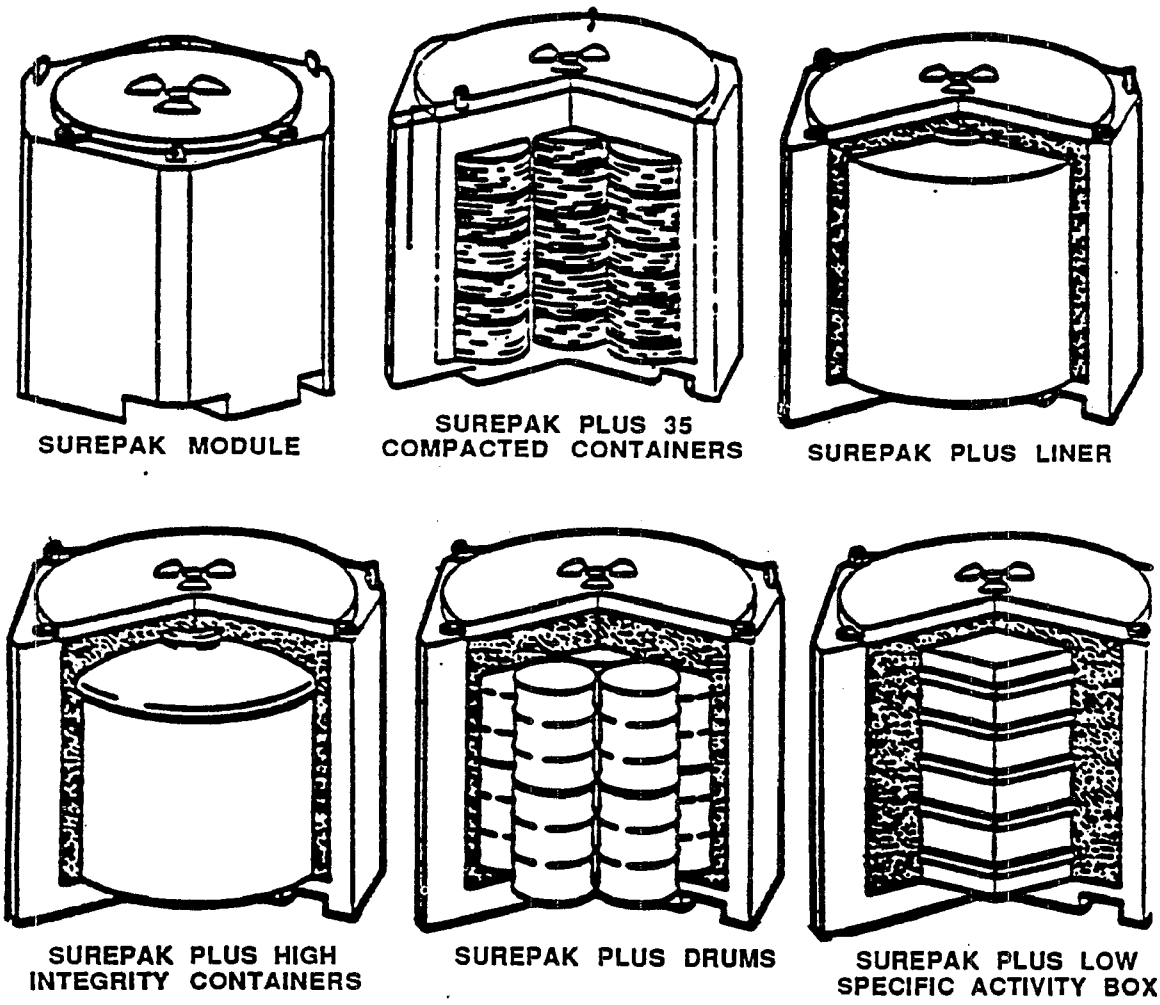

Figure 4-9. SUREPAK Module and Contents

When wastes arrive onsite, the shipment documents (manifests) are processed and the waste packages are inspected by the health physics personnel to ensure compliance with Federal and State regulations. If the packages satisfy the appropriate regulations, the transport vehicle is directed to the waste packaging and documentation building. All wastes are re-packaged into SUREPAK modules that hold from 3 to 5 cubic meters of waste and grout. This is equivalent to 14 standard drums (each 0.2 cubic meter in volume) or up to 35 compacted 200-liter drum containers. The SUREPAK will also accept standard low specific activity 1.2 m x 1.2 m x 1.8 m boxes, special liners, and high integrity containers.

The filled SUREPAKs are then placed in a trench. A typical disposal trench with waste in place is shown in Figure 4-10 (WEC85). The trench is 154 m x 37 m x 7.8 m and has a volume of 44,500 m^3. The trench walls are assumed to have a fairly steep slope of 1:4, representative of cohesive soils. The bottom of the trench slopes gently toward a French drain that extends the whole length of the trench. Water is pumped from this drain. The SUREPAKs are stacked in the trench in approximately 72 rows, with 57 modules in each row. Using this maximum disposal configuration, a total of approximately 4,100 modules can be disposed of in one trench.

Wastes that have been packaged in the SUREPAK module are placed in the trench by means of forklifts and arranged in rows as described above. The modules are self-shielding, enabling emplacement of high-activity waste in the same trench as low-activity waste, with minimal disruption of disposal operations. This disposal technique, with warning labels and index numbers on each module, makes it possible to retrieve the waste.

As the wastes are placed in the trench, the trench is backfilled with dirt that was removed during trench excavation. The backfill material fills in the voids between the waste packages prior to building up the cap with layers of material to protect the waste and to control water runoff. The cap is 4.3 meters thick and is composed of (1) an alluvium structural cap, (2) a gravel capillary barrier, (3) a cobble drain trench, (4) a silt layer, and (5) a graded riprap and topsoil layer. When the trench is filled and the cap is completed, the boundaries and location of each trench are mapped by means of a land survey, as required by 10 CFR 61.

During the post-operational period, the facilities and equipment are decontaminated to remove all residual radioactive material using standard techniques. All buildings and structures onsite are demolished and the refuse is disposed of in accordance with the regulatory requirements for such wastes. Closure and post-closure operations are the same as for SLD (Section 4.2.2).

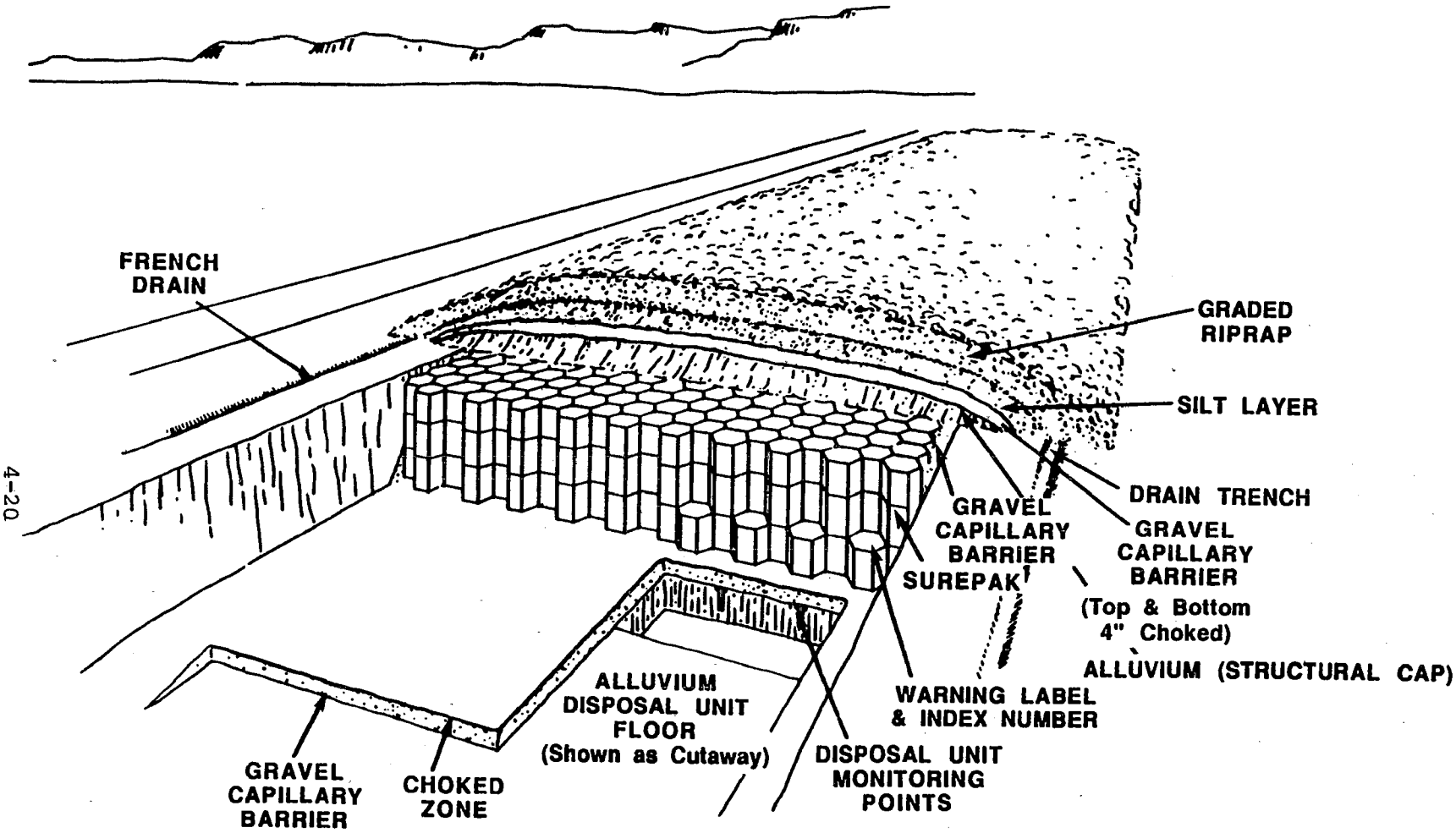

Figure 4-10. A SUREPAK LLW Disposal Unit (WEC85)

4.2.8 Deep-Well Injection

Deep-well injection (DWI) consists of injecting liquid wastes into a deep (300 to 3,000 meters), permeable formation, the liquid contents of which have no fresh water or mineral value (En84a). Deep-well injection (see Figure 4-11) is a technique developed by the oil industry for the disposal of oil field brines. As a disposal method for hazardous wastes, DWI is distinctly different from most hazardous waste disposal concepts. The basis of most disposal options is the immobilization of the waste in a region isolated from the biosphere. The objective of DWI is not to immobilize the waste per se, but to pump it into a porous formation that is confined by impermeable layers. With DWI, the waste remains in a liquid form and may disperse within the formation. Some disposal of liquid LLW through DWI has been done in the past and numerous studies were made to evaluate the method (Do64, Wa65, Te72, Re77, Wa77, EPA77, Pe82).

That the waste is not immobilized may not be a real deficiency of DWI. If the impermeable layers between the waste formation and useful ground water remain intact, the natural pathways for the waste to enter the biosphere will remain blocked. In some cases, wastes can be injected into a recharge area (the direction of water motion is down) of a deep basin where the overlying formations may have some moderate permeability. These cases rely on the extremely long transit times (thousands of years) and the dilution effects of the in-situ water to protect the biosphere.

In both cases, drilling into the wastes should be avoided. It is assumed that the probability of this event is proportional to the size of the area throughout which the waste has dispersed and the likelihood of natural resources in the vicinity. It is also assumed that the likelihood of this event can be minimized by restricting DWI to those areas and horizons in which deeper formations have little or no economic value.

A potential problem with DWI is the question of land ownership and mineral rights. Since DWI wastes can disperse over a considerable area, it is difficult to predict the extent of required land ownership.

Some other difficulties of DWI are:

- the availability of sufficient wastes in a chemical and physical form suitable for DWI to provide the necessary economies of scale;
- the availability of suitable rock formations; and
- transportation of the liquid waste from the point of generation to the DWI site.

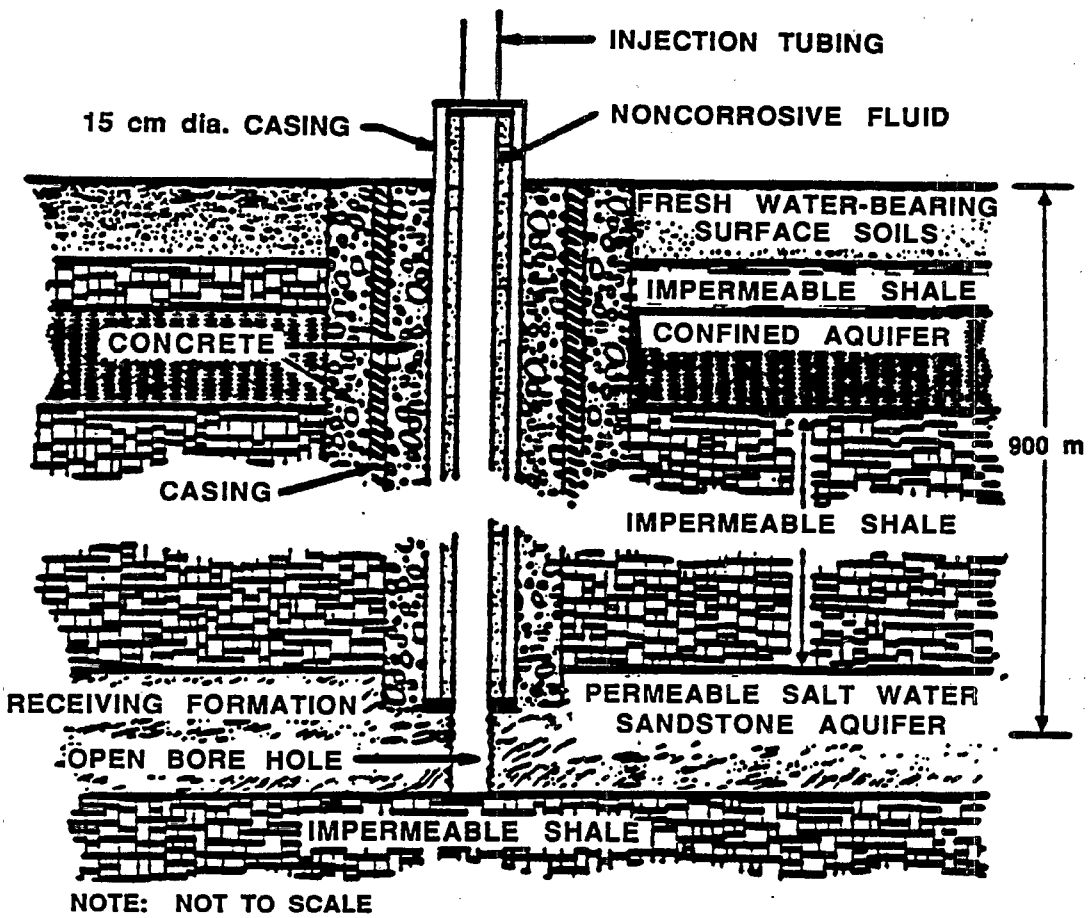

Figure 4-11. Profile of a Deep-Well Injection Facility

Because of the severe restrictions on transporting liquid nuclear wastes, it is assumed the DWI site will be located on or adjacent to the site of a light water reactor, DOE facility, or other licensed facility. The DWI site is assumed to have an operating life equal to the facility, which is assumed to be 40 years.

The DWI site is assumed to be located on the site, with the wellhead 800 meters from the facility. Liquid wastes are transported to the DWI site through a buried pipeline which can feed directly either a set of tanks for temporary storage or the wellhead. The well extends 900 meters into a porous sandstone formation having thick, impermeable shale layers above and below it. The hydraulic pressure in the sandstone is 600 meters of water column.

DWI is currently regulated by either State or Federal agencies using the EPA standards (EPA74, EPA79) in the Underground Injection Control (UIC) Program (40 CFR 146). The requirements for a UIC Class I well have been used to design the generic DWI facility. A Class I well injects hazardous wastes into a formation located beneath the lowermost formation containing an underground source of drinking water.

We recognize the limitations of transporting liquids, but for our analyses we assumed an NRC-licensed or DOE-regulated facility using the following liquid waste streams: I-ABSLIQD, L-CONCLIQ, and L-DECONRS.

The well design is shown in Figure 4-11. A hole is drilled through the shallow aquifers and a surface casing is cemented in place. A smaller diameter hole is then drilled within 168 meters of the target sandstone formation. The last 168 meters are cored to provide samples of the overburden. The coring operation is stopped at the top of the sandstone, and the hole is logged with a gamma, density, neutron, and acoustic geophysical open hole log. A 15-centimeter diameter casing is then cemented in the hole. A pilot core hole is continued 15 meters into the sandstone to provide an adequate drainage area. The hole is completed by lowering a 7-centimeter plastic-lined injection tubing into the hole with an inflatable packer at the end of the tubing. The packer is inflated to provide a seal between casing and tubing, and the tubing annulus is filled with a noncorrosive fluid.

The wellhead is sheltered by a modular metal building on a cement pad to allow winter operation. This building is large enough to accommodate the use of forklifts and winch trucks near the wellhead. A small metal warehouse is also located at the DWI site for supplies, as well as two 37 m^3 fiberglass tanks for the temporary storage of wastes. These buildings are surrounded by a high fence for security. A gravel road is required to transport the wastes from the facility to the DWI site, to provide access to the DWI site, and to allow heavy equipment access to the pipeline for maintenance.

The liquid LLW's are pumped to the process building located adjacent to the DWI facility. The LLW's have already been concentrated in evaporators and may be combined with nonradioactive wastes. In the process building, some of these liquids are chemically treated to minimize their corrosion properties and maximize their compatibility with the sandstone formation. The wastes are either temporarily stored at the process building or pumped directly to the DWI site.

The DWI site operates for 8 hours a day, 5 days a week. During typical operations, the LLW flows directly from the pipeline into the wellhead. Since the 600-meter water column in the wellbore is sufficiently larger than the 400-meter backpressure in the sandstone, no pumping is required to inject the liquid waste. If problems develop near the wellhead, the waste in the pipeline can be diverted to the storage tanks.

An alarm system automatically shuts the pipeline down if a leak is detected. Periodically, the pipeline is hydrostatically tested to ensure there are no leaks. During waste injection operations, the fluid in the annulus between the injection tubing and well casing is pressurized. This pressure is also monitored to detect leaks in the downhole tubing or casing.

At closure, the DWI surface facility will be decontaminated and the resulting liquid wastes pumped into the well. The injection string will then be removed from the well casing, and a sonic log survey will be conducted to ensure adequate bonding between the formation and the original cement job outside the casing. In zones where the bond is not satisfactory, the casing can be perforated and a squeeze cement operation can be initiated. Finally, the inside of the casing is sealed with a combination of clay, cement, and additives to ensure a good bond and minimize shrinkage. The clay also increases the ion exchange capacity of the grout column.

4.2.9 Hydrofracture

Hydrofracture (HF) consists of mixing liquid and pulverized solid wastes with a grout and pumping this grout into an extremely impermeable rock formation such as shale. The grout forms a thin, horizontal sheet in the rock formation where it solidifies and immobilizes the wastes.

A novel disposal method, HF has been used at Oak Ridge National Laboratory to dispose of higher activity (270 Ci/m^3) liquid LLW (AEC74, La70, We83, DOE85). Hydrofracture tests were also conducted at West Valley, New York, from 1969 to 1971.

Because of the restrictions on transporting liquid nuclear wastes, the HF site is assumed to be located 1,600 meters from a facility. This distance is larger than the 800 meters used for DWI to ensure that the facility is not affected by the surface uplift from the HF grout injections. The DWI process does not cause a measurable uplift. The HF site has an assumed operating life of 40 years.

A wider variety of waste forms are compatible with HF than with DWI. Wastes can be in either a liquid or a pulverized solid (40 mesh or smaller) form for HF. We recognize the restrictions on transporting liquids; however, for our HF analysis we assumed an NRC-licensed or DOE-regulated facility and used the following liquid or semi-liquid waste streams for analysis: L-CONCLIQ, I-ABSLIQD, L-DECONRS, L-FSLUDGE, L-IXRESIN, and R-RAIXRSN.

The HF site consists of an injection well, two monitor wells, a small warehouse, and a shelter over the injection wellhead. It is assumed that the HF well will be regulated as a Class I well under the UIC Program (40 CFR 146). A Class I well injects hazardous wastes into a formation located beneath the lowermost formation containing a source of drinking water within 400 meters of the wellbore.

The injection well design is shown in Figure 4-12. A hole is drilled through the shallow aquifers and a surface casing is cemented in place. A smaller diameter hole is then drilled to 300 meters through the target shale formation and cored over the last 90 meters. This hole is then surveyed with a full suite of geophysical logs, and a 15-centimeter diameter casing is cemented in the hole.

A modular metal building on a cement pad houses the wellhead and the HF injection equipment. This building has ample room for using forklifts and winch trucks near the wellhead. A small, metal warehouse is also located at the HF site. These buildings are surrounded by a high fence for security. A buried pipeline is used to transport the wastes from the facility to the HF site. A gravel road is required to provide access to the HF site and allow heavy equipment access to the pipeline for maintenance.

The LLW is sent to a waste process building located adjacent to the HF facility. Resins and sludge are pulverized, combined with concentrated liquids, and temporarily stored in fiberglass tanks. When 250 cubic meters of wastes are accumulated, the slurry is chemically treated to obtain a pH between 8.0 and 9.0, filtered, and pumped to storage tanks at the HF site.

In preparation for a grout injection, the casing is circumferentially perforated with a sand drill at the injection depth. The slurry is blended with fresh water, cement, flyash, and additives and injected into the well at approximately 950 liters (L) per minute (L/min) and 210 kilograms per square centimeter (kg/cm^2). This grout forms a thin (1-centimeter thick), horizontal pancake in the formation, with an average radius of about 110 meters and a maximum radius of 230 meters. It typically solidifies in 1 to 7 days. Observation wells are used to ensure that the grouts are horizontal rather than vertical.

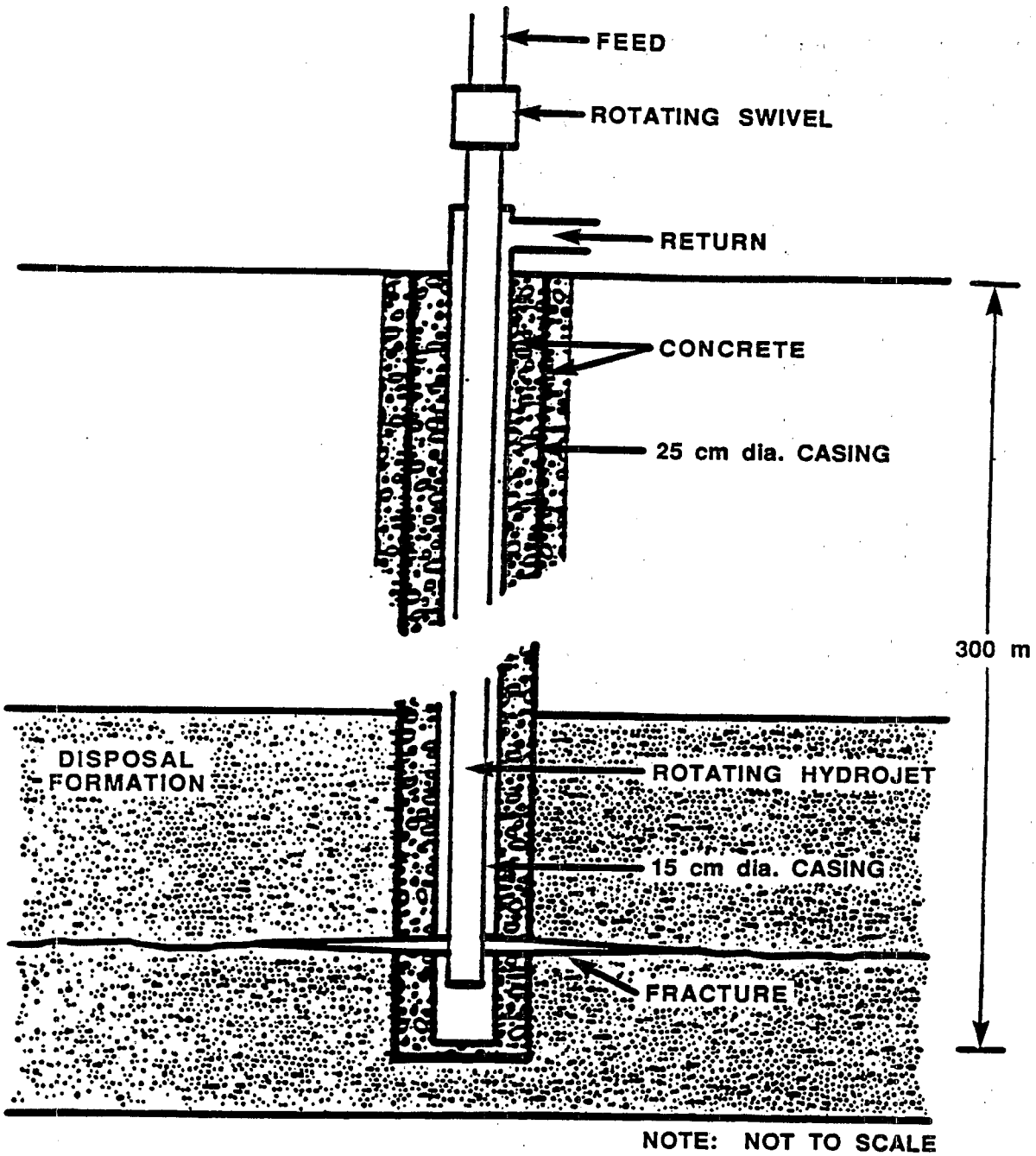

Figure 4-12. Hydrofracture Well

Each injection consists of approximately 530 cubic meters of grout, which contains approximately 1 kilogram of cement, 1 kilogram of flyash, and 0.2 kilogram of pottery clay per 3.8 liters of waste slurry. This waste slurry is assumed to be 2.3 kilograms of fresh water and 1.8 kilogram of wastes. Four separate 530-cubic meter injections are made into one perforation. Then the wellbore is grouted above the old perforation and a new perforation is made 3 meters above the old one. This continues until the usable section of the formation is exhausted.

A typical injection sequence consists of the following:

1. 1,900 liters of fresh water
2. 530 cubic meters of grout with wastes
3. 9.5 cubic meters of grout with fresh water
4. 1,900 liters of fresh water
5. Wiper plug
6. Well sealed under pressure.

This grout formulation was developed by ORNL to provide acceptable workability, compressive strength, and fluid loss, as well as high ion exchange capacity to immobilize the radionuclides. The use of flyash allowed use of a lower amount of cement, which decreased the soluble calcium compounds in the mix and increased the absorption coefficients for radionuclides such as strontium-90. The grout mix can be adjusted to optimize its performance for a given set of wastes.

For our analysis (based on previous ORNL operational history and waste stream quantities), it is assumed the facility will require 61 injections over a 40-year period or one every 9.5 months. These 61 injections will occupy a height of 50 meters in the host formation. Each injection contains 250 cubic meters of LLW and has a total activity of approximately 2,400 curies. Assuming 4.4 mW/Ci, each injection initially generates 10.6 watts of heat in the formation. Since iron-55 with a half-life of 2.6 years accounts for about half of this activity, the heat generation rate will decay significantly over the 40-year life of the HF facility. Even if the rate remained constant, all the injections would generate only 600 watts distributed over a 4-million metric ton mass of shale.

If the heat source is assumed to be a sphere with a radius of 25 meters and a uniform heat source of 0.009 watt/m^3, then the maximum temperature increase in the earth mass is approximately 0.2 degree centigrade (C) (assuming a shale conductivity of 0.148 watt/cm-degrees C). Since most shales remain stable up to 100 degrees C, even a two order of magnitude temperature increase over this calculated value will not be a serious problem. The HF site operations are closed in the same way as those at the DWI site.

4.2.10 Deep Geological Disposal

Deep geologic disposal (DGD) isolates the LLW from the biosphere by using a mined cavity in an impermeable formation. Proposed host rocks include shale, basalt, salt, granite, and others. Depths of the cavity could range from 30 to 1,000 meters.

This facility is modeled as a cavity located at a depth of 300 meters in either salt or shale. This DGD has a 5-year design and construction period, a 20-year operating life, a 2-year closure period, and a 100-year long-term care period. The total waste disposal capacity is assumed to be 168,000 cubic meters, since only LLW with high activity would be disposed of here. The underground facility is rectangular in shape with two long main drifts (600 meters each) and three lateral drifts (300 meters each). Each main drift is 10 m wide and 3 m high. Located off the perimeter of the lateral drifts are rooms that are 5 m wide, 3 m high, and 150 m long. A 10-meter thick pillar is maintained between these exterior rooms. In addition, rooms are dug between the long drifts, with 20-meter thick pillars separating these interior rooms.

Three shafts connect the underground facility with the surface (see Figure 4-13). The main shaft is 7.5 meters in diameter and contains both a material lift and a personnel lift. The material lift is 5 m by 3 m. It is used for raising and lowering equipment (e.g., forklifts), excavated material, backfill, and wastes. Since the rate of waste emplacement is anticipated to be approximately 4 m^3/h when assuming a single shift operation, one material lift should suffice for all these purposes. For redundancy (safety) there are three personnel lifts, one in the main shaft and one in each vent shaft. They measure 3 m by 2 m. The vent shafts are 5 meters in diameter and are used for air circulation, utilities, and communications.

Forced air circulation is required to prevent the buildup of pollutants in the mine air, ensure an adequate supply of oxygen for the crews, and control moisture and temperature. To ensure an adequate supply of fresh air, there are redundant blowers at the surface facilities, and air is moved around at the disposal level by forced air ducts or blowers. Return air is filtered before being exhausted at the surface. The surface facilities also have redundant sources of power to provide reliable lift operation.

The surface facilities include administration, health physics/ security, warehouse/garage, and waste activities buildings, as well as shelters for the surface ventilation equipment. The surface facilities are surrounded by a high fence.

When a waste shipment arrives on the site, its shipment documents (manifests) are processed while the waste packages are inspected by health physics personnel to ensure compliance with Federal and State regulations. If the packages meet the appropriate regulations, the

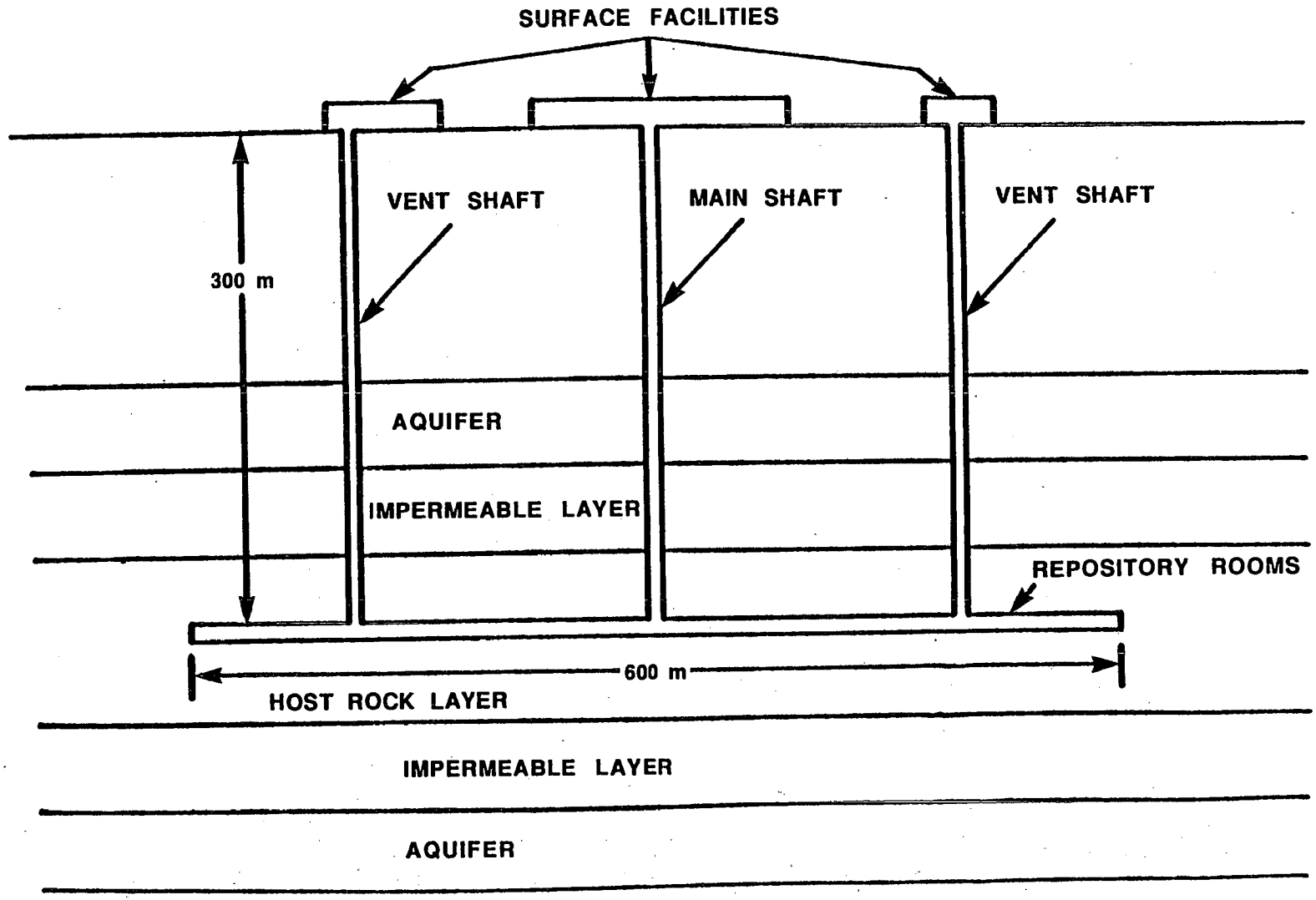

Figure 4-13. Profile of a Deep Geological Disposal Facility

transport vehicle is directed to a parking area near the main shaft. The waste packages can be either unloaded immediately for placement on the material lift or stored temporarily on the transport vehicle in this parking area. The waste activities building also contains a limited amount of waste storage space. If the waste packages do not meet the appropriate requirements, remedial action is taken using the facilities in the waste activities building. These facilities include a decontamination bay, a liquid treatment area, and a waste solidification/packaging area.

The wastes are lowered into the mine on the material lift, unloaded with forklifts, and transported to a room for waste emplacement. A waste emplacement efficiency of 50 percent is assumed in each room. After two days of waste emplacement operations in a given room, the area filled with wastes is backfilled using the fines generated by the mining process.

The mining of the rooms is conducted by a continuous miner with belt haulage from the operating face to screening equipment which separates the fines from the bulk rocks. The fines are used for room backfill; the bulk rocks are removed from the mine on the material lift and placed in a rock storage pile. The rock storage pile is a 1-hectare area with a 0.2-hectare settling pond adjacent to it. The storage area and pond are underlain by a hypalon liner and 0.6 meter of clay.

After 20 years of operation, closure operations begin. All the surface facilities are dismantled except for the health physics/security building and the fence. The contaminated debris from the surface facilities is placed in the last room of the mine and backfilling of the main tunnels begins. Backfilling consists of filling rooms and tunnels with crushed rock. Thick, grout barriers are emplaced at selected locations. All shafts are grouted from the mine to the surface. The surface rock storage area is covered with 7.5 cubic meters of asphalt to minimize the infiltration of water through the rock pile.

4.3 BRC Waste Disposal Methods Considered

In order to assess the radionuclide transport and potential health risks resulting from disposal of surrogate BRC waste streams by typical solid waste disposal practices, six types of generic "disposal methods" have been defined. These methods are identified in Table 4-1. Detailed information concerning conceptual design parameters and other modeling assumptions that are common to each disposal site regardless of hydrogeologic/climatic setting is presented in this section. In Chapter 5, data and modeling assumptions that normally change depending on site-specific hydrogeology and climate are discussed in detail. The area populations were assumptions made by EPA.

Table 4-1. Generic BRC waste disposal methods

Disposal site	Acronym	Area population
Suburban sanitary landfill	SF	175,000
Suburban sanitary landfill with onsite incinerator	SI	175,000
Urban sanitary landfill	UF	1,000,000
Urban sanitary landfill with onsite incinerator	UI	1,000,000
Rural municipal dump	MD	60,000
Suburban landfill on the waste generator's property with pathological incinerator	LURO	175,000

Parameters affecting the simulation of radionuclide transport from a disposal site include the size, volume, and area of the disposal facility; depth of waste and cover; cap failure rates; types of wastes accepted; operational, closure, and post-closure care periods; permeability, porosity, and density of trench contents; and incinerator type, controls, and incineration rates (where applicable). The assessment of health effects caused by disposal of surrogate BRC wastes includes risks for disposal site workers and visitors. Therefore, the conceptual design for each site also includes the number of workers and onsite visitors; their level(s) of exposure to airborne dust and surface (gamma) radiation; and rates of disturbance of disposal site surface by mechanical and natural means.

If disposal of a waste stream was not subject to regulation on the basis of radioactivity, generators would choose between disposal options available to them based on economic criteria and attempt to minimize transportation, processing, and disposal costs. Of course, conformance with Federal, State, and local regulations governing the storage, transportation, and disposal of hazardous wastes would still be required.

In this assessment, it is assumed that the waste characteristics of individual streams allow application of the disposal alternatives being evaluated. Conservatively, generators are assumed to ship the BRC-surrogate wastes by themselves to locally situated disposal sites where they would be mixed with much larger quantities of nonradioactive materials. None of the wastes are assumed to be contained in drums or other packaging. Free liquids are assumed to be mixed with absorbent materials, because most municipal facilities will not accept liquid wastes of any type.

The disposal methods identified in Table 4-1 imply a distinction between "sanitary landfills" and "municipal dumps." EPA has promulgated "Guidelines for the Land Disposal of Solid Wastes" (40 CFR 241-257). The facility design and modeling assumptions used to simulate disposal of surrogate BRC waste streams in a sanitary landfill are meant to coincide with conditions at a facility in compliance with these guidelines. Less stringent operational conditions, such as those that might be present at a facility not in complete compliance with these guidelines, are used to simulate operational conditions at a municipal "dump," realizing that some do exist. (This is not meant to imply that EPA is encouraging noncompliance with existing regulations governing land disposal of solid wastes.) The major differences between the modeling assumptions used for municipal dumps versus sanitary landfills include: (1) the assumed initial and final values for the percentage of the trench cap that failed; (2) depth of cover; (3) distance from the bottom of the trench to the aquifer; (4) concentration of airborne dust onsite; (5) rate of mechanical disturbance of the disposal site surface; (6) fraction of surrogate BRC wastes spilled directly onto the surface of the landfill; and (7) the number of onsite workers and visitors exposed to airborne dust and surface gamma radiation.

Disposal methods evaluated also include onsite disposal on the generator's property after onsite incineration. In these cases they are institutional generators using a pathological incinerator.

4.3.1 Data Requirements

Numerous waste-specific parameters must be assigned to the PRESTO-EPA-BRC and PATHRAE codes. The more general parameters pertaining to specific BRC scenarios are discussed here, while Appendix C includes the more specific code parameters.

It is assumed that wastes placed in the landfill are a homogeneous mixture from a variety of sources. The mixture is composed primarily of nonradioactive wastes, such as those typically disposed of in a sanitary landfill or municipal dump (e.g., paper, rubble, metal, glass, plastic, and biodegradable wastes). A relatively small fraction of the waste is composed of radioactive surrogate BRC waste streams. The identity and quantity of each BRC waste depends on the scenario under analysis. Because the waste composition at the disposal site is primarily nonradioactive, the nonradioactive constituents predominate in interactions among the waste, soil, and rainfall that infiltrate through the landfill.

It is assumed that none of the wastes are containerized for disposal. No waste processing such as compaction, solidification, or incineration is assumed to occur at the generator. Two waste forms are used in the simulation. Wastes are placed as-received (waste form = trash) or after incineration at the landfill (waste form = ash).

Sanitary landfills are currently used for the disposal of non-hazardous solid wastes. This disposal method involves daily replacement of a dirt cover over the disposed refuse material in a manner designed to minimize environmental pollution. As previously mentioned, the standards for design and operation of a municipal sanitary landfill have been established by EPA (40 CFR 241-257); however, the actual licensing of such a landfill is normally conducted by municipal or county agencies.

4.3.2 Suburban Sanitary Landfill

In this assessment, a suburban sanitary landfill (SF) is assumed to receive the normal variety of nonradioactive solid wastes, together with a relatively small quantity of surrogate BRC waste streams.

(A) Site Design and Operations

The reference SF has an assumed period of active operation of 20 years. Its capacity is 6,000,000 cubic meters. Land requirements include approximately 100 hectares for the operational zone (1,000 m x 1,000 m) and a 50-meter buffer zone surrounding the disposal facility.

Assuming the SF serves a population of 175,000 with a daily trash emplacement rate of 2.6 kg/person/day, the suburban landfill will process approximately 454 tonnes (t) per day. The operating schedule is assumed to consist of one 8-hour shift per day, 5 days per week, 52 weeks per year. The facility, therefore, operates approximately 23 percent of the time during its 20-year lifetime.

Operations at the facility consist of the daily receipt of wastes, which are dumped onto the working face of the landfill. A bulldozer spreads and compacts the waste onto the slope left from the previous day's cover. At the end of each day's activity, the waste is covered with 0.15 meter of soil obtained directly in front of the working face. When a large enough area has been filled, an additional 0.45 meter of soil is placed over the waste cells, resulting in an average cover depth of 0.6 meter. The average depth of waste material in the trench is assumed to be 6 meters. Typical volumetric ratios of waste to cover material range from 3:1 up to 4:1.

The density, porosity, and permeability of the trench cover are assumed to be re-worked to original site conditions. If necessary, the site design may include a gas vent system to control local air quality. The conceptual sanitary landfill is pictured in cross-section in Figure 4-2.

Because of the spreading and compaction of wastes with a bulldozer and economic considerations at the generator, it is assumed that all wastes (including the surrogate BRC wastes) are not containerized upon receipt, and are homogeneously mixed with the daily cover. The porosity, density, and permeability of the trench materials are 0.25, 0.59 g/cm^3,

and 31.54 m/yr, respectively. Approximately 1 percent of the activity present in the BRC wastes is assumed to be mingled with the cover and exposed at the surface of the landfill (spillage). The remaining 99 percent (of radioactivity) is blanketed below the cover.

During the operational period, the SF is assumed to receive wastes at a constant rate until its capacity is reached after 20 years. The radioactive waste inventory of the SF is decayed throughout the operational period. Throughout the operational period, transport of radionuclides is assumed to be possible through natural and mechanical disturbance of the cover. After closure, additional transport mechanisms, including leaching and trench overflow, begin.

Prior to closure and the license termination, the landfill will be inspected by the regulating agency. There is no period of post-closure maintenance.

The trench cap area is assumed to begin to fail from the first year after closure, directly exposing that fraction of the surface area of the waste contents of the trench. The trench cap failure increases at a constant rate until it reaches 30 percent of the area of the cover 40 years after closure. Trench cap failure is then assumed to remain constant at 30 percent for the remainder of the period of analysis.

(B) Staffing Requirements and Site Visitors

In order to compute health effects caused by inhalation of radioactive dust and gamma exposures to workers situated in proximity to the surrogate BRC waste streams, assumptions concerning the number of onsite workers and their degree of exposure are required. Methods to estimate these parameters are based on work reported by Oztunali and Roles (Oz84).

Staff requirements for a 454-t/d facility are estimated to be two equipment operators and one weighmaster. Their exposure to airborne dust and proximity to gamma sources are summarized below (Oz84).

Worker classification	Number	Dust exposure ($\mu g/m^3$)	Gamma source distance (m)
Equipment operator	2	High (400)	1
Weighmaster	1	Low (100)	50

To estimate the number of visitors to the SF during its operational period, the following assumptions are made:

1. Four percent of all wastes generated are "bulky wastes" brought by visitors (e.g., "spring cleaning").

2. The average "visitor" brings 182 kilograms of waste to the disposal site, stays 30 minutes, and experiences moderate exposure to airborne dust (200 $\mu g/m^3$) and moderate proximity to gamma sources (distance = 10 meters).

Then, to arrive at the number of visitors, the following calculations are made:

0.04 x 454,000 kg/d = 18,200 kg/d
18,200 kg/d ÷ 182 kg/person = 100 visitors/d
100 visitors/d x 313 d/yr = 31,300 visitors/yr
31,300 visitors/yr x 0.5 h/visitor = 16,650 visitor h/yr (moderate exposure).

A weighted average of worker- and visitor-hours of exposure to high airborne dust loadings and close proximity to gamma sources is used to compute total health effects to onsite personnel.

4.3.3 Suburban Sanitary Landfill with Onsite Incineration

At some sanitary landfills, wastes are incinerated prior to the land disposal of ash and rubble. This section provides basic design and operational criteria for such a facility.

(A) Site Design and Operations

The reference suburban sanitary landfill with incineration (SI) has an assumed period of active operation of 20 years. Its capacity is 1,000,000 cubic meters. Land requirements are approximately 16 hectares (400 m x 400 m), and are reduced by a factor of 6 compared with the SF, due to volume reduction achieved through incineration. There is also a 50-meter buffer zone surrounding the disposal site.

The daily rate of waste processing and the daily schedule of land disposal operations are identical to those described for the SF in Section 4.3.2.

Operations at the facility are similar to those described by Oztunali (Oz84). Operations begin with the daily receipt of wastes in bulk carriers (e.g., compactor or dump trucks). The wastes may be stored for several days in pits prior to incineration. The waste is incinerated at 700° to 980°C. Residue consists of ash, cans, glass, rocks, etc., and a volume reduction factor of approximately 6.0 is typical. In order to process wastes received at a rate of 454 t/d, two incinerators feeding a

common stack are assumed. The incinerator stack has a height of about 61 m, diameter of 2.2 m, and exit velocity of 15.9 m/sec. See Appendix C for incineration parameters.

(B) Staffing Requirements and Site Visitors

Staffing requirements are greatly increased at the SI. In addition to the two equipment operators and one weighmaster involved in landfill operations, many additional incinerator personnel are required. These include a superintendent, assistant superintendent, office manager, secretary, two additional weighmasters, two crane operators, six charging floor operators, four process controllers, four residual handlers, and eight other laborers. These requirements are based on estimates by Oztunali (Oz84). Total staffing requirements, together with airborne dust exposures and proximity to gamma sources, are presented below.

Dust exposures

Number of workers	Dust environment ($\mu g/m^3$)
Incinerator workers:	
12	High (400)
14	Medium (200)
4	Low (100)
Sanitary landfill workers:	
2	High (400)
1	Low (100)

Gamma exposures

Number of workers	Proximity to working face (m)
Incinerator workers:	
18	Close (1)
10	Moderate (10)
2	Far (30)
Sanitary landfill workers:	
2	Close (1)
1	Far (30)

The number of disposal site visitors and their levels of exposure are assumed to be identical to those presented in Section 4.3.2.

4.3.4 Urban Sanitary Landfill

This section identifies design parameters and modeling assumptions used to simulate disposal of BRC surrogate waste streams in an urban sanitary landfill (UF).

(A) Site Design and Operations

The UF is designed and operated very similarly to the SF described in Section 4.3.2. The major differences are described below.

The capacity of the UF is 34,700,000 cubic meters. The land requirements for such a facility are approximately 576 hectares (2,400 m x 2,400 m), making this alternative prohibitively expensive in many areas. Assuming the urban sanitary landfill serves a population of 1,000,000 and has a daily trash emplacement rate of 2.6 kg/person/d, the UF will process approximately 2,600 t/d.

Operational procedures and modeling assumptions at the UF are identical to those at the SF, but are simply on a larger scale. Assumptions concerning the depth of cover, depth of waste, density, porosity, and permeability of trench cover and contents, noncontainerization of wastes, percent spillage, operational period, nonuse of the landfill for residential or agricultural purposes after closure, and trench cap failure rates, are the same as those described previously for the SF.

(B) Staffing Requirements and Site Visitors

Staff requirements for the UF are estimated at ten equipment operators, three weighmasters, two laborers, and one foreman. The number of employees and their exposure levels are estimated based on the discussion by Oztunali (Oz84). These are summarized below:

Worker classification	Number	Dust exposure ($\mu g/m^3$)	Gamma source distance (m)
Equipment operator	10	High (400)	High (1)
Weighmaster	3	Low (100)	Low (50)
Laborer	2	Moderate (200)	Moderate (30)
Foreman	1	Moderate (200)	Moderate (30)

The number of visitors to the facility and their levels of exposure are computed similarly to those for the SF, and proportionally to the volume of waste received. The number of visitor-hours per year is projected at 95,000. The level of airborne dust exposure is assumed to be moderate (200 $\mu g/m^3$) and 30 meters to the open face.

4.3.5 Urban Sanitary Landfill with Onsite Incineration

This section describes operations at an urban sanitary landfill with onsite incineration (UI) of the wastes prior to disposal. Conceptual design and operations are similar to the SI described in Section 4.3.3. The differences are noted below.

(A) Site Design and Operations

The land-disposal capacity of the UI is 5,780,000 cubic meters. Land requirements are approximately 96 hectares, including a 980-m x 980-m disposal zone surrounded by a 50-m buffer zone.

Wastes are received at a rate of approximately 2,600 t/d and are temporarily stored onsite until they are incinerated. Six identical incinerators are assumed with 24-hour operation over the entire 20-year operational lifetime.

Two stacks each venting three incinerators are assumed. Stack height is 76 meters, diameter 2.6 meters, and exit velocity 18.9 m/sec. Operating temperatures, volume reduction achieved, and volatility of each radionuclide are identical to those described in Section 4.3.3. After incineration, the ash and rubble are landfilled consistent with the method used at the SF (see Section 4.3.2).

(B) Staffing Requirements and Site Visitors

Staffing requirements are divided into incinerator and landfill personnel. Labor requirements for landfilling are assumed to be identical to those described for the UF in Section 4.3.3. Personnel requirements for incinerator operations are based on estimates for the SF (see Section 4.3.3). However, staffing requirements per tonne of waste incinerated are assumed to decrease by 50 percent as the amount of waste incinerated increases from 454 to 2,600 t/d (Oz84). An estimate of labor requirements by classification includes 3 superintendents, 3 assistant superintendents, 3 office managers, 3 secretaries, 3 additional weighmasters, 6 crane operators, 18 charging floor operators, 12 process controllers, 12 residue handlers, and 24 other laborers. Their levels of airborne dust concentrations and distance to surface radiation sources are summarized below.

Dust concentrations

Number of workers	Dust exposure ($\mu g/m^3$)
Incinerator workers:	
35	High (400)
40	Medium (200)
12	Low (100)

Sanitary landfill workers:

10	High (400)
3	Medium (200)
3	Low (100)

Gamma exposures

Number of workers	Proximity to working face (m)
Incinerator workers:	
52	Close (1)
29	Moderate (10)
6	Far (30)
Sanitary landfill workers:	
10	Close (1)
3	Moderate (10)
3	Far (30)

The number of disposal site visitors and the parameter values for dust and gamma radiation are assumed to be identical to those described in Section 4.3.3.

4.3.6 Rural Municipal Dump

This section provides conceptual design and modeling assumptions for a rural municipal dump (MD) serving a population of 60,000. Criteria for definition of this site are based on the assumption that it is not in complete compliance with EPA regulations 40 CFR 241-257.

(A) Site Design and Operation

The reference MD has an active period of operation of 20 years. Its capacity is 2,100,000 cubic meters of waste and fill material. Land requirements are approximately 35 hectares, including a 590-m x 590-m operational zone and a 50-m buffer zone surrounding the facility.

Wastes are received at the rate of approximately 154 t/d. The operating schedule is 8 hours per day, 5 days per week, 52 weeks per year. Operations consist of the daily receipt of wastes, which are dumped onto the working face of the landfill. A bulldozer spreads and compacts the waste onto the slope, but the cover may not be applied on a daily basis. The final depth of cover is only 0.3 meter. The percentage of wastes uncovered is greater than at the SF and averages 2 percent.

Cap failure rates are greater than at the SF. Forty percent of the cap is assumed to fail the first year after closure, increasing to 60 percent 20 years after closure.

(B) Staffing Requirements and Site Visitors

Labor requirements at the MD are minimal. The number of employees and their level of exposure are based on Oztunali (Oz84). Because of insufficient dust control methods, levels of exposure to airborne dust are somewhat greater than at the SF described previously. Dust concentration and gamma source (working face) distance parameter values are summarized below.

Worker classification	Number	Dust exposure ($\mu g/m^3$)	Proximity to working face (m)
Equipment operator	1	High (500)	High (1)
Weighmaster	1	Low (150)	Low (50)

The number of visitors to the MD during its operational period is estimated similarly to the SF, but proportionally to the size of the population served. The number of annual visitor-hours is estimated to be 5,700. The concentration of airborne dust is higher than at the SF (250 $\mu g/m^3$) because of the lack of dust control measures. The distance from the visitors to the working face is estimated at 30 meters.

4.3.7 Suburban Incineration and Disposal on the Generator's Property

This section describes those modeling assumptions that change in instances in which BRC surrogate wastes are incinerated and placed in a landfill on the generator's property.

(A) Site Design and Operations

The reference suburban onsite incineration and land disposal (LURO) facility is assumed to be located in an area with a population of 175,000. The facility has a period of active operation of 20 years. Its capacity is 170,000 cubic meters. Land requirements for land disposal are approximately 2.8 hectares. In addition, a 50-meter buffer zone surrounds the facility.

Prior to land disposal, a pathological incinerator is used to combust trash, liquid scintillation vials, biological wastes, and non-radioactive wastes. The incineration rate is assumed to be 227 kg/h, 8 hours per day, 260 days per year, but for purposes of modeling, the source term and thus the emission rate are spread out to be continuous (24 h/d) over the 20-year lifetime of the facility.

Operating conditions and pollution control equipment for the pathological incinerator result in lower emission rates for some of the radionuclides emitted during incineration. The volatility of radionuclides other than hydrogen-3, carbon-14, technetium-99, ruthenium-106, and iodine-129 is 0.25 percent. Volatility of the five

radionuclides identified above is the same as for the SI (see Appendix C for FVOLAT numbers).

After incineration, the ash and residue are transported to the landfill and worked into the fill on a daily basis. The thickness of cover is assumed to be the same as the SF. The thickness of waste is 6 meters. The porosity and density of the trench contents are 0.35 and 0.89 g/cm^3, respectively. The rate of spillage is the same as at the SF.

After closure, there is assumed to be no restricted site use. The trench cap is assumed to begin to fail during the first year after closure, increasing continuously until it reaches 30 percent 40 years after closure. Trench cap failure remains at 30 percent for the remainder of the period of analysis.

(B) Staffing Requirements and Site Visitors

Because of the relatively small amount of wastes being processed, staffing requirements are minimal. The assumed level of staffing is two employees, with moderate dust concentration exposure (200 $\mu g/m^3$) and moderate proximity to surface gamma radiation (10 meters distance). Because of the private nature of the operations, no visitors are exposed to airborne or surface radioactive dust.

4.4 Below Regulatory Concern (BRC) Localized Waste Disposal Scenarios Considered

To provide a basis for the estimation of health effects that would result if the surrogate waste streams described in Chapter 3 were disposed of by one of the generic disposal methods described in Section 4.3, 15 representative specific disposal scenarios have been hypothesized. Each of these scenarios is defined in this section in terms of its waste stream inventory, appropriate generic disposal method, and rationale for inclusion in the analysis. Each disposal scenario is assumed to take place in each of the three hydrogeologic/climatic settings of humid permeable, humid impermeable, and arid permeable.

Each scenario consists of a generator or combination of generators sending BRC surrogate waste streams to a disposal site during that site's entire active lifetime. The generators include nuclear power plants and other fuel-cycle facilities, industrial facilities, universities, medical facilities, and consumers. The combination of generators and the disposal site to which they would ship waste is based on actual situations that are currently known to exist and realistic approximations of situations which could be expected to occur if some LLW was classified as BRC.

The overall health impacts as shown in Chapter 10 include (1) cumulative population health effects over a period of 10,000 years, and (2) maximum annual exposures to the critical population group that can be converted to an annual or lifetime risk over the same time period. The analytical methods for determining these impacts are different and are explained in Chapter 8. The computation of health effects resulting from each scenario is meant to be an analytical tool used to identify waste stream characteristics that may make them inappropriate for BRC designation, examine the cumulative health risks from multiple BRC surrogate wastes, and compare health risks among BRC surrogate streams in terms of existing regulations and disposal practices.

Table 3-10 presents a listing of the surrogate BRC waste streams used in our assessment. Tables 4-2 and 4-3 summarize the 15 scenarios and waste volumes. Figure 4-14 shows the generic formula used to determine the various scenarios.

4.4.1 Scenario 1: Three-Unit Pressurized-Water Power Reactor Complex (PWR-MD)

Waste inventory

Wastes	Total volume disposed (m^3/20 yr)
P-CONDRSN	3.47E+2
P-COTRASH	1.25E+4
L-WASTOIL	4.45E+2
TOTAL	1.33E+4

Generic Disposal Method

A rural municipal dump with a surrounding population of 60,000.

Reason for Inclusion

Since many nuclear power reactors are located in rural areas, this is believed to be a realistic case.

4.4.2 Scenario 2: Two-Unit Boiling-Water Power Reactor Complex (BWR-MD)

Waste inventory

Wastes	Total volume disposed (m^3/20 yr)
B-COTRASH	3.24E+4
L-WASTOIL	1.15E+3
TOTAL	3.36E+4

Table 4-2. Waste disposal scenario alternatives and related acronyms in the BRC analysis

Scenario No.	Description	Acronym
1.	3-unit pressurized-water power reactor complex - municipal dump	PWR-MD
2.	2-unit boiling-water power reactor complex - municipal dump	BWR-MD
3.	University and medical center complex - urban sanitary landfill	LUMC-UF
4.	Metro area and fuel cycle facility - suburban sanitary landfill	MAFC-SF
5.	Metro area and fuel cycle facility - suburban sanitary landfill with incineration	MAFC-SI
6.	2-unit power reactor, institutional, and industrial facility - municipal dump	PWRHU-MD
7.	Uranium hexafluoride facility - municipal dump	UHX-MD
8.	Uranium foundry - municipal dump	UF-MD
9.	Large university/medical center; volatilization of 90% H-3 and 75% C-14 - onsite landfill with onsite incineration	LUR03
10.	Large metropolitan area with consumer wastes - suburban sanitary landfill with incineration	LMACW-SI
11.	Large metropolitan area with consumer wastes - urban sanitary landfill with incineration	LMACW-UI
12.*	Consumer product wastes - suburban sanitary landfill	CW-SF
13.*	Consumer product wastes - urban sanitary landfill	CW-UF
14.*	Large university/medical center; 100% volatilization of H-3 and C-14 - onsite landfill with onsite incineration	LUR01
15.*	Large university/medical center; 50% volatilization of H-3 and C-14 - onsite landfill with onsite incineration	LUR02

*Indicates those scenarios where the waste streams are already deregulated.

Table 4-3. As-generated BRC waste volumes (m^3) used in the analysis (20-year totals)

	Disposal scenario														
	1	2	3	4	5	6	7	8	9[d]	10	11	12	13	14	15
P-COTRASH	12500					8333				8333	8333				
B-COTRASH		32413													
I-COTRASH			7146	3573	3573	893			3573	3573	7146				
I-ABSLIQD			282	141	141	35			141	141	282				
I-BIOWAST			190	95	95	24			95	95	190				
I-LIQSCVL			382	191	191	48			191	191	382			4000[a,b]	4000[b,c]
N-LOTRASH				7165	7165	896				3583	7165				
N-LOWASTE				4259	4259	532				2130	4259				
N-SSTRASH								49000							
N-SSWASTE								8647							
F-PROCESS				12189	12189										
U-PROCESS							10693								
F-COTRASH				36794	36794										
F-NCTRASH				6504	6504										
P-CONDRSN	347					231				231	231				
L-WASTOIL	445	1153				297				297	297				
C-TIMEPCS										7	38	7	38		
C-SMOKDET										60	335	60	335		
TOTALS	13292	33566	8000	70911	70911	11289	10693	57647	4000	18640	28558	67	373	4000	4000

[a] Assumes 100 percent volatilization of H-3 and C-14 (no stack recovery).
[b] Represents a waste at BIOMED Rule limits (10 CFR 20.306), i.e., H-3 and C-14 concentrations should each be 4.45 E-02 Ci/m^3.
[c] Assumes 50 percent stack recovery of H-3 and C-14.
[d] Assumes 90 percent volatilization of H-3 and 75 percent volatilization of C-14.

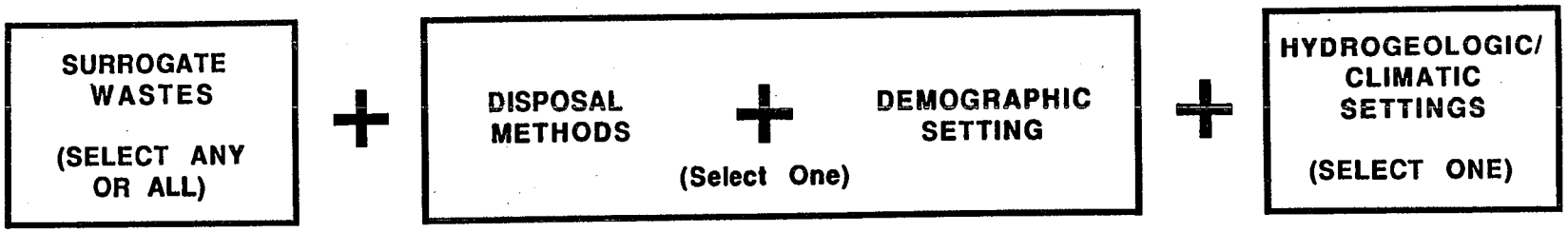

Figure 4-14. Generic Formula for BRC Disposal Scenarios

Generic Disposal Method

A rural municipal dump with a surrounding population of 60,000.

Reason for Inclusion

Since many nuclear power reactors are located in rural areas, this is believed to be a realistic case.

4.4.3 Scenario 3: University and Medical Center Complex (LUMC-UF)

Waste inventory

Wastes	Total volume disposed (m^3/20 yr)
I-COTRASH	7.15E+3
I-BIOWAST	1.90E+2
I-ABSLIQD	2.82E+2
I-LIQSCVL	3.82E+2
TOTAL	8.00E+3

Generic Disposal Method

An urban sanitary landfill with a surrounding population of 1,000,000.

Reasons for Inclusion

This scenario represents two large universities with a medical center, medical school, or a hospital located in an urban setting.

4.4.4 Scenario 4: Metropolitan Area and Fuel-Cycle Facility (MAFC-SF)

Waste inventory

Wastes	Total volume disposed (m^3/20 yr)
F-NCTRASH	6.50E+3
F-COTRASH	3.68E+4
F-PROCESS	1.22E+4
N-LOWASTE	4.26E+3
N-LOTRASH	7.16E+3
I-COTRASH	3.57E+3
I-BIOWAST	9.5 E+1
I-ABSLIQD	1.41E+2
I-LIQSCVL	1.91E+2
TOTAL	7.09E+4

Generic Disposal Method

A suburban sanitary landfill with a surrounding population of 175,000.

Reason for Inclusion

This scenario represents a large university or a number of medical centers or hospitals, several industrial radionuclide generators, and one fuel fabrication facility in a suburban setting. Industrial waste volumes represent one-half of the largest contribution by any single State. Fuel fabrication waste volumes represent an actual facility (for example, the Westinghouse plant in South Carolina).

4.4.5 Scenario 5: Metropolitan Area and Fuel-Cycle Facility with Incineration (MAFC-SI)

Generic Disposal Method

A suburban sanitary landfill with incineration capability and a surrounding population of 175,000. The waste inventory is the same as for Scenario 4.

Reason for Inclusion

This scenario provides a comparison between a landfill facility that uses incineration and a landfill that does not (Scenario 4).

4.4.6 Scenario 6: Two-Unit Power Reactor, Institutional, and Industrial Facilities (PWRHU-MD)

Waste inventory

Wastes	Total volume disposed (m^3/20 yr)
N-LOWASTE	5.32E+2
N-LOTRASH	8.96E+2
P-CONDRSN	2.31E+2
P-COTRASH	8.33E+3
L-WASTOIL	2.97E+2
I-COTRASH	8.93E+2
I-BIOWAST	2.40E+1
I-ABSLIQD	3.50E+1
I-LIQSCVL	4.80E+1
TOTAL	1.13E+4

Generic Disposal Method

A rural municipal dump with a surrounding population of 60,000.

Reason for Inclusion

This scenario represents a two-unit PWR complex, either a medium university or several hospitals, and a small industrial radionuclide generator in a rural setting. PWR volumes are two-thirds of Scenario 1 volumes; other volumes represent one-fourth to one-eighth of the volumes in Scenario 4.

4.4.7 Scenario 7: Uranium Hexafluoride Facility (UHX-MD)

Waste inventory

Wastes	Total volume disposed (m^3/20 yr)
U-PROCESS	1.07E+4
TOTAL	1.07E+4

Generic Disposal Method

A rural municipal dump with a surrounding population of 60,000.

Reason for Inclusion

This scenario represents a single uranium hexafluoride processing facility. Volumes reflect estimates for an actual facility derived from U.S. total volume projections. A rural setting is used (to reflect, for example, the Kerr-McGee facility in Sequoyah County, Oklahoma).

4.4.8 Scenario 8: Uranium Foundry (UF-MD)

Waste inventory

Wastes	Total volume disposed (m^3/20 yr)
N-SSWASTE	8.65E+3
N-SSTRASH	4.90E+4
TOTAL	5.76E+4

Generic Disposal Method

A rural municipal dump with a surrounding population of 60,000.

Reason for Inclusion

In this scenario, the wastes from a uranium foundry are placed for disposal in a rural municipal dump. It has been included to estimate an upper bound of the long-term radiological impacts of BRC disposal of these waste streams.

4.4.9 Scenario 9: Large University and Medical Center with Onsite Incineration and Disposal (LURO 3)

Waste inventory

Wastes	Total volume disposed (m^3/20 yr)
I-COTRASH	3.57E+3
I-BIOWAST	9.50E+1
I-LQSCNVL	1.91E+2
I-ABSLIQD	1.41E+2
TOTAL	4.00E+3

Generic Disposal Method

The incinerator and landfill are located on the generator's property. The university is assumed to be located in a suburban area with a surrounding population of 175,000.

Reason for Inclusion

This scenario represents a large university with a possible medical center and/or hospital with onsite incineration and a dedicated landfill. It is believed that this scenario is currently in practice.

4.4.10 Scenario 10: Large Metropolitan Area with Consumer Wastes (LMACW-SI)

Waste inventory

Wastes	Total volume disposed (m^3/20 yr)
N-LOWASTE	2.13E+3
N-LOTRASH	3.58E+3
P-CONDRSN	2.31E+2
P-COTRASH	8.33E+3
L-WASTOIL	2.97E+2
I-COTRASH	3.57E+3

I-BIOWAST	9.50E+1
I-LQSCNVL	1.91E+2
I-ABSLIQD	1.41E+2
C-SMOKDET	6.00E+1
C-TIMEPCS	7.00E+0
TOTAL	1.86E+4

Generic Disposal Method

A suburban sanitary landfill with incineration capability and a surrounding population of 175,000.

Reason for Inclusion

This scenario represents a realistic metropolitan area with consumer wastes containing deregulated radioactive materials, a two-unit PWR complex, and one large university or medical center or hospitals, and one or two radionuclide generators. The purpose of the scenario is to combine specific source terms that are co-generated in a large metropolitan area. The area is served by a large suburban sanitary landfill equipped with an incinerator.

4.4.11 Scenario 11: Large Metropolitan Area with Consumer Wastes (LMACW-UI)

Waste inventory

Wastes	Total volume disposed (m^3/20 yr)
N-LOWASTE	4.30E+3
N-LOTRASH	7.17E+3
P-CONDRSN	2.31E+2
P-COTRASH	8.33E+3
L-WASTOIL	2.97E+2
I-COTRASH	7.15E+3
I-BIOWAST	1.90E+2
I-LQSCNVL	3.82E+2
I-ABSLIQD	2.82E+2
C-SMOKDET	3.35E+2
C-TIMEPCS	3.80E+1
TOTAL	2.86E+4

Generic Disposal Method

An urban sanitary landfill with incineration capability and a surrounding population of 1,000,000.

Reason for Inclusion

This scenario is applied to an urban landfill to provide a direct comparison of resulting effects associated with suburban and urban settings. It represents a realistic, large urban metropolitan area with waste generation from a two-unit PWR complex, several industrial radionuclide producers, and a combination of several institutional facilities (hospitals/medical centers/universities), along with a large quantity of consumer wastes containing deregulated radioactive materials.

4.4.12 Scenario 12: Consumer Product Wastes (CW-SF)

Waste inventory

Wastes	Total volume disposed (m^3/20 yr)
C-SMOKDET	6.0E+1
C-TIMEPCS	7.0E+0
TOTAL	6.7E+1

Generic Disposal Method

A suburban sanitary landfill with a surrounding population of 175,000.

Reason for Inclusion

In this scenario, disposal of the wastes from two common consumer products containing deregulated radioactive materials is considered. The two products are smoke detectors (containing Am-241) and luminous-dial time pieces (containing H-3). This scenario has been defined to assess the impact of these well-known consumer products and provide a comparison with other BRC surrogate waste streams.

4.4.13 Scenario 13: Consumer Product Wastes (CW-UF)

Waste inventory

Wastes	Total volume disposed (m^3/20 yr)
C-SMOKDET	3.35E+2
C-TIMEPCS	3.80E+1
TOTAL	3.73E+2

Generic Disposal Method

An urban sanitary landfill with a surrounding population of 1,000,000.

Reason for Inclusion

In this scenario, disposal of the wastes from two common consumer products in a UF is considered. The two products are smoke detectors and luminous-dial time pieces containing deregulated radioactive materials. This scenario has been defined to assess the impact of these well-known consumer products and provide a comparison with other BRC surrogate waste streams, as well as provide a direct comparison of resulting effects associated with suburban and urban landfill settings. The urban area is assumed to generate 5.5 times as much waste as a suburban area.

4.4.14 Scenario 14: Large University and Medical Center with Onsite Incineration and Disposal (LURO-1)

Waste inventory

Wastes	Total volume disposed (m^3/20 yr)
Institutional (I-LQSCNVL and I-BIOWAST)	4.03+3
TOTAL	4.0E+3

Generic Disposal Method

An institution with incineration capability and an onsite sanitary landfill. The institution is located in a suburban setting with a surrounding population of 175,000.

Reason for Inclusion

In this scenario, disposal of institutional wastes (liquid scintillation vials and biomedical wastes) through incineration and onsite disposal in a suburban setting is considered.

This scenario has been included for comparison to NRC's biomedical waste disposal rule (10 CFR 20.306) where certain radionuclide wastes are deregulated based on concentrations. The concentrations of H-3 and C-14 in I-LQSCNVL and I-BIOWAST in this specific scenario are equivalent to the maximum allowable for disposal without regard to radioactivity as defined in the rule cited above. The concentrations of H-3 and C-14 are each set at 4.30E-02 Ci/m^3.

This scenario assumes a 100 percent volatilization of the C-14 and H-3 in the wastes during incineration. This also provides a direct comparison with the other BRC surrogate waste streams.

4.4.15 Scenario 15: Large University and Medical Center with Onsite Incineration and Disposal (LURO-2)

The waste inventory and generic disposal method is the same as for Scenario 14.

The reason for inclusion is the same as for Scenario 14 except that this scenario assumes a 50 percent volatilization of the C-14 and H-3 in the wastes during incineration. This also provides a direct comparison with the other BRC surrogate waste streams and Scenario 14.

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Chapter 5: HYDROGEOLOGIC/CLIMATIC SETTINGS

5.1 Introduction

One of the more significant pathways for transporting the released radionuclides from a LLW disposal site to the biosphere is the hydrologic pathway. The radionuclides can be released by leaching, by breaching of the disposal site, by slow degradation of the disposal facility, by trench flooding, or by waste spillage during disposal. The wastes can contaminate surface water, ground water, atmospheric water, or any combination of these parts of the hydrosphere.

A large number of hydrogeologic and climatic parameters are necessary to characterize the release scenarios. Climatic factors are most important in determining possible land use, such as through reuse of the land for farming or pasture. Concentration of radionuclides in the soils and plants of these areas could bring a wider portion of the population into contact with the released radionuclides. For these release pathways, the near surface, microclimatological parameters are the most useful, yet are the ones least understood and most difficult to quantify.

The hydrologic parameters that are least understood are those concerned with the movement of fluid through the unsaturated zone. Unsaturated zone flow mechanisms are not only poorly understood, but quantifying the parameters which control this flow is a controversial subject on which there is little agreement. Hydrologic processes and pathways in saturated formations are better understood, and quantifying these parameters is an established practice.

It is expected that disposal facilities will be established in a wide variety of hydrogeologic/climatic settings. For the purpose of this risk assessment, which was designed to supply databases for determining generally applicable environmental standards for the United States, three representative sites were selected to cover all potential sites possibly selected by the State compacts. These three sites have their own distinct hydrogeological/climatic conditions and were designated as (1) humid permeable, (2) humid impermeable, and (3) arid permeable.

Theoretically, the characteristics of each representative site should be characterized by a statistical analysis of each parameter randomly selected from actual or potential sites that have similar characteristics as the representative site. This approach would involve characterizing the entire 50 States and would require collecting data presently not available. Therefore, hydrogeological/climatic conditions at the Barnwell, Beatty, and West Valley sites were used as the settings of the representative sites. The hydrogeologic and climatic settings at these sites have been well characterized by the U.S. Geological Survey (USGS) and are believed to bracket the range of actual settings that will be present at LLW disposal sites in the United States. However, the settings

used in the modeling effort are meant to be generic rather than site specific. To maintain a generic approach, data thought to be more accurately representative of a geographic region have been used rather than specific site data.

Given these considerations, the objectives are to:

- Describe hydrogeologic/climatic settings of conditions for three regional commercial disposal facilities, as characterized by the Barnwell, Beatty, and West Valley sites.
- Use information from the three commercial sites to develop generic but realistic hydrogeologic/climatic settings for disposal options in both near-surface and deep scenarios.
- Present data sufficient to complete the hydrogeologic/climatic input data matrix necessary to run the PRESTO-EPA computer model (see Chapter 8) for the various disposal options.
- Specify hydrogeologic pathways for each of the disposal options appropriate to each site.

The descriptions of general geology, hydrogeology, surface water hydrology, climatic settings, and potential hydrogeologic pathways for the Barnwell, Beatty, and West Valley disposal facilities are presented in a general, qualitative manner in Appendix D. Based on this information, generic site descriptions and data requirements are then developed.

5.2 Generic Site Descriptions and Data Requirements

Three generic site hydrogeologic/climatic settings have been chosen for analysis: humid climate with permeable soil; arid climate with permeable soil; and humid climate with impermeable soil. It is important to note that, as a consequence of this generic site approach, the results of these analyses do not represent any single site or facility and should not be construed as such. Soil characteristics, aquifer depth, and other parameters have been chosen to represent a range of potential disposal sites. Precipitation and temperature data were taken from actual records of disposal sites. The air turbulence stability class formulations were defined and set the same for all sites. Some site-specific data were used for the model directly, while other parameter values reflected average regional conditions.

5.2.1 General Site Characterization

As stated earlier, the purpose of the risk assessments is intended to supply databases for determining the generally applicable environmental standard for the disposal of LLW to be generated in the United States. Three generic sites were selected to represent all potential sites possibly selected by the State compacts.

All three sites are assumed to have some common features. The depth of the trench, the cover thickness, etc., differ according to disposal option. A buffer zone is assumed to exist around the trench. Beyond this area, there may be any number of communities of various sizes and distances from the trench. Runoff from the disposal facility is collected in a channel which surrounds the disposal area. This runoff is carried to the nearest stream or river and enters the surface water system at a point downstream of the local population. The local population is situated between the disposal site and the stream. A well is located between the disposal trench and the stream in such a way that a ground-water gradient line would pass through the trench and the well before reaching the stream. Thus, if contaminated water percolates from the trench to the aquifer, it will move through the aquifer to the well and ultimately be discharged to the stream.

This last assumption, that the aquifer discharges to the surface stream, facilitates health risk analyses for regional basin populations. Transport time of the radionuclides through the aquifer to the discharge point is computed, based on the distance, ground-water velocity, and retardation factors. The resulting residual radioactivity will become a water-pathway exposure to populations residing in the regional segment of the river basin.

The parameter data presented here and used in the analysis were obtained from several sources. These include unpublished reports by the USGS and information obtained from the NRC and the disposal site operators, which were incorporated into several reports (EPA79, EPA83). Atmospheric data were obtained from several sources other than the site operators (AEC68, EPA79, NOAA80).

Site-specific data were collected describing temperature, precipitation, hours of sunlight, and hydraulic characteristics of the soil. The temperature and precipitation data used for the humid permeable site were originally generated by the USGS meteorological station at the Barnwell LLW disposal site in South Carolina in 1982. Data for the humid impermeable site came from USGS data for Colden, New York, collected in 1978. Precipitation and temperature data for the arid permeable site were supplied by the USGS from data collected at the Beatty, Nevada, LLW disposal site. A complete listing of input data for all three sites, as well as definitions of the parameters and their values, is contained in Appendix C.

5.2.2 Humid Permeable Site

It is assumed that ground water in the vicinity of the humid permeable site is the only source of water for human and animal consumption. Half of the water for the animals is assumed to be potentially contaminated ground water. The remainder of the water for the animals is assumed to come from an uncontaminated source such as a farm pond, a stream, or a well outside the range of the contaminated ground-water plume. A schematic illustration is shown in Figure 5-1.

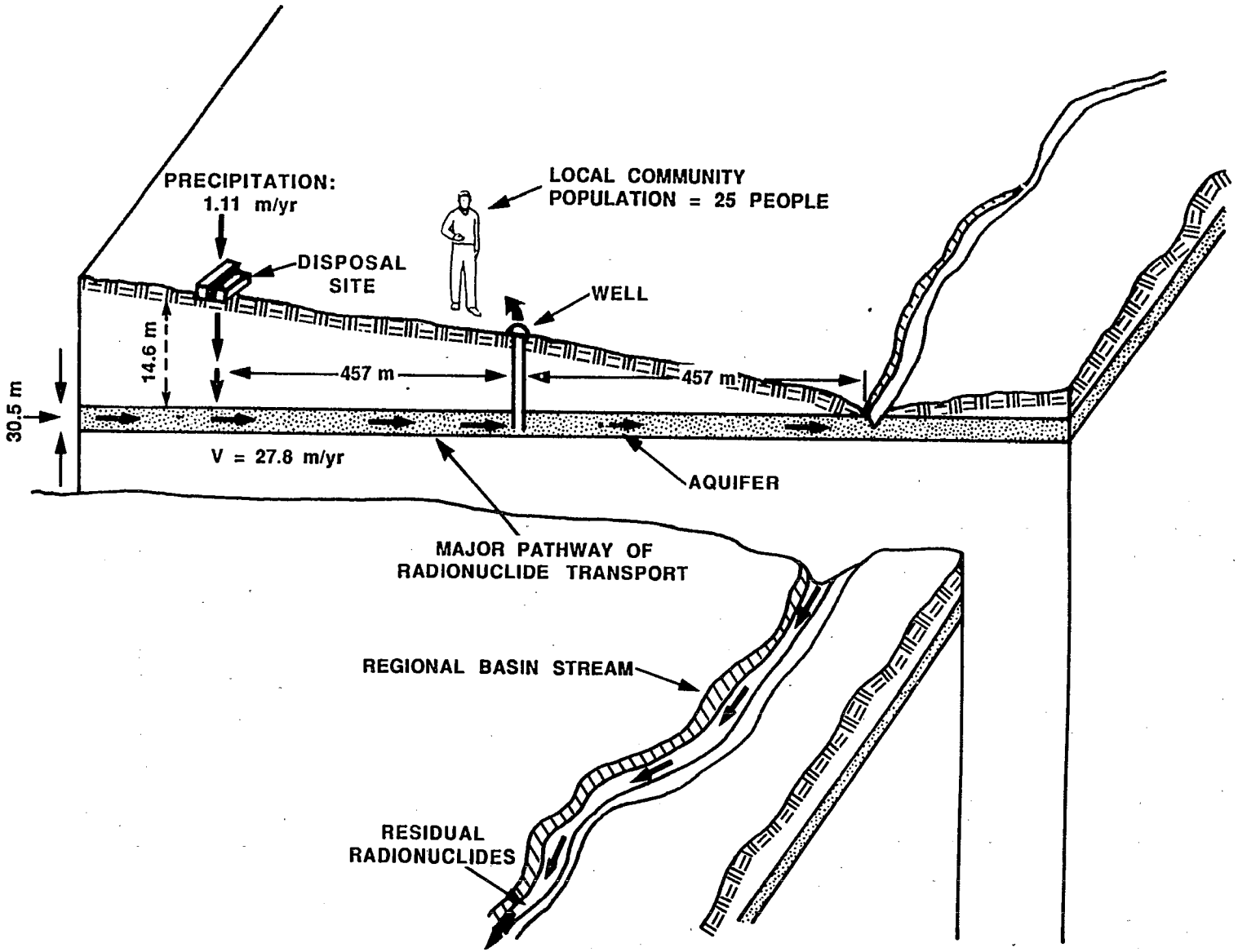

Figure 5-1. Schematic of Geographic and Demographic Assumptions Used in Modeling the Humid Permeable Site.

The aquifer is 14.6 meters below the trench bottom for the shallow-land disposal options. The soil below the trench has a porosity of 0.35 and a permeability of 2.2 m/yr. The aquifer thickness is 30.5 meters with a dispersion angle of 0.3 radian and a porosity of 0.39. Water moves through the aquifer with a velocity equal to 27.8 m/yr. The horizontal travel distance between the trench and the well is 457 meters. The distance between the well and the stream is also 457 meters. These two distances are used to compute travel times through the aquifer and may not represent actual straight-line distances.

The atmospheric parameters for the humid permeable site include annual wind speed, duration, and direction data. Also, each population center is defined by direction, distance, and size of population. The atmospheric source height is equal to 1.0 meter. The gravitational fall velocity is 0.01 m/sec. Mean wind speed equals 2.01 m/sec, the deposition velocity is 0.1 m/sec, and the source-to-receptor distance (trench to population) is 480 meters. The atmospheric lid height is 300 meters and the Hosker roughness factor is 0.01 meter (a roughness condition of the ground surface).

Surface soil characteristics for the humid permeable site are defined by a porosity of 0.39 and a bulk density of 1.6 g/cm^3. The local stream has a flow rate of 3.57E+05 m^3/yr. The down-slope distance from the trench to the stream is 460 meters. The cross-slope extent of spillage is 0.2 meter for the unit volume disposal scenario. These last two parameters define the maximum area which could be contaminated by overflow of trench water. The depth of soil that would be affected by soluble contaminants is 0.1 meter. The fraction of precipitation that runs off the site is 0.29.

5.2.3 Arid Permeable Site

It is assumed that the ground water in the vicinity of the arid permeable site is used exclusively for human and animal consumption and irrigation. The depth of the aquifer is 80 meters below the trench bottom for the shallow-land disposal options. The soil below the trench has a porosity of 0.40 and a permeability of 63.4 m/yr. The aquifer thickness is 37 meters with a dispersion angle of 0.3 radian and a porosity of 0.40. Water moves through the aquifer with a velocity equal to 90 m/yr. The horizontal travel distance between the trench and the well is 29,000 meters. The distance between the well and the stream is 30,000 meters. These two distances are used to compute travel times through the aquifer and may not represent actual straight-line distances. A schematic drawing is shown in Figure 5-2.

The atmospheric source height is equal to 1 meter. The gravitational fall velocity is 0.027 m/sec. Mean wind speed equals 4.8 m/sec, the deposition velocity is 0.027 m/sec, and the source-to-receptor distance (trench to population) is 29,000 meters. The atmospheric lid height is 300 meters and the Hosker roughness factor is 0.01 meter.

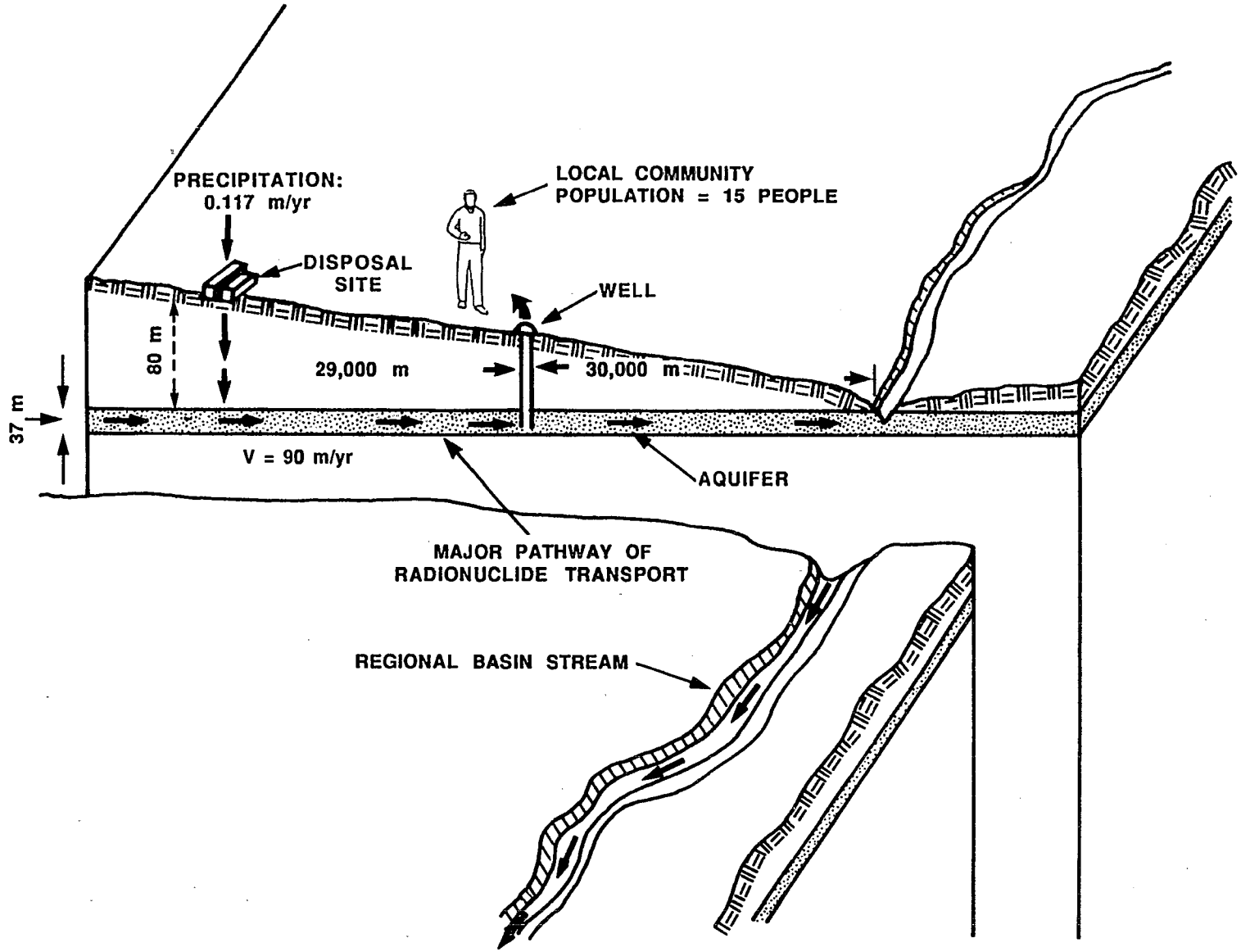

Figure 5-2. Schematic of Geographic and Demographic Assumptions Used in Modeling the Arid Permeable Site.

Surface soil characteristics for the arid permeable site are defined by a porosity of 0.3 and a bulk density of 1.55 g/cm^3. The local stream has a flow rate of 1,000 m^3/yr. The down-slope distance from the trench to the stream is 4,000 meters. The cross-slope extent of spillage is 0.2 meter for the unit volume disposal scenario. These last two parameters define the maximum area that could be contaminated by overflow of trench water. The depth of soil that would be affected by soluble contaminants is 0.1 meter. The fraction of precipitation that runs off the site is 0.005.

5.2.4 Humid Impermeable Site

It is assumed that the surface water in the vicinity of the humid impermeable site is the source for human and animal consumption and irrigation. One-tenth of the water for the animals and irrigation is assumed to be contaminated surface water. The remainder of the water for the animals and irrigation is assumed to come from an uncontaminated source such as a farm pond or a stream or well outside the range of a contaminated ground-water plume and upstream of the contaminated surface water. The aquifer is 21 meters below the trench bottom for the shallow-land disposal options. The soil below the trench has a porosity of 0.32 and a permeability of 0.019 m/yr. The aquifer thickness is 11 meters with a dispersion angle of 0.1 radian and a porosity of 0.25. Water moves through the aquifer with a velocity equal to 0.03 m/yr. The horizontal travel distance between the trench and the well is 250 meters. The distance between the well and the stream is also 250 meters. These two distances are used to compute travel times through the aquifer and may not represent actual straight-line distances. A schematic drawing is shown in Figure 5-3.

The atmospheric source height is equal to 1.0 meter. The gravitational fall velocity is 0.01 m/sec. Mean wind speed equals 5.0 m/sec. The deposition velocity is 0.01 m/sec, and the source-to-receptor distance (trench to population) is 7,240 meters. The atmospheric lid height is 300 meters and the Hosker roughness factor is 0.01 meter.

Surface soil characteristics for the humid impermeable site are defined by a porosity of 0.3 and a bulk density of 1.49 g/cm^3. The local stream has a flow rate of 3.65E+08 m^3/yr. The down-slope distance from the trench to the stream is 100 meters. The cross-slope extent of spillage is 0.2 meter for the unit volume disposal scenario. These last two parameters define the maximum area that could be contaminated by overflow of trench water. The depth of soil that would be affected by soluble contamination is 0.1 meter. The fraction of precipitation that runs off the site is 0.56.

Radionuclide exposure pathways include the consumption of contaminated food. This may occur through the use of contaminated irrigation water on crops or through air pathway deposition of

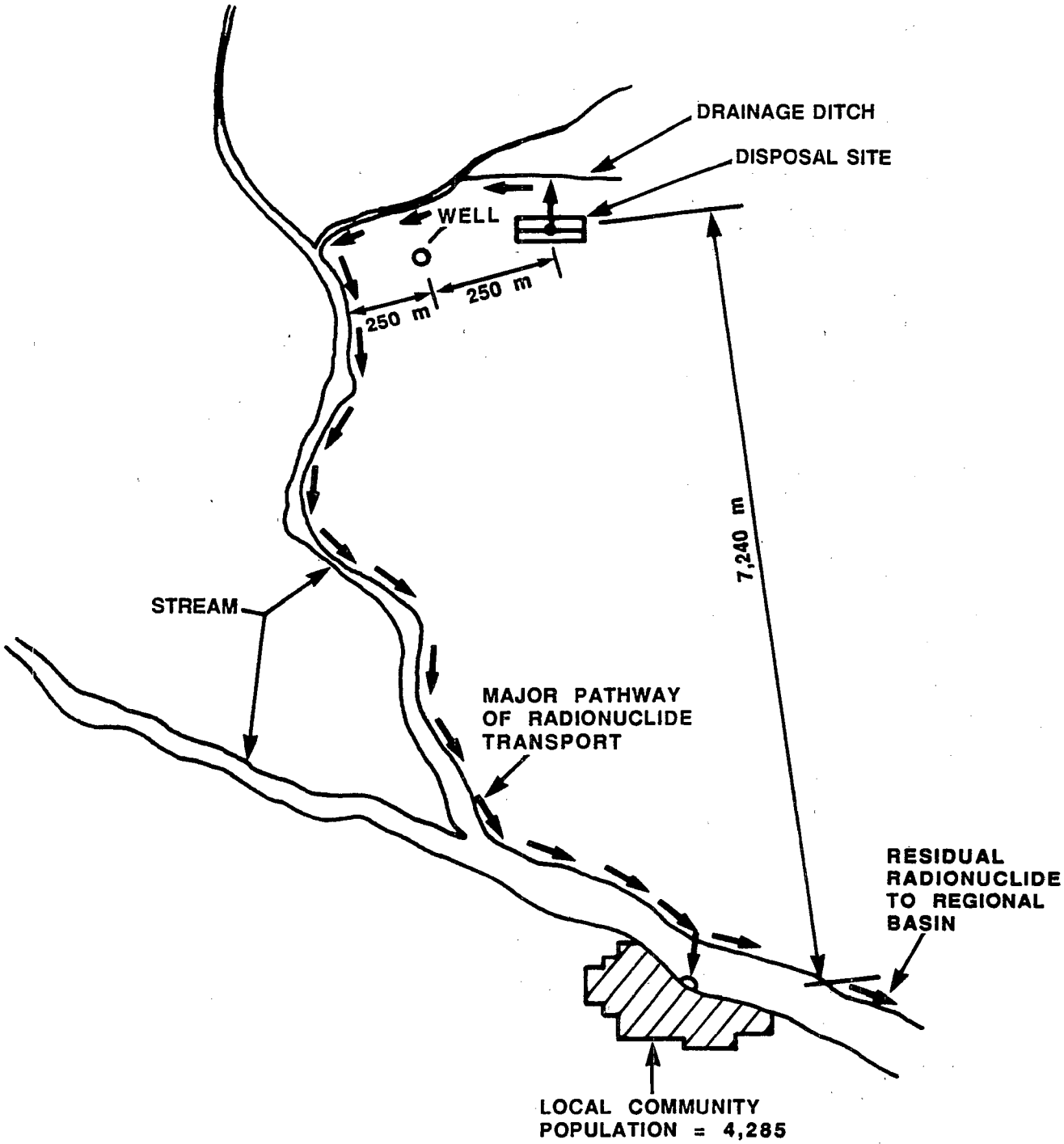

Figure 5-3. Schematic of Geographic and Demographic Assumptions Used in Modeling the Humid Impermeable Site.

contaminants on the crops. Animals may, in turn, be raised on feed that has been contaminated or may be given contaminated drinking water. The resulting exposure to humans is dependent on the consumption of these contaminated foods. In most communities, only a fraction of the food consumed by the population is grown locally. The fraction is higher for rural communities than for urban areas. Food uptake parameter values for all three sites are presented in Appendix C.

5.3 BRC Waste Disposal Settings

The concept of the generic hydrogeologic/climatic site is also applied in this analysis for surrogate BRC waste streams. In order to provide a basis for comparison of the results of the two studies, the site characteristics of the BRC waste disposal sites were kept as similar to the LLW disposal sites as possible. Naturally, the siting requirements for a municipal dump or a sanitary landfill are far less stringent than those for a LLW disposal site. This influences the choice of values for parameters such as aquifer depth and trench cover thickness. The input data selected for each of the three generic sites are described in this chapter.

The three generic disposal sites are representative of three broad sections of the United States: the humid impermeable, the humid permeable, and the arid permeable. The database used to create these generic sites for the BRC analysis was the same data used for the LLW analysis. In fact, large portions of the LLW data sets were incorporated intact.

For each site, a variety of populations were modeled. The population size influences parameters such as the distance from the disposal trench to the population, the consumption of locally grown foods, and the distance from the trench to the well. Each of those items will be discussed in this chapter. The three sites all have certain general characteristics. The disposal facility is assumed to be centrally located with respect to the population. This enables the air pathway exposures to be analyzed 360 degrees around the site. Placement of the disposal facility at the outer edge of the population area would have limited the air pathway exposures to only 180 degrees around the site. It would not have been possible to use actual wind speed/direction data in this case without violating the generic site concept. The effect of the central placement on the results for the air pathway analysis assumed that the health impacts would tend to be slightly higher than with an outer edge placement. The other exposure pathways are not affected by this assumption.

The analysis includes the effects of consuming water from a well and/or a stream in the vicinity of the disposal facility. The ground water in the aquifer that passes under the disposal site feeds the local well, and a portion of it discharges into the downstream basin. Thus, any contamination originating at the disposal site due to

exfiltration of trench water will eventually reach the well and the stream. The time of travel for radionuclides leached from the trench to the point of discharge is dependent on a number of variables. The characteristics of the soil beneath the trench that limit vertical migration are cited for each specific area. Also, aquifer characteristics and individual radionuclide sorption coefficients are specific for the different regional areas.

5.3.1 Site-Specific Data

Site-specific data include climatic, meteorologic, hydrologic, and geologic parameters. Demographics are also included in the disposal-option data since multiple population sizes are being analyzed for each site. The population size influences the air pathway dispersion factors, the distances to the well and the stream, water usage patterns, and consumption rates for locally grown foods.

(A) Humid Permeable Site

For the BRC humid permeable site, the aquifer is 9.7 meters below the ground surface. The other hydrogeologic and climatic parameters are the same as those noted in Section 5.2.2.

Certain hydrologic parameters are utilized to determine the fraction of precipitation that runs off the site, the erodability of the site, and the quantity of sediment that is carried to the stream from the site. Other parameters that determine the erodability of the soil include: the rainfall factor, 250; the soil erodability factor, 0.23 ton/acre-R (where R is the rainfall factor); the slope steepness-length factor, 0.27; the crop management factor, 0.3; the erosion control factor, 0.3; and a sediment delivery factor, 1.0. These terms are further explained in the PRESTO documentation reports (EPA87a,b,c,d).

The surface soil characteristics at the BRC humid permeable site are the same as in Section 5.2.2 except for: the cross-slope extent of spillage, which is 0.45 meter for the unit volume disposal scenario, and the distance from the trench to the nearest drainage ditch or channel is 50 meters.

The agricultural productivity of the humid permeable site is summarized in Appendix C.

(B) Humid Impermeable Site

For the BRC humid impermeable site, the aquifer is 12.9 meters below the ground surface. The other hydrogeologic and climatic parameters are the same as noted in Section 5.2.4.

For the BRC humid impermeable site, the hydrologic parameters that determine the erodability of the site include: the rainfall factor, 100; the soil erodability factor, 0.19 ton/acre-R (where R is the rainfall factor); the slope steepness-length factor, 0.54; the crop management factor, 0.1; the erosion control factor, 1.0; and the sediment delivery factor, 1.0 (EPA87a,b,c,d).

The surface soil characteristics at the BRC humid impermeable site are the same as in Section 5.2.4 except that the cross-slope extent of spillage is 0.45 meter for the unit volume disposal scenario, and the distance from the trench to the nearest drainage ditch or channel is 50 meters.

The agricultural productivity of the humid impermeable site is summarized in Appendix C.

(C) Arid Permeable Site

For the BRC arid permeable site, the aquifer is 43 meters below the ground surface. The other hydrogeologic and climatic parameters are the same as noted in Section 5.2.3.

For the BRC arid permeable site, the hydrologic parameters that determine the erodability of the site include: the rainfall factor, 20; the soil erodability factor, 0.5 ton/a-R (where R is the rainfall factor); the slope steepness-length factor, 0.26; the crop management factor, 0.3; the erosion control factor, 0.4; and the sediment delivery factor, 1.0.

The surface soil characteristics at the BRC arid permeable site are the same as in Section 5.2.3, except that the cross-slope extent of spillage is 0.45 meter for the unit volume disposal scenario and the distance from the trench to the nearest drainage ditch or channel is 50 meters.

The agricultural productivity of the arid permeable site is summarized in Appendix C.

5.3.2 Data Related to Demographics

Unlike the disposal scenarios analyzed for LLW disposal, the BRC disposal facilities are located in or very near population centers. The effect of population density and consumption patterns on the computed health effects is analyzed by modeling three populations and appropriate disposal options. The population at a site influences the total human consumption of locally grown contaminated foods, water usage patterns, the total potential exposure to airborne contamination, and gamma exposures.

For the atmospheric pathway of the population health effects assessment, it was necessary to determine the distance from the disposal trench to the center of population. To do this, 3 representative populations were chosen: 60,000 for the rural area, 175,000 for the suburban area, 1,000,000 for the urban area, and 175,000 for the scenario involving onsite pathological incineration of institutional wastes followed by onsite landfill disposal. The next step was to collect population density data for communities of these sizes in the United States (DOC77). After calculating an average population density for each community, a circular area was determined. Then the area occupied by one-half of that population was found and a new, smaller concentric circular area was assumed. The radius of this smaller area was designated as XG, the distance from the disposal trench to the population, since half of the population would be closer to and half farther away than XG (see Appendix C). Table 5-1 shows the population and area data. For the hydrologic pathway, the distances between the trench and the well, and the well and the stream were chosen as 1,610 meters and 3,220 meters, respectively (see Appendix C).

In addition, in large metropolitan or urban areas, the assumption is made that no large-scale farming occurs in the region of potential contamination. Small individual gardens may be present, but they do not produce food for the general public. Therefore, the human consumption rates are all zero. In the institutional setting, the suburban area population of 175,000 is assumed to obtain 10 percent of its food requirements locally. Likewise, in the other scenarios, the suburban population obtains 10 percent of its food requirements locally, and 22 percent of the food consumed by the rural population is locally grown.

Water usage patterns for the various populations were developed based on the data for the LLW analyses. Basically, it was assumed that larger populations are likely to have more than one source of drinking water. Therefore, if two well fields are needed to supply drinking water to a community, it is reasonable to assume that only one of the two sources would be in the path of the contamination plume from the disposal site. Table 5-2 lists the water usage patterns for each site and population.

5.3.3 Airborne Transport

Data input requirements related to air pathways are affected by site-specific, option-specific, and demographic considerations.

Emission and transport of radionuclides result from various site operations, including natural and mechanical disturbance of the surface of the landfill and incineration of BRC wastes. These emissions potentially result in exposures to onsite workers and visitors, as well as the population of the local community. Important air pathway parameters and values used for these parameters for each of the eight disposal alternatives and at each of the three generic sites, are presented in Appendix C (EPA87a,b,c,d).

Table 5-1. Summary of demographic influences on BRC modeling parameters

Population	Density (No.km^2)	Total area[1] (km^2)	Radius[1] (km)	Area[2] (km^2)	Radius[2] (XG)* (km)
60K	249	1619	22.7	810	16
175K	600	1953	24.3	976	17.7
1000K	2189	3064	31.2	1532	22.1

[1]Refers to larger circular area.
[2]Refers to smaller circular area.
*XG parameter listed in Appendix C.

Table 5-2. Fraction of water consumption from contaminated sources

Population	Well water			Surface water		
	Irrigation	Animals	Human	Irrigation	Animals	Human
Humid Permeable						
60,000	0.0	0.5	1.0	0.0	0.0	0.0
175,000	0.0	0.5	1.0	0.0	0.0	0.0
1,000,000	0.0	0.0	1.0	0.0	0.0	0.0
Humid Impermeable						
60,000	0.0	0.0	0.0	0.1	0.1	1.0
175,000	0.0	0.0	0.0	0.1	0.1	1.0
1,000,000	0.0	0.0	0.0	0.0	0.0	1.0
Arid Permeable						
60,000	1.0	1.0	1.0	0.0	0.0	0.0
175,000	0.7	0.7	1.0	0.0	0.0	0.0
1,000,000	0.0	0.0	1.0	0.0	0.0	0.0

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DOC77 U.S. Department of Commerce, County and City Data Book, 1977, Bureau of the Census, Washington, D.C., 1977.

EPA79 U.S. Environmental Protection Agency, AIRDOS-EPA: A Computerized Methodology for Estimating Environmental Concentrations and Dose to Man from Airborne Releases of Radionuclides, EPA 520/1-79-009 (Contract report ORNL-5532-June 1979), Office of Radiation Programs, Washington, D.C., December 1979.

EPA87a U.S. Environmental Protection Agency, in press, PRESTO-EPA-POP: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code, Volume I, Methodology Manual, RAE-8706-1, Rogers and Associates Engineering Corporation, Salt Lake City, Utah, 1987.

EPA87b U.S. Environmental Protection Agency, in press, PRESTO-EPA-POP: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code, Volume II, User's Manual, RAE-8706-2, Rogers and Associates Engineering Corporation, Salt Lake City, Utah, 1987.

EPA87c U.S. Environmental Protection Agency, in press, PRESTO-EPA-CPG: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code, Documentation and User's Manual, RAE-8706-4, Rogers and Associates Engineering Corporation, Salt Lake City, Utah, 1987.

EPA87d U.S. Environmental Protection Agency, in press, PRESTO-EPA-BRC: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code, Documentation and User's Manual, RAE-8706-5, Rogers and Associates Engineering Corporation, Salt Lake City, Utah, 1987.

NOAA80 U.S. National Oceanic and Atmospheric Administration, Local Climatological Data, Annual Services for 1979, Environmental Data and Information Services, National Climatic Center, Asheville, N.C.

Chapter 6: RADIATION DOSIMETRY

6.1 Introduction

The setting of LLW standards requires an assessment of the doses received by individuals who are exposed by coming into contact with radiation from LLW sites. Two forms of potential radiation exposures can occur from these sites--internal and external. Internal exposures can result from the inhalation of contaminated air or the ingestion of contaminated food or water. External exposures can occur when individuals are immersed in contaminated air or water or are standing on contaminated ground surfaces. Internal or external doses can result from either direct contact with the radiation from radionuclides at the site area or from radionuclides that have been transported from these sites to other locations in the environment. The quantification of the doses received by individuals from these radiation exposures is called radiation dosimetry. This chapter highlights the internal and external dosimetric models used by EPA to assess the dose to individuals exposed to LLW products.

The models for internal dosimetry consider the quantity of radionuclides entering the body, the factors affecting their movement or transport through the body, and the energy deposited in organs and tissues from the radiation that is emitted during spontaneous decay processes. The models for external dosimetry consider only the photon doses to organs of individuals who are immersed in air or are exposed to a contaminated ground surface. In addition, the uncertainties associated with each model will be discussed.

6.2 Basic Concepts

Radioactive materials produce radiation as their constituent radioactive nuclides undergo spontaneous radioactive decay. The forms of emitted energy are characteristic of the decay process and include energetic particles such as alpha and beta particles, and photons, such as gamma and x rays. Alpha particles are nuclei of helium atoms and carry a positive charge two times that of an electron. These particles can produce dense ionization tracks in the biological material in which they traverse. Beta particles are positively or negatively charged electrons. Their penetration power in material is greater than alpha particles. Gamma and x rays are electromagnetic radiation and are distinguishable from alpha and beta particles by their greater penetrating power in material.

The purpose of this section is to introduce the reader to some of the terms or concepts used in Chapters 6 and 7 to describe internal and external dosimetry. For a more detailed explanation, the reader is referred to reports published in this area by the International Commission on Radiation Units and Measurements (ICRU80), International Commission on Radiological Protection (ICRP84), and National Council on Radiation Protection and Measurements (NRCP71).

6.2.1 Activity

The activity of a sample of any radionuclide of species, i, is the rate at which the unstable nuclei spontaneously decay. If N is the number of unstable nuclei present at a certain time, t, its activity, $A_i(t)$, is given by

$$A_i(t) = -dN/dt = -\lambda_i^R N \ , \qquad (1)$$

where λ_i^R, is the radioactive decay constant. The customary unit of activity is the curie (Ci); its submultiples, the millicurie (mCi), the microcurie (μCi), and the picocurie (pCi), can also be used. The curie, which is defined as 3.7×10^{10} disintegrations per second, is the approximate activity of 1 gm of radium.

The time variation of the activity can be expressed in the form:

$$A_i(t) = A_{oi} \exp(-\lambda_i^R t). \qquad (2)$$

A_{oi} is the activity of the nuclide at time t=0. For a sample of radioactive material containing more than one radionuclide, the total activity is determined by summing the activities for each radionuclide:

$$A(t) = \sum_i A_{oi} \exp(-\lambda_i^R t). \qquad (3)$$

6.2.2 Radioactive Half-Life

From the above equations, it is apparent that the activity exponentially decays with time. The time when the activity of a sample of radioactive material containing species i, becomes one-half its original value (i.e., any time t such that $A_i(t) = A_{oi}/2$) is called its radioactive half-life, T_i^R, and is defined as:

$$T_i^R = (\ln 2)/\lambda_i^R. \qquad (4)$$

The unit for the radioactive half-life is any suitable unit of time such as seconds, days, or years. The specific activity of a radionuclide (the activity per unit mass) is inversely proportional to the half-life and can vary over many orders of magnitude.

6.2.3 Radionuclide Chains

Radionuclides decay either to stable atoms or to other radioactive species called daughters. For some species, a decay chain of daughter products may be produced until stable atoms are formed. For example, strontium-90 decays by emitting a beta-particle, producing the daughter yttrium-90, which also decays by beta emission to form the stable atom zirconium-90:

$$^{90}Sr(28.6\ yr) \xrightarrow{\beta^-} {}^{90}Y(64.0h) \xrightarrow{\beta^-} {}^{90}Zr(stable) \qquad (5)$$

The radioactive half-lives for strontium-90 and yttrium-90 are 28.6 yr and 64.0 h, respectively.

6.2.4 Internal and External Exposures to Radionuclides

The term exposure, in the context of this report, denotes physical interaction of the radiation emitted from the radioactive material with cells and tissues of the human body. An exposure can be "acute" or "chronic" depending on how long an individual or organ is exposed to the radiation. Internal exposures occur when radionuclides, which have entered the body through the inhalation or ingestion pathway, deposit energy to organ tissues from the emitted gamma, beta, and alpha radiation. External exposures occur when radiation enters the body directly from sources located outside the body, such as radiation from material on ground surfaces, dissolved in water, or dispersed in the air. In general, for sources of concern in this report, external exposures are from material emitting gamma radiation. Gamma rays are the most penetrating of these radiations and external gamma ray exposure may contribute heavily to radiation doses to the internal organs. Beta and alpha particles are far less penetrating and deposit their energy primarily on the skin's outer layer. Consequently, their contribution to the absorbed dose to the total body, compared to that deposited by gamma rays, is negligible and will not be considered in this report.

6.2.5 Absorbed Dose and Absorbed Dose Rate

The radiological quantity absorbed dose, D, denotes the mean energy, imparted $\Delta\bar{\varepsilon}$, by ionizing radiation to a small finite mass of organ tissue with a mass, Δm, and is expressed as

$$D = d\bar{\varepsilon}/dm = \lim_{\Delta m \to 0} (\Delta\bar{\varepsilon}/\Delta m). \qquad (6)$$

The customary unit of absorbed dose is the rad, which is equivalent to 100 erg/g. For convenience, the millirad (mrad) is commonly used; it is one-thousandth of a rad. The mean absorbed dose to the total organ can be determined by averaging the absorbed dose distribution over the entire organ mass.

Internal and external exposures produced by radioactive waste products are not usually instantaneous, but are distributed over extended periods of time. The resulting time rate of change of the absorbed dose to a small volume of mass is referred to as the absorbed dose rate, $\dot{D}$:

$$\dot{D} = dD/dt = \lim_{\Delta t \to 0} (\Delta D/\Delta t). \qquad (7)$$

The customary unit of absorbed dose rate is any quotient of the rad (or its multiple or submultiple) and a suitable unit of time. In this report, absorbed dose rates are generally given in mrad/yr.

6.2.6 Linear Energy Transfer (LET)

The linear energy transfer, L_∞, is a quantity that represents the energy lost per unit length by charged particles in an absorbing medium. It represents the increment of the mean energy lost, ΔE, to tissue by a charged particle of specified energy in traversing a distance, ΔX:

$$L_\infty = dE/dX = \lim_{\Delta X \to 0} (\Delta E/\Delta X) \qquad (8)$$

For photons, L_∞ represents the energy imparted by the secondary electrons (electrons that are knocked out of their orbitals by primary radiation) resulting from secondary interactions between the photons and tissue material. High-LET radiation (alpha particles) deposits more energy per unit length of organ tissue than low-LET radiation (x rays, gamma rays, and beta particles). Consequently, the former are more effective per unit dose in causing biological damage. Customarily, L_∞ is expressed in keV/μm.

6.2.7 Dose Equivalent and Dose Equivalent Rate

Dose equivalent is a special radiation protection quantity that is used to express the absorbed dose in a manner which considers the difference in biological effectiveness of various kinds of ionizing radiation. The ICRU has defined the dose equivalent, H, as the product of the absorbed dose, D, the quality factor, Q, and all other modifying factors, N, at the point of interest in biological tissue (ICRU80). This relationship can be expressed in the following manner:

$$H = D\ Q\ N. \qquad (9)$$

The customary unit of dose equivalent is the rem. The quality factor is a dimensionless quantity that depends on the collision stopping power for charged particles, and it accounts for the differences in biological effectiveness found among varying types of radiation. By definition, it is independent of tissue and biological endpoint. EPA currently uses the ICRP-recommended value Q = 1 for x or gamma rays, while for alpha particles from nuclear transformations, a value of 20 is used (ICRP77). The product of all other modifying factors, N, such as dose rate, fractionation, etc., is taken as 1.

The dose equivalent rate, $\dot{H}$, is the time rate of change of the dose equivalent to organs and tissues and is expressed as:

$$\dot{H} = dH/dt = \lim_{\Delta t \to 0} (\Delta H/\Delta t). \qquad (10)$$

The customary unit of dose equivalent rate is mrem/yr.

6.2.8 Effective Dose Equivalent and Effective Dose Equivalent Rate

The ICRP has defined the effective dose equivalent, H_E, as:

$$H_E = \sum_T w_T H_T, \tag{11}$$

where H_T is the dose equivalent in tissue, T, and w_T is the weighting factor, which represents the proportion of the stochastic risk resulting from tissue, T, to the stochastic risk when the whole body is uniformly irradiated (ICRP77).

The customary unit of effective dose equivalent is the rem. The effective dose equivalent rate is the time derivative of the dose equivalent and is expressed as, $\dot{H}_E$, where:

$$\dot{H}_E = \sum_T w_T \dot{H}_T. \tag{12}$$

The customary unit of the effective dose equivalent rate is the rem divided by a suitable unit of time such as mrem/yr.

6.2.9 Working Levels and Working Level Months

Working levels is a unit that has been used to describe the radon decay-product activities in air in terms of potential alpha energy. It is defined as any combination of short-lived radon daughters (through ^{214}Po) per liter of air that will result in the emission of 1.3×10^5 MeV of alpha energy. An activity concentration of 100 pCi/l of ^{222}Rn, in equilibrium with its daughters, corresponds approximately to a potential alpha-energy concentration of 1 WL. The WL unit could also be used for thoron daughters. In this case, $1.3\ 10^5$ MeV of alpha energy (1 WL) is released by the thoron daughters in equilibrium with 7.5 pCi of thoron per liter. The potential alpha energy exposure of miners is commonly expressed in the unit Working Level Month (WLM). One WLM corresponds to an exposure to a concentration of 1 WL for the reference period of 170 hours.

6.2.10 Customary and SI Units

The relationship between the customary units used in this text and the international system of units (SI) for radiological quantities is shown in Table 6-1. While the SI radiological units are almost universally used in other countries for radiation protection regulation, the U.S. has not yet officially adopted their use for such purposes.

6.3 EPA Dosimetric Models

The EPA dosimetric models, to be discussed in the following sections, have been described in detail in previous publications (Du80, Su81). Information on the elements treated in these sections was taken directly from those documents or reports. With their permission, we have

Table 6-1. Comparison of customary and SI special units for radiation quantities

Quantity	Customary units	Definition	SI units	Definition
Activity (A)	Curie (Ci)	$3.7 \times 10^{10} s^{-1}$	becquerel (Bq)	s^{-1}
Absorbed Dose (D)	rad (rad)	$10^{-2} Jkg^{-1}$	gray (Gy)	Jkg^{-1}
Absorbed Dose Rate ($\dot{D}$)	rad per second (rad s^{-1})	$10^{-2} Jkg^{-1} s^{-1}$	gray per second (Gy s^{-1})	$Jkg^{-1} s^{-1}$
Dose Equivalent (H)	rem (rem)	$10^{-2} Jkg^{-1}$	sievert (Sv)	Jkg^{-1}
Dose Equivalent Rate ($\dot{H}$)	rem per second (rem s^{-1})	$10^{-2} Jkg^{-1} s^{-1}$	sievert per second (Sv s^{-1})	$Jkg^{-1} s^{-1}$
Linear Energy Transfer (L_∞)	kiloelectron volts per micrometer ($keV\mu m^{-1}$)	$1.602 \times 10^{-10} Jm^{-1}$	kiloelectron volts per micrometer ($keV\mu m^{-1}$)	$1.602 \times 10^{-10} Jm^{-1}$

adopted many edited passages from a variety of reports to explain our dosimetric models. The metabolic models and parameters employed in EPA internal dose models have been previously described (Su81). In most cases, the models are similar or identical to those recommended by the ICRP (ICRP79; ICRP80; ICRP81). However, differences in model parameters do exist for some radionuclides (Su81). The basic physiological and metabolic data used by EPA in calculating radiation doses are taken from ICRP reports (ICRP75, ICRP79).

EPA uses RADRISK for calculating radiation doses and risks to indvidiuals resulting from a unit intake of a radionuclide, at a constant rate, for a life time exposure (70-y dose committment). These calculations are done for the inhalation and ingestion pathways of those individuals who are exposed by immersion in contaminated air or by contaminated ground surfaces.

RADRISK computes doses for both chronic and acute exposures. Following an acute intake, it is assured the activity in the body decreases monotonically, particularly for radionuclides with rapid radiological decay rates or rapid biological clearance. In the case of chronic exposure, the activity in each organ of the body increases monotonically until a steady state is achieved, at which time the activity remains constant. The instantaneous dose rates at various times after the start of chronic exposure provide a reasonably accurate (and conservative) estimate of the total annual dose for chronic exposure conditions. However, the instantaneous dose rates may err substantially in the estimation of annual dose from an acute exposure, particularly if the activity levels decrease rapidly.

Since the rate of change in activity levels in various organs is more rapid at early times after exposure, doses are computed annually for the first several years, and for progressively longer periods thereafter, dividing by the length of the interval to estimate the average annual dose. This method produces estimates of risk that are similar to those computed by the original RADRISK methodology for chronic exposures and provides a more accurate estimate of the risks from acute intakes.

6.3.1 Internal Dose Models

EPA implements contemporary internal dosimetric models to estimate absorbed dose rates as a function of time to specified reference organs in the body caused by the ingestion or inhalation of radionuclides from low-level radioactive waste products. Estimates concerning the deposition and retention of inhaled particulates in the lung and their subsequent absorption into the blood and clearance into the gastrointestinal (GI) tract are made using the ICRP Task Group Lung Model (ICRP66). Residence times of radionuclides in the GI tract and transfer to the blood are estimated using a four-segment model of the GI tract that involves first-order mass transport and absorption of activity. Retention functions for activity in other organs are approximated by linear combinations of exponential functions.

The use of dosimetric factors together with estimates of activity in the various organs permits an estimate of dose rates from cross-irradiation when penetrating radiation is present. The models permit the consideration of the different absorption and retention properties of the various radionuclides in a decay chain. It is also assumed that activity transferred to other organs and tissues does not return to the blood.

(A) Generalized Scheme for Estimating Organ Absorbed Dose Rates

(1) Distribution of Activity of Radionuclides in the Body

The complex behavior of radionuclides is simplified conceptually by considering the body as a set of compartments. A compartment may be any anatomical, physiological, or physical subdivision of the body throughout which the concentration of a radionuclide is assumed to be uniform at any given time. The terms "compartment" and "organ" are often used interchangeably, although some of the compartments considered in this report may represent only portions of a structure usually considered to be an organ, while some compartments may represent portions of the body usually not associated with organs. Examples of compartments used in this report are the stomach, the pulmonary lung, the blood, or the bone. Within a compartment there may be more than one "pool" of activity.

A pool is defined to be any fraction of the activity within a compartment that has a biological half-life which is distinguishable from the half-time(s) of the remainder of activity within the compartment. Activity entering the body by ingestion is assumed to originate in the stomach compartment: activity entering through inhalation is assumed to originate in a compartment within the lung (the tracheo-bronchial, pulmonary, or naso-pharyngeal region). From the stomach, the activity is viewed as passing in series through the small intestine, the upper large intestine, and the lower large intestine, from which it may be excreted. Also, activity reaching the small intestine may be absorbed through the wall into the bloodstream, from which it may be taken in parallel into any of several compartments within the skeleton, liver, kidney, thyroid, and other organs and tissues. The list of organs or regions for which dose rates are calculated is found in Table 6-2. Activity in the lung may reach the bloodstream either directly or indirectly through the stomach or lymphatic system. The respiratory system and gastrointestinal tract models are discussed further in later sections. Figure 6-1 illustrates the EPA model used to represent the movement of radioactivity in the body.

EPA models separately consider the intake and subsequent behavior of each radionuclide in the environment. The models also allow for the formation of radioactive decay products within the body, and it is assumed that the movement of internally produced radioactive daughters is governed by their own metabolic properties rather than those of the parent. This is in contrast to the ICRP assumption that daughters behave exactly as the parent.

If $A_{ik}(t)$ denotes the activity of the ith species of the chain in organ k and if that activity is divided among several "pools" or "compartments" indexed by subscript l, then the time rate of change of activity can be modeled by a system of differential equations of the following form:

$$\dot{A}_{ilk} = -(\lambda_i^R + \lambda_{ilk}^B)A_{ilk} + c_{ilk}\left(\lambda_i^R \sum_{j=1}^{i-1} B_{ij} \sum_{r=1}^{L_{jk}} A_{jr} + p_{ik}\right),$$

$$l = 1,\ldots\ldots,L_{ik} \qquad (13)$$

where compartment l is assumed to have L_{ik} separate pools of activity, and where:

A_{ilk} = the activity of species i in compartment l of organ K;

λ_i^R = $(\ln 2)/T_i^R$, where T_i^R = radioactive half-life of species i;

λ_{ilk}^B = rate coefficient ($time^{-1}$) for biological removal of species i from the compartment l of organ k;

L_{ik} = number of exponential terms in the retention function for species in organ k;

B_{ij} = branching ratio of the nuclide j to species i;

p_{ik} = inflow rate of the ith species into organ k; and

c_{ilk} = the fractional coefficient for nuclide i in the lth compartment of organ k.

The subsystem described by these L_{ik} equations can be interpreted as a biological compartment in which the fractional retention of radioactive species is governed by exponential decay. Radioactivity that enters an organ may be lost by both radioactive decay and biological removal processes. For each source organ, the fraction of the initial activity remaining at any time after uptake at time t = 0 is described by a retention function consisting of one or more exponentially decaying terms:

$$R_{ik}(t) = \sum_{l=1}^{L_{ik}} c_{ilk} \exp[-(\lambda_i^R + \lambda_{ilk}^B)t] \qquad (14)$$

Table 6-2. Target organ or regions for which dose rates are calculated

Red bone marrow	Small intestine wall
Bone surfaces	Upper large intestine wall
Lungs	Lower large intestine wall
Kidneys	Bladder wall
Breast	Stomach wall
Pancreas	Other[a]

[a]Esophagus, lymphatic system, pharynx, larynx, salivary glands, brain, ovaries, uterus, spleen.

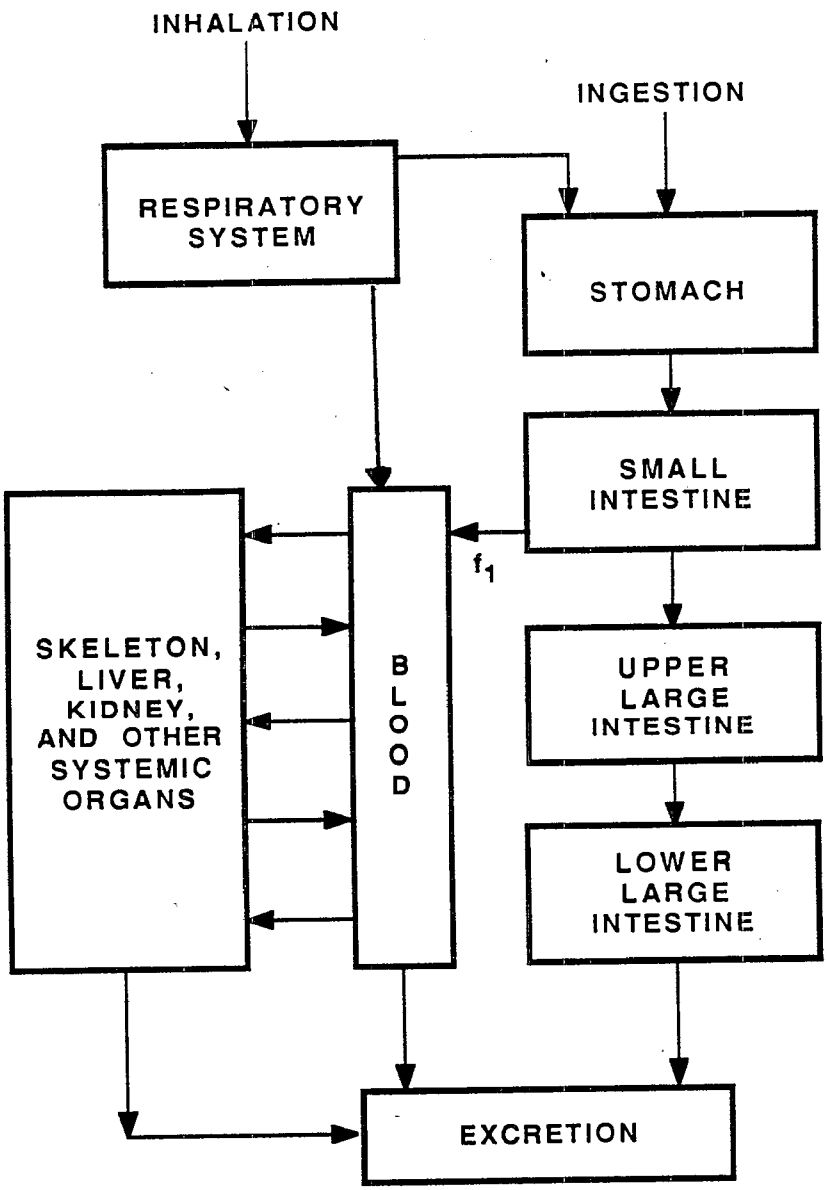

Figure 6-1. EPA Schematic Representation of Radioactivity Movement in the Human Body.

The subscript l in the above equation represents the lth term of the retention function, and the coefficients c_{ilk} can be considered as "pathway fractions." Figure 6-2 illustrates an example of the decline of activity in an organ as a function of time t.

(2) Dose Rates to Target Organs

The activity of a radionuclide in a compartment is a measure of the rate of energy being emitted in that compartment, at any time, t , and can be related to the dose rate to a specific organ at that time. This requires estimating the fraction of the energy emitted by the decay of the radionuclide in each compartment that is absorbed by the specific organ.

The absorbed dose rate, $D_i(X;t)$ to target organ X at time t due to radionuclide species i in source organs $Y_1, Y_2, \ldots, Y_M$ is estimated by the following equation:

$$\dot{D}_i(X;t) = \sum_{k=1}^{M} \dot{D}_i(X \leftarrow Y_k;t) \qquad (15)$$

where: $\dot{D}_i(X \leftarrow Y_k; t) = S_i(X \leftarrow Y_k)\, A_{ik}(t)$; and
$A_{ik}(t)$ is the activity, at time t of species i in source organ Y_k.

$S_i(X \leftarrow Y_k)$, called the S-factor, represents the average dose rate to target organ X from one unit of activity of the radionuclide uniformly distributed in source organ or compartment Y_k. It is expressed in the following manner:

$$S_i(X \leftarrow Y_k) = c \sum_m f_m E_m \Phi_m(X \leftarrow Y_k) \qquad (16)$$

where:

c = a constant that depends on the units of dose, energy, and time being used;

f_m = intensity of decay event (number per disintegration);

E_m = average energy of decay event (Mev); and

$\Phi m(X \leftarrow Y_k)$ = specific absorbed fraction, i.e., the fraction of emitted energy from source organ Y_k absorbed by target organ X per gram of X,

where the summation is taken over all events of type m. The units for S-factors depend on the units used for activity and time; thus, the S-factor units may be rad/Ci-day.

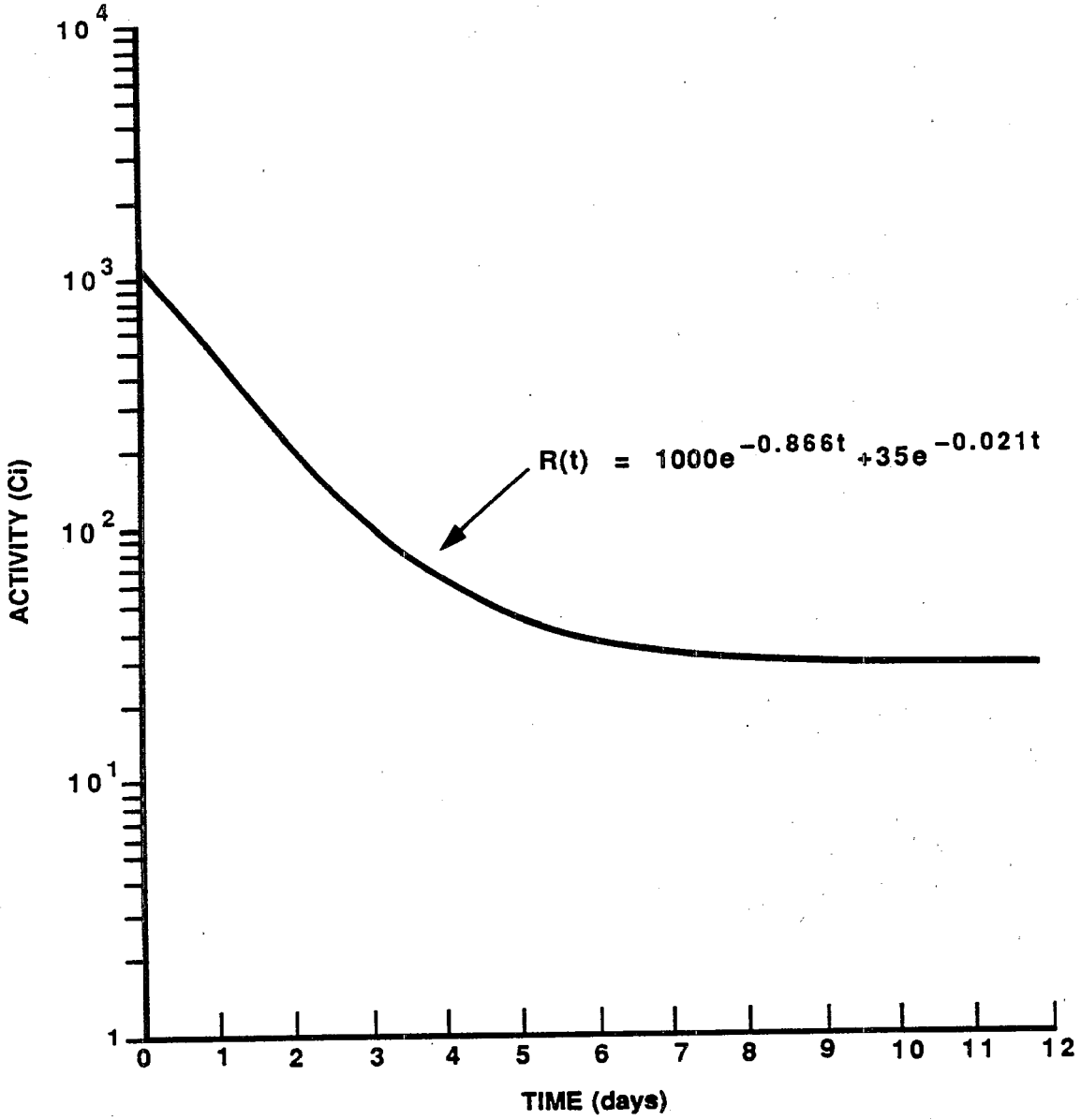

Figure 6-2. An Example (on a Log-Linear Axis) of the Decline of Activity of a Radionuclide in an Organ, Assuming an Initial Activity in the Organ and No Additional Uptake of Radionuclide by the Organ

The S-factor is similar in concept to the SEE factor (specific effective energy) used by the ICRP Committee 2 in Publication 30. However, the SEE factor includes a quality factor for the radiation emitted during the transformation. The above equations are combined to produce the following expressions for the absorbed dose rates to target organs at any time due to one unit of activity of radionuclide species, i, uniformly distributed in source organs $Y_1 \ldots Y_k$:

$$\dot{D}_i(X;t) = \sum_k \sum_m A_{ik}(t)\, S_{im}(X \leftarrow Y_k) \qquad (17)$$

The corresponding dose equivalent rate, $\dot{H}_i(X;t)$, can be estimated by inclusion of the quality factor , Q_m, and the modifying factor, $N_m(Y_k)$:

$$\dot{H}_i(X;t) = \sum_k \sum_m A_{ik}(t) Q_m N_m(Y_k)\, S_{im}(X \leftarrow Y_k) \qquad (18)$$

Implicit in the above equations is the assumption that the absorbed dose rate to an organ is determined by averaging absorbed dose distributions over its entire mass.

Alpha and beta particles are usually not sufficiently energetic to contribute a significant cross-irradiation dose to targets separate from the source organ. Thus, the absorbed fraction for these radiations is generally assumed to be just the inverse of the mass of organ X, or if the source and target are separated, then $\Phi_m(X \leftarrow Y) = 0$. Exceptions occur when the source and target are in very close proximity, as is the case with various skeletal tissues. Absorbed fractions for cross irradiations among skeletal tissues are computed as a function of energy, using a method described by Eckerman (Ec85). The energy of alpha particles and their associated recoil nuclei is generally assumed to be absorbed in the source organ. Therefore, $\Phi_m(X \leftarrow X)$ is taken to be the inverse of the organ mass, and $\Phi_m(X \leftarrow Y) = 0$ if X and Y are separated. Special calculations are performed for active marrow and endosteal cells in bone, based on the method of Thorne (Th77).

(3) Monte Carlo Methodology to Estimate Photon Doses to Organs

The Monte Carlo method uses a computerized approach to estimate the probability of photons interacting within target organ X after emission from source organ Y. The method is carried out for all combinations of source and target organs and for several photon energies. The body is represented by an idealized phantom in which the internal organs are assigned masses, shapes, positions, and attenuation coefficients based on their chemical composition. A mass attenuation coefficient μ_o is chosen, where μ_o is greater than or equal to the mass attenuation coefficients for any region of the body. Photon courses are simulated in randomly chosen directions, and potential sites of interactions are selected by taking distances traversed by them as $-\ln r/\mu_o$, where r is a random number distributed between 0 and 1. The process is terminated when either the total energy of photons has been deposited or the photon escapes from the body. The energy deposition for an interaction is determined according to standard equations (ORNL74).

(4) Effects of Decay Products

In calculating doses from internal and external exposures, the ingrowth of radioactive decay products (or daughters) must be considered for some radionuclides. When an atom undergoes radioactive decay, the new atom created in the process, which may also be radioactive, can contribute to the radiation dose to organs or tissues in the body. Although these decay products may be treated as independent radionuclides in external exposure, the decay products of each parent must be followed through the body in internal exposure situations. The decay product contributions to the absorbed dose rates, which are included in EPA calculations, are based on the metabolic properties of the individual daughters and the organ in which they occur.

(B) Inhalation Dosimetry

As was stated earlier, individuals immersed in contaminated air will breathe radioactive aerosols or particulates, which can lead to a radiation dose to lung tissue or other organ tissues in the body. The total internal dose cause by inhalation can depend on a variety of factors, including such parameters as breathing rate, particle size, and physical activity.

In order to estimate the deposition probability and the total integrated dose over a specific time period, a set of assumptions is required specifying the distribution of particle depositions in the respiratory tract and the mathematical characteristics of the clearance parameters. The EPA currently uses the set of assumptions and kinetic equations that were earlier incorporated and used in the model of lung deposition and clearance of radioactive aerosols developed by the ICRP Task Group on Lung Dynamics (TGLM)(ICRP66). This section will summarize the essential features of that model. For a more comprehensive treatment, the reader is referred to the actual report.

(1) ICRP Respiratory Tract Model

The basic features of the ICRP lung compartmental model are shown in Figure 6-3. According to this model, the respiratory tract is divided into four regions: naso-pharyngeal (N-P), tracheo-bronchial (T-B), pulmonary (P), and lymphatic tissues.

In the model, the regions N-P, T-B, and P are assumed to receive fractions D_3, D_4, and D_5 of the inhaled particulates, where the sum of these is less than 1 (some particles are removed by prompt exhalation). The values D_3, D_4, and D_5 depend on the activity median aerodynamic diameter (AMAD) of the inspired particles, and for purposes of risk calculations, EPA uses AMAD's of 1 micron. The lung model employs three clearance classes, D, W, and Y, corresponding to rapid, intermediate, and low clearance, respectively, of material deposited in the respiratory passages. The clearance class depends on chemical properties of the inhaled particles.

COMPARTMENT		CLASS					
		D		W		Y	
		T	F	T	F	T	F
N•P	a	0.01	0.5	0.01	0.1	0.01	0.01
(D_3 = 0.30)	b	0.01	0.5	0.4	0.9	0.4	0.99
T•B	c	0.01	0.95	0.01	0.5	0.01	0.01
(D_4 = 0.08)	d	0.2	0.05	0.2	0.5	0.2	0.99
	e	0.5	0.8	50	0.15	500	0.05
P	f	n.a.	n.a.	1.0	0.4	1.0	0.4
		n.a.	n.a.	50	0.4	500	0.4
(D_5 = 0.25)	g	0.5	0.2	50	0.05	500	0.15
	h						
L	i	0.5	1.0	50	1.0	1000	0.9

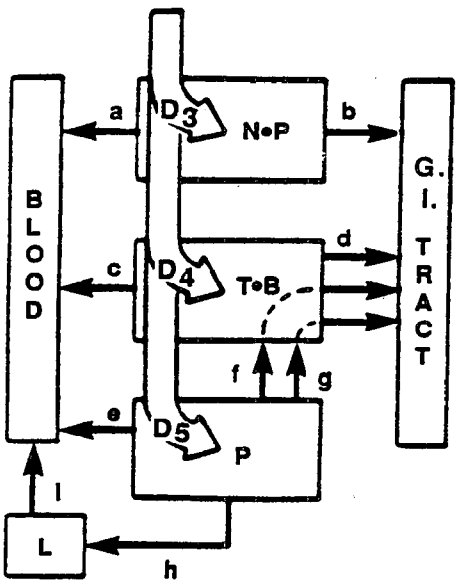

Figure 6-3. The ICRP Task Group Lung Deposition and Clearance Model for Particulates

(a) Lung Compartmental Retention Functions

The retention functions for the four regions of the human lung are given in Table 6-3 for the case of an instantaneous acute intake of a single radionuclide species, i.

Equation 6-3a defines the retention function for the naso-pharyngeal region. The first term is assigned to pathway a, that is, clearance to the blood; the second term is assigned to pathway b, that is, mechanical removal to the stomach. In this equation, as in the equations that follow, $\lambda_{v,i}$ is the sum of the radiological decay constant and the biological removal constant $\lambda^B_{v,i} = (\ln 2)/T^B_{v,i}$, where $T_{v,i}$, is the biological half-time associated with clearance pathway v(=a,b,...., i* in Figure 6-3. Included also are the pathway partition fractions, $F_{v,i}$, for the three clearance classes D, W, and Y.

The retention function describing activity deposited directly in the the tracheo-bronchial tree takes the form of Equation 6-3b, where the first term represents removal through pathway c (clearance to the blood) and the second term is assigned c to pathway d (removal to the stomach). Activity in the tracheo-bronchial tree can also result from recirculation from the pulmonary region (Equation 6-3c). The clearance of the activity in transit from the pulmonary pathways f and g to the stomach via pathways k and l, respectively, is described by the retention function in Equation 6-3d. The clearance of activity from the pulmonary region and respiratory lymph through pathways e, f, g, h, and i is represented by Equation 6-3e.

In Figure 6-3 the columns labeled D, W, and Y correspond, respectively, to rapid, intermediate, and slow clearance of the inspired material (in days, weeks, or years). The symbols T and F denote the biological half-time (days) and coefficient, respectively, of a term in the appropriate retention function. The values shown for D_3, D_4, and D_5 correspond to activity median aerodynamic diameter AMAD = 1 μm and represent the fraction of the inspired material depositing in the lung regions.

(b) Estimating the Absorbed Dose Rate to the Lung

EPA estimates the absorbed dose rate to the lung due to the inhalation of species i from:

$$D_i(\text{lung};t) = \sum_k \sum_m A_{ik}(t)\, S_{im}(\text{lung} \leftarrow Y_k) \qquad (19)$$

Table 6-3. Compartment retention functions of the ICRP lung model

$$R_i^{N-P}(t) = F_{a,i}\exp(-\lambda_{a,i}t) + F_{b,i}\exp(-\lambda_{b,i}t) \quad (6\text{-}3a)$$

$$R_{1,i}^{T-B}(t) = F_{c,i}\exp(-\lambda_{c,i}t) + F_{d,i}\exp(-\lambda_{d,i}t) \quad (6\text{-}3b)$$

$$R_{2,i}^{T-B}(t) = \exp(-\lambda_{d,i}t) \quad (6\text{-}3c)$$

$$R_i^{P}(t) = F_{e,i}\exp(-\lambda_{e,i}t) + F_{f,i}\exp(-\lambda_{f,i}t) + F_{g,i}\exp(-\lambda_{g,i}t) + F_{h,i}\exp(-\lambda_{h,i}t) \quad (6\text{-}3d)$$

$$R_i^{L}(t) = F_{i^*,i}\exp(-\lambda_{i^*,i}t) + (1 - F_{i^*,i})\exp(-\lambda_i^R t) \quad (6\text{-}3e)$$

Like the ICRP, EPA assumes that the absorbed dose rate to the N-P region can be neglected. Unlike the ICRP, however, EPA averages the dose over the pulmonary region of the lung model (compartments e through h), to which is assigned a mass of 570 g, including capillary blood (ICRP75). In addition, it is assumed that the total volume of air breathed in one day by a typical member of the general population is 2.2E+04 liters. This value was determined by averaging the 23 ICRP adult male and female values based on 8 hours of working "light activity," 8 hours of nonoccupational activity, and 8 hours of resting. EPA uses the anatomical model that was devised by Weibel (Model A (We63)) in dosimetric calculations.

D_i(lung;t) has two components: the absorbed dose rate from radioactive material located in the lung (i.e., in target tissues comprising the P-region) and the dose rate from photons arising in other organs and tissues of the body:

$$D_i(\text{lung};t) = \left[\sum_m A_m(t)\, S_{im}(\text{lung} \leftarrow \text{lung}) + \sum_k \sum_m A_{ik}(t)\, S_{im}(\text{lung} \leftarrow Y_k)\right]$$

The EPA approach yields lung dose estimates that are 50 percent higher than those of the ICRP for class D and W aerosols and 20 percent lower for class Y particles (ORNL85).

(C) Ingestion Dosimetry

(1) ICRP GI Tract Model

According to the ICRP 30 GI tract model, the gastrointestinal tract consists of four compartments, the stomach (S), small intestine (SI), upper large intestine (ULI), and lower large intestine (LLI). The fundamental features of the model are shown in Figure 6-4. It is assumed that absorption into the blood occurs only from the small intestine (SI).

(a) GI Tract Compartment Retention Functions

This model postulates that the radioactive material that enters the compartments of the GI tract is exponentially removed by both radioactive decay and biological removal processes, and that there is no feedback.

Absorption of a particular nuclide from the GI tract is characterized by f_1, which represents that fraction of the nuclide ingested which is absorbed into body fluids, if no radiological decay occurs:

$$f_1 = \lambda_{ab} VA_{SI}/(\lambda_{ab} + \lambda_{SI})A_{SI} \qquad (20)$$

from which an expression for λ_{ab} in terms of f_1 can be derived:

$$\lambda_{ab} = f_1 \lambda_{SI}/(1 - f_1) \qquad (21)$$

The kinetic model, as formulated by the ICRP, does not permit total absorption of a nuclide ($f_1 = 1$).

The retention functions for the four sections of the gastrointestinal tract are found in Table 6-4. Equation 6-4a defines the retention function for the stomach. In this equation, as with the equations that follow, $\lambda_{\sigma,i}$, is the sum of the radiological decay constant, λ_i^R, and the biological removal constant $\lambda_{S,i}$ (for stable radionuclide) = $\ln 2/T_{S,i}$, where $T_{S,i}$ = the mean residence time associated with section σ (=S, SI, ULI, or LLI). Equation 6-4b represents retention of radionuclides in the SI. Equations 6-4c and 6-4d are retention functions that define the loss of stable nuclide from the upper and lower large intestine, respectively.

(b) Estimating the Absorbed Dose Rates to Sections of the Gastrointestinal Tract

The absorbed dose rate to a section of the GI tract resulting from the ingestion or inhalation of species i is estimated using the following equation:

$$\dot{D}_i(\sigma;t) = \sum_k \sum_m A_{ik}(t)\, S_{im}(\sigma \leftarrow Y_k) \qquad (22)$$

where σ=S, SI, ULI, or LLI are the segments of the tract.

$\dot{D}_i(\sigma;t)$ is estimated for the wall for each segment of the GI tract.

The coefficients A_S, A_{SI}, A_{ULI}, A_{ULI}, are expressed in terms of the elimination rate constants and can be calcuated easily.

(D) Bone Dosimetry

(1) Estimating the Absorbed Dose Rate to the Bone

The absorbed dose rate to the bone resulting from the inhalation and ingestion of species, i, is estimated using the following equation:

$$\dot{D}_i(\text{bone};t) = \sum_k \sum_m A_{ik}(t) Q_m S_{im}(\text{bone} \leftarrow Y_k) \qquad (23)$$

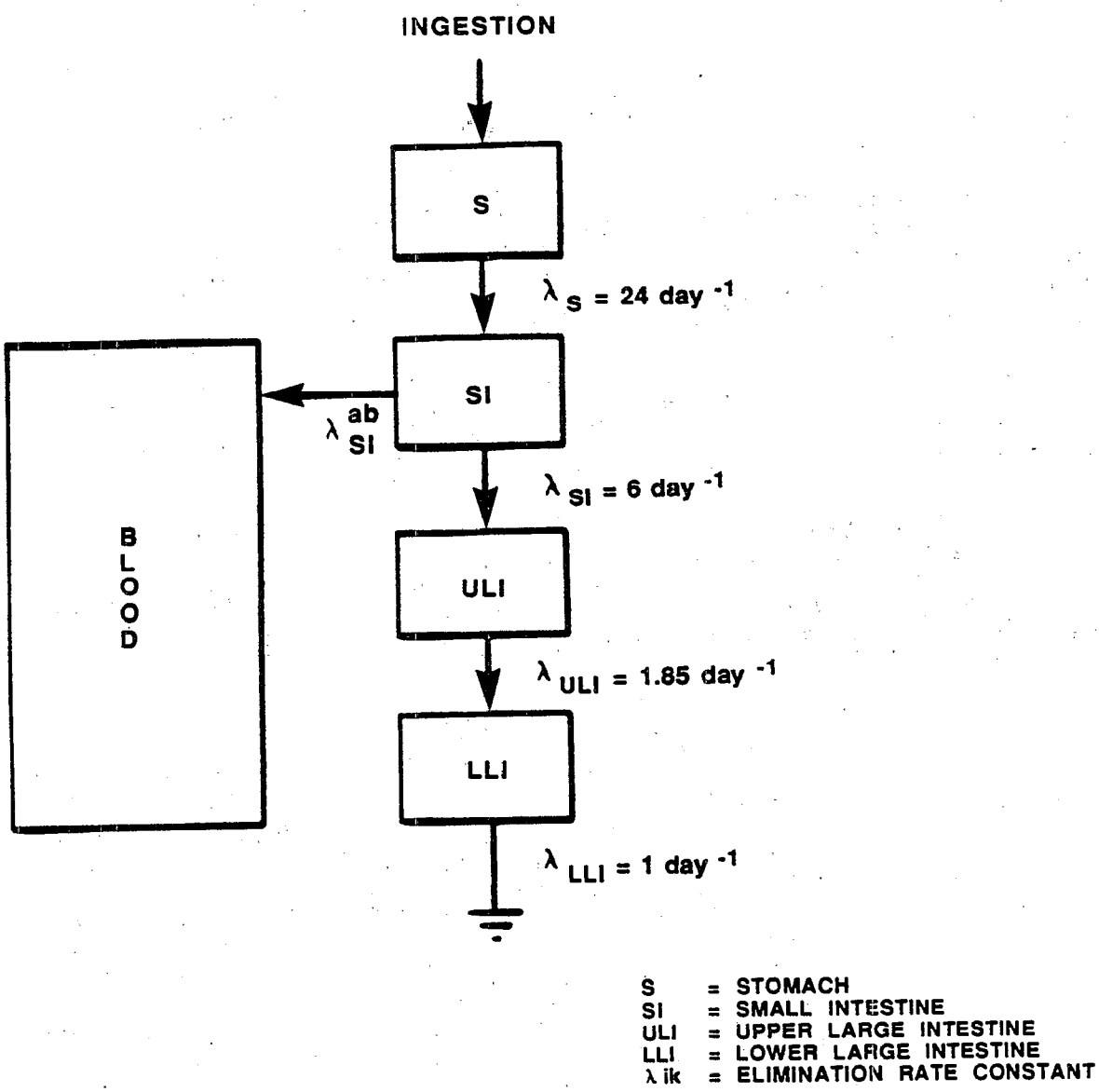

Figure 6-4. Schematic Representation of Radioactivity Movement in the Gastrointestinal Tract and Blood

Table 6-4. Compartment retention functions of the ICRP GI tract model

$$R_i^S(t) = A_S \exp(-\lambda_{S,i}\, t) \qquad (6\text{-}4a)$$

$$R_i^{SI}(t) = A_{SI}[\exp(-\lambda_{S,i}\, t) - \exp(-(\lambda_{SI,i} + \lambda_{ab})t)] \qquad (6\text{-}4b)$$

$$R_i^{ULI}(t) = A'_{ULI} \exp(-\lambda_{S,i}\, t) - A''_{ULI}\exp[-(\lambda_{SI,i} + \lambda_{ab})t] + A'''_{ULI}\exp(-\lambda_{ULI,i}\, t) \qquad (6\text{-}4c)$$

$$R_i^{LLI}(t) = A'_{LLI} \exp(-\lambda_{S,i}\, t) - A''_{LLI}\exp[-(\lambda_{SI,i} + \lambda_{ab})t] + A'''_{LLI} \exp(-\lambda_{ULI,i}\, t) + A''''_{LLI} \exp(-\lambda_{LLI,i}\, t) \qquad (6\text{-}4d)$$

The active marrow space is contained within the trabecular bone of the skeleton, and the endosteal tissues are associated with both cortical and trabecular bone. Cortical bone is the hard mineral region on the exterior of the bones, while trabecular bone is the soft, spongy mineral lying in the interior of bone, particularly the vertebrae, ribs, flat bones, and the ends of the long bones of the skeleton. To implement the dosimetric formulations for these two target regions, EPA considers radionuclide activity occurring within both cortical and trabecular bone and the distribution of the activity within the volume or along the bone surface.

The 10 μm thick layer on bone surfaces over which the dose equivalent rate is averaged has a mass of 120 g. The mass of the active red marrow region and the mineral bone are 1,500 g and 5,000 g, respectively. The actual bone compartmental model used by EPA depends on the type of radionuclide being considered and its metabolic behavior in the body. As an example, for strontium, EPA views the mineralized skeleton as consisting of two main compartments: trabecular and cortical bone. Two subcompartments, surface and volume, are considered within each of the main compartments. The four subcompartments of the bone and the movement of strontium in the bone are shown in Figure 6-5. To describe retention of plutonium in the skeleton, however, EPA views the skeleton as consisting of a cortical compartment and a trabecular compartment, with each of these further divided into three subcompartments: bone surface, bone volume, and a transfer compartment. The transfer compartment, which includes the bone marrow, may receive plutonium that is removed from bone surface or volume; plutonium may reside in this compartment temporarily before being returned either to the bloodstream or to bone surfaces (Figure 6-6). Because of the large amount of recycling of plutonium among the skeletal compartments, blood, and other organs, recycling is considered explicitly in this model.

(E) Uncertainties in Internal Dose Calculations

All internal dosimetric models are inherently uncertain. At best, they can only approximate the real situation. The uncertainties associated with the EPA internal dose estimates are extremely difficult to quantify because the models used are limited by the lack of data for the model parameters. In estimating doses to individuals in the general population, a number of sources of uncertainties will arise. These sources can be attributed both to ICRP model formulation and to parameter variability caused by measurement errors, sampling errors, or natural variation.

The purpose of this section is to discuss some of the uncertainty factors that are associated with EPA internal dosimetric modeling and to illustrate how parameter variability can affect the estimates of the doses to organs in the body.

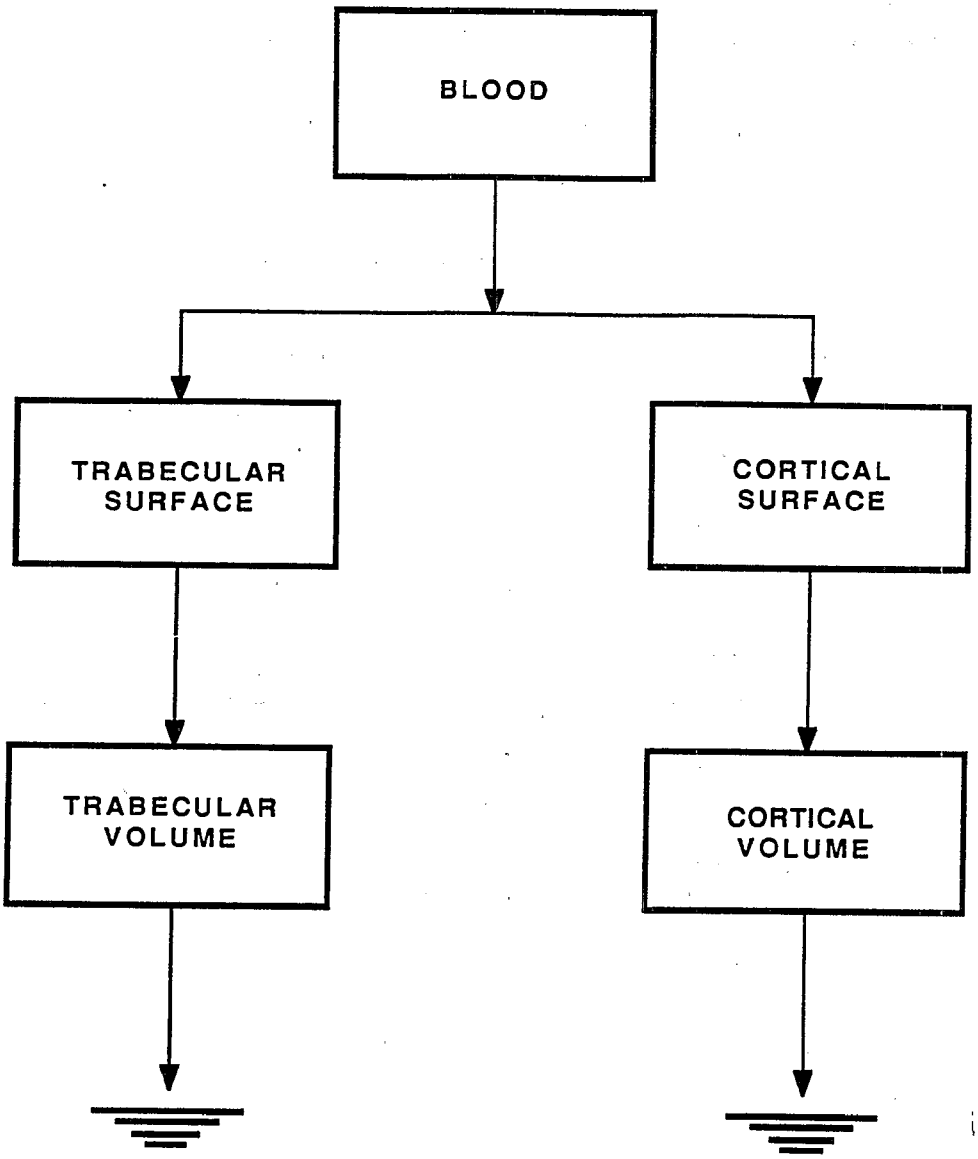

Figure 6-5. Compartments and Pathways in Model for Strontium in the Skeleton.

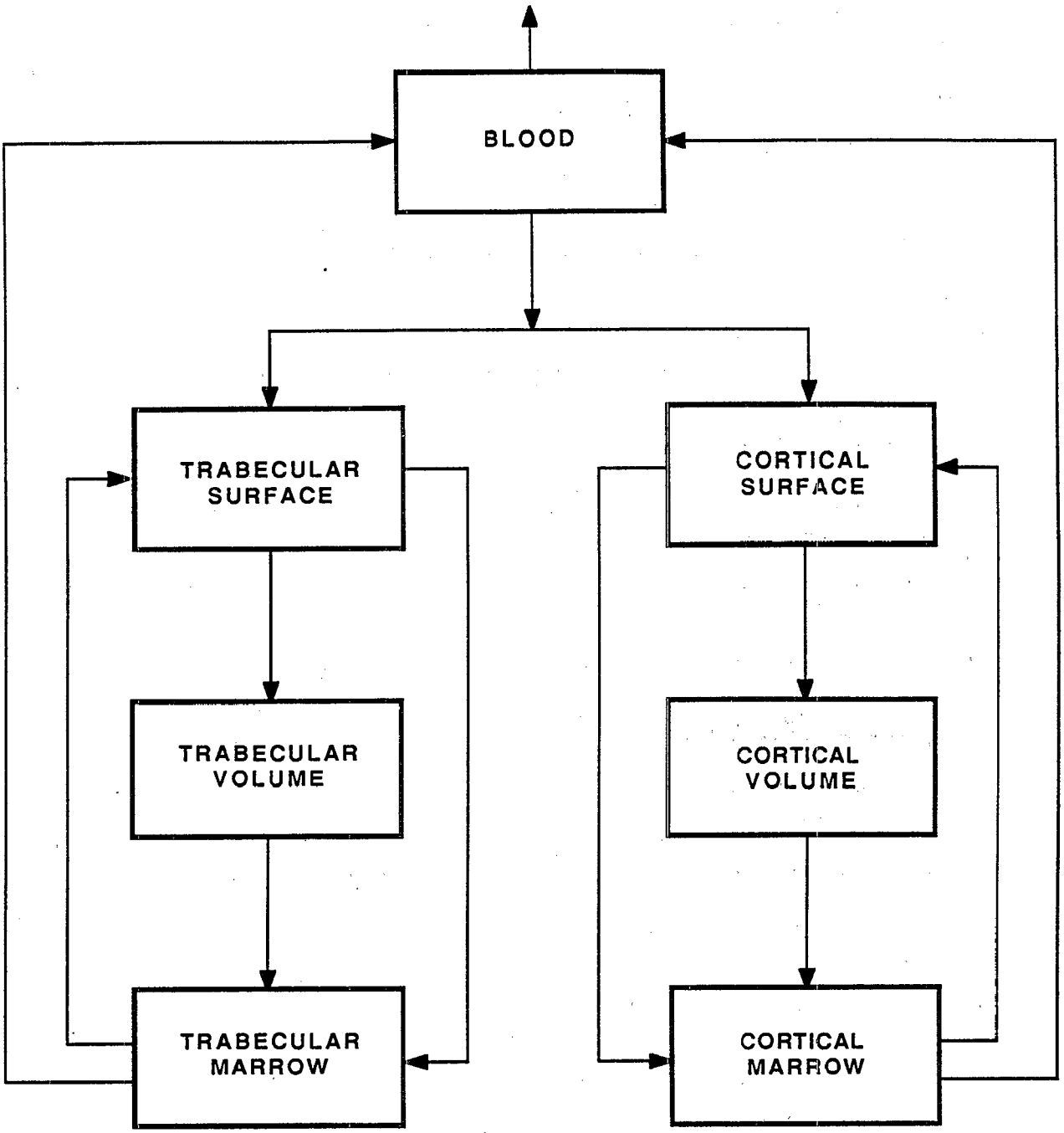

Figure 6-6. Compartments and Pathways in Model for Plutonium in the Skeleton.

(1) Uncertainties Associated with ICRP Model Formulation

In general, the ICRP estimates of the dose to tissue of various radiosensitive organs, following internal exposure to a given radionuclide, are derived by considering three models: (1) a model for the retention and translocation to blood of inhaled material by the respiratory or gastrointestinal tract; (2) a "metabolic model" of the allotment of activity among the various organs and retention in those organs; and (3) a model of the dose received by each organ from the given distribution of the radionuclide and its radioactive progeny. ICRP metabolic models were derived by restricting attention to the average adult, considering only integrated doses over relatively long periods, not explicitly considering the recirculation of radionuclides among the organs, and assuming that daughter radionuclides produced from their parent within the body stay with and behave metabolically like their parent.

The uncertainties associated with the ICRP 30 modeling approach can be summarized in the following: (a) Animal data were used extensively in constructing the models, and there is often no sound basis to support extrapolation to humans. (b) Dose to heterogeneously distributed radiosensitive tissues of an organ (e.g., skeleton) cannot be estimated accurately, since the actual movement of radionuclides in the body is usually not accurately tracked, even in cases where the whole-body retention is estimated fairly well. (c) Some radionuclides are assigned the model of an apparently related nuclide, although certain differences in metabolism are known. (d) The ingrowth of radioactive daughters is often not handled realistically, and the format of the models makes it difficult to do so. (e) The models often do not yield accurate estimates of excretion, even for the average adult. (f) The models cannot be extended to a person with anatomical or metabolic characteristics different from "Reference Man," such as a child. This is because the components of the models were derived as fits to experimental data and do not correspond to identifiable anatomical or physiological entities. There is generally insufficient data with which to develop new models for special subgroups by the fitting techniques that characterize most of the standard-man models.

(a) Uncertainties in the ICRP Respiratory Tract Model

According to some researchers it can be concluded that the experimental data for most individuals indicate that the ICRP respiratory model overestimates pulmonary deposition, underestimates tracheo-bronchial deposition, and provides an adequate estimate of naso-pharyngeal deposition (Cr80).

Deposition in the respiratory tract of individuals exposed to radioactive aerosols is controlled by three main mechanisms: sedimentation, impaction, and Brownian diffusion. The uncertainties associated with these mechanisms are due to: (1) uncertainty in the anatomical model of the respiratory tract, (2) uncertainty in the effective aerodynamic diameter of the inhaled particles, and (3) uncertainty in the breathing pattern and rates.

Deposition in the respiratory tract is largely controlled by the anatomical structure of the lung. The number of particles deposited in the lung depends essentially on the number of airways and their diameters and lengths, branching and gravitational angles, and the distances to the alveolar walls. The ICRP respiratory tract model TGLM (ICRP66) uses the anatomical model devised by Findeisen (Fi35). Current dosimetric models use Weibel Model A (We63) in dosimetric calculations. It, like other models, assumes that the lung airways are rigid tubes with symmetric dichotomous branching patterns and that their morphometric properties are those of the average adult male. In reality, however, the airways have circular ridges or longitudinal grooves (FRC67); thus, the trachea may be quite irregular in shape (Br52). Moreover, airways change in diameter and length during inspiration and expiration (Ho75, Hu72, Th78), which will also have some effect on branching and gravitational angles.

In addition, there is some evidence that not all aleveoli are open all the time but are recruited as necessary (Th78), which would influence the active lung volume. Finally, there are branching pattern and other differences in lung morphology which can depend on the age of the individual at the time of exposure (lengths, diameters, and gravity angles) (Ph85).

According to the model, the physico-chemical properties of the inhaled aerosol governing the rate deposited material is cleared from the lung are reflected in three clearance classes. Radionuclides are assigned to a clearance class based primarily on the solubility of the element in an artificial lung fluid of relevant chemical compounds. This procedure, while quite reasonable for some situations, is questionable if the radioactive material is attached to ambient environmental aerosols. In this case, the physico-chemical nature of the carrier aerosol rather than the chemical form of the radionuclide may influence clearance rates. Very limited information exists to evaluate potential uncertainties associated with the assignment of clearance classifications for complex mixtures.

(b) Uncertainties in the ICRP GI Tract Model

As was stated earlier, the ICRP GI tract model assumes that ingested material moves in sequence through the stomach, small intestine, upper large intestine, and lower large intestine. The model depicts an exponential removal from each compartment, characterized by a single removal rate that depends only on the compartment. For most radionuclides, estimates of dose to all organs except for the GI tract are fairly insensitive to assumptions concerning the rate of transport through the GI tract. Exceptions are those nuclides with such short half-lives that a substantial fraction of the activity entering the stomach decays before being absorbed into the blood.

Radionuclides are assumed to be absorbed into the blood from the SI, even though radionuclides transported through the GI tract may be absorbed into the bloodstream from any of the four major compartments.

The fractions absorbed from the stomach and large intestine are usually considered negligible compared with f_1, the fraction from the small intestine. The latter varies considerably depending on the radionuclide and on the material to which it is attached. The variability in f_1 probably represents the greatest uncertainty associated with the GI tract model for most radionuclides and will be further discussed in the next section.

(2) Uncertainties Due to Parameter Variability

Parameters employed in internal dosimetric models are often quantified by values that represent "best estimates" or "average" values from parameter distributions and ignore the recognized variability among individuals. As a result, there are limitations inherent in taking a deterministic approach in applying "Reference Man" parameters to assess the dose to individuals in the general population. When assessing the uncertainties associated with using EPA dosimetric models, the variability of such parameters as radionuclide intake rate, organ mass, blood transfer factor, organ uptake rate, and biological half-times of the ingested radionuclides must be considered. These parameters vary among individuals in the general population primarily because of age and sex differences. Other factors that contribute are biological, environmental, and geographical differences.

In order to fully assess the uncertainty in model predictions of the doses to organs due to parameter variability (assuming that the model structure is correct), a parametric uncertainty analysis must be done. This process involves taking frequency distributions of values for each model parameter to produce a frequency distribution of model predictions of the doses.

Parameter values used in radiological assessments are generally taken from the literature. However, a wide range of reported values is expected for some parameters, implying a large uncertainty in the estimated dose.

As a numerical illustration, we will consider the uncertainties in parameters associated with age variations for the simple case of chronic ingestion of a single radionuclide. The model used to define the absorbed dose rate to a target organ X due to radioactivity located in source organ Y_k can be expressed as:

$$\dot{D}(X \leftarrow Y;t) = c\, I\, f_1\, f_2\, E\, [1-\exp(-\lambda_{ik})]/mV\lambda_{ik} \qquad (24)$$

where:

$\dot{D}(X \leftarrow Y;t)$ = absorbed dose rate (rad/day);

I = radionuclide intake rate (Ci/day);

f_1 = fraction of ingested activity transferred to the blood;

f_2' = fraction of blood activity transferred to the organ;

m = target organ mass (g);

λ_{ik} = elimination constant (day^{-1});

E = energy absorbed by the target organ for each radioactive transformation;

c = proportionality constant (51.2×10^6 g rad Ci^{-1} $Mev^{-1}d^{-1}$.

For simplicity, we will consider the case where t is very large compared to the biological half-life of the incorporated radionuclide, so that the term in the bracket is approximately 1:

$$D\ (X \leftarrow Y\ ;t) = c\ I\ f_1\ f_2'\ E/mV\lambda_{ik} \qquad (25)$$

In addition, it is assumed that the parameters remain constant throughout the period of investigation.

Equation 25 is a simplified form of the actual equations used by EPA to estimate the absorbed dose rates to target organs resulting from the ingestion of radioactive material. It represents the absorbed dose rate to a target organ from particulate radiation due to radioactivity that is uniformly distributed in that organ (i.e., $\phi(X \leftarrow Y_k) = \phi(X \leftarrow X)$).

For this analysis, we will consider the chronic intake of iodine-131 assuming that it behaves metabolically the same as stable iodine. It is further assumed that iodine is rapidly and almost completely absorbed into the bloodstream following inhalation or ingestion. From the blood, iodine enters the extracellular fluid and quickly becomes concentrated in the salivary, gastric, and thyroid glands. It is rapidly secreted from the salivary and gastric glands, but is retained in the thyroid for relatively long periods.

The intake and metabolism of iodine have been reviewed extensively in the literature. There are two principal parameter data bases to be used in this analysis. The first is found in an article, published by Dunning and Schwarz (Du81), in which the authors compiled and evaluated the variability in three of the principal biological parameters contained in Equation 25: m, $T_{1/2}$, and f_2'. The second is taken from a paper by Bryant (Br69), which provides age-specific values for most of the model parameters. These data will serve as a means of illustrating how parameter variabilities in the above model can affect absorbed dose rate estimates to members of the general population.

(a) Intake Rate, I

The amount of radioactive material taken into the body over a specified period of time, by ingestion or inhalation, is expected to be proportional to the rate of intake of food, water, or air by individuals, and it would depend

on such factors as age, sex, diet, and geographical location. Therefore, understanding the patterns of food intake for individuals in the population is important in assessing the possible range of intake rates for radionuclides.

Recent EPA studies were done to assess the daily intake rates of food and water for individuals in the general population. These studies showed that age and sex played an important role (Ne84). Age significantly affects food intake rates for all of the major food classes and, with one exception, subclasses. The relationships between food intake and age are, in most cases, similar to growth curves: there is a rapid increase in intake at an early stage of physical development, then a plateau is reached in adulthood, followed by an occasional decrease after age 60.

When sex differences were significant, males, without exception, consumed more than females. The study also showed that relative consumption rates for children and adults depend on the type of food consumed. The amount of radioactivity taken into the body per unit intake of food, air, and water depends on its relative density (amount of radioactivity contained in the material per unit volume). The most likely pathway to organs in the body for the ingestion of radioactive iodine comes from drinking milk. According to the above study, the daily intake rate for milk for children (under 1 yr) to that of an adult (25 to 29 yr) for males, varied by a factor of 2. Thus, if milk contains radioactive iodine, the absorbed dose rate to the thyroid due to the milk intake rate alone would also vary by a factor of 2. The intake rates for milk used in this analysis are 0.7l/day and 0.5l/day for the child and adult, respectively.

(b) Transfer Fraction, f_1

While uncertainty in f_1 is not an important consideration for iodine, it can be very significant for other elements. Experimental studies suggest that the f_1 value for some radionuclides may be orders of magnitude higher in newborns than in adult mammals, with the largest relative changes with age occurring for those nuclides with small adult f_1 values (Le83). For some radionuclides there appears to be a rapid decrease in the f_1 value in the first year of life. This can be related to the change in diet during this time period, which could affect both the removal rate from the small intestine to the upper large intestine and the absorption rate from the small intestine to the bloodstream. Studies have indicated that the wall of the small intestine is a selective tissue and that absorption of nutrients is to a large extent controlled by the body's needs (Le83). In particular, the fraction of calcium or iron absorbed depends on the body's needs for these elements, so the f_1 value for these elements and for related elements such as strontium, radium, and barium (in the case of calcium) and plutonium (in the case of iron) may change as the need for calcium or iron changes during various stages of life.

In the case of some essential elements such as potassium and chemically similar radioelements such as rubidium and cesium, however, absorption into the bloodstream is nearly complete at all ages, so that changes with age and possible homeostatic adaptations in absorption are not discernible. The fraction of a radioelement that is transferred to the blood depends on its

chemical form, and wide ranges of values are found in the literature for individuals who ingest the material under different working conditions. For example, f_1 values for uranium were found to range from 0.005 to 0.05 for industrial workers, but a higher value of 0.2 is indicated by dietary data from persons not occupationally exposed (ICRP79). EPA has used the 0.2 value for uranium ingestion by the general population. It appears that all iodine entering the small intestine is absorbed into the blood, and hence the f_1 value is taken as 1 for all ages, which is the value we will use for this analysis.

(c) Organ Masses, m

To a large extent, the variability in organ masses among individuals in the general population is related to age. For most of the target organs listed in Table 6-2, the mass increases during childhood and continues to increase until adulthood, at which time the net growth of the organ ceases; there may be a gradual decrease in mass (for some organs) in later years.

The associated uncertainty in estimating the dose to the thyroid of exposed individuals can be estimated by considering how the mass varies with age, as well as among individuals of the same age. Based on data reviewed by Dunning and Schwarz (Du81), the mass of an adult thyroid ranges from 2 to 62 g. As a result, the absorbed dose rate to the thyroid would vary by a factor of 31 just among adults. In comparing estimates for children and adults, children in the age group from .5 to 2 yr were found to have a mean thyroid mass of 2.1 g, while the adult group had a mean mass of 18.3 g. Based on these values, the absorbed dose rate to the thyroid of the average child and adult would differ by about a factor of 9. For this analysis, we have used the same values employed by the ICRP (20 g for the adult thyroid mass and 1.8 g for that of a 6-month-old child), which are also consistent with the recommendation of Bryant (Br69).

(d) Organ Uptake Fraction, f_2'

The fraction of a radionuclide taken up from the blood in an organ is strongly correlated with the size of the organ, its metabolic activity, and the amount of material ingested. Iodine introduced into the bloodstream is rapidly deposited in the thyroid, usually reaching a peak within 24 hours. The uptake of iodine-131 by the thyroid is similar to that of stable iodine in the diet (Do71), and can be influenced by sex and dietary differences. There can be considerable variation among populations.

Dunning and Schwarz (Du81) found a mean f_2' value of 0.47 for newborns, 0.39 for infants, 0.47 for adolescents, and 0.19 for adults. Other data (Pi69, Be70, Wo77) would suggest that a value of 0.10 to 0.20 may better represent adult populations in the United States. For purposes of this analysis, we have used f_2' values of .35 and .30 for a child and adult, respectively.

(e) Biological Half-Life, $T_{1/2}$

Data suggest that there is a strong correlation between biological half-lives of radionuclides in organs in the body and the age of the individual. Children are expected to exhibit smaller values of $T_{1/2}$ and greater uptakes (Ro58), and this relationship appears to be independent of the type of radionuclide ingested (Br85). For iodine-131, a range of 21 to 200 days for adults was observed and a similarly wide range would be expected for other age groups (Du81). Rosenberg (Ro58) found a significant correlation between the biological half-life and the age of the individual, and an inverse relationship between uptake and age in subjects from 22 to 50 yr of age. Dunning and Schwarz (Du81) concluded that for adults the observed range was from 21 to 372 days, implying for adults about an 18-fold variation in absorbed dose rate, other factors being held constant. For children in the age group from .5 to 2 yr, the range was 4 to 39 days, which would affect the absorbed dose rate estimate by about a factor of 10.

In light of the possible inverse relation between the biological half-life and the f_2' value, we will, for the purposes of this analysis, use biological half-lives of 24 and 129 days, respectively, for children and adults, based on the paper by Bryant (Br69).

(f) Effective Energy per Disintegration, E

The effective energy per disintegration of a radionuclide within an organ is dependent upon the decay energy of the radionuclide and the effective radius of the organ containing the radionuclide (ICRP59). It is expected, therefore, that E is an age-dependent parameter which could vary as the size of the organ changes. While very little work has been done in determining E values for the radionuclides found in low-level radioactive waste products, some information has been published for iodine-131 and cesium-137. Considering the differences between the child and the adult thyroid, Bryant (Br69) derives E values of 0.18 MeV/dis for the child and 0.19 MeV/disintegration for the adult. The above values correspond to a 6-month-old child with a mass of 1.8 g and an f_2' value of 0.35. The corresponding E value for the adult was calculated for a 20 g thyroid with an f_2' value of 0.3.

(3) Differences in Child and Adult Doses Associated with Age-Dependent Changes in Model Parameters

To examine the uncertainties in thyroid dose associated with changes in model parameters with age, values for child and adult parameters were chosen as discussed above and are listed in Tables 6-5 and 6-6.

Using Equation 25, the absorbed dose rate to the thyroid of a child D_c, can be compared to that of an adult D_a, by the following:

$$\dot{D}c/\dot{D}a = \frac{(0.7)(1)(0.35)(0.18)(20)(24)}{(0.5)(1)(0.30)(0.19)(1.8)(139)} = 2.96 \qquad (26)$$

Table 6-5. Model parameters for iodine metabolism in the thyroid of a child (Age 0.5 to 2 yr)

Parameters	Values	Reference
I	0.7/day x $C^{@}$	Br69
f_1	1	ICRP59
f_2^1	0.35	Br69
E_m	0.18 Mev/transformation	Br69
m	1.8 g	Br69
$T_{1/2}$	24 days	Br69

$C^{@}$ = iodine concentration in milk = 1 μCi/L

Table 6-6. Model parameters for iodine metabolism in the thyroid of an adult (Age > 18 yr)

Parameters	Values	Reference
I	0.5/day x $C^{@}$	Br69
f_1	1	ICRP59
f_2^1	0.30	Br69
E	0.19 Mev/transformation	Br69
m	20 g	Br69
$T_{1/2}$	139 days	Br69

$C^{@}$ = iodine concentration in milk = 1 μCi/L

Based on these parameters, therefore, the analysis indicates that, for a given concentration of I-131 in milk, the estimated absorbed dose rate to the thyroid of a 6-month-old child would be a factor of approximately 3 times that to the adult thyroid. In other words, use of adult parameters would underestimate the thyroid dose to the child by almost a factor of 3. This difference is expected to change with age with other radionuclides, however.

Depending on the type of radionuclide ingested, the age and element dependency in the metabolic and physiological processes determines how the dose to target organs varies with age. For example, strontium tends to follow the calcium pathways in the body and deposits to a large extent in the skeleton. In fact, the fraction of ingested strontium eventually reaching the skeleton at a given age depends largely on the skeletal needs for calcium at that age, although the body is able to discriminate somewhat against strontium in favor of calcium after the first few weeks of life.

In summary, it is difficult to make generalizations concerning the uncertainty involved in making calculations. More work is necessary to properly characterize the effect of age and individual dependent morphological and metabolic changes on dose.

6.3.2 Special Radionuclides

The following paragraphs briefly summarize some of the special considerations for particular elements and radionuclides.

(A) Tritium and Carbon-14

Most radionuclides are nuclides of elements found only in trace quantities in the body. Others like tritium (hydrogen-3) or carbon-14
must be treated differently since they are long-lived nuclides of elements that are ubiquitous in tissue. An intake of tritium is assumed to be completely absorbed and will rapidly mix with the water content of the body (Ki78a).

The estimates for inhalation include consideration of absorption through the skin. Organ dose estimates are based on the steady-state specific-activity model described by Killough et al. (Ki78a).

Carbon-14 is assumed to be inhaled as CO_2 or ingested in a biologically bound form. Inhaled carbon-14 is assumed to be diluted by stable carbon from ingestion (Ki78b). This approach allows separate consideration of the ingestion and inhalation pathways. The specific-activity model used for organ dose estimates is also that of Killough et al. (Ki78a). Short-lived carbon radionuclides (e.g., carbon-11 or carbon-15) are treated as trace elements and the organ doses are calculated accordingly.

(B) Noble Gases

The retention in the lung of noble gases uses the approach described by Dunning et al. (Du79). The inhaled gas is assumed to remain in the lungs until it is lost by radiological decay or respiratory exchange. Translocation of the noble gas to systemic organs is not considered, but is included for any decay products produced in the lungs. The inhalation of the short-lived decay products of radon is assessed using a potential alpha energy exposure model (see Chapter 7) rather than by calculating the doses to lung tissues from these radionuclides.

(C) Transuranics

The metabolic models for transuranic elements (Po, Np, Pu, Am, and Cm) are consistent with those used for the EPA transuranic guidance (EPA 77). Basically, a GI tract to blood absorption factor of 10^{-3} is used for the short-lived nuclides of plutonium (plutonium-239, -240, and -242), while a value of 10^{-4} is used for other transuranics. For soluble forms of uranium, a GI tract to blood absorption factor of 0.2 is used in accordance with the high levels of absorption observed for low-level environmental exposures by Hursh and Spoor (Hu73 and Sp73).

6.3.3 External Dose Models

This section is concerned with the calculation of dose rates for external exposure to photons from radionuclides dispersed in the environment. Two exposure models are discussed: (1) immersion in contaminated air and (2) irradiation from material deposited on the ground surface. The immersion source is considered to be a uniform semi-infinite radionuclide concentration in air, while the ground surface irradiation source is viewed as a uniform radionuclide concentration on an infinite plane. In both exposure modes, the dose rates to organs are calculated from the dose rate in air. For low-level waste assessments, ground surface irradiation is, almost without exception, more significant than air immersion.

Dose rates are calculated as the product of a dose rate factor which is specific for each radionuclide, tissue, and exposure mode and the corresponding air or surface concentration. The dose rate factors used in the low-level waste modeling assessments were calculated with the DOSE FACTOR code (Ko81). Note that the dose rate factors for each radionuclide do not include any contribution for decay products. For example, the ground surface dose factors for cesium-137 are all zero, since no photons are emitted in its decay. To assess surface deposition of cesium-137, one must first calculate the ingrowth of its decay product, metastable barium-137, which is a photon emitter.

(A) Immersion

For immersion exposure to the photons from radionuclides in air, EPA assumes that an individual is standing at the base of a semi-infinite cloud of uniform radionuclide concentration. We first calculate the dose rate factor (the dose rate for a unit concentration) in air for a source

of photons with energy E_γ. At all points in an infinite uniform source, conservation of energy considerations requires that the rates of absorbed and emitted energy per unit mass be equal. The absorbed energy rate per unit mass at the boundary of a semi-infinite cloud is just half that value. Hence

$$DRF_\gamma^a(E_\gamma) = \frac{1}{2} k\ E_\gamma/\rho \qquad (27)$$

where:

DRF_γ^a = the immersion dose rate per unit air concentration (rad m^3/Ci s);

E_γ = emitted photon energy (MeV);

k = units conversion factor

= 1.602 10^{-13} (J/MeV) x 3.7 10^{10} (1/Ci s) x 10^3 (g/kg) x 10^2 (rad kg/J)

= 5.93 10^2 (g rad/MeV Ci s); and

ρ = density of air (g/m^3).

The above equation presumes that for each nuclide transformation, one photon with energy E_γ is emitted. The dose rate factor for a nuclide is obtained by adding together the contributions from each photon associated with the transformation process for that radionuclide.

(B) Ground Surface Irradiation

In the case of air immersion, the radiation field was the same throughout the source region. This allows the dose rate factor to be calculated on the basis of energy conservation without having to explicitly consider the scattering processes taking place. For ground surface irradiation, the radiation field depends on the height of the receptor above the surface, and the dose rate factor calculation is more complicated. The radiation flux per unit solid angle is strongly dependent on the angle of incidence. It increases from the value for photons incident from immediately below the receptor to a maximum close to the horizon. Attenuation and buildup due to scattering must be considered to calculate the dose rate factor. Secondary scattering provides a distribution of photon energies at the receptor, which increases the radiation flux above that calculated on the basis of attenuation. Trabey (Tr66) has provided a useful and reasonably accurate expression to approximate this buildup:

$$B_{en}^a(\mu_a r) = 1 + C_a \mu_a r\ \exp(D_a \mu_a r) \qquad (28)$$

where B_{en}^a is the buildup factor (i.e., the quotient of the total energy flux and that calculated for attenuation) only for energy in air; μ_a is the attenuation coefficient at the energy of the released photon (m^{-1}); r is the distance between the photon source and the receptor;

and the Berger buildup coefficients C_a and D_a are dependent on energy and the scattering medium. The buildup factor is dimensionless and always has a value greater than unity. The resulting expression for the dose rate factor at a height z (m) above a uniform plane is

$$DRF_{\gamma}^{a}(z,E_{\gamma}) = \frac{1}{2}\, k(E_{\gamma}/\rho)(\mu_{en}/\rho)_a \left\{E_1(\mu_a z) + \frac{C_a}{1-D_a} \exp[-(1-D_a)\mu_a z]\right\} \qquad (29)$$

where $(\mu_{en}/\rho)_a$ is the mass energy-absorption coefficient (m^2/g) for air at photon energy E_γ (MeV); E_1 is the first order exponential integral function, i.e.,

$$E_1(x) = \int_x^{\infty} \frac{\exp(-u)}{u}\, du \qquad (30)$$

C_a and D_a are the buildup coefficients in air at energy E_γ; and $k=5.93\ 10^2$ (g rad/MeV Ci s) as for the immersion calculation.

As for immersion, the dose rate factor for a nuclide combines the contribution from each photon energy released in the transformation process.

(C) Organ Doses

The dose rate factors in the preceding two sections are for the absorbed dose in air. For a radiological assessment, the absorbed doses in specific tissues and organs are needed. For this purpose, Kerr and Eckerman (Ke80, Ke80a) have calculated organ dose factors for immersion in contaminated air. Their calculations are based on Monte Carlo simulations of the absorbed dose in each tissue or organ for the spectrum of scattered photons in air resulting from a uniform concentration of monoenergetic photon sources. Kocher (Ko81) has used these data to calculate values of the ratio of the organ dose factor to the air dose factor, $G^k(E_\gamma)$, for 24 organs and tissues at 15 values of E_γ ranging from 0.01 to 10.0 MeV.

The resulting organ-specific dose rate factor for immersion is

$$DRF_{\gamma}^{k}(E_{\gamma}) = G^{k}(E_{\gamma})\ DRF_{\gamma}^{a}(E_{\gamma}) \qquad (31)$$

For a specific nuclide, the dose rate factor is obtained by taking the sum of the contributions from each photon energy associated with the radionuclide decay.

Ideally, a separate set of $G^k(E_\gamma)$ values would be used for the angular and spectral distributions of incident photons from a uniform plane source. Since these data are not available, Kocher (Ko81) has used the same set of $G^k(E_\gamma)$ values for calculating organ dose rate factors for both types of exposure.

(D) Uncertainty Considerations in External Dose Rate Factors

In computing the immersion dose rate factor in air, the factor of 1/2 in Equation (27), which accounts for the semi-infinite geometry of the source region, does not provide a rigorously correct representation of the air ground interface. However, Dillman (Di74) has concluded that this result is within the accuracy of available calculations. The radiation field between the feet and the head of a person standing on contaminated ground is not uniform, but for source photon energies greater than about 10 keV, the variation about the value at 1 meter becomes minimal. A more significant source of error is the assumption of a uniform concentration. Kocher (Ko81) has shown that sources would have to be approximately uniform over distances of as much as a few hundred meters from the receptor for the dose rate factors to be accurate for either ground surface or immersion exposures. Penetration of deposited materials into the ground surface, surface roughness, terrain irregularities, as well as the shielding provided by buildings to their inhabitants, all serve to reduce doses.

The effect of using the same factors to relate organ doses to the dose in air for ground surface as for immersion photon sources has not been studied. The assumptions that the radiation field for the ground surface source is isotropic and has the same energy distribution as for immersion clearly do not hold true, but more precise estimates of these distributions are not likely to change the organ dose rate factors substantially.

Kocher (Ko81) has noted that the idealized photon dose rate factors are "likely to be used quite extensively even for exposure conditions for which they are not strictly applicable... because more realistic estimates are considerably more difficult and expensive [to make]."

6.4 Distribution of Doses in the General Population

Although the use of extreme parameter values in a sensitivity analysis indicates that large uncertainties in calculated doses are possible, this uncertainty is not usually reflected in the general population. There are several reasons for this: the parameter values chosen are intended to be typical of an individual in the population; it is improbable that the "worst case" parameters would be chosen for all terms in the equation; and not all of the terms are mutually independent, e.g., an increased intake may be offset by more rapid excretion.

This smaller range of uncertainty in real populations is demonstrated by studies performed on various human and animal populations. It should be noted that there is always some variability in observed doses that results primarily from differences in the characteristics of individuals. The usual way of specifying the dose, or activity, variability in an organ is in terms of the deviation from the average, or mean, value.

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Chapter 7: ESTIMATING THE RISK OF HEALTH EFFECTS RESULTING FROM EXPOSURE TO LOW LEVELS OF IONIZING RADIATION

7.1 Introduction

This chapter describes how EPA estimates the risk of fatal cancer, serious genetic effects, and other detrimental health effects caused by exposure to low levels of ionizing radiation.

Ionizing radiation refers to radiation that strips electrons from atoms in a medium through which it passes. The highly reactive electrons and ions created by this process in a living cell can produce, through a series of chemical reactions, permanent changes (mutations) in the cell's genetic material, the DNA. These may result in cell death or in an abnormally functioning cell. A mutation in a germ cell (sperm or ovum) may be transmitted to an offspring and be expressed as a genetic defect in that offspring or in an individual of a subsequent generation: such a defect is commonly referred to as a genetic effect. There is also strong evidence that the induction of a mutation by ionizing radiation in a non-germ (somatic) cell can serve as a step in the development of a cancer. Finally, mutational or other events, including possible cell killing, produced by ionizing radiation in rapidly growing and differentiating tissues of an embryo or fetus, can give rise to birth defects: these are referred to as teratological effects. At acute doses above about 25 rads, radiation induces other deleterious effects in man; however, for the low doses and dose rates of interest in this document only those three kinds of effects referred to above are thought to be of significance.

Most important from the standpoint of the total societal risk from exposures to low-level ionizing radiation are the risks of cancer and genetic mutations. Consistent with our current understanding of their origins in terms of DNA damage, these are believed to be stochastic effects; i.e., the probability (risk) of these effects increases with the absorbed dose of radiation, but the severity of the effects is independent of dose. For neither induction of cancer nor genetic effects, moreover, is there any convincing evidence for a "threshold," i.e., some dose level below which the risk is zero. Hence, so far as we know, any dose of ionizing radiation, no matter how small, might give rise to a cancer or to a genetic effect in future generations. Conversely, there is no way to be certain that a given dose of radiation, no matter how large, has caused an observed cancer or will cause one in the future.

In summary, knowledge of the radiation dose absorbed by an individual allows us to estimate the probability that the dose will result in a cancer or a genetic effect (or somewhat more precisely - to estimate the number of excess cancers and genetic effects resulting from the same dose to a large group of similar individuals).

Beginning nearly with the discovery of x rays in 1895 but especially since World War II, there has been an enormous amount of research conducted into the biological effects of ionizing radiation. This research continues at the level of the molecule, the cell, the tissue, the whole laboratory animal, and man. There are two fundamental aspects to most of this work:

1. Estimating the radiation dose to a target (cell, tissue, etc.). This aspect (dosimetry), which may involve consideration of physiological, metabolic, and other factors, is discussed more fully in Chapter 6.

2. Measuring the number of effects of a given type associated with a certain dose (or exposure).

For the purpose of assessing the risk to man from exposures to ionizing radiation, the most important information comes from human epidemiological studies in which the number of health effects observed in an irradiated population is compared to that in an unirradiated control population. The human epidemiological data regarding radiation-induced cancer are extensive. As a result, the risk can be estimated to within an order of magnitude with a high degree of confidence. Perhaps for only one other carcinogen - tobacco smoke - are we in a better position with regard to the reliability of risk estimates.

Nevertheless, there are serious gaps in the human data on radiation risks. No clear-cut evidence of excess genetic effects has been found in irradiated human populations, for example. Likewise, no statistically significant excess of cancers has been demonstrated below about 10 rads, the dose range of interest from the standpoint of environmental exposures. Since the epidemiological data are incomplete in many respects, risk assessors must rely on mathematical models to estimate the risk from exposures to low-level ionizing radiation. The choice of models, of necessity, involves subjective judgments, but should be based on all relevant sources of data collected by both laboratory scientists and epidemiologists. Thus, radiation risk assessment is a process that continues to evolve as new scientific information becomes available.

The EPA's estimates of cancer and genetic risks in this BID are based largely on the results of a National Academy of Sciences (NAS) study as given in the BEIR-3 report (NAS80). The study assessed radiation risks at low exposure levels. As phrased by the President of the Academy, "We believe that the report will be helpful to the EPA and other agencies as they reassess radiation protection standards. It provides the scientific bases upon which standards may be decided after nonscientific social values have been taken into account."

In this discussion, we outline the various assumptions made in calculating radiation risks based on the 1980 NAS report, and compare these risk estimates with those prepared by other scientific groups, such

as the 1972 NAS BEIR Committee (NAS72), the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR77, 82), and the ICRP (ICRP77). We recognize that information on radiation risks is incomplete and do not argue that any of the estimates derived by the 1980 NAS BEIR Committee on the basis of alternative assumptions is highly accurate. Rather, we discuss some of the deficiencies in the available data base and point out possible sources of bias in current risk estimates. Nevertheless, we believe the risk estimates made by EPA are reasonable in light of current evidence.

In the sections below, we first consider (Sections 7.2-7.2.8) the cancer risk resulting from whole-body exposure to low-LET (see Chapter 6) radiation, i.e., sparsely ionizing radiation like the energetic electrons produced by x rays or gamma rays. Environmental contamination by radioactive materials also leads to the ingestion or inhalation of the material and subsequent concentration of the radioactivity in selected body organs. Therefore, the cancer risk resulting from low-LET irradiation of specific organs is examined next (Sections 7.2.9-7.2.11). Organ doses can also result from high-LET radiation, such as that associated with alpha particles. The estimation of cancer risks for situations where high-LET radiation is distributed more or less uniformly within a body organ is the third situation considered (Section 7.3). Because densely ionizing alpha particles have a very short range in tissue, there are exposure situations where the dose distribution to particular organs is extremely nonuniform. An example is the case of inhaled radon progeny, polonium-218, lead-214, and polonium-214. For these radionuclides we base our cancer risk estimates on the amount of radon progeny inhaled rather than the estimated dose, which is highly nonuniform and cannot be well quantified. Therefore, risk estimates of radon exposure are examined separately (Section 7.4). In Section 7.5, we review the causes of uncertainty in the cancer risk estimates and the magnitude of this uncertainty so that both the public and EPA decision makers have a proper understanding of the degree of confidence to place in them. In Section 7.6, we review and quantify the risk of deleterious genetic effects from radiation and the effects of exposure in utero on the developing fetus. Finally, in Section 7.7, we calculate cancer and genetic risks from background radiation using the models described in this chapter.

7.2 Cancer Risk Estimates for Low-LET Radiations

Most of the observations of radiation-induced carcinogenesis in humans are on groups exposed to low-LET radiations. These groups include the Japanese A-bomb survivors and medical patients treated with diagnostic or therapeutic radiation, most notably for ankylosing spondylitis in England from 1935 to 1954 (Sm78). Comprehensive reviews of these and other data on the carcinogenic effects of human exposures are available (UNSCEAR77, NAS80).

The most important source of epidemiological data on radiogenic cancer is the population of Japanese A-bomb survivors. The A-bomb survivors have been studied for more than 38 years and most of them, the Life Span Study Sample, have been followed since 1950 in a carefully planned and monitored epidemiological survey (Ka82, Wa83). They are the largest group that has been studied, and they provide the most detailed information on the response pattern for organs by age and sex over a wide range of doses of low-LET radiation. Unfortunately, the doses received by various individuals in the Life Span Study Sample are not yet known accurately. The 1980 BEIR Committee's analysis of the A-bomb survivor data collected up to 1974 was prepared before bias in the dose estimates for the A-bomb survivors (the tentative 1965 dose estimates, T65) became widely recognized (Lo81). It is now clear that the T65 dose equivalents to organs tended, on average, to be overestimated (Bo82, RERF83,84) so that the BEIR Committee's estimates of the risk per unit dose are likely to be too low. A detailed reevaluation of current risk estimates is indicated when the A-bomb survivor data have been reanalyzed on the basis of new and better estimates of the dose to individual survivors. These estimates should become available during 1988.

Uncertainties in radiation risk estimates do not result just from the uncertainties about the Japanese and other epidemiological studies. As discussed below, risk projections based on these studies require certain assumptions (e.g., with regard to low dose extrapolation). The degree of uncertainty associated with these assumptions is probably greater than the uncertainty of the estimated risk per unit dose among the A-bomb survivors or other sources of risk estimates for radiogenic cancer in humans.

7.2.1 Assumptions Needed to Make Risk Estimates

A number of assumptions must be made on how to extrapolate observations made at high doses to estimate effects from low doses and low dose rates. Excess cancers have been observed, for the most part, only following doses of ionizing radiation that are relatively high when compared to those likely to occur as a result of the combination of background radiation and environmental contamination from controllable sources of radiation. Therefore, a dose response model must be chosen to allow extrapolation from the number of radiogenic cancers observed at high doses to the number of cancers at low doses resulting from all causes including background radiation.

The range of extrapolation is not the same for all kinds of cancer because it depends upon the radiosensitivity of a particular tissue. For example, the most probable radiogenic cancer for women is breast cancer. As described below, the incidence of radiogenic breast cancer does not seem to diminish when the dose is protracted over a long period of time. For example, the number of excess cancers per unit dose among Japanese women, who received acute doses, is about the same per unit dose as women exposed to small periodic doses of x rays over many years. If this is

actually the case, background radiation is as carcinogenic per unit dose for breast tissue as the acute exposures from A-bomb gamma radiation. Moreover, the female A-bomb survivors show an excess of breast cancer at doses below 20 rads which is linearly proportional to that observed at several hundred rads (To84). [Evidence of a nonlinear dose response relationship for induction of breast cancer has been obtained in a study of Canadian fluoroscopy patients, but only at doses above about 500 rads (Ho84).] Women in their 40's, the youngest age group in which breast cancer is common, have received about 4 rads of whole-body low-LET background radiation and usually some additional dose incurred for diagnostic medical purposes. Therefore, for this cancer, the difference between the lowest dose at which radiogenic cancers are observed, less than 20 rads, and the dose resulting from background radiation is less than a factor of 5, not several orders of magnitude as is sometimes claimed. Based on data from irradiated tinea capitis patients, induction of thyroid cancer also seems to be linear with doses down to 10 rads or lower (NCRP85). However, for most other cancers, a statistically significant excess has not been observed at doses below 50 rads of low-LET radiation. Therefore, the range of dose extrapolation is often large.

7.2.2 Dose Response Functions

The 1980 NAS report (NAS80) examined only three dose response functions in detail: (1) linear, in which the number of effects (risk) is directly proportional to dose at all doses; (2) linear-quadratic, in which risk is very nearly proportional to dose at very low doses and proportional to the square of the dose at high doses; and (3) a quadratic dose response function, where the risk varies as the square of the dose at all dose levels.

We believe the first two of these functions are compatible with most of the data on human cancer. Information which became available only after the BEIR-3 report was published indicates that a quadratic response function is inconsistent with the observed excess risk of solid cancers at Nagasaki, where the estimated gamma-ray doses are not seriously confounded by an assumed neutron dose component. The chance that a quadratic response function underlies the excess cancer observed in the Nagasaki incidence data has been reported as only 1 in 10,000 (Wa83). Although a quadratic response function is not incompatible with the Life Span Study Sample data on leukemia incidence at Nagasaki, Beebe and others (Be78, E177) have pointed out how unrepresentative these data are of the total observed dose response for leukemia in that city. There is no evidence that a quadratic response function provides a better fit to the observed leukemia excess among all A-bomb survivors in the Life Span Study Sample than a simple linear model (NAS80). Based on these considerations, we do not believe a quadratic response can be used in a serious effort to estimate cancer risks due to ionizing radiation.

The 1980 NAS BEIR Committee considered only the Japanese mortality data in their analysis of possible dose response functions (NAS80). Based on the T65 dose estimates, this Committee concluded that the excess mortality from solid cancers and leukemia among the A-bomb survivors is compatible with either a linear or linear-quadratic dose response to the low-LET radiation component and a linear response to the high-LET neutron component (NAS80). Although the 1980 BEIR report indicated risk estimates for low-LET radiation based on a linear-quadratic response were "preferred" by most of the scientists who prepared that report, opinion was not unanimous, and we believe the subsequent reassessment of the A-bomb dose seriously weakens the Committee's conclusion. The Committee's analysis of dose response functions was based on the assumption that most of the observed excess leukemia and solid cancers among survivors in Hiroshima resulted from neutrons (see Tables V-13, A-7, Equations V-10, V-11 in NAS80). Current evidence, however, is conclusive that neutrons were only a minor component of the dose among all but a few survivors in both Hiroshima and Nagasaki (Bo82, RERF83,84). Therefore, it is likely that most of the response attributed to neutrons was caused by the gamma dose, not the dose from neutrons. This point is discussed further in Section 7.3.

The revised dosimetry will involve changes in individual absorbed doses that vary with distance from the explosion in each of the two cities and with shielding characteristics. As a consequence, though it seems clear that there will generally be a higher response per unit dose, there will also be an unpredictable change in the shape of the dose response exhibited by the data. Reanalysis of the Japanese experience after completion of the dose reassessment may then provide more definitive information on the dose response of the A-bomb survivors; nevertheless, it is unlikely to produce a consensus on the dose response at environmental levels, i.e., about 100 mrad/yr. This is because at low enough doses there will always be sampling variations in the observed risks so that observations are compatible, in a statistical sense, with a variety of dose response functions. In the absence of empirical evidence or a strong theoretical basis, a choice between dose response functions must be based on other considerations.

Although there is evidence for a nonlinear response to low-LET radiations in some, but not all, studies of animal radiocarcinogenesis (see below), we are not aware of any data on human cancers that are incompatible with the linear model. In such a case, it may be preferable to adopt the simplest hypothesis that adequately models the observed radiation effect. Moreover, EPA believes that risk estimates, for the purpose of assessing radiation impacts on public health, should be based on scientifically credible risk models that are unlikely to understate the risk. The linear model fulfills this criterion. Given the current bias in the doses assigned to A-bomb survivors (see Section 7.5.1 below), such an approach seems reasonable as well as prudent. Therefore, EPA has primarily used the BEIR-3 linear dose response model for estimating the risk of radiogenic cancer due to low-LET radiations.

For low-LET radiations, the BEIR-3 Committee preferred the linear-quadratic dose response model. In this model, the risk from an acute dose, D, of low-LET radiation is assumed to be of the form $\alpha D + D^2$. The BEIR-3 Committee assumed that the linear and quadratic terms were equal at 116 rads, leading to a linear coefficient α, which was a factor of 2.5 times lower than the coefficient obtained from the linear model (NAS80). At low doses the quadratic term becomes negligible; at chronic low-dose rates it is ignored, for reasons discussed below. For environmental exposures, therefore, risk estimates based on the BEIR-3 linear-quadratic dose response model are only 40 percent of those based on the BEIR-3 linear model.

A theoretical basis for the linear-quadratic dose response model has been put forth by Kellerer and Rossi (Ke72). In this theory of "dual radiation action," events leading to "lesions" (i.e., permanent changes) in cellular DNA require the formation of interacting pairs of "sublesions." The interacting pairs can be produced by a single traversing particle, or track, or by two separate tracks, giving rise, respectively, to a linear and quadratic term in the dose response relationship. According to the theory, a sublesion may be repaired before it can interact to form a lesion, the probability of such repair increasing with time. Consequently, as dose rate is reduced, the formation of lesions from sublesions caused by separate tracks becomes less important, and the magnitude of the D^2 term diminishes. Hence, the theory predicts that at sufficiently low doses or dose rates the response should be a linear function of dose. Moreover, the constant of proportionality is the same in both cases, i.e., α.

Results of many animal experiments are qualitatively consistent with the theory: low-LET radiation often seems to have a reduced effectiveness per unit dose at low dose rates (NCRP80); however, it is usually not possible from the data to verify that the dose response curve has the linear-quadratic form. Another success of the dual action theory has been in explaining observed differences between the effects of low-LET and high-LET radiations. In this view, the densely ionizing nature of the latter results in a much greater production of interacting pairs of sublesions by single tracks, leading in turn to higher relative biological effectiveness at low doses and a linear dose response relationship for high-LET radiation (except for possible cell-killing effects).

The dual action theory has nevertheless been challenged on experimental grounds, and observed variations in response with dose, dose rate (see below), and LET can also be explained in terms of a theory involving only single lesions and a "saturable" repair mechanism that decreases in effectiveness at high dose rates on the microscopic scale (Go82). One property of such a theory is that the effectiveness of repair, and therefore the shape of the dose response curve, can in principle vary substantially with cell type and species. Hence, results obtained on laboratory animals would not necessarily be entirely applicable to people.

Finally, some mention should be made of "supralinear models" in which the risk coefficient decreases with increasing dose (downward bending, or convex, dose response curve). Such models imply that the risk at low doses would actually be greater than predicted by linear interpolation from higher doses.

The evidence from radiation biology investigations, at the cellular as well as the whole animal level, indicates that the dose response curve for induction of mutations or cancer by low-LET radiation is either linear or concave upward for doses to mammalian systems below about 250 rads (NCRP80). Somewhere above this point the dose response curve often begins to bend over: this is commonly attributed to "cell-killing." Analysis of the A-bomb survivor data, upon which most of our risk estimates depend, is dominated by individuals receiving about 250 rads or less. Consequently, the cell-killing phenomenon should not produce a substantial underestimate of the risk at low doses.

Noting that human beings, in contrast to pure strains of laboratory animals, may be highly heterogeneous with respect to radiation sensitivity, Baum (Ba73) proposed an alternative mechanism by which a convex dose response relationship could arise. He pointed out that sensitive subgroups may exist in the population who are at very high risk from radiation. The result could be a steep upward slope in the response at low doses, predominantly reflecting the elevated risk to members of these subgroups, but a decreasing slope at higher doses as the risk to these highly sensitive individuals approaches unity.

Based on current evidence, however, it seems unlikely that the effect postulated by Baum would lead to substantial overestimation of the risk at low doses. While there may indeed be small subgroups at very high risk, it is difficult to reconcile the A-bomb survivor data with a strongly convex dose response relationship. For example, if most of the leukemias found among the cohort receiving about 200 rads or more in fact arose from subgroups whose risk saturated below 200 rads, then many more leukemias ought to have occurred in lower dose cohorts than were actually observed (Ro78). The U.S. population, it could be argued, may be more heterogeneous with respect to radiation sensitivity than the Japanese. The risk of radiation-induced breast cancer appears, however, to be similar in the two populations, so it is difficult to see how the size of

the hypothetical sensitive group could be large enough in the former to alter the conclusion reached above. The linear dose-response relationship seen for radiogenic breast cancer in several populations (NIH85) further argues against Baum's hypothesis.

7.2.3 The Possible Effects of Dose Rate on Radiocarcinogenesis

The BEIR-3 Committee limited its risk estimates to a minimum dose rate of 1 rad per year and stated that it "does not know if dose rates of gamma rays and x rays of about 100 mrad/yr are detrimental to man." At dose rates comparable to the background everyone receives from naturally-occurring radioactive materials, a considerable body of scientific opinion holds that the effects of radiation are reduced compared to high dose rates. NCRP Committee 40 has suggested that carcinogenic effects of low-LET radiations may be a factor of from 2 to 10 times less per unit dose for small doses and dose rates than have been observed at high doses and dose rates (NCRP80).

The low dose and low dose rate effectiveness factors estimated by NCRP Committee 40 are based on their analysis of a large body of plant and animal data that showed reduced effects at low doses for a number of biological endpoints, including radiogenic cancer in animals, chiefly rodents. However, no data for cancer in humans confirm these findings as yet; indeed, a few human studies seem to contradict them. Highly fractionated small doses to human breast tissue are apparently as carcinogenic as large acute doses (NAS80, La80). Furthermore, small acute (less than 10 rads) doses to the thyroid have been found to be as effective per rad as much larger doses in initiating thyroid cancer (UNSCEAR77, NAS80). Moreover, the increased breast cancer resulting from chronic, low dose, occupational, gamma ray exposures among British dial painters is comparable to, or larger than, that expected on the basis of acute, high dose exposures (Ba81).

While none of these examples is persuasive by itself, collectively they indicate that it may not be prudent to assume that all kinds of cancers are reduced at low dose rates and/or low doses. However, it may be overly conservative* to estimate the risk of all cancers on the basis of the linearity observed for breast and thyroid cancer. The ICRP and UNSCEAR have used a dose rate effectiveness factor (DREF) of about 2.5 to estimate the risks from occupational (ICRP77) and environmental exposures (UNSCEAR77). That choice of a DREF is fully consistent with and equivalent to the reduction of risk at low doses obtained by substituting the BEIR-3 linear-quadratic response model for their linear model (see above). Therefore, use of both a DREF and a linear-quadratic model for risk estimation in the low-dose region is inappropriate (NCRP80).

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* In carrying out risk assessments, one is often forced to choose among alternative assumptions, none of which can be definitively shown to be more accurate than the others. A conservative choice, in this connection, is one leading to higher estimates of risk.

7.2.4 Risk Projection Models

None of the exposed populations have been observed long enough to assess the full effects of their exposures if, as currently thought, most radiogenic cancers occur throughout an exposed person's lifetime (NAS80). Therefore, another major choice that must be made in assessing the lifetime cancer risk due to radiation is to select a risk projection model to estimate the risk for a longer period of time than currently available observational data will allow.

To estimate the risk of radiation exposure that is beyond the years of observation, either a relative risk or an absolute risk projection model (or suitable variations) may be used. These models are described at length in Chapter 4 of the 1980 NAS report (NAS80). The relative risk projection model projects the currently observed percentage increase in annual cancer risk per unit dose into future years, i.e., the increase is proportional to the underlying (baseline) risk. An absolute risk model projects the average annual number of excess cancers per unit dose into future years at risk, independent of the baseline risk.

Because the underlying risk of most types of cancer increases rapidly with age, the relative risk model predicts a larger probability of excess cancer toward the end of a person's lifetime. In contrast, the absolute risk model predicts a constant incidence of excess cancer across time. Therefore, given the incomplete data we have now, less than lifetime follow-up, a relative risk model projects a somewhat greater total lifetime cancer risk than that estimated using an absolute risk model.

Neither the NAS BEIR Committee nor other scientific groups (e.g., UNSCEAR) have concluded which projection model is the appropriate choice for most radiogenic cancers. However, recent evidence favors the relative risk projection model for most solid cancers. As pointed out by the 1980 NAS BEIR Committee:

> "If the relative-risk model applies, then the age of the exposed groups, both at the time of exposure and as they move through life, becomes very important. There is now considerable evidence in nearly all the adult human populations studied that persons irradiated at higher ages have, in general, a greater excess risk of cancer than those irradiated at lower ages, or at least they develop cancer sooner. Furthermore, if they are irradiated at a particular age, the excess risk tends to rise

pari passu (at equal pace) with the risk of the population at large. In other words, the relative-risk model with respect to cancer susceptibility at least as a function of age, evidently applies to some kinds of cancer that have been observed to result from radiation exposure." (NAS80, p.33)

This observation is confirmed by the Ninth A-bomb Survivor Life Span Study, published two years after the 1980 Academy report. This latest report indicates that, for solid cancers, relative risks have continued to remain constant in recent years, while absolute risks have increased substantially (Ka82). Smith and Doll (Sm78) reached similar conclusions on the trend in excess cancer with time among the irradiated spondylitic patients.

Although we believe considerable weight should be given to the relative risk model for most solid cancers (see below), the model does not necessarily give an accurate projection of lifetime risk. The mix of tumor types varies with age so that the relative frequency of some common radiogenic tumors, such as thyroid cancer, decreases for older ages. Land has pointed out that this may result in overestimates of the lifetime risk when they are based on a projection model using relative risks (La83). While this may turn out to be true for estimates of cancer incidence that include cancers less likely to be fatal, e.g., thyroid, it may not be too important in estimating the lifetime risk of fatal cancers, since the incidence of most of the common fatal cancers, e.g., breast and lung cancers, increases with age.

Leukemia and bone cancer are exceptions to the general validity of a lifetime expression period for radiogenic cancers. Most, if not all, of the leukemia risk has apparently already been expressed in both the A-bomb survivors and the spondylitics (Ka82, Sm78). Similarly, bone sarcoma from acute exposure appears to have a limited expression period (NAS80, Ma83). For these diseases, the BEIR-3 Committee believed that an absolute risk projection model with a limited expression period is adequate for estimating lifetime risk (NAS80).

Note that, unlike the NAS BEIR-1 report (NAS72), the BEIR-3 Committee's relative and absolute risk models are age dependent; that is, the risk coefficient changes, depending on the age of the exposed persons. Observational data on how cancer risk resulting from radiation changes with age are sparse, particularly so in the case of childhood exposures. Nevertheless, the explicit consideration of the variation in radiosensitivity with age at exposure is a significant improvement in methodology. It is important to differentiate between age sensitivity at exposure and the age dependence of cancer expression. In general, people seem to be most sensitive to radiation when they are young. In contrast most radiogenic cancers seem to occur late in life, much like cancers resulting from other causes. In this chapter we present lifetime cancer risk estimates for a lifetime exposure of equal annual doses. However, it is important to note that the calculated lifetime risk of developing a

fatal cancer from a single year of exposure varies with the age of the recipient at the time of exposure.

7.2.5 Effect of Various Assumptions on the Numerical Risk Estimates

Differences between risk estimates made by using various combinations of the assumptions described above were examined in the 1980 NAS report. Table 7-1 below, taken from Table V-25 (NAS80), shows the range of cancer fatalities that are induced by a single 10-rad dose as estimated using linear, linear-quadratic, and quadratic dose response functions and two projection models, relative and absolute risk (NAS80).

As illustrated in Table 7-1, estimating the cancer risk for a given projection model on the basis of a quadratic as compared to a linear dose response reduces the estimated risk of fatal cancer by a factor of about 18. Between the more credible linear and linear-quadratic response functions, the difference is less, a factor of about 2.2. For a given dose response model, results obtained with the two projection models for solid cancers, differ by about a factor of 3.

Even though the 1980 NAS analysis estimated lower risks for a linear-quadratic response at 10 rads, it should not be concluded that this response function always provides smaller risk estimates. In contrast to the 1980 NAS analysis where the proportion of risk due to the dose-squared term (e.g., C_3 in Footnote c of Table 7-1) was constrained to positive values, the linear-quadratic function that agrees best with Nagasaki cancer incidence data has a negative coefficient for the dose-squared term (Wa83). Although this negative coefficient is small and indeed may not be significant, the computational result is a larger linear term, which leads to higher risk estimates at low doses than would be estimated using a simple linear model (Wa83).

Differences in the estimated cancer risk introduced by the choice of the risk projection model are also appreciable. As pointed out above, the 1980 NAS analysis indicates that relative lifetime risk estimates exceed absolute risk estimates by about a factor of 3 (see Table 7-1). However, relative risk estimates are quite sensitive to how the risk resulting from exposure during childhood persists throughout life. This question is addressed in the next section, where we compare risk estimates made by the 1972 and 1980 NAS BEIR Committees with those of the ICRP and UNSCEAR.

7.2.6 Comparison of Cancer Risk Estimates for Low-LET Radiation

A number of estimates of the risk of fatal cancer following lifetime exposure are compared in Table 7-2. The BEIR-1 and BEIR-3 values were calculated for this table using risk model data from NAS72 and NAS80. The BEIR-3 values in this table differ slightly from those in NAS80 and Table 7-1 because of some minor calculational corrections including revised age-specific mortality data. The risk estimates in this table

Table 7-1. Range of cancer fatalities induced by a single 10-rad, low-LET radiation exposure to the general population (Average value per rad per million persons exposed)

Dose response functions	Lifetime risk projection model	
	Relative[a]	Absolute
Linear[b]	501	167
Linear Quadratic[c]	226	77
Quadratic[d]	28	10

[a] Relative risk projection for all solid cancers except leukemia and bone cancer fatalities, which are projected by means of the absolute risk model (NAS80).

[b] Response R varies as a constant times the dose, i.e.:

$R=C_1D.$ $(L,\overline{L})$

[c] $R=C_2D+C_3D^2$ $(LQ,\overline{LQ})$ See text for model notation.

[d] $R=C_4D^2$ $(Q,\overline{Q})$

Source: NAS80, Table V-25.

Table 7-2. A comparison of estimates of the risk of fatal cancer from low-LET radiation

Source of estimate	Reference	Fatalities per 10^6 person-rad	Projection model
BEIR-1	NAS72	118	Absolute[a]
BEIR-1	NAS80	622	Relative[a]
BEIR-3	NAS80	168	Absolute[a,b]
BEIR-3	NAS80	395	Relative[a,b,c]
BEIR-3	NAS80	71	Absolute[a,d]
BEIR-3	NAS80	163	Relative[a,c,d]
UNSCEAR	UNSCEAR77	200-300	None[e] - high dose ($\geq$100 rad)
UNSCEAR	UNSCEAR77	75-175	None[e] - low dose, dose rate
CLM	Ch83	100-400	None - UNSCEAR77 without A-bomb data
ICRP	ICRP77	125	None - Occupational low dose, low dose rate

[a] Lifetime projection for constant dose rate calculated for 1970 U.S. general population lifetable and mortality rates; see text.

[b] Linear model (L-L for leukemia and bone, $\overline{\text{L-L}}$ for all other sites; notation explained in text).

[c] Leukemia and bone are calculated with absolute risk model.

[d] Linear-Quadratic model (LQ-L for leukemia and bone, $\overline{\text{LQ-L}}$ for all other sites; notation explained in text).

[e] Taken from paragraphs 317 and 318 in UNSCEAR77.

are based on different assumptions regarding the extrapolation to low doses and dose rates; they also differ considerably because of other assumptions. In contrast with absolute risk estimates, which have increased since the 1972 NAS BEIR-1 Committee report (NAS72), the 1980 NAS BEIR-3 Committee's estimates of the relative risk, as shown in Table 7-2, have decreased relative to those in the BEIR-1 report. This illustrates the sensitivity of risk projections to changes in modeling assumptions. For the NAS80 report, the relative risk coefficient determined for ages 10 to 19 was substituted for the considerably higher relative risk coefficient that would be calculated for those exposed during childhood, ages 0 to 9. In addition, the relative risk coefficients used in the BEIR-3 analysis are based on matching excess cancer for a 30-year follow-up of Japanese A-bomb survivors with 1970 U.S. lifetime and cancer mortality rates. In the 1972 NAS report this excess was compared to cancer mortality in Japan.

By comparing the three relative risk estimates from Table 7-2, it is apparent that the relative risk estimates are fairly sensitive to the assumptions made as to what extent the observed high relative risk of cancer from childhood exposure continues throughout adult life. The Life Span Study (Ka82) indicates that the high-risk adult cancer caused by childhood exposures is continuing, although perhaps not to the extent predicted by the NAS BEIR-1 Committee in 1972.

The major reason the risk estimates in Table 7-2 differ is because of the underlying assumption in each set of risk estimates. The NAS BEIR estimates are for lifetime exposure and lifetime expression of induced cancers (NAS72, NAS80). Neither the age distribution of the population at risk nor the projection models (if any) have been specified by either the UNSCEAR (UNSCEAR77) or the ICRP (ICRP77). UNSCEAR apparently presumes the same age distributions as had occurred in the epidemiological studies they cited, mainly the A-bomb survivors, and a 40-year period of cancer expression. The ICRP risk estimates are for adult workers, presumably exposed between ages 18 and 65, and a similar expression period. These are essentially age-independent absolute risk models with less than lifetime expression of induced cancer mortality. For these reasons, risks estimated by ICRP and UNSCEAR are expected to be smaller than those made on the basis of a lifetime relative risk model in the BEIR-3 report.

The next to the last entry in Table 7-2 (Ch83) is of interest because it specifically excludes the A-bomb survivor data based on T65 dose estimates. The authors reanalyzed the information on radiogenic cancer in UNSCEAR77 so as to exclude all data based on the Japanese experience. Their estimate of fatalities ranges from 100 to 440 per 10^6 person-rad based on data from exposure at high doses and dose rates. As indicated in Table 7-2, this is somewhat greater but comparable to the UNSCEAR estimate, which includes the A-bomb survivor data. The upper bound estimate for the number of fatalities given in Ch83 is 400 per 10^6 person-rem, which is nearly identical to the value

EPA has used in this report for a linear dose response model--395 fatalities per 10^6 person-rad (see below).

7.2.7 EPA Assumptions About Cancer Risks Resulting from Low-LET Radiations

The EPA estimates of radiation risks, presented in Section 7.2.8 below, are based on a presumed linear dose response function. We believe, however, that the linear-quadratic model is also credible. Using the BEIR-3 linear-quadratic model is equivalent to using a dose rate effectiveness factor of 2.5; thus, at low doses, it would project 2.5 times lower risk than the linear model.

Except for leukemia and bone cancer, where we use a 25-year expression period for radiogenic cancer, we use a lifetime expression period, as was done in the NAS report (NAS80). Because the most recent Life Span Study Report (Ka82) indicates absolute risks for solid cancers are continuing to increase 33 years after exposure, the 1980 NAS Committee choice of a lifetime expression period appears to be well founded. We do not believe limiting cancer expression to 40 years (as has been done by the ICRP and UNSCEAR) is compatible with the continuing increase in solid cancers that has occurred among irradiated populations (Ka82).

To project the number of fatalities resulting from leukemia and bone cancer, EPA uses an absolute risk model, a minimum induction period of 2 years, and a 25-year expression period. To estimate the number of fatalities resulting from other cancers, EPA has used the arithmetic average of absolute and relative risk projection models (EPA84). For these cancers, we assume a 10-year minimum induction period and expression of radiation-induced cancer for the balance of an exposed person's lifetime after the minimum induction period.

7.2.8 Methodology for Assessing the Risk of Radiogenic Cancer

EPA uses a life table analysis to estimate the number of fatal radiogenic cancers in an exposed population of 100,000 persons. This analysis considers not only death due to radiogenic cancer, but also the probabilities of other competing causes of death which are, of course, much larger and vary considerably with age (Bu81, Co78). Basically, it calculates for ages 0 to 110 the risk of death due to all causes by applying the 1970 mortality data from the National Center for Health Statistics (NCHS75) to a cohort of 100,000 persons. Additional information on the details of the life table analysis is provided in Appendix B. It should be noted that a life table analysis is required to use the age-dependent risk coefficients in the BEIR-3 report. For relative risk estimates, we have used age-specific cancer mortality data also provided by NCHS (NCHS73). The EPA computer program we use for the life table analysis was furnished to the NAS BEIR-3 Committee by EPA and

used by the Committee to prepare its risk estimates. Therefore, the population base and calculations should be essentially the same in both the NAS and EPA analyses.

We have considered both absolute and relative risk models to project the observed risks of most solid radiogenic cancers beyond the period of current observation. As indicated in Table 7-2, the range of estimated fatal cancers resulting from the choice of a particular projection model and its internal assumptions is about a factor of 3. Although the relative risk model has only been tested in some detail for lung and breast cancer (La78), based on current evidence, it appears to be the better projection model for solid cancers. We have, therefore, adopted it for our risk estimates in this report. Previously, we have used an average of the risks calculated by the absolute and relative risk projection models (EPA84).

To estimate the cancer risk from low-LET, whole-body, lifetime exposure, we use the relative risk projections (the BEIR-3 $\overline{\text{L-L}}$ model) for solid cancers and the absolute risk projection for leukemia and bone cancer (the BEIR-3 L-L model). Since the expression period for leukemia and bone cancer is less than the follow-up period, the same risk values would be calculated for these cancers using either projection method. For a dose to the whole body, this procedure yields about 400 fatalities per million person-rad. For the BEIR-3 linear-quadratic model, which is equivalent to applying a DREF of about 2.5 to the linear model, a low-LET whole-body dose yields an estimated lifetime risk of about 160 fatalities per million person-rad.

BEIR-3 also presented estimates of excess soft tissue cancer incidence for specific sites, as a function of age at exposure, in their Table V-14. By summing the site-specific risks, they then arrived at an estimate for the whole-body risk of cancer incidence (other than leukemia and bone cancer) as given in Table V-30. Finally, by using the weighted incidence/mortality ratios given in Table V-15 of the same report (NAS80), the results in Table V-30 can be expressed in terms of mortality to yield (for lifetime exposure) a risk estimate of about 242 and 776 cancer fatalities per 10^6 person-rad, depending on whether an absolute or a relative risk projection model, respectively, is used to estimate lifetime risk. These values are about 1.6 and 2.1 times their counterparts for the BEIR-3 $\overline{\text{L-L}}$ model and 3.9 and 9.1 times the $\overline{\text{LQ-L}}$ values.

These models all presume a uniform dose to all tissues at risk in the body. In practice, such uniform whole-body exposures seldom occur, particularly for ingested or inhaled radioactivity. The next section describes how we apportion this risk estimate for whole-body exposure when considering the risks following the exposure of specific organs.

7.2.9 Organ Risks

For most sources of environmental contamination, inhalation and ingestion of radioactivity are more common than external exposure. In

many cases, depending on the chemical and physical characteristics of the radioactive material, inhalation and ingestion result in a nonuniform distribution of radioactive materials within the body so that some organ systems receive much higher doses than others. For example, since iodine isotopes concentrate preferentially in the thyroid gland, the dose to this organ can be orders of magnitude larger than the average dose to the body.

Fatal Cancer at Specific Sites

To determine the probability that fatal cancer occurs at a particular site, we have performed life table analyses for each cancer type using the information on cancer incidence and mortality in NAS80. For cancer other than leukemia and bone cancer, we have used NAS80 Table V-14 (Age Weighted Cancer Incidence by Site Excluding Leukemia and Bone Cancer) and NAS80 Table V-15*, which lists the BEIR Committee's estimates of the ratio of cancer fatality to cancer incidence for these various sites, to calculate a set of site-specific mortality risk coefficients. The excess mortality for the $\overline{L\text{-}L}$ model was presumed to be distributed similarly. The proportions of leukemia and fatal bone cancer caused by low-LET radiation were estimated using the results of the models given in Table V-17** of NAS80. Normalized results, which give the proportion of fatal radiogenic cancers resulting from uniform whole-body irradiation, by cancer site, are listed in Table 7-3. These proportions were calculated for the average of the absolute and relative risk projections as in EPA84. Since it was not practicable to reanalyze all the scenarios considered for this report, we have adjusted the original risk estimate by the factor 395/280 to approximate the effects of using a relative risk projection model for solid cancers. As noted above, these proportions are assumed to be the same for the BEIR-3 linear-quadratic dose response model.

Information on the distribution of fatal, radiogenic cancers by organ is not precise. One reason is that the data in NAS80 (and Table 7-3) are based on whole-body exposures, and it is possible that the incidence of radiogenic cancer varies depending on the number of exposed organs. Except for breast and thyroid cancer, very little information is available on radiogenic cancer resulting from exposure of only one region in the body.

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* The mortality to incidence ratio for thyroid (male: 0.18, female: 0.20) in NAS80 is high compared to other references, e.g., NCRP80 uses a mortality to incidence ratio for thyroid of 0.1 for both males and females.

** The low-LET risk rate coefficient for bone has been changed to 0.125×10^{-6} sarcoma/yr per person-rad to be consistent with an alpha particle RBE of 8.

Table 7-3. Proportion of the total risk of fatal radiogenic cancer among different sites as given in EPA 84[e]

Site	Proportion of total risk	Fatalities per 10^6 person-rad[d]
Lung	0.207	58.2
Breast[a]	0.130	36.4
Red bone marrow[b]	0.150	42.1
Thyroid	0.099	27.7
Bone surface	0.009	2.4
Liver	0.085	23.9
Stomach	0.084	23.6
Intestines	0.039	10.9
Pancreas	0.059	16.4
Kidneys and urinary tract	0.025	7.0
Other[c]	0.113	31.8
Total		280.4

[a] Average for both sexes.

[b] Leukemia.

[c] Total risk for all other organs, including the esophagus, lymphatic system, pharynx, larynx, salivary gland, and brain.

[d] Lifetime exposure and cancer expression. Total risk for all sites calculated using the L-L absolute risk model for leukemia and bone and the $\overline{\text{L-L}}$ model for the total of the remaining sites averaged for absolute and relative risk projections (EPA84). The risks for these remaining sites have been apportioned as for the site specific absolute risk model using NAS80 Tables V-14 and V-15.

[e] In projecting cancer deaths for this proposed rulemaking, the organ risks above have been scaled up by a factor of 395/280 (see text).

Another reason is that most epidemiological studies use mortality data from death certificates, which often provide questionable information on the site of the primary cancer. Moreover, when the existing data are subdivided into specific cancer sites, the number of cases becomes small, and sampling variability is increased. The net result of these factors is that numerical estimates of the total cancer risk are more reliable than those for most single sites.

The 1977 UNSCEAR Committee's estimated risks (UNSCEAR77) to different organs are shown in Table 7-4. For all of the organs, except the breast, a high and low estimate was made. This range varies by a factor of 2 or more for most organs (Table 7-4). Other site-specific estimates show a similar degree of uncertainty (Ka82), and it is clear that any system for allocating the risk of fatal cancer on an organ-specific basis is inexact. Table 7-5 compares proportional risks by the NAS BEIR-3 Committee, UNSCEAR, and the ICRP. ICRP Report 26 provides organ-specific weights for assessing combined genetic and cancer risks from occupational exposure (ICRP77). In Table 7-5, we have renormalized ICRP risks so that they pertain to cancer alone.

Considering that the cancer risk for a particular site is usually uncertain by a factor of 2 or more, as indicated by the range of UNSCEAR estimates in Table 7-4, we would not expect perfect agreement in apportionment of total body risks. Table 7-5, however, does indicate reasonable agreement among the three sets of estimates considered here.

The differences between the proportions of the total risk of fatal cancer shown in Table 7-5 are, for the most part, small in comparison to their uncertainty. We have used the BEIR-3 organ risks in preference to those made by other groups such as UNSCEAR or the ICRP for several reasons. BEIR estimates of organ risk are based on a projection of lifetime risk using age-specific risk coefficients, rather than just observations to date. Moreover, the 1980 BEIR Committee considered cancer incidence data as well as mortality data. This gives added confidence that the diagnostic basis for their estimates is correct. And, finally, because we apply these proportional organ risk estimates to the NAS80 cancer risk estimates for whole-body exposures, we believe it is consistent to use a single set of related risk estimates. The way we have used NAS80 to estimate mortality resulting from cancer at a particular site is outlined in the next section.

7.2.10 Thyroid Cancer from Iodine-131 and Iodine-129

Iodine-131 has been reported to be only one-tenth as effective as x rays or gamma rays in inducing thyroid cancer (NAS72, NCRP77, NCRP85). On this basis, EPA has employed a thyroid cancer risk coefficient for internal exposures to iodine-131 and iodine-129 which is one-tenth that used for gamma rays or beta radiations from other radionuclides.

Table 7-4. UNSCEAR estimates of cancer risks (fatalities per 10^6 person-rad) at specified sites

Site	Range	Average	Proportion of total risk
Lung	25-50	37.5	.245
Breast[a]	25	25.0	.164
Red bone marrow[b]	15-25	20.0	.131
Thyroid	5-15	10.0	.066
Bone	2-5	3.5	.023
Liver	10-15	12.5	.082
Stomach	10-15	12.5	.082
Intestines	14-23	17.5	.115
Pancreas	2-5	3.5	.023
Kidneys and urinary tract	2-5	3.5	.023
Other[c]	4-10	7.0	.046

[a] Average for both sexes.

[b] Leukemia.

[c] Includes esophagus and lymphatic tissues.

Source: Adapted from UNSCEAR77.

Table 7-5. Comparison of proportion* of the total risk of radiogenic cancer fatalities by body organ

Site/Source	EPA84[a,b]	UNSCEAR77	ICRP77[c]
Lung	0.21	0.25	0.16
Breast	0.13	0.16	0.20
Red Marrow	0.16	0.13	0.16
Thyroid	0.10	0.07	0.04
Bone Surface	0.01	0.02	0.04
Remainder	--	--	0.40[d]
Liver	0.08	0.08	---
Stomach	0.08	0.08	---
Intestine	0.04	0.12	---
Pancreas	0.06	0.02	---
Urinary	0.02	0.02	---
Other[e]	0.11	0.05	---

* Values rounded to 2 decimal places.

[a] Lifetime exposure and cancer expression. UNSCEAR and ICRP estimates use different age distributions and periods of expression.

[b] EPA Radionuclides Background Information Document; EPA 520/1-84-022-1 (EPA84). Also see Table 7-3 and text.

[c] Normalized for risk of fatal cancer (see text).

[d] Five additional target organs which have the highest doses are assigned 0.08 each for a total of 0.4.

[e] Other includes esophagus, lymphatic system, pharynx, larynx, salivary gland, and brain.

7.2.11 Cancer Risks for a Constant Intake Rate

The fatal cancer risks shown in the tables of this chapter presume a lifetime exposure at a constant dose rate. Even for a dosimetric model with age invariant parameters, dose rates vary with time for a constant intake rate. This variation reflects the time dependent activity levels associated with the retention of the radionuclide in the organs and tissues. The ingrowth of radioactive decay products can also contribute further to the time dependence of dose rates.

Traditionally, risk estimates for chronic intake of a radionuclide have been determined using a dose commitment model to calculate dose rates following a fixed period (e.g., a 70-year average lifespan). For the purpose of estimating risk, these dose rates are considered to be invariant over the individual's lifetime. This approach is overly conservative for estimating risk for many long-lived radionuclides. Therefore, EPA estimates risks for constant radionuclide intakes by first determining dose rates to each radiosensitive organ or tissue as a function of time. Then these dose rates and the risk models of this chapter are used to calculate lifetime risk based on 1970 life table data. The resulting risks are consistent with both the dosimetric and risk models, and the arbitrary choice of a dose commitment period is avoided.

7.3 Fatal Cancer Risk Resulting from High-LET Radiations

In this section we explain how EPA estimates the risk of fatal cancer resulting from exposure to high-LET radiations. Unlike exposures to x rays and gamma rays where the resultant charged particle flux results in linear energy transfers (LET) of the order of 0.2 to 2 keV per □m in tissue, 5-MeV alpha particles result in energy deposition at a track average rate of more than 100 keV per □m. High-LET radiations have a larger biological effect per unit dose (rad) than low-LET radiations. How much greater depends on the particular biological endpoint being considered. For cell killing and other readily observed endpoints, the relative biological effectiveness (RBE) of high-LET alpha radiations is often 10 or more times greater than low-LET radiations. The RBE may also depend on the dose level; for example, if linear and linear-quadratic dose response functions are appropriate for high- and low-LET irradiations, respectively, then the RBE will decrease with increasing dose.

7.3.1 Quality Factors and RBE for Alpha Particles

For purposes of calculating dose equivalent, each type of biologically important ionizing radiation has been assigned a quality factor, Q, to account for its relative efficiency in producing biological damage. Unlike an RBE value, which is for a specific tissue and well-defined endpoint, a quality factor is based on an average overall assessment by radiation protection experts of potential harm of a given

radiation relative to x- or gamma radiation. In 1977, the ICRP assigned a quality factor of 20 to alpha particle irradiation from radionuclides (ICRP77). However, the appropriateness of this numerical factor for estimating fatal radiogenic cancers is still unclear -- particularly for individual sites.

The dose equivalent, in rem, is the dose, in rad, times the appropriate quality factor for a specified kind of radiation. For the case of internally deposited alpha-particle emitters, the dose equivalent from a one-rad dose is 20 rem. It should be noted that prior to ICRP Report 26 (ICRP79), the quality factor assigned to alpha particle irradiation was 10. That is, the biological effect from a given dose of alpha particles was estimated to be 10 times that from an acute dose of low-LET x rays or gamma rays of the same magnitude in rad. The ICRP decision to increase this quality factor to 20 followed from their decision to estimate the risk of low-LET radiations, in occupational situations, on the assumption that biological effects were reduced at low dose rates. There is general agreement that dose rate effects do not occur for high-LET (alpha) radiations. Implicit in ICRP's risk estimates for low dose/dose rate gamma radiation is a dose rate reduction factor of about 2.5. The EPA (linear) risk model for low-LET radiation does not involve such a DREF; therefore, in order to avoid an artifactual inflation in our high-LET risk estimates, we have assumed an RBE of 8 (20/2.5) for calculating the risks from alpha particles (see Section 7.3.3).

In 1980 the ICRP published the task group report "Biological Effects of Inhaled Radionuclides," which compared the results of animal experiments on radiocarcinogenesis following the inhalation of alpha-particle and beta-particle emitters (ICRP80). The task group concluded that: "...the experimental animal data tend to support the decision by the ICRP to change the recommended quality factor from 10 to 20 for alpha radiation."

7.3.2 Dose Response Function

In the case of high-LET radiation, a linear dose response is commonly observed in both human and animal studies. This response is not reduced at low dose rates (NCRP80). Some data on human lung cancer indicate that the carcinogenic response per unit dose of alpha radiation is higher at low doses than higher ones (Ar81, Ho81, Wh83); in addition, some studies with animals show the same response (Ch81, Ul82). We agree with the NAS BEIR-3 Committee that: "For high-LET radiation, such as from internally deposited alpha-emitting radionuclides, the linear hypothesis is less likely to lead to overestimates of the risk and may, in fact, lead to underestimates" (NAS80). However, at low doses, departures from linearity are small compared to the uncertainty in the human epidemiological data, and we believe a linear response provides an adequate model for evaluating risks in the general environment.

A possible exception to a linear response is provided by the data for bone sarcoma (but not sinus carcinoma) among U.S. dial painters who ingested alpha-emitting radium-226 (NAS80). These data are consistent with a dose-squared response (Ro78). Consequently, the NAS BEIR-3 Committee estimated bone cancer risk on the basis of both linear and quadratic dose response functions. However, as pointed out in NAS80, the number of U.S. dial painters at risk who received less than 1,000 rads was so small that the absence of excess bone cancer at low doses is not statistically significant. Therefore, the consistency of these data with a quadratic (or threshold) response is not remarkable and, perhaps, not relevant to evaluating risks at low doses. In contrast to the dial painter data, the incidence of bone cancer following short-lived radium-224 irradiation, observed in spondylitics by Mays and Spiess (Ma83, NAS80), in a larger sample at much lower doses, is consistent with a linear response. Therefore, for high-LET radiations EPA has used a linear response function to evaluate the risk of bone cancer.

Closely related to the choice of a dose response function is what effect the rate at which a dose of high-LET radiation is delivered has on its carcinogenic potential. This is an active area of current research. There is good empirical evidence, from both human and animal studies, that repeated exposures to radium-224 alpha particles is 5 times more effective in inducing bone sarcomas than a single exposure which delivers the same dose (Ma83, NAS80). The 1980 NAS BEIR Committee took this into account in its estimates of bone cancer fatalities, which EPA is using. We do not know to what extent, if any, a similar enhancement of carcinogenicity may occur for other cancers resulting from internally deposited alpha-particle emitters.

7.3.3 Assumptions Made by EPA for Evaluating the Dose from Alpha-Particle Emitters

We have evaluated the risk to specific body organs by applying an RBE of 8 for alpha radiations to the risk estimates for low dose rate low-LET radiations as described above. For some organs, this factor may be too large. Several authors have noted that estimates of leukemia based on an RBE of 20 for bone marrow alpha irradiation (relative to a low dose rate low-LET risk model which includes a DREF of 2.5) overpredicts the observed incidence of leukemia in persons receiving thorotrast (thorium oxides) (Mo79) and in the U.S. radium dial painters (Sp83). Nevertheless, in view of the paucity of applicable human data and the uncertainties discussed above, the ICRP quality factor provides a reasonable and prudent way of evaluating the risk due to alpha emitters deposited within body organs.

All EPA risk estimates for high-LET radiations are based on a linear dose response function. For bone cancer and leukemia we use the absolute risk projection model described in the previous section. For other cancers we use relative risk projections.

Table 7-6 indicates the Agency's estimates of the risk of fatal cancer due to a uniform organ dose in various organs from internally deposited alpha-particle emitters. It was prepared by multiplying the average risk based on the linear model for a uniformly distributed whole-body dose of low-LET radiation by an RBE of 8 and then apportioning this risk by organ, as indicated in Table 7-6. These estimates are for lifetime doses at a constant dose rate. This procedure was not followed for bone cancer. As outlined above, the risk estimate for this cancer in the BEIR-3 report is based directly on data for high-LET (alpha) radiation.

Some readers may note that the risk estimate in Table 7-6, 19 bone cancer fatalities per 10^6 person-rad, is less than the 27 fatalities listed in Table A-27 of (NAS80) for alpha particles. This is because the analysis in Appendix A of NAS80 (but not Chapter V of that report) assumes that in addition to a 2-year minimum induction period, 27 years are available for cancer expression. This is usually not the case for doses received beyond about age 50. Hence, the estimated lifetime risk is smaller when it is based on a life table analysis that considers lifetime exposure in conjunction with competing causes of death.

In the next section, we describe how we estimate the risk due to inhalation of alpha-emitting radon progeny, a situation where the organ dose is highly nonuniform.

7.4 Estimating the Risk Resulting from Lifetime Population Exposures from Radon-222 Progeny

The Agency estimates of the risk of lung cancer due to inhaled radon progeny do not utilize the dosimetric approach, outlined above, but rather are based on what is sometimes called an epidemiological approach, that is, on the excess human lung cancer in groups known to have been exposed to radon progeny.

When radon-222, a radioactive noble gas, decays, a number of short half-life radionuclides, principally polonium-218, lead-214, bismuth-214, and polonium-214, are formed, some of which attach to inhalable dust particles in air. When inhaled, the radon progeny are deposited on the surfaces of the larger bronchi of the lung. Since two of these radionuclides decay by alpha-particle emission, the bronchial epithelium is irradiated by high-LET radiation. There is a wealth of data indicating that a range of exposures to the bronchial epithelium of underground miners causes an increase in bronchial lung cancer, both in smoking and in nonsmoking miners. Two recent reviews on the underground miner experience are of particular interest. The 1980 NAS BEIR-3 Report (NAS80) contains a review of the epidemiological studies on these miners. A lengthy report, "Risk Estimates for the Health Effects of Alpha Radiation" by D. C. Thomas and K. C. McNeil for the Atomic Energy Control Board (AECB) of Canada (Th82), reanalyzes many of these epidemiological studies in a consistent fashion, so that the modeling assumptions are similar for all of the data sets.

Table 7-6. Estimated number of cancer fatalities from a lifetime exposure to internally deposited alpha-particle emitters as given in EPA 84[f]

Site	Proportional risk[a]	Fatalities per 10^6 person-rad[b]
Lung	0.207	466
Breast[c]	0.130	291
Red bone marrow[d]	0.150	337
Thyroid	0.099	222
Bone surface[e]	0.009	19
Liver	0.085	191
Stomach	0.084	189
Intestine	0.039	87
Pancreas	0.059	131
Kidneys and urinary tract	0.028	56
Other	0.113	254
Total		2243

[a] Proportion of whole-body risk from Table 7-3.

[b] Rounded to two figures.

[c] Average for both sexes.

[d] Leukemia.

[e] Bone surface (endosteum) as defined in ICRP-30 (ICRP79).

[f] As in the case of low-LET radiation, the organ risks above have been scaled up by a factor of 395/280 for this proposed rulemaking.

Although considerable progress has been made in modeling the deposition of radon daughters in the lung (Ha82, Ja80, Ja81), it is not yet possible to adequately characterize the bronchial dose delivered by alpha radiation from inhaled radon-222 progeny. This is in part due to the uncertainty concerning the kinds of cells in which bronchial cancer is initiated (Mc78) and the depth of these cells in the bronchial epithelium. Current estimates of the dose actually causing radiogenic cancer due to inhaled radon-222 progeny are based on average doses which may or may not be relevant.

Even if accurate estimates of the dose delivered to the target cells in the bronchial epithelium could be obtained for both low-LET and alpha radiations, they would probably not be adopted as the basis for estimating the risk to the public from airborne radon daughters. To do so would mean extrapolating risk estimates derived from observations on populations (particularly, the A-bomb survivors) receiving acute doses of low-LET radiation over the whole lung to the case of chronic, nonuniform lung doses from high-LET alpha irradiation. It would appear that more reliable estimates of the risk can be derived on the basis of observed cancers following occupational exposure to radon progeny, i.e., through the epidemiological approach. Dosimetric considerations may nevertheless be helpful in refining the risk estimates for the general population. In particular, they were used, as discussed below, in formulating our age-specific risk estimates for members of the general public through the use of an "exposure equivalent."

7.4.1 Characterizing Exposures to the General Population vis-a-vis Underground Miners

Exposures to radon progeny under working conditions are commonly reported in a special unit called the working level (WL). One working level is any combination of short half-life radon-222 progeny having 1.3×10^5 MeV per liter of potential alpha energy (FRC67). This unit was developed because the concentration of specific radon progeny depends on ventilation rates and other factors. A working level month (WLM) is the unit used to characterize a miner's exposure to one working level of radon progeny for a working month of about 170 h. Because the results of epidemiological studies are expressed in units of WL and WLM, we outline below how they can be interpreted for members of the general population exposed to radon progeny.

For a given concentration of radon progeny, the amount of potential alpha energy inhaled in a month by a member of the general population is more than that received in a miner's working month. These individuals are exposed longer, up to 24 h/da, 7 da/wk. However, the average amount of air inhaled per minute (minute volume) by a member of the general population is less than the amount for a working miner when such activities as sleeping and resting are taken into account. To compare the radon progeny exposure of a working miner to a member of the general population, we have calculated the amount of potential alpha energy each inhales per year.

We have assumed that (averaged over a work day) a miner inhales 30 L/min. This average corresponds to about 4 h of light activity and 4 h of moderately heavy work per day (ICRP75). We recognize that the new ICRP radon model assumes a 20-L/min volume for miners, which corresponds to 8 h of light activity per day (ICRP81). Although this may be appropriate for nuclear workers, studies of the metabolic rate of working miners clearly show that they are not engaged only in light activity (Sp56, ICRP75, NASA73). Therefore, we have chosen 30 L as a more realistic estimate of their average minute volume. A working miner with this minute volume inhales 3.6E+03 m^3 in a working year of 2,000 h (ICRP79). One working level of radon-222 progeny is 2.08E-05 joules/m^3. Therefore, in a working year the potential alpha energy inhaled by a miner exposed to one working level is 7.5E-02 joules.

For adult males and females in the general population, we follow the ICRP Task Group on Reference Man (ICRP75) in assuming an inhaled air volume of 23 m^3/da for males and 21 m^3/da for adult females. We use the average of these two values, 22 m^3/da, for an adult member of the general public. This average volume results in 1.67E-01 joules/yr of inhaled potential alpha energy from a continuous exposure to 1 WL of radon-222 progeny for 365.25 da. Although it may be technically inappropriate to quantify the amount of potential alpha particle energy inhaled by a member of the general population in WLM, this corresponds to about the same inhaled potential alpha energy as a 27 WLM exposure would to a miner. Hence, a one WL concentration of radon progeny provides an adult a 27 WLM annual exposure equivalent (see Table 7-7). For indoor exposure, we assume an occupancy factor of 0.75, so that an indoor exposure to 1 WL results in an annual exposure equivalent to 20 WLM, (EPA78) in terms of the amount of potential alpha energy actually inhaled.

Children have a smaller bronchial area than adults, which more than offsets their lower minute volume, so that the bronchial deposition and expected dose, for a given concentration of radon progeny, is greater. This problem has been addressed by Hofmann and Steinhausler (Ho77). Their results indicate that doses received during childhood are about 50 percent greater than adult doses for a given air concentration of radon daughters. We have used the information in (Ho77) to prepare Table 7-7, which lists the age-dependent exposure equivalents we have used in the risk assessments described below. (The assumptions on minute volume, etc., for miners and the general population described above are the same as those used in the preparation of EPA79,82,83a,b.) The results in Table 7-7 have been rounded to two significant figures. The larger effective exposure to children relative to adults increases the estimated mortality due to lifetime exposure from birth by about 20 percent.

7.4.2 The EPA Model

Since 1978, the Agency has based risk estimates due to inhaled radon-222 progeny on a linear dose response function, a relative risk projection model, and a minimum induction period of 10 years. The life

table analysis described in Appendix E is used to project this risk over a full life span. Lifetime risks were initially projected on the assumption that an effective exposure of 1 WLM increases the age-specific risk of lung cancer by 3 percent over the age-specific rate in the U.S. population as a whole (EPA79).

The initial EPA model for calculating radon risks has been described in detail (EPA79, E179). In reviewing this model in terms of the more recent information described below, we have found that our major assumptions, linear response and relative risk projection, have been affirmed. Data on the A-bomb survivors clearly indicate that for low-LET radiation their absolute risk of radiogenic lung cancer has continued to increase while their relative risk has remained reasonably constant (Ka82). The UNSCEAR, ICRP, and 1980 NAS Committee have continued to use a linear dose response to estimate the risk of lung cancer due to inhaled radon progeny. Thomas and McNeill's analysis (Th82) indicates that the use of linearity is not unduly conservative and may, in fact, underestimate the risk at low doses. As noted above, the 1980 NAS BEIR Committee reached a similar conclusion.

A major limitation of the EPA model is the uncertainty in the choice of relative risk coefficient, the percent increase per WLM. This value is based on the excess mortality due to lung cancer among exposed miners of various ages, many of whom smoked. Therefore, it is an average value for a mixed population of smokers and nonsmokers. Smoking was more prevalent among some of the groups of miners studied than it is among the U.S. general population today; this may inflate the risk estimate, as discussed below.

Radford and Renard (Ra84) reported on the results of a long-term study of Swedish iron miners who were exposed to radon progeny. This study is unique in that most of the miners were exposed to less than 100 WLM, and the risks to smokers and nonsmokers were considered separately. The absolute risk of the two groups was similar, 20 fatalities per 10^6 person WLM year for smokers compared to 16 for nonsmokers. While absolute risks were comparable for the smoking and nonsmoking miners, relative risks were not. Nonsmokers have a much lower baseline incidence of lung cancer mortality than smokers. As a result, the relative risk coefficient for nonsmoking miners was about 4 times larger than for smoking miners. In each case, the risk was calculated relative to baseline rates in nonsmokers and smokers, respectively.

Although occupational exposures to pollutants other than radon-222 progeny are probably not important factors in the observed lung cancer risk for underground miners (E179, Th82, Mu83, Ra84), the use of occupational risk data to estimate the risk of a general population is far from optimal, as it provides no information on the effect of radon progeny exposures to children and women. While we have continued to assume that the risk per unit dose during childhood is no more effective than that occurring to adults, this assumption may not be correct. The A-bomb survivor data indicate that, in general, the risk from childhood

Table 7-7. Annual exposure equivalent (WLM) by age for members of the general public continuously exposed to radon progeny at 1 WL (2.08×10^{-5} joules per cubic meter)

Age (yr)	Exposure equivalent (WLM)
0-2	35
3-5	43
6-11	49
12-15	43
16-19	38
20-22	32
23 or more	27
Lifetime Average	31.4

exposure to low-LET radiation is greater than from exposure of adults and continues for at least 33 years, the time over which A-bomb survivors have been observed (Ka82). There are not, as yet, specific data for lung cancer (Ka82). Another limitation of the underground miner data is the absence of women in the studied populations. The A-bomb survivor data indicate women are as sensitive as men to radiogenic lung cancer from low-LET radiation even though, on the whole, they smoke less (Pr83). These data are not conclusive, however.

7.4.3 Comparison of Risk Estimates

Several estimates of the risk due to radon progeny have been published since the EPA model was developed. One of particular interest was expounded by the BEIR Committee in NAS80. The BEIR-3 Committee formulated an age-dependent absolute risk model with increasing risk for older age groups. The Committee estimates of the risk per WLM for various ages are listed on page 325 in NAS80 and its estimated minimum induction period for lung cancer following exposure on page 327. We have used these data, summarized in Table 7-8, to calculate the lifetime risk of lung cancer mortality from lifetime exposure to persons in the general population by means of the same life table analysis used to calculate other EPA risk estimates.

It should be noted that the zero risk shown in Table 7-8 for those under 35 years of age at exposure does not mean no harm occurs, but rather that it is not expressed until the person is at least 35 years old, i.e., only after the minimum induction period. The sequence of increasing risk with age shown in Table 7-8 is not unlike the increase in lung cancer with age observed in unexposed populations, so that the pattern of excess risk over time is similar to that found using a relative risk projection model.

Recently, Thomas and McNeil conducted a thorough analysis of lung cancer among uranium and other hard rock miners for the AECB of Canada (Th82). These investigators tested a number of risk models on all of the epidemiological studies that contained enough data to define a dose-response function. They concluded that for males a 2.3 percent increase in lung cancer per WLM and a relative risk projection model were more consistent with the excess lung cancer incidence observed in underground miner groups than other models they tested. This is the only analysis we are aware of that treated each data set in a consistent fashion and utilized modern epidemiological techniques, such as controlling, to the extent possible, for age at exposure and duration of follow-up.

The initial EPA risk estimates for lifetime exposure to a general population, along with AECB, NAS, UNSCEAR, ICRP, and NCRP estimates of the risk of lung cancer resulting from inhaled radon progeny, are listed in Table 7-9. The AECB estimate for lifetime exposure to Canadian males is 830 fatalities per million person-WLM (Th82). In Table 7-9 this estimate has been adjusted for the U.S. 1970 male and female population.

There is good agreement between the EPA, NAS80 (BEIR-3), and the AECB estimates shown in Table 7-9. Each of these estimates is based on lifetime exposure and lifetime expression of the incurred risk. In contrast, the ICRP and UNSCEAR risk estimates in Table 7-9 do not explicitly include these factors.

The ICRP estimates are for occupational exposure to working adults. The larger ICRP estimate is based on their epidemiological approach, that is, the exposure to miners in WLM and the risk per WLM observed in epidemiological studies of underground miners. The ICRP epidemiological approach assumes an average expression period of 30 years for lung cancer. Children, who have a much longer average expression period, are excluded from this estimate. The ICRP has not explicitly projected the risk to miners beyond the years of observation even though most of the miners on whom its estimates are based are still alive and continuing to die of lung cancer.

The smaller of the two ICRP estimates listed in Table 7-9 is based on this dosimetric approach. The ICRP assumes that the risk per rad for lung tissue is 0.12 of the risk of cancer and genetic damage following whole-body exposure (ICRP77). For the case of exposure to radon progeny, the ICRP divided this factor of 0.12 into two equal parts. A weighting factor of 0.06 was used to assess the risk from the high dose to bronchial tissue, where radiogenic lung cancer is observed in exposed underground miners. The other half of the lung weighting factor, another 0.06 of the total body risk, was used to assess the risk to the pulmonary region which receives a comparatively small dose from radon-222 progeny and where human lung cancer is seldom, if ever, observed.

The UNSCEAR estimate is for a general population and assumes an expression time of 40 years. Like the ICRP, UNSCEAR did not make use of an explicit projection of risk of fatal lung cancer over a full lifetime.

The last entry in Table 7-9, the NCRP risk estimate based on an analysis by Harley and Pasternack (Ha82), is of particular interest since, like those of EPA and AECB, it is based on a life table analysis of the lifetime risk due to lifetime exposure. This estimate utilizes an absolute risk projection model with a relatively low risk coefficient, 10 cases per 10^6 person-WLM per year at risk, the smallest of those listed by the NAS BEIR-3 Committee: cf. Table 7-8. Moreover, they have assumed that the risk of lung cancer following irradiation decreases exponentially with a 20-year half-life, so that exposures occurring early in life have very little risk. The NCRP assumption of a 20-year half-life for radiation injury reduces the estimated lifetime risk by about a factor of 2.5. Without this assumption the NCRP risk estimate would be the same as the midpoint of the UNSCEAR estimate, about 325 fatalities per million person-WLM. Note that if lung cancer risk from low-LET radiation decreased over time with a 20-year half-life, the excess lung cancer observed in Japanese A-bomb survivors should have decreased during the period they have been followed, 1950-1978. During this period the absolute lung cancer risk in every age cohort has markedly increased (Ka82).

Table 7-8. Age-dependent risk coefficients and minimum induction period for lung cancer due to inhaling Radon-222 progeny (NAS80)

Age (yr)	Excess (cases per 10^6 WLM person-years)	Minimum induction period (years)
0-14	0	25
15-34	0	15-20
35-49	10	10
50-65	20	10
65 or more	50	10

Table 7-9. Risk estimate for exposures to radon progeny

Organization	Fatalities per 10^6 person-WLM	Exposure period	Expression period
EPA[a]	760 (460)[b]	Lifetime	Lifetime
NAS BEIR-3[a]	730 (440)[b]	Lifetime	Lifetime
AECB[c]	600 (300)[b]	Lifetime	Lifetime
ICRP	150-450	Working Lifetime	30 years
UNSCEAR	200-450	Lifetime	40 years
NCRP[d]	130	Lifetime	Lifetime

[a] The number of fatalities per 10^6 person-WLM listed for EPA and NAS80 in this table differs from figures we have previously published (e.g., EPA83b) because we have now included, correctly we believe, the increased potential alpha energy exposure during childhood in the denominator of this ratio. Our risk estimates for various sources of radon in the environment have not changed, because all were calculated via a life table analysis yielding deaths per 100,000 exposed, not deaths per 10^6 person-WLM.

[b] EPA and AECB based their estimates of risk for the general population on an exposure equivalent, corrected for breathing rate (and other factors). For comparison purposes, the values in parentheses express the risk in more customary units, in which a continuous annual exposure to 1 WL corresponds to 51.6 WLM.

[c] Adjusted for U.S. General Population, see text.

[d] NCRP84: Table 10.2; assumes risk diminishes exponentially with a 20-year halftime.

Sources: EPA83b; NAS80; Th82; ICRP81; UNSCEAR77; NCRP84; USRPC80.

Good agreement exists among the EPA, NAS (BEIR-3), and the AECB estimates listed in Table 7-9. Each of these estimates is based on lifetime exposure and lifetime expression of the incurred risk. Conversely, the three lower risk estimates shown in Table 7-9 either do not explicitly include these conditions or they include other modifying factors. Nevertheless, Table 7-9 indicates a divergence, by a factor of about 6, in risk estimates for exposure to radon-222 progeny. Thus, the use of a single risk coefficient may not be appropriate, as it could give the impression that the risk is known more precisely than is warranted by available information. The EPA, BEIR-3, and AECB estimates may be slightly high because they represent relative risks based on adult males, many of whom smoked. The actual risk may be smaller for a population that includes adult females, children, and nonsmokers. The UNSCEAR and ICRP estimates are probably low because they represent absolute risk estimates that do not completely take into account the duration of the exposure and/or the duration of the risk during a lifetime. The NCRP estimate is likely to be very low, as a low risk coefficient was used in an absolute risk model, and it was assumed that the risk decreases exponentially after the exposure.

7.4.4 Selection of Risk Coefficients

To estimate the range of reasonable risks from exposure to radon-222 progeny for use in the Background Information Document for Underground Uranium Mines (EPA85), EPA averaged the estimates of BEIR-3, the EPA model, and the AECB to establish an upper bound of the range. The lower bound of the range was established by averaging the UNSCEAR and ICRP estimates. The Agency chose not to include the NCRP estimate in its determination of the lower bound because this estimate is believed to be outside the lower bound. Therefore, the EPA chose relative risk coefficients of 1.2 to 2.8 percent per WLM exposure equivalent (300 to 700 fatalities per million person-WLM exposure equivalent) as estimates of the possible range of effects from inhaling radon-222 progeny for a full lifetime. Although these risk estimates do not encompass the full range of uncertainty, they seemed to illustrate the breadth of much of current scientific opinion.

The lower limit of the range of 1985 EPA relative risk coefficients, 1.2 percent per effective WLM, is similar to that derived by the Ad Hoc Working Group to Develop Radioepidemiological Tables, which also used 1.2 percent per WLM (NIH85). However, some other estimates based only on U.S. and Czech miner data averaged 1 percent per WLM (Ja85) or 1.1 percent per WLM (St85). On the other hand, three studies, two on miners (Ra84, Ho86) and one on residential exposure (Ed83, 84), indicate a relative risk coefficient greater than 3 percent per WLM, perhaps as large as 3.6 percent.

The EPA has, therefore, increased the upper limit of its estimated range of relative risk coefficients. To estimate the risk due to radon-222 progeny, the EPA now uses the range of relative risk coefficients of 1 to 4 percent per WLM. [See EPA86 for a more detailed

discussion.] Based on 1980 vital statistics, this yields, for members of the general public, a range of lifetime risks from 380 to 1,520 fatal cases per 10^6 WLM (expressed in exposure equivalents). In standard exposure units, uncorrected for breathing rate and age, this corresponds to 230 to 920 cases per 10^6 WLM. Coincidentally, the geometric mean estimate obtained in this way, 4.6E-04/WLM in standard units of exposure, is numerically the same as that obtained using a 3 percent relative risk coefficient and 1970 vital statistics (see Table 7-9).

7.5 Uncertainties in Risk Estimates for Radiogenic Cancer

As pointed out in the Introduction of this chapter, numerical estimates of risks due to radiation are not precise. A numerical evaluation of radiogenic cancer risks depends both on epidemiological observations and on a number of *ad hoc* assumptions which are largely external to the observed data. These assumptions include such factors as the expected duration of risk expression and variations in radiosensitivity as a function of age and demographic characteristics. A major assumption is the shape and slope of the dose response curve, particularly at low doses, i.e., below 1 rad, where there is insufficient epidemiological data to directly base risk estimates. In 1971, the BEIR Committee based its estimates of cancer risk on the assumption that effects at low doses are directly proportional to those observed at high doses, the so called linear-nonthreshold hypothesis. As described above in Section 7.2, the BEIR-3 Committee considered three dose response models and indicated a preference for the linear-quadratic model. The risk coefficients the BEIR-3 Committee derived for its linear-quadratic model, and to a lesser extent its linear model, are subject to considerable uncertainty primarily because of two factors: (1) systematic errors in the estimated doses of the individual A-bomb survivors, and (2) statistical uncertainty because of the small number of cancers observed at various dose levels.

7.5.1 The BEIR-3 Analysis of the A-bomb Survivor Data

For its analysis of the A-bomb survivor data, the BEIR-3 Committee expanded the equations for low-LET radiations to include a linear dose response function for neutrons:

$$F(D_g, D_n) = C_1 D_g + K_1 D_n \qquad (7\text{-}1)$$

$$F(D_g, D_n) = C_2 D_g^2 + K_2 D_n \qquad (7\text{-}2)$$

$$F(D_g, D_n) = C_3 D_g + C_4 D_g^2 + K_3 D_n \qquad (7\text{-}3)$$

where D_g is the gamma dose and D_n is that part of the dose due to high-LET radiations from neutron interactions. Note that Equation 7-1

and Equation 7-3 each have two linear terms: one for neutrons and one for gamma radiation. In analyzing approximately linear data in terms of these equations, the decision as to how much of the observed linearity should be assigned to the neutron or the gamma component is crucial. As discussed below, the BEIR-3 Committee attributed much of the observed radiogenic cancer to a linear response from neutron doses that did not occur.

The BEIR-3 Committee's general plan was to examine the dose response for leukemia and for solid cancer separately to find statistically valid estimates of the coefficients $C_1 \ldots\ldots C_4$ and $K_1 \ldots\ldots K_3$ by means of regression analyses. The T65 neutron and gamma doses to individual survivors are highly correlated since both are strongly decreasing functions of distance. This makes accurate determination of the coefficients in Equation 7-3 by means of a regression analysis extremely difficult. In addition, there is considerable sampling variation in the A-bomb survivor data due to small sample size, which exacerbates the regression problem (He83). Because of these and other problems, agreement between the observed response and that predicted by any of the dose response functions examined by the BEIR-3 Committee provides little basis for a choice between models.

The Committee analyzed the A-bomb survivor data in two separate sets: first, leukemia; second, all cancers excluding leukemia (solid cancers). Its treatment of these two cases was not equivalent. The Committee's analysis of leukemia considered the Nagasaki and Hiroshima data separately. The Committee's regression analysis of the leukemia mortality data provided stable values for all of the coefficients in Equation 7-3, and hence for the neutron RBE and the ratio of linear to dose-squared terms for leukemia induction by gamma rays, as a function of dose.

Estimating the linear-quadratic response coefficients for solid cancers proved to be less straightforward, however, and it was

decided that the observations on solid cancers were "not strong enough to provide stable estimates of low dose, low-LET cancer risk when analyzed in this fashion" (NAS80,p.186).

As outlined in the BEIR-3 Report, the Committee decided to use a constrained regression analysis, carrying over some of the parameters for Equation 7-3 found in its analysis of leukemia deaths to the regression analysis of the dose response for solid cancers. Specifically, both the neutron RBE at low dose (the ratio of the coefficient K_3 to C_3) and the ratio of C_3 to C_4, as estimated from the leukemia data, were assumed to apply to the induction of fatal solid cancers. These estimates became the basis for the "preferred" linear-quadratic (LQ-L) risk estimates for solid cancers presented in BEIR-3 (NAS80):p. 187.

7.5.2 Uncertainty of the Dose Response Models Due to Bias in the A-bomb Dosimetry

A careful state-of-the-art evaluation of the dose to A-bomb survivors was carried out by investigators from Oak Ridge National Laboratory in the early 1960's (Au67, Au77). The results of these studies resulted in a "T65" dose being assigned to the dose (kerma) in free air at the location of each survivor for both gamma rays and neutrons. A major conclusion of the ORNL study was that the mix of gamma ray and neutron radiations was quite different in the two cities where A-bombing occurred. These results indicated that at Hiroshima the neutron dose was more important than the gamma dose when the greater biological efficiency of the high-LET radiations produced by neutrons was taken into account. Conversely, the neutron dose at Nagasaki was shown to be negligible compared to the gamma dose for that range of doses where there were significant numbers of survivors. Therefore, the 1980 BEIR Committee evaluated the cancer risks to the survivors at Hiroshima on the assumption that the combined effects of gamma rays and particularly neutrons caused the observed cancer response.

Since the BEIR-3 report was published, it has become evident that the organ doses due to neutrons at Hiroshima were overestimated by about an order of magnitude, at distances where most of the irradiated persons survived bomb blast and yet received significant doses (1,000-1,500 m).

In fact, the neutron doses at Hiroshima are quite comparable to those previously assigned, at similar distances, to Nagasaki survivors (Ke81a, Ke81b, RERF83, RERF84). Moreover, there are now grounds to believe the T65 estimates of gamma-ray doses in both cities are also incorrect (RERF83, RERF84).

At the time of this writing, a major effort is underway to reassess the dosimetry in both cities (RERF83, RERF84). Preliminary indications are that gamma-ray doses in air will decrease in Nagasaki, but only slightly. In Hiroshima there may be substantial increases in the gamma-ray kerma beyond about 1500 m, but only small increases closer to the hypocenter, where most of the collective dose was received. In addition, recalculation of shielding from structures and body tissue is expected to decrease the average gamma-ray organ doses somewhat. The net effect of these changes in gamma-ray doses is still unclear, but they are unlikely to result in more than a 50 percent change in risk estimates for gamma irradiation. More important, it seems, is the anticipated effect of revised estimates of the neutron dose to the Hiroshima survivors.

Given the information discussed above, it is possible to see, at least qualitatively, how the large bias in the estimated T65 neutron dose to the Japanese survivors affects the 1980 BEIR Committee's estimates of the risk coefficients for leukemia. The Committee's age-adjusted risk coefficients for leukemia are listed in Table V-8. For the linear fit, the neutron RBE (K_1/C_1) was 11.3, while for the linear-quadratic case the neutron RBE (K_3/C_3) was 27.8. Tables A-11 and V-13 provide the estimates of neutron and gamma doses to the bone marrow of Hiroshima survivors that were used by the Committee. Substituting these doses in its risk equations (Table V-8) indicates that, for either model, almost 50 percent of the total leukemia deaths were ascribed to the neutron dose component then thought to be present at Hiroshima. (At first sight, it might appear that a substantially larger fraction of leukemias would be attributed to neutrons in the LQ-L model, in view of its higher neutron RBE. However, when one takes into account the D_g^2 term in Equation 7-3, it turns out that the contribution of gamma rays is about the same as in the L-L model.)

In a similar way, the conversion factors given in Table V-13 for obtaining tissue dose from kerma can be used to derive the fraction of all solid tumors attributed to neutrons. Here again almost half the cancers were attributed to the neutron component. Since, as noted above, the neutron dose was overestimated, almost all of the excess leukemias and solid tumors will probably now have to be attributed to gamma rays.

There is no simple way of adjusting the 1980 BEIR risk estimates to account for the risk they attributed to neutrons. Adjustment of neutron doses alone is clearly inappropriate, since there is good reason to believe that T65 estimates of the dose due to gamma rays are also subject to considerable change. Moreover, not all of the individuals in a given T65 dose category will, necessarily, remain grouped together after new

estimates of neutron and gamma doses are obtained. Both the numerator and denominator in the ratio of observed to expected cases are subject to change and indeed could change in opposite directions, a fact not considered in some preliminary analyses (St81). Nevertheless, it is reasonable to conclude that bias in the estimated neutron doses at Hiroshima has led to considerable uncertainty in the BEIR-3 risk estimates and probably to a systematic underestimation of the risk due to low-LET radiations. In addition, random errors in dose estimates will contribute to the uncertainty in risk coefficients. As discussed by Gilbert (Gi84), these errors also tend to bias risk estimates downward. In light of these biases arising from errors in dosimetry, we believe that estimates based on the more conservative linear dose response should be given considerable weight vis a vis those made using the BEIR-3 linear-quadratic models.

In conclusion, the overall effect of the revised dosimetric calculations will probably be to increase the estimated risk per unit dose of low-LET radiation in the A-bomb survivor population. The magnitude of the increase is unknown, but will probably not be more than about a factor of 2.

From the standpoint of estimating risks from low-level, low-LET radiation, however, the most important result of the new dosimetric calculations may be in helping to determine which models best describe the data on human radiation carcinogenesis. After all, the greatest uncertainties in radiation risk estimation generally reflect model uncertainties, not uncertainties in the magnitude of risk coefficients.

7.5.3 Sampling Variation

Besides the systematic bias in the BEIR-3 risk estimates for low-LET radiation outlined above, the precision of the estimated linear and linear-quadratic risk coefficients in the BEIR-3 report is limited by statistical fluctuations due to sample size. The uncertainty bounds ($\pm$ 1 SD) attached to the gamma-ray risk coefficient in the BEIR-3 linear model are about $\pm$25 percent, for either leukemia (Table V-8) or for all other cancers (Table V-11). For the latter groups of cancers, however, the neutron RBE was constrained to the value obtained from analysis of the leukemia data. If this constraint is removed, the uncertainty in the estimate increases to $\pm$150 percent (Table V-9). This increase reflects the large uncertainty associated with the neutron contribution in the analysis and the strong correlation between neutron and gamma-ray doses. Following the dosimetry reassessment, neutron doses will decrease markedly, but will remain correlated with gamma-ray doses. It is also likely that the estimated risk per unit dose will still turn out to be significantly different between Hiroshima and Nagasaki. Attributing this difference to the much smaller neutron fluxes may imply a biologically implausible value for the neutron RBE (<0 or >100, for example).

An alternative approach would then be to impose a constraint: in particular, it might be assumed that the neutron contribution to the excess cancer is negligible, the apparent difference between the two cities being due to some residual systematic errors in dosimetry, or to other unknown causes. If the fit is constrained in this way, the standard deviation in the linear coefficient obtained from the combined data may be reduced back down to approximately that obtained from the constrained analysis based on the T65 dosimetry, i.e., to about ±25 percent; there could, however, be a residual uncertainty relating to any unexplained differences between the two cities.

With the linear-quadratic model, there is the additional uncertainty over the relative magnitudes of the linear and quadratic coefficients. If the quadratic term is constrained to be non-negative, then the linear model estimate provides an upper bound on the magnitude of the risk at low doses. On the other hand, a pure quadratic model (linear coefficient equal to zero) based on the T65 dosimetry is consistent with the A-bomb survivor data on leukemia as well as on solid tumors.

7.5.4 Low Dose Extrapolation

As discussed above, the A-bomb survivor data on leukemia and all solid tumors, when analyzed in terms of the linear-quadratic model, are consistent with a very small, possibly zero, linear coefficient and thus a risk at low doses/dose rates, which is much smaller than what would be predicted from the linear model. A reasonable lower bound on the risk coefficient at low doses and dose rates can, however, be derived from other considerations.

Results from animal and cellular studies often show decreasing effects (e.g., cancers, mutations, or transformations) per rad of low-LET radiation at low doses and dose rates. Based on a review of this literature, the National Council on Radiation Protection (NCRP80) has concluded that "linear interpolation from high doses (150 to 300 rads) and dose rates (>5 rads min^{-1}) may overestimate the effects of either low doses (0-20 rads or less) or of any dose delivered at dose rates of 5 rad y^{-1} or less by a factor of 2 to 10." Judged solely from laboratory experiments, therefore, about a factor of 10 reduction from the linear prediction would seem to constitute a plausible lower limit on the effectiveness of low-LET radiation under chronic low dose conditions. Epidemiological evidence, however, would seem to argue against such a large DREF for human cancer induction.

Data on the A-bomb survivors and patients irradiated for medical reasons indicate that excess breast cancer incidence is proportional to dose and independent of dose fractionation (NAS80, NIH85). The evidence regarding thyroid cancer induction is less firm, but the data would again suggest a linear dependence on dose (NAS80, NIH85). The only other cancer for which there are human data "good enough" to provide any test of dose response models is leukemia. An analysis of the A-bomb survivor

data based on T65 dosimetry suggests a quadratic component; however, the best estimate of the linear coefficient obtained from the linear quadratic fit to the data is only about a factor of 2.5 less than the coefficient derived from the linear model.

A lower bound estimate of risk might be constructed by assuming that a linear dose-response function holds for breast cancer induction, but that for low dose rates the pure linear model overpredicts other cancers by a factor of 10 (DREF = 10). Using a linear model for all cancers, it was estimated (see Table 7-3) that about 17 percent of all fatal cancers resulting from a uniform whole-body dose to the general population are breast cancers. Thus, under the assumption above, the lower bound estimate is 23 percent of the linear estimate (1 x 14 percent + 0.1 x 86 percent). This would still seem to be an extreme lower bound estimate of the risk, especially in light of the evidence on thyroid cancer and leukemia referred to above. We believe a reasonable lower bound on the effectiveness of low-LET radiation in causing fatal cancers at low doses and dose rates is about 30 percent of that computed by linear extrapolation from high acute doses (equivalent to DREF=3.3).

7.5.5 Other Uncertainties Arising from Model Selection

In addition to a dose response model, a "transportation model" is needed to apply the risks from an observed irradiated group to another population having different demographic characteristics. A typical example is the application of the Japanese data for A-bomb survivors to Western people. Seymour Jablon (Director of the Medical Follow-up Agency of the National Research Council, NAS) has called this the "transportation problem," a helpful designation because it is often confused with the risk projection problem described below. However, there is more than a geographic aspect to the "transportation problem." Risk estimates for one sex must sometimes be based on data for the other. In transporting risk estimates from one group to another, one may have to consider habits influencing health status, such as differences between smokers and nonsmokers, as described in Section 7.4 for the case of risk estimates for radon progeny.

The BEIR-3 Committee addressed this problem in its 1980 report and concluded, based largely on the breast cancer evidence, that the appropriate way to transport the Japanese risk to the U.S. population was to assume that the absolute risk over a given observation period was transferrable but that relative risk was not. Therefore, the Committee calculated what the relative risk would be if the same number of excess cancer deaths were observed in a U.S. population having the same age characteristics as the A-bomb survivors. A constant absolute risk model, as postulated by the Committee, would imply that, whatever the factors are which cause Japanese and U.S. baseline cancer rates to differ, they have no effect on the incidence of radiation-induced cancers; i.e., the effects of radiation and these factors are purely additive.

An alternative approach to solving the "transportation problem" is that of the 1972 NAS BEIR-1 Committee. This Committee assumed relative risks would be the same in the United States and Japan and transferred the observed percentage increase directly to the U.S. population. Since the U.S. and Japanese baseline rates differ drastically with respect to mortality from specific cancers, this approach implies some large differences in the predicted number of specific cancers resulting from a given dose of radiation in the two countries. The most important differences relate to cancers of the breast, lung, and stomach. Baseline rates of breast and lung cancers are higher in the U.S. by factors of about 4 and 2, respectively, while the risk of stomach cancer is about 8 times higher in Japan (Gi85). As noted above, it now appears that the absolute risk should be transported for breast cancer. Evidence is lacking regarding the other diseases, however. If lung cancer risk were to be transported with a relative risk model, retaining the absolute model for other cancers, the estimated risk from a whole-body exposure would increase by about 20 percent; on the other hand, applying the relative risk model to stomach cancer alone would lower the whole-body risk by about 8 percent. Based on these considerations, including the tendency for changes in specific cancers to cancel one another, we believe that using the absolute risk "transportation model" is unlikely to cause errors of more than $\pm$20 percent in the total risk estimate. Thus, in the case of uniform whole-body doses, the amount of uncertainty introduced by transporting cancer risks observed in Japan to the U.S. population appears to be small compared to other sources of uncertainty in this risk assessment.

The last of the models needed to estimate risk is a risk projection model. As outlined in Section 7.2, such models are used to project what future risks will be as an exposed population ages. For leukemia and bone cancer, where the expression time is not for a full lifetime but rather 25 years, absolute and relative risk projection models yield the same number of radiogenic cancers, but would distribute them somewhat differently by time after exposure, and hence by age. For solid cancers, other than bone, the BEIR-3 Committee assumed that radiogenic cancers would occur throughout the estimated lifetime. This makes the choice of projection model more critical, because the relative risk projection yields estimated risks about three times larger than those obtained with an absolute risk projection, as shown in Table 7-2. Recent follow-up of the A-bomb survivor population strongly suggests that the relative risk projection model better describes the variation in risk of solid tumors over time (NIH85). However, there may be some cancers, apart from leukemia and bone cancers, for which the absolute risk projection model is a better approximation to reality. For other cancers, the relative risk may have been roughly constant for the current period of follow-up, but may eventually decrease over time. Thus, while the relative risk model was used in this report for calculating a "best estimate" of the lifetime risk of solid tumors, it may overestimate the risk by as much as a factor of 2.

Similarly, there is as yet insufficient information on radiosensitivity as a function of the age at exposure. The age-dependent risk coefficients we have used are those presented in the BEIR-3 report. As yet, there is little information on the ultimate effects of exposure during childhood. As the A-bomb survivors' population ages, more information will become available on the cancer mortality of persons irradiated when they were young. Table 7-2 indicates that the more conservative BEIR-1 assumption for the effect of childhood exposures would increase BEIR-3 risk estimates by about 40 percent. This is probably an upper bound. A lower bound can be estimated by assuming that the relative risk coefficient for those irradiated between ages 0-19 is actually only as large as that calculated for the next higher age category (20-34). This assumption leads to about a 20 percent decrease in the lifetime risk as compared to the BEIR-3 calculation. Therefore, the lack of precise information concerning the dependence of risk on age at exposure does not appear to be a major source of uncertainty in estimates of risk caused by either lifetime exposure or by a single exposure to the general population. Similarly, the BEIR-3 Committee did not include in utero exposures when calculating population risks for radiogenic cancer because they felt the estimate of the effect of in utero radiation is uncertain. We have deferred to their judgment in this regard. The BEIR-1 report did include in utero cancer risk. These had little effect, 1 to 10 percent, on the lifetime risk of cancer from lifetime exposure. An effect this small is not significant relative to other sources of uncertainty in the risk assessment.

7.5.6 Summary

We can only semi-quantitatively estimate the overall uncertainty in the risk per rad for low-LET radiations. We expect that more quantitative estimates of the uncertainty will be possible only after the A-bomb dose reassessment is completed and the A-bomb survivor data are reanalyzed on the basis of the new dose estimates. It should be noted, however, that even if all systematic bias is removed from the new dose estimates, there will still be considerable random error in the dose estimate for each survivor. This random error biases the estimated slope of the dose response curve so that it is smaller than the true dose response (Da75, Gi84, Ma59). The amount of bias introduced depends on the size of the random errors in the dose estimates and their distribution, which are unknown quantities at this stage of the dose reassessment.

Table 7-10 summarizes the various sources of uncertainty, as discussed above. The numerical entries represent multiplicative factors by which our estimates might have to be adjusted due to each source. To fully assess the magnitude of the combined uncertainty from all these sources, one must first characterize the underlying distribution of uncertainty relating to each source. This is beyond the scope of this report. However, a rough estimate of the overall uncertainty can be derived employing the general approach outlined in the Report of the

Table 7-10. Uncertainties in fatal cancer risk estimates

Source of uncertainty	Factor for limit	
	lower	upper
Use of linear model to extrapolate from acute high dose to chronic low dose exposures	0.3	1.0
Slope of dose response resulting from sampling variation	0.5[b] (.6)[c]	1.5[b](1.6)[c]
Use of T65 dosimetry	1.2	2.0
Use of lifetime relative risk projection model	0.5	1.0
Use of absolute risk "transportation model"[a]	0.8	1.2
Influence of age at exposure	0.8	1.4
Overall uncertainty[d]	0.23	1.6

[a] For the total of all cancers resulting from a uniform whole-body exposure to low-LET radiation; uncertainties relating to specific cancers may be considerably larger.

[b] Estimated 95% confidence limits based on a normal distribution.

[c] Estimated 95% confidence limits based on a lognormal distribution having the same mean and variance as the normal distribution defined by b.

[d] Combined uncertainty estimate from sources listed above; 95% confidence interval assuming the uncertainties from each source are independent and lognormally distributed (see text).

Ad Hoc Working Group to Develop Radioepidemiological Tables (NIH85). [It might be noted that the sources of uncertainty listed here differ somewhat from those in the NIH Report. To a large extent this reflects a difference in focus: here, it is on estimating the total number of cancers from whole-body exposures to a population; there, it is on estimating the probability that a particular cancer was caused by a given exposure. In addition, we have tried to incorporate sources of uncertainty (viz., those relating to sampling variation and choice of transportation model) not included by the Working Group in its calculation of combined uncertainty.]

As in the NIH Report, the uncertainties due to each source are assumed to be independent and lognormally distributed, the geometric mean of the distribution being set equal to the geometric mean of the upper and lower bounds. The respective upper and lower bounds are further assumed to be commensurate with one another; in particular, all are taken to be 95 percent confidence interval limits. The combined standard deviation or confidence interval can then be readily calculated. Denoting the geometric standard deviation of each source i ($i = 1, 2, \ldots k$) by S_i, the geometric standard deviation of the combined distribution is given by

$$\ln^2 S = \ln^2 S_i + \ldots \ln^2 S_k.$$

This procedure is highly arbitrary since there is generally no objective information on the actual underlying distribution of uncertainty for each source, the lognormal distribution being adopted, in large part, for calculational ease. Even the choice of upper and lower bounds involves a largely subjective judgment in most cases. One partial exception is the uncertainty relating to sampling variation (second entry in Table 7-10). This uncertainty is directly derived from a linear regression analysis of the A-bomb survivor data, the upper and lower bounds reflecting ± 2 standard deviations about the best estimate of the risk coefficient. Given the properties of the data, these bounds should indeed represent approximate 95 percent confidence limits; however, the underlying distribution of uncertainty is expected to be normal rather than lognormal. To better reflect this fact, while retaining the calculational simplicity of the lognormal assumption, a lognormal distribution having the same arithmetic mean and standard deviation as the normal distribution of uncertainty was constructed for this source and used for the purpose of computing the combined uncertainty. As seen in Table 7-10, the lognormal construct had upper and lower bounds shifted upward slightly with respect to the normal distribution, the geometric mean of the former (0.97) falling very close to the arithmetic mean of the latter (1.0).

Scaled to our estimate of the average risk from whole-body low-LET radiation, the upper and lower confidence limits, calculated as described above, span the range from 0.23 to 1.6. This corresponds to a range of 91 to 630 fatal cancers/10^6 person-rad. The geometric mean of the range is about 240 fatal cancers/10^6 person-rad, suggesting that our estimate of 395 fatal cancers/10^6 person-rad may be biased high slightly. However, our estimate falls well within the range of uncertainty, and we believe it represents a prudent and reasonable choice for the purposes of radiation protection.

Finally, it should be noted that the analysis above pertains to whole-body, low-LET radiation exposures. The uncertainties in risk to specific organs may be considerably larger. This is particularly important for internal emitters that concentrate in certain organs. Often the dose estimates for these radionuclides are more uncertain as well.

7.6 Other Radiation-Induced Health Effects

The earliest report of radiation-induced health effects was in 1896 (Mo67), and it dealt with acute effects in skin generally caused by very large x-ray exposures. Within the six-year period following, 170 radiation-reiated skin damage cases had been reported. Such injury, like many other acute effects, is the result of exposure to hundreds or thousands of rads. Under normal situations, environmental exposure does not cause such large doses, so possible acute effects will not need to be considered in assessing the risk to the general population from non-accidental radionuclide emissions.

Radiation-induced carcinogenesis was the first delayed health effect described: the first case was reported in 1902 (Vo02), and 94 cases of skin cancer and 5 of leukemia were reported by 1911 (Up75). Radiation-induced genetic changes were noted soon afterward. In 1927, H.J. Muller described x-ray-induced mutations in animals (in the insect, Drosophila) and in 1928, L.J. Stadler reported a similar finding in plants (Ki62). At about the same time, radiation effects on the developing embryo were observed. Case reports in 1929 showed a high rate of microcephaly (small head size) and central nervous system disturbance and one case of skeletal defects in children irradiated in utero (UNSCEAR69). These effects, at unrecorded but high exposures and at generally unrecorded gestational ages, appeared to produce central nervous system and eye defects similar to those reported in rats as early as 1922 (Ru50).

For purposes of assessing the risks of environmental exposure to radionuclide emissions, the genetic effects and in utero developmental effects are the only health hazards other than cancer that are addressed in this Background Information Document (BID).

7.6.1 Types of Genetic Harm and Duration of Expression

Genetic harm or the genetic effects of radiation exposure are those effects induced in the germ cells (eggs or sperm) of exposed individuals, which are transmitted to and expressed only in their progeny and future generations.

Of the possible consequences of radiation exposure, the genetic risk is more subtle than the somatic risk, since it does not affect the persons exposed, but relates only to subsequent progeny. Hence, the time scales for expression of the risk are very different. Somatic effects are expressed over a period on the order of a lifetime, while about 30 subsequent generations (about 1,000 yr) are needed for near complete expression of genetic effects. Genetic risk is incurred by fertile

people when radiation damages the nucleus of the cells which become their eggs or sperm. The damage, in the form of a mutation or a chromosome aberration, is transmitted to, and may be expressed in, a child conceived after the radiation exposure or in subsequent generations. However, the damage may be expressed only after many generations or, alternatively, it may never be expressed because of failure to reproduce or failure of the chance to reproduce.

EPA treats genetic risk as independent of somatic risk even though somatic risk may be caused by mutations in somatic cells because, whereas somatic risk is expressed in the person exposed, genetic risk is expressed only in progeny and, in general, over many subsequent generations. Moreover, the types of damage incurred often differ in kind from cancer and cancer death. Historically, research on genetic effects and development of risk estimates have proceeded independently of the research on carcinogenesis. Neither the dose response models nor the risk estimates of genetic harm are derived from data on studies of carcinogenesis.

Although genetic effects may vary greatly in severity, the genetic risks considered by the Agency evaluating the hazard of radiation exposure include only those "disorders and traits that cause a serious handicap at some time during lifetime" (NAS80). Genetic risk may result from one of several types of damage that ionizing radiation can cause in the DNA within gonidial cells or eggs and sperm. The types of damage usually considered are: dominant and recessive mutations in autosomal chromosomes, mutations in sex-linked (x-linked) chromosomes, chromosome aberrations (physical rearrangement or removal of part of the genetic message on the chromosome or abnormal numbers of chromosomes), and irregularly inherited disorders (genetic conditions with complex causes, constitutional and degenerative diseases, etc.).

Estimates of the genetic risk per generation are conventionally based on a 30-yr reproductive generation. That is, the median parental age for production of children is age 30 (one-half the children are produced by persons less than age 30, the other half by persons over age 30). Thus, the radiation dose accumulated up to age 30 is used to estimate the genetic risks. Using this accumulated dose and the number of live births in the population along with the estimated genetic risk per unit dose, it is possible to estimate the total number of genetic effects per year, those in the first generation and the total across all time. Most genetic risk analyses have provided such data. EPA assessment of risks of genetic effects includes both first generation estimates and total genetic burden estimates.

(A) Direct and Indirect Methods of Obtaining Risk Coefficients for Genetic Effects

Genetic effects, as noted above, may occur in the offspring of the exposed individuals or they may be spread across all succeeding

generations. Two methods have been used to estimate the frequency of mutations in the offspring of exposed persons, direct and indirect. In either case, the starting point is data from animal studies, not data obtained from studies of human populations. This is required since the human evidence available is inadequate to provide statistically valid estimates of the dose response relationship for radiation-induced mutations in humans.

For a direct estimate, the starting point is the frequency of a mutation per unit exposure in some experimental animal study. The 1982 UNSCEAR (UNSCEAR82) report gave an example of the direct method for estimating induction of balanced reciprocal translocations (a type of chromosomal aberration) in males per rad of low-level, low-LET radiation.

	Induction rate/rad
1) Rate of induction in rhesus monkey spermatogonia: cytogenetic data [x rays delivered at ∿30 rad/min]	8.6E-05
2) Rate of induction that relates to recoverable translocations in the F_1 (1st filial generation) progeny [divide (1) by 4] [based on mouse data, UNSCEAR 1977]	2.15E-05
3) Rate after low dose rate x rays: based on mouse cytogenetic observations [divide (2) by 2]	1.075E-05
4) Rate after chronic gamma-irradiation: based on mouse cytogenetic observations [divide (2) by 10]	2.2E-06
5) Expected rate of unbalanced products: [multiply (3) and (4) by 2] for (3):	2.15E-05
for (4):	4.3E-06
6) Expected frequency of congenitally malformed children in the F_1, assuming that about 6% of unbalanced products [item (5) above] contribute to this for low dose rate x rays:	1.3E-06
for chronic gamma radiation:	∿3E-07

For humans, UNSCEAR (UNSCEAR 82) estimates that as a consequence of induced balanced reciprocal translocations in exposed fathers, an estimated 0.3 to 1.3 congenitally malformed children would occur in each 10^6 live births for every rad of paternal low-level radiation exposure.

A complete direct estimate of genetic effects would include estimates derived in a manner similar to that shown above for each type of genetic damage. These direct estimates can be used to calculate the risk of genetic effects in the first generation (F_1) children of exposed parents.

The indirect (or doubling dose) method of estimating genetic risk also uses animal data but in a different way. The 1980 BEIR-3 report (NAS80) demonstrates how such estimates are obtained.

	Induction rate/rad
1) Average radiation-induced mutation per gene for both sexes in mice [based on 12 locus data in male mice adjusted to a chronic gamma radiation estimate]: induction rate per rad, observed in the F_1 generation	2.5E-08
2) Estimated human spontaneous mutation rate per gene	5E-07 5E-06
3) Relative mutation risk in humans [divide (1) by (2)]	0.005 to 0.05
4) Doubling dose: the exposure needed to double the human mutation rate	200 to 20 rads

The doubling dose can then be used to estimate the equilibrium genetic effects or the genetic burden in all future generations caused by the exposure of parents. Since the genetic component of congenital defects occurring in the population can be estimated by epidemiological surveys, and this component is considered to be maintained at an equilibrium level by mutations, a doubling dose of ionizing radiation would double these genetic effects. Dividing the number of the various genetic effects in 10^6 live-births by the doubling dose yields the estimate of genetic effects per rad. For example:

1) Autosomal dominant and x-linked diseases, current incidence	10,000 per 10^6 live births
2) Estimated doubling dose	20 to 200 rads
3) Estimate of induced autosomal dominant and x-linked diseases	50 to 500 per 10^6 live births per rad of parental exposure.

A doubling dose estimate assumes that the total population of both sexes is equally irradiated, as occurs from background radiation, and that the population exposed is large enough so that all genetic damage can be expressed in future offspring. Although it is basically an estimate of the total genetic burden across all future generations, it can also provide an estimate of effects that occur in the first generation. Usually a fraction of the total genetic burden for each type of damage is assigned to the first generation using population genetics data as a basis to determine the fraction. For example, the BEIR-3 committee geneticists estimated that one-sixth of the total genetic burden of x-linked mutations would be expressed in the first generation, five-sixths across all subsequent generations. EPA assessment of risks of genetic effects includes both first generation estimates and total genetic burden estimates.

7.6.2 Estimates of Genetic Harm Resulting from Low-LET Radiations

One of the first estimates of genetic risk was made in 1956 by the NAS Committee on the Biological Effects of Atomic Radiation (BEAR Committee). Based on Drosophila (fruit fly) data and other considerations, the BEAR Genetics Committee estimated that 10 roentgens (10 R*) per generation continued indefinitely would lead to about 5,000 new instances of "tangible inherited defects" per 10^6 births, and about one-tenth of them would occur in the first generation after the irradiation began (NAS72). The UNSCEAR addressed genetic risk in their 1958, 1962, and 1966 reports (UNSCEAR58, 62, 66). During this period, they estimated one rad of low-LET radiation would cause a 1 to 10 percent increase in the spontaneous incidence of genetic effects.

In 1972, both the NAS BEIR Committee (NAS72) and UNSCEAR (UNSCEAR72) reexamined the question of genetic risks. Although there were no definitive human data, additional information was available on the genetic effects of radiation on mammals and insects. In 1977, UNSCEAR reevaluated the 1972 genetics estimates (UNSCEAR77). These new estimates used recent information on the current incidence of various genetic conditions, along with additional data on radiation exposure of mice and marmosets and other considerations.

In 1980, an ICRP Task Group [ICRPTG] summarized recommendations that formed the basis for the genetic risk estimates published in ICRP Report 26 (Of80). These risk estimates are based on data similar to those used by the BEIR and UNSCEAR Committees, but with slightly different assumptions and effect categories (Table 7-11).

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* R is the symbol for roentgen, a unit of measurement of x-radiation exposure, equivalent to an absorbed dose in soft tissue of approximately 0.9 rad.

Table 7-11. ICRP Task Group estimate of number of cases of serious genetic ill health in liveborn from parents irradiated with 10^6 person-rem in a population of constant size[a] (Assumed doubling dose = 100 rad) [low level radiation exposure]

Category of genetic effect	First generation	Equilibrium
Unbalanced translocations: risk of malformed liveborn	23	30
Trisomics and XO	30	30
Simple dominants and sex-linked mutations	20	100
Dominants of incomplete penetrance and multifactorial disease maintained by mutation	16	160
Multifactorial disease not maintained by mutation	0	0
Recessive disease	--	--
Total	89	320

[a] This is equivalent to effects per 10^6 liveborn following an average parental population exposure of 1 rem per 30-yr generation, as used by BEIR and UNSCEAR.

Source: Of80.

The 1980 NAS BEIR Committee revised genetic risk estimates (NAS80). The revision considered much of the same material that was in BEIR-1 (NAS72), the newer material considered by UNSCEAR in 1977 (UNSCEAR77), and some additional data. Estimates for the first generation are about a factor of 2 smaller than those reported in the BEIR-1 report. For all generations, the new estimates are essentially the same (Table 7-12).

The most recent genetic risk estimate, in the 1982 UNSCEAR Report (UNSCEAR82), includes some new data on cells in culture and the results of genetic experiments using primates rather than rodents (Table 7-13).

Although all of the reports described above used somewhat different sources of information, there is reasonable agreement in the estimates. However, all these estimates have a considerable margin of error, both inherent in the original observations and in the extrapolations from experimental species to man. Some of the committee reports assessing the situation have attempted to indicate the range of uncertainty; others have simply used a central estimate. The same uncertainties exist for the latter (central estimates) as for the former (see Table 7-14). Most of the difference is caused by the newer information used in each report. Note that all of these estimates are based on the extrapolation of animal data to humans. Groups differ in their interpretation of how genetic experiments in animals might be expressed in humans. While there are no comparable human data at present, information on hereditary defects among the children of A-bomb survivors provides a degree of confidence that the animal data do not lead to underestimates of the genetic risk following exposure to humans. (See "Observations on Human Populations," which follows.)

It should be noted that the genetic risk estimates summarized in Table 7-14 are for low-LET, low-dose, and low-dose-rate irradiation. Much of the data was obtained from high dose rate studies, and most authors have used a sex-averaged factor of 0.3 to correct for the change from high-dose rate, low-LET to low-dose rate, low-LET exposure (NAS72, 80, UNSCEAR72,77). However, factors of 0.5 to 0.1 have also been used in estimates of specific types of genetic damage (UNSCEAR72,77,82).

(A) Beta Particles

Studies with the beta-particle-emitting isotopes carbon-14 and tritium yielded RBEs of 1.0 and 0.7 to about 2.0, respectively, in comparison to high-dose rate, high-dose exposure to x rays (UNSCEAR82). At the present time, the RBE for genetic endpoints due to beta particles is taken as one (UNSCEAR77,82).

7.6.3 Estimates of Genetic Harm from High-LET Radiations

Although genetic risk estimates are made for low-LET radiation, some radioactive elements, deposited in the ovary or testis, can irradiate the germ cells with alpha particles. The relative biological effectiveness

Table 7-12. BEIR-3 estimates of genetic effects of an average population exposure of 1 rem per 30-yr generation [chronic x-ray or gamma radiation exposure]

Type of genetic disorder	Current incidence per 10^6 liveborn	Effects per 10^6 liveborn per rem per generation	
		First generation*	Equilibrium**
Autosomal dominant and x-linked	10,000	5-65	40-200
Irregularly inherited	90,000	(not estimated)	20-900
Recessive	1,100	Very few	Very slow increase
Chromosomal aberrations	6,000	Fewer than 10	Increases only slightly
Total	107,100	5-75	60-1100

* First generation effects estimates are reduced from acute fractionated exposure estimates by a factor of 3 for dose rate effects and 1.9 for fractionation effects (NAS80, p. 117).

** Equilibrium effects estimates are based on low dose rate studies in mice (NAS80, pp. 109-110).

Source: NAS80.

Table 7-13. UNSCEAR 1982 estimated effect of 1 rad per generation of low-dose or low-dose rate, low-LET radiation on a population of 10^6 liveborn according to the doubling dose method (Assumed doubling dose = 100 rad) [low level, low-LET radiation]

Disease classification	Current incidence	Effect of 1 rad per generation	
		First generation	Equilibrium
Autosomal dominant and x-linked diseases	10,000	15	100
Recessive diseases increase	2,500	Slight	Slow
Chromosomal diseases			
Structural	400	2.4	4
Numerical	3,000	Probably very small	
Congenital anomalies, anomalies expressed later, constitutional and degenerative diseases	90,000	4.5	45
Total	105,900	22	149

Source: (UNSCEAR82).

Table 7-14. Summary of genetic risk estimates per 10^6 liveborn for an average population exposure of 1 rad of low-dose or low-dose rate, low-LET radiation in a 30-yr generation

Source	Serious hereditary effects	
	First generation	Equilibrium (all generations)
BEAR, 1956 (NAS72)	---	500
BEIR-I, 1972 (NAS72)	49[a] (12-200)	300[a] (60-1500)
UNSCEAR, 1972 (UNSCEAR72)	9[a] (6-15)	300
UNSCEAR, 1977 (UNSCEAR77)	63	185
ICRP, 1980 (Of80)	89	320
BEIR-3, 1980 (NAS80)	19[a] (5-75)	257[a] (60-1100)
UNSCEAR, 1982 (UNSCEAR82)	22	149

Numbers in parentheses are the range of estimates.

[a] Geometric Mean. The geometric mean of two numbers is the square root of their product; in general, it is the N^{th} root of the product of N numbers.

(RBE) of high-LET radiation, such as alpha particles, is defined as the ratio of the dose (rad) of low-LET radiation to the dose of high-LET radiation producing the same specific patho-physiological endpoint.

Studies of the RBE for alpha-emitting elements in germinal tissue have been carried out only with plutonium-239. Studies comparing cytogenetic endpoints after chronic low-dose-rate gamma radiation exposure, or incorporation of plutonium-239 in the mouse testis, have yielded RBEs of 23 to 50 for the type of genetic injury (reciprocal translocations) that might be transmitted to liveborn offspring (NAS80, UNSCEAR77,82). However, an RBE of 4 for plutonium-239 compared to chronic gamma radiation was reported for specific locus mutations observed in neonate mice (NAS80). Neutron RBE, determined from cytogenetic studies in mice, also ranges from about 4 to 50 (UNSCEAR82, Gr83a, Ga82). Most reports use an RBE of 20 to convert risk estimates for low-dose rate, low-LET radiation to risk estimates for high-LET radiation.

7.6.4 Uncertainty in Estimates of Radiogenetic Harm

Chromosomal damage and mutations have been demonstrated in cells in culture, in plants, in insects, and in mammals (UNSCEAR72,77,82). Chromosome studies in peripheral blood lymphocytes of persons exposed to radiation have shown a dose-related increase in chromosome aberrations (structural damage to chromosome) (UNSCEAR82). In a study of nuclear dockyard workers exposed to external x-radiation at rates of less than 5 rad/yr, Evans et al. (Ev79) found a significant increase in the incidence of chromosome aberrations in peripheral lymphocytes. The increase appeared to have a linear dependence on cumulative dose. In a study of people working and living in a high natural background area where there was both external gamma-radiation and internal alpha-radiation, Pohl-Ruling et al. (Po78) reported a complex dose response curve. For mainly gamma-radiation exposure (less than 10 percent alpha-radiation), they reported that chromosome aberrations increased linearly from 100 to 200 mrad/yr, plateaued from 300 mrad to 2 rad/yr, and then increased linearly again for doses above 2 rad/yr.

Although chromosomal damage in peripheral blood lymphocytes cannot be used for predicting genetic risk in progeny of exposed persons, it is believed by some to be a direct expression of the damage, analogous to that induced in germ cells, resulting from the radiation exposure. It is at least evidence that chromosome damage can occur *in vivo* in humans.

Since human data are so sparse, they can be used only to develop upper bounds of some classes of genetic risks following radiation exposure. Most numerical risk estimates are based on extrapolations from animal data. As genetic studies proceeded, emphasis shifted from *Drosophila* (fruit flies) to mammalian species in attempts to find an experimental system that would reasonably project what might happen in humans.

For example, Van Buul (Va80) reported the slope (b) of the linear regression, Y = a + bD, for induction of reciprocal translocations in spermatogonia (one of the stages of sperm development) in various species as follows:

	b x E+04 ± sd x E+04
Rhesus monkey	0.86 ± 0.04
Mouse	1.29 ± 0.02 to 2.90 ± 0.34
Rabbit	1.48 ± 0.13
Guinea Pig	0.91 ± 0.10
Marmoset	7.44 ± 0.95
Human	3.40 ± 0.72

These data indicate that animal-based estimates for this type of genetic effect would be within a factor of 4 of the true human value. In this case, most of the animal results would underestimate the risk in humans.

However, when risk estimates such as this are used in direct estimation of risk for the first generation, the total uncertainty in the estimate becomes indeterminate. Even if studies have been made in a species that can predict the dose response and risk coefficient for a specific radiation-induced genetic damage, there is no certainty that it predicts the response for all genetic damage of that type. In addition, as shown in the example from the 1982 UNSCEAR report (UNSCEAR82) in Section 7.6.1, additional assumptions based on observations, usually in other animal species, are used to adjust the risk coefficient to what is expected for humans. The uncertainty in these extrapolations has not been quantified.

A rough estimate of the uncertainty can be obtained by comparing direct estimates of risk for the first generation with doubling-dose estimates in the 1977 UNSCEAR report (UNSCEAR77). The estimates differ by a factor between 2 and 6, with the direct estimate usually smaller than the doubling-dose estimate.

A basic assumption in the doubling-dose method of estimation is that there is a proportionality between radiation-induced and spontaneous mutation rates. Some of the uncertainty was removed in the 1982 UNSCEAR report with the observation that in two-test systems (fruit flies and bacteria), there is a proportionality between spontaneous and induced mutation rates at a number of individual gene sites. There is still some question as to whether the sites that have been examined are representative of all sites and all gene loci or not. The doubling-dose estimate dose, however, seems better supported than the direct estimate.

While there is still some uncertainty as to which hereditary conditions would be doubled by a doubling dose, future studies on genetic conditions and diseases can only increase the total number of such conditions. Every report, from the 1972 BEIR and UNSCEAR reports to the most recent, has listed an increased number of conditions and diseases which have a genetic component and hence may be increased by exposure to ionizing radiations.

(A) Observations on Human Populations

As noted earlier, the genetic risk estimates are based on interpretation of animal experiments as applied to data on naturally-occurring hereditary diseases and defects in man. A study of the birth cohort consisting of children of the Japanese A-bomb survivors was initiated in mid-1946. In a detailed monograph, Neel and Schull (Ne56) outlined the background of this first study and made a detailed analysis of the findings to January 1954 when the study terminated. The study was designed to determine: (1) if during the first year of life, any differences could be observed in children born to exposed parents when compared to children born to suitable control parents, and (2) if differences existed, how should they be interpreted (Ne56). At the time the study started, there were data on spontaneous and radiation-induced mutation in Drosophila, but little was known concerning spontaneous mutation rates in mammals and less on the effects of radiation on mammals. The authors concluded that, based on the human data, it was improbable that human genes were so sensitive that exposures as low as 3 R, or even 10 R, would double the mutation rate.

While this first study addressed a number of endpoints, including sex ratio, malformations, perinatal data and anthropometric data, subsequent studies have addressed other endpoints. The most recent reports on this birth cohort of 70,082 persons have reported data on six endpoints. Frequency of stillbirths, major congenital defects, prenatal death, and frequency of death prior to age 17 have been examined in the entire cohort. Frequency of cytogenetic aberrations (sex chromosome aneuploidy) and frequency of biochemical variants (a variant enzyme or protein electrophoresis pattern) have been measured on large subsets of this cohort.

There are small but statistically insignificant differences between the number of effects in the children of the proximally and distally exposed with respect to these various indicators. These differences are in the direction of the hypothesis that mutations were produced by the parental exposure. Taking these differences then as the point of departure for an estimate of the human doubling dose, an estimated doubling dose for low-LET radiation at high doses and dose rates for human genetic effects of about 156 rem (Sc81) or 250 rem (Sa82) was obtained as an unweighted average. When each individual estimate was weighted by the inverse of its variance, an average of 139 rem was found (Sc84). Because of the assumptions necessary for these calculations, as

well as the inherent statistical errors, the errors associated with these estimates are rather large. As a result, a reasonable lower bound to the human estimate overlaps much of the range based on extrapolation from mouse data.

As noted above, animal studies indicate that chronic exposures to low-LET radiation would be less hazardous than acute exposures by a factor of about 3 (NAS72, 80). If applicable to the Japanese A-bomb survivors, this would increase the estimated doubling doses cited above to 468 rem, 750 rem, and 417 rem, respectively. These recent reports thus suggest the minimum doubling dose for humans may be 4 to 7 times higher than those in Table 7-14 (based on animal data). It would be premature to estimate the exact magnitude since these reports are based on the T65 dosimetry in Japan (see Section 7.2), which is being revised.

The EPA is using the geometric mean of the BEIR-3 range of doubling doses, about 110 rads. Although the best estimate of the minimum doubling dose derived from human data is 4 to 7 times greater than the EPA estimate, the 95 percent lower confidence limit averages about 70 rem. Therefore, EPA believes the estimate of doubling of about 100 rads is on the conservative side; however, it is compatible with both human and mouse data and should not be changed at this time. However, the EPA estimates of genetic risks will be reviewed and revised, if necessary, when the dosimetry of A-bomb survivors is revised.

(B) Ranges of Estimates Provided by Various Models

EPA has continued to follow the recommendations of the 1980 BEIR-3 and earlier committees and uses a linear nonthreshold model for estimating genetic effects, although, as pointed out by the 1982 UNSCEAR committee, a number of models other than linear ($Y = c + aD$) have been proposed: e.g., linear-quadratic ($Y = C + bD + eD^2$), quadratic ($Y = k + fD^2$), or even a power function ($Y = K + gD^h$)*.

Some data on specific genetic endpoints obtained with acute low-LET exposures are well described by a linear-quadratic function. Moreover, in some of these cases, it has been found that a reduction in dose rate (or fractionation of dose) produced a reduction in the quadratic term seen at high doses with little or no effect on the linear component. Such observations can be qualitatively explained, as previously discussed in reference to somatic effects (Section 7.2.2), in terms of the dual radiation action theory of Kellerer and Rossi (Ke72), as well as alternative theories, e.g., one involving enzyme saturation (Go80, Ru58).

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* Y is yield of genetic effects; D is radiation dose; c, C, k, and K are spontaneous incidence constants for genetic effects; and a, b, e, f, g, and h are the rate constants for radiation-induced genetic effects.

The linear model adopted by BEIR-3 and EPA incorporates a factor of 3 reduction in extrapolating results obtained with high acute exposures to low dose rates. For this reason, the predictions obtained with the model in the low dose region, are roughly consistent with what one would obtain with a linear-quadratic model based on the same data.

Most of the arguments for a nonlinear dose response have been based on target theory (Le62) or microdosimetry site theory (Ke72, NAS80). However, other theories based on biology [e.g., enzyme induction-saturation (Go80,82), repair-misrepair (To80)] could also provide models that fit the observed data. There is still much disagreement on which dose response model is appropriate for estimating genetic effects in humans. Until there is consensus, EPA will continue to use the linear nonthreshold model.

Even though genetic risk estimates made by different committees based on the linear non-threshold model vary, the agreement is reasonably good. While the authors of the reports used different animal models, interpreted them in different ways, and had different estimates of the level of human genetic conditions in the population, the range of risk coefficients is about an order of magnitude (see Table 7-14). For the most recent, more comparable estimates, the range is a factor of 2 to 4 (see ICRP, BEIR-3, and UNSCEAR 1982 in Table 7-14).

7.6.5 The EPA Genetic Risk Estimate

There is no compelling evidence for preferring any one set of the genetic risk estimates listed in Table 7-14. EPA has used the estimates from BEIR-3 (NAS80). These "indirect" estimates are calculated using the normal prevalence of genetic defects and the dose that is considered to double this risk. The NAS estimates that EPA uses are based on a "doubling dose" range with a lower bound of 50 rem and an upper bound of 250 rem. We prefer these risk estimates to those made by the ICRP task group (Of80), which used "direct" estimates for some types of genetic damage with doubling-dose estimates for others. We also prefer the BEIR-3 risk estimates to the "direct" estimates of UNSCEAR 1982, which tabulates genetic risk separately by the direct method and by the doubling-dose method.

Our reasons are as follows: mutation rates for all gene loci affected by ionizing radiation are not known nor have loci associated with "serious" genetic conditions been identified. Therefore, the risk estimated by the direct method, at this time, is incomplete, does not include the same types of damage estimated by doubling doses, and was not considered further. Moreover, the BEIR-3 genetic risk estimates provide a better estimate of uncertainty than the UNSCEAR 1982 and ICRPTG estimates because the BEIR-3 Committee assigned a range of uncertainty for multifactorial diseases (>5 percent to <50 percent) that reflects the uncertainty in the numbers better than the other estimates (5 percent and 10 percent, respectively).

In developing the average mutation rate for the two sexes used in the calculation of the relative mutation risk, the BEIR-3 Committee postulated that the induced mutation rate in females was about 40 percent of that in males (NAS80). Recent studies by Dobson et al. suggest that the assumption was invalid and that human oocytes should have a risk equivalent to that of human spermatogonia. This would increase the risk estimate obtained from doubling-dose methods by a factor of 1.43 (Do83a, Do83b, Do84a, Do84b).

We recognize, however, that the use of the doubling-dose concept does assume that radiation-induced genetic damage is in some way proportional to "spontaneous" damage. As noted earlier, the recent evidence obtained in insects (Drosophila) and bacteria (E. coli) supports the hypothesis that, with the exception of "hot spots" for mutation, the radiation-induced mutation rate is proportional to the spontaneous rate (UNSCEAR82). No proof that this is also true in mammals is available yet.

The BEIR-3 estimates for low-LET radiations give a considerable range. To express the range as a single estimate, the geometric mean of the range is used, a method first recommended by UNSCEAR (UNSCEAR58) for purposes of calculating genetic risk. The factor of 3 increase in risk for high-dose rate, low-LET radiation, noted earlier, is also used.

The question of RBE for high-LET radiation is more difficult. As noted above, estimated RBEs for plutonium-239 alphas versus chronic gamma radiation for reciprocal translocations as determined by cytogenetic analyses are between 23 and 50 (NAS80, UNSCEAR82). However, the observed RBE for single locus mutations in developing offspring of male mice given plutonium-239 compared to those given chronic gamma irradiation is 4 (NAS80). The average of RBEs for reciprocal translocations and for specific locus mutations is 20. Since reported neutron RBEs are similar to those listed above for plutonium-239 alpha radiation, we use an RBE of 20 to estimate genetic risks for all high-LET radiations. This is consistent with the RBE for high-LET particles recommended for estimated genetic risks associated with space flight (Gr83b).

Genetic risk estimates used by EPA for high- and low-LET radiations are listed in Table 7-15. As noted above, EPA uses the dose received before age 30 in assessing genetic risks.

The EPA estimates in Table 7-15 are limited, like all other human genetic risk estimates, by the lack of confirming evidence of genetic effects in humans. These estimates depend on a presumed resemblance of radiation effects in animals to those in humans. The magnitude of the possible error is indeterminable. The largest source of data, the Japanese A-bomb survivors, appears, at best, to provide only an estimate of the minimum doubling dose for calculating the maximum genetic risk in man. However, doubling-dose estimates are also uncertain since the number of human disorders having a recognized genetic component is constantly increasing, and the type of genetic damage implicated in a

Table 7-15. Estimated frequency of genetic disorders in a birth cohort due to exposure of the parents to 1 rad per generation

Radiation	Serious heritable disorders (Cases per 10^6 liveborn)			
	First generation		All generations	
	low[a]	high[b]	low[a]	high[b]
Low Dose Rate, Low-LET	20	30	260	370
High Dose Rate, Low-LET	60	90	780	1110
High-LET	400	600	5200	7400

[a] Female sensitivity to induction of genetic effects is 40 percent as great as that of males.

[b] Female sensitivity to induction of genetic effects is equal to that of males.

specific disorder may change. The combined uncertainties in doubling-dose estimates and the magnitude of genetic contributions to various disorders probably introduce an overall uncertainty of about an order of magnitude in the risk estimates. Moreover, the BEIR Committee in deriving its estimate has assumed that almost all of the risk was due to recessive mutations which would eventually be eliminated. To what extent this occurs will depend on medical practices in the future. It is possible, as our knowledge of medicine improves, that recessive hereditary defects will be carried on for many more generations than assumed by the BEIR Committee.

The relative risk of high-LET radiation compared to low-dose-rate, low-LET radiation (RBE) is also uncertain. The data are sparse, and different studies often used different endpoints. In addition, the microscopic dosimetry, i.e., the actual absorbed dose in the cells at risk, is poorly known. However, the RBE estimate used by EPA should be within a factor of 5 of the true RBE for high-LET radiation.

7.6.6 Effects of Multigeneration Exposure

As noted earlier, while the somatic effects, i.e., cancer, occur in persons exposed to ionizing radiation, the genetic effects occur in progeny, perhaps generations later. The number of effects appearing in the first generation is based on direct estimates of the mutations induced by irradiation and should not change appreciably regardless of the background or "spontaneous" mutation rate in the exposed population. The estimate for total genetic effects, or the equilibrium estimate, is based on the doubling-dose concept. For these estimates, the background mutation rate is important: it is the background rate that is being "doubled."

If there is long-lived environmental contamination, such that 30 generations or more are exposed (≈1000 years), the background mutation rate will change and come into equilibrium with the new level of radiation background. There will be an accumulation of new radiation-induced mutations until the background mutation rate has reached equilibrium with this continued insult.

While predicting 1,000 yr in the future is chancy, at best, if it is assumed that there are no medical advances, and no changes in man or his environment, then an estimate can be made. In Table 7-15, it is estimated that exposure to 1 rad per generation of low-dose-rate low-LET radiation will induce 260 cases of serious heritable disorders per 10^6 live births in all generations. This is for a background mutation rate leading to 29,120 cases of serious heritable disorders per 10^6 live births. The "all generations" estimate in Table 7-15 is equal to the "equilibrium" estimate in Table 7-12. The "all generations" estimate is used for exposures to a single generation; the same number is employed as the "equilibrium" estimate for multigeneration exposures (see NAS80, p. 126, note 16). Thus, the risk estimate can be reexpressed as an

estimate of the effects expected for a given change in the level of background radiation (Table 7-16). Since these calculations are based both on the background level mutations and the doubling dose, changes in either must be reflected in new calculations.

7.6.7 Uncertainties in Risk Estimates for Radiogenic Genetic Effects

As noted throughout the preceding sections, there are sources of uncertainty in the genetic risk estimates. The overall uncertainty can be addressed only in a semi-quantitative manner. The identified sources of uncertainty are listed in Table 7-17. Uncertainties listed in Table 7-17 are likely to be independent of each other and therefore unlikely to be correlated in sign. Although the root mean square sum of the numerical uncertainties suggests the true risk could be a factor of 4 higher or lower [x/÷ by a factor of 4], it is unlikely in light of the Japanese A-bomb survivor data that the upper bound is correct.

7.6.8 Teratogenic Effects

Although human teratogenesis (congenital abnormalities or defects) associated with x-ray exposure has a long history, the early literature deals mostly with case reports. Stettner reported a case in 1921 (St21) and Murphy and Goldstein (Mu29, Go129) studied a series of pregnancies in which 18 of the children born to 76 irradiated mothers had microcephaly (reduced head circumference). However, the irradiation exposures were high.

In 1930, Murphy exposed rats to x rays at doses of 200 R to 1600 R. Thirty-four of 120 exposed females had litters, and five of the litters had animals with developmental defects (Mu30). He felt that this study confirmed his clinical observations and earlier reports of animal studies. Although there were additional studies of radiation-induced mammalian teratogenesis before 1950, the majority of the studies were done after that time (see Ru53 for a review), perhaps reflecting concerns about radiation hazards caused by the explosion of nuclear weapons in 1945 (Ja70).

Much of the work done after World War II was done using mice (Ru50, Ru54, Ru56) and rats (Wi54, Hi54). Early studies, at relatively high radiation exposures, 25 R and above, established some dose response relationships. More important, they established the timetable of sensitivity of the developing rodent embryo and fetus to radiation effects (Ru54, Hi53, Se69, Hi66).

Rugh, in his review of radiation teratogenesis (Ru70), listed the reported mammalian anomalies and the exposure causing them. The lowest reported exposure was 12.5 R for structural defects and 1 R for functional defects. He also suggested human exposure between ovulation

Table 7-16. Increase in background level of genetic effects after 30 generations or more

Increase in background radiation (mrad/y)	Increase in serious heritable disorders per 10^6 live births	
	Low-dose-rate, low-LET radiation	High-LET radiation
0.1	0.8	16
1.0	8	160
10.0	80	1600

Table 7-17. A list of the causes of uncertainty in the genetic risk estimates

Source of uncertainty	Degree of uncertainty in risk estimates
Selection of species to use in developing a direct estimate	x/÷, a factor of 4
Selection of species and loci to use in developing a doubling-dose estimate	-100% to +indeterminate [a]
Use of - division by a factor of 3 - to convert acute, high-dose low-LET estimates to chronic low-LET estimates	x/÷, a factor of 3
Sensitivity of oogonia compared to spermatogonia as described in BEIR-3	0.5[b,d] -1.0[c,d]
Background rate selected for use with a doubling dose	+/-, indeterminate
Selection of RBE for high-LET radiation compared to an RBE of 20	x/÷, a factor of 5
Underestimate of the doubling dose required in man	÷, a factor of 3[e]

[a] The risk estimate cannot go below zero, -100%, but it may not be possible to determine the upper bound; indeterminate.

[b] Assumes no radiation-induced mutations from oocytes.

[c] Assumes equal radiation-induced mutations from oocytes and spermatocytes.

[d] In reference to the high estimate in Table 7-15.

[e] If the most recent analysis of the Japanese A-bomb survivors is correct, the lower bound for an estimate of the doubling dose in man is at least 3 times greater than the average doubling dose in the mouse.

and about 7 weeks gestational age could lead to structural defects, and exposures from about 6 weeks gestational age until birth could lead to functional defects. In a later review (Ru71), Rugh suggested structural defects in the skeleton might be induced as late as the 10th week of gestation and functional defects as early as the 4th week. It should be noted that the gestation period in mice is much shorter than that in humans and that weeks of gestation referred to above are in terms of equivalent stages of mouse-human development. However, estimates of equivalent gestational age are not very accurate.

Rugh (Ru71) suggested there may be no threshold for radiation-induced congenital effects in the early human fetus. In the case of human microcephaly (small head size) and mental retardation, at least, there are some data to support this theory (Ot83, 84). However, for most teratogenic effects, the dose response at low doses is not known. In 1978, Michel and Fritz-Niggli (Mi78) reported induction of a significant increase in growth retardation, eye and nervous system abnormalities, and post-implantation losses in mice exposed to 1 R. The increase was still greater if there was concurrent exposure to radiosensitizing chemicals such as iodoacetimide or tetracycline (Mi78).

In other reports of animal studies it appeared as if teratologic effects, other than perhaps growth retardation, had a threshold for induction of effects (Ru54, Ru53, Wi54). However, Ohzu (Oh65) showed that doses as low as 5 R to preimplantation mouse embryos caused increased resorption of implanted embryos and structural abnormalities in survivors. Then in 1970, Jacobsen (Ja70) reported a study in which mice were exposed to 5, 20, or 100 R on the 8th day of pregnancy. He concluded that the dose response function for induction of skeletal effects was linear, or nearly linear, with no observable threshold. This appears consistent with a report by Russell (Ru57), which suggested a threshold for some effects whereas others appeared to be linearly proportional to dose.

One of the problems with the teratologic studies in animals is the difficulty of determining how dose response data should be interpreted. Russell (Ru54) pointed out some aspects of the problem: (1) although radiation is absorbed throughout the embryo, it causes selective damage that is consistently dependent on the stage of embryonic development at the time of irradiation, and (2) the damaged parts respond, in a consistent manner, within a narrow time range. However, while low-dose irradiation at a certain stage of development produces changes only in those tissues and systems which are most sensitive at that time, higher doses may induce additional abnormalities in components which are most sensitive at other stages of development, and may further modify expression of the changes induced in parts of the embryo at maximum sensitivity during the time of irradiation. In the first case, damage may be to primordial cells themselves, while in the second, the damage may lead indirectly to the same or different endpoints.

The human embryo/fetus starts as a single, fertilized egg and divides and differentiates to produce the normal infant at term. (The embryonic period, when organs develop, is the period from conception through 7 weeks gestational age. The fetal period, a time of in utero growth, is the period from 8 weeks gestational age to birth.) The different organ and tissue primordia develop independently and at different rates. However, they are in contact through chemical induction or evocation (Ar54). These chemical messages between cells are important in bringing about orderly development and the correct timing and fitting together of parts of organs or organisms. While radiation can disrupt this pattern, interpretation of the response may be difficult. Since the cells in the embryo/fetus differentiate, divide, and proliferate at different times during gestation and at different rates, gestational times when cells of specific organs or tissues reach maximum sensitivity to radiation are different. Each embryo/fetus has a different timetable. In fact, each half (left/right) of an embryo/fetus may have a slightly different timetable.

In addition, there is a continuum of variation from the hypothetical normal to the extreme deviant which is obviously recognizable. There is no logical place to draw a line of separation between normal and abnormal. The distinction between minor variations of normal and frank malformation, therefore, is an arbitrary one, and each investigator must establish his or her own criteria and apply them to spontaneous and induced abnormalities alike (HWC73).

The limitations of the human data available make the use of animals in both descriptive and experimental studies inevitable. However, this gives rise to speculation about the possible relevance of such studies to man. There are species differences in development attributable partly to the differing complexity of the adult organs, but especially to differences in growth rates and timing of birth in relation to the developmental events. For example, the histological structure of the brain is, in general, surprisingly similar, both in composition and in function, from one mammalian species to another; and the sequence of events is also similar (Ref.). However, the processes of brain development that occur from conception to about the second year of life in man are qualitatively similar to those seen in the rat during the first six weeks after conception (Do79, 81).

For example, a major landmark, the transition from the principal phase of multiplication of the neuronal precursors to that of glial multiplication, occurs shortly before midgestation in man, but at about the time of birth in the rat (Do73). In this respect, then, the rat is much less neurologically mature at birth than the newborn human infant. Many other species are more mature at birth; the spectrum ranges from the late-maturing mouse and rat to the early-maturing guinea pig, with non-human primates much closer to the guinea pig than to man (Do79, 81). As a consequence, it is unreasonable to compare a newborn rat's brain, which has not begun to myelinate (Do79, 81), with that of a newborn

human, which has (Do79, 81), or with that of a newborn guinea pig in which myelination has been completed (Do79, 81).

Nevertheless, in the study of teratogenic effects of prenatal exposure to ionizing radiation, in which the timing of the exposure in relation to the program of developmental events dictates the consequences of that insult, it is only necessary to apply the experimental exposure at the appropriate stage (rather than at a similar age) of embryonic or fetal development in any species to produce similar results in all (Do79, 81). The duration of exposure must, however, match the different time scales in the different species. Unless these elementary rules of cross-species adjustments are followed, extrapolation of even qualitative estimates of effects will be of dubious relevance and worth.

Because of the problems in interpretation listed above, a pragmatic approach to evaluation of studies is useful. The dose response should be given as the simplest function that fits the data (often linear or linear with a threshold). No attempt should be made to develop complex dose response models unless the evidence is unequivocal.

(A) Teratologic Effects: Mental Retardation in Humans

The first report of congenital abnormalities in children exposed in utero to radiation from atomic bombs was that of Plummer (P152). Twelve children with microcephaly, of which 10 also had mental retardation, had been identified in Hiroshima in a small set of the in utero exposed survivors. They were found as part of a program started in 1950 to study children exposed in the first trimester of gestation. However, not all of the in utero exposed survivors were examined. In 1955, the program was expanded to include all survivors exposed in utero.

Studies initiated during the program have shown radiation-related (1) growth retardation; (2) increased microcephaly; (3) increased mortality, especially infant mortality; (4) temporary suppression of antibody production against influenza; and (5) increased frequency of chromosomal aberrations in peripheral lymphocytes (Ka73).

Although there have been a number of studies of Japanese A-bomb survivors, including one showing a dose- and gestational age-related increase in postnatal mortality (Ka73), only the incidences of microcephaly and mental retardation have been investigated to any great extent. In the most recent report, Otake and Schull (Ot83, 84) showed that mental retardation was particularly associated with exposure between 8 and 15 weeks of gestation (10 to 17 weeks of gestation if counted from the last menstrual period). They further found the data suggested little, if any, non-linearity and were consistent with a linear dose-response relationship for induction of mental retardation that yielded a probability of occurrence of severe mental retardation of 4.16 ± 0.4 cases per 1,000 live births per rad of exposure (Ot84). A child was classified as severely mentally retarded if he or she was "unable to

perform simple calculations, to make simple conversation, to care for himself or herself, or if he or she was completely unmanageable or had been institutionalized" (Ot83, 84). There was, however, no evidence of an effect in those exposed at 0 to 7 weeks of gestation (Ot83). Exposure at 16 weeks or more of gestation was about a factor of 4 less effective, with only a weak relationship between exposure and risk, and with few cases below 50 rads exposure (Ot84).

Mental retardation can be classified as mild (IQ 50-70), moderate (IQ 35-49), severe (IQ 20-34), and profound (IQ<20) (WHO75). However, some investigators use only mild mental retardation (IQ 50-70) and severe mental retardation (IQ<50) as classes (Gu77b, Ha81a, St84). Mental retardation is not usually diagnosed at birth but at some later time, often at school age. Since the mental retardation may have been caused before or during gestation, at the time of birth, or at some time after birth, that fraction caused before or during gestation must be estimated. In like manner, since mental retardation caused before birth may be due to genetic conditions, infections, physiologic conditions, etc., the fraction related to unknown causes during gestation must be estimated. This is the fraction that might possibly be related to radiation exposure.

Estimates of the risk of mental retardation for a rad of embryo/fetus exposure in the U.S. population can be derived using the absolute risk calculated by Otake and Schull for the Japanese survivors (Ot84). Otake and Schull (Ot84) gave an estimate for one case entitled, "The Relationship of Mental Retardation to Absorbed Fetal Exposure in the 'Sensitive' Period When All 'Controls' Are Combined." This estimate of frequency of mental retardation, 0.416 per 100 rads, could be directly applicable to a U.S. population. In this case, the risk estimate would be about:

> Four cases of severe mental retardation per 1,000 live births per rad of exposure during the 8th and 15th week of gestation.

Data on mental retardation in school age populations in developed countries suggest a prevalence of 2.8 cases/1,000 (Uppsala County, Sweden) to 7.4 cases/1,000 (Amsterdam, Holland) of severe mental retardation, with a mean of about 4.3 $\pm$ 1.3 cases/1,000 (St84). Where data are available for males and females separately, the male rate is about 30 percent higher than the female rate (St84). Historically, the prevalence of mild mental retardation has been 6 to 10 times greater than that of severe mental retardation. However, in recent Swedish studies, the rates of prevalence of mild and severe mental retardation have been similar (St84). This was suggested to be due to a decline in the "cultural-familial syndrome." That is, improved nutrition, decline in infection and diseases of childhood, increased social and intellectual stimulation, etc., combined to reduce the proportion of nonorganic mental retardation and, therefore, the prevalence of mild mental retardation (St84).

In studies of the causes of mental retardation, 23 to 42 percent of the cases have no identified cause (Gu77a, Ha81b, St84). It is this (idiopathic) portion of the mental retardation that may be susceptible to increase from radiation exposure of the embryo/fetus and should be used as the "background" incidence for comparison with radiation-induced effects. In that case, the prevalence of idiopathic mental retardation would be 0.6 to 3.1 cases per 1,000 of severe mental retardation and perhaps an equal number of cases of mild mental retardation. This estimate may be biased low because mental retardation induced during gestation is often associated with a high childhood death rate (St84). If this is generally true for idiopathic causes of mental retardation, it would cause an underestimation of the risk.

The risk of increased mental retardation per rad of embryo/fetus exposure during the 8- to 15-week gestational period estimated as 4×10^{-3} cases per live birth, compares with an earlier UNSCEAR (UNSCEAR77) estimate of 1×10^{-3} excess cases of mental retardation per rad per live birth. The UNSCEAR estimate, however, did not consider gestational age at the time of exposure. The Otake and Schull report (Ot84) did address gestational age and estimated a higher risk, but with what appears to be a narrower window of maximum susceptibility.

If the estimate is applicable, the low-LET background radiation (about 15 mrads) delivered during the 8- to 15-week gestational age-sensitive period could induce a risk of 6×10^{-5} cases of severe mental retardation per live birth. This can be compared to an estimate of a spontaneous occurrence of 0.6×10^{-3} to 3.1×10^{-3} cases of idiopathic severe mental retardation per live birth.

(B) Teratologic Effects: Microcephaly in Humans

Plummer (Pl52) reported microcephaly associated with mental retardation in Japanese A-bomb survivors exposed in utero. Wood (Wo65, 66) reported both were increased. The diagnosis of reduced head circumference was based on "normal distribution" statistical theory (Wo66); i.e., in a population, the probability of having a given head circumference is expected to be normally distributed around the mean head circumference for that population.

For example, in a population of live born children, 2.275 percent will have a head circumference 2 standard deviations or more smaller than the mean, 0.621 percent will have a head circumference 2.5 standard deviations or more smaller than the mean, and 0.135 percent will have a head circumference 3 standard deviations or more smaller than the mean (statistical estimates based on a normal distribution).

For most of the studies of the Japanese A-bomb survivors exposed in utero, if the head circumference was two or more standard deviations smaller than the mean for the appropriate controls in the unexposed population, the case was classified as having reduced head circumference

even if the data had not been adjusted for differences in stature (Ta67, Mi72, Wo65). While a definitive relationship between reduced head circumference and mental retardation has not been established, there is evidence that they are related.

For example, there is evidence in a nonselected group of 9,379 children that mental retardation may be estimated using incidence of microcephaly, even though head circumference in the absence of other supporting data, e.g., height or proportion, is an uncertain indicator of mental retardation. Based on this study of 9,379 children, Nelson and Deutschberger (Ne70) concluded that about half of the children with a head circumference 2.5 standard deviations or more smaller than average had IQs of 79 or lower. Since 0.67 percent of those studied were in this size group, the observed number is about what would be expected based on a normal distribution of head size in a population (0.62 percent). The estimated incidence of mental retardation per live birth in a population would then be:

$$(6.7 \text{ cases of microcephaly per } 1{,}000 \text{ live births}) \times 0.5 \frac{\text{case of mental retardation}}{\text{case of microcephaly}}$$

or about 3.4 cases of mental retardation per 1,000 live births. This might be divided roughly into 1.7 cases each of mild and severe mental retardation.

Studies of the Japanese survivors also show a relationship between reduced head size and mental retardation, but all these studies are based on subsets of the total in utero population. The fraction of mentally retarded with reduced head circumference has been reported as 50 percent (RERF78) to 70 percent (Wo66). While the fraction of those selected for reduced head circumference who had mental retardation has been reported as 11 percent (Wo66) to 22 percent (Mi72). Thus, while the relationship appears present, the quantitative relationship is not known.

The majority of the cases of reduced head size are observed in those exposed in the first trimester of gestation, particularly the 6th or 7th to 15th weeks of gestation (Mi59, Wo66, Mi72, Wo65, Ta67). Most recently, it has been shown that reduction in head circumference was a linear function of dose (Is84). However, the authors noted that the analysis was based on T65 dosimetry, and the data should be reanalyzed after completion of the dosimetry reassessment currently in progress.

These findings of reduction in head circumference, with a window of effect in the same time period of gestation as mental retardation, help support the observations on mental retardation. Although the exact dose response functions are still uncertain, data on both types of effects have so far been consistent with a linear, no-threshold dose response.

(C) Teratologic Effects: Other

Japanese A-bomb survivors exposed in utero also showed a number of structural abnormalities and, particularly in those who were microcephalic, retarded growth (Wo65). No estimate has been made of the radiation-related incidence or dose-response relationships for these abnormalities. However, UNSCEAR (UNSCEAR77) made a very tentative estimate based on animal studies that the increased incidence of structural abnormalities in animals may be 5E-03 cases per R per live born, but stated that projection to humans was unwarranted. In any event, the available human data cannot show whether the risk estimates derived from high-dose animal data overestimates the risk in humans.

It should be noted that all of the above estimates are based on high-dose-rate, low-LET exposure. In 1977, UNSCEAR also investigated the dose rate question and stated:

> "In conclusion, the majority of the data available for most species indicate a decrease of the cellular and malformature effects by lowering the dose rate or by fractionating the dose. However, deviations from this trend have been well documented in a few instances and are not inconsistent with the knowledge about mechanisms of the teratogenic effects. It is therefore impossible to assume that dose rate and fractionation factors have the same influence on all teratological effects." (UNSCEAR77).

From this analysis, EPA has concluded that there is risk of 4E-03 cases of mental retardation per live birth per rad of low-LET radiation delivered between weeks 8 and 15 of gestation with no threshold identified at this time.

No attempt can be made now to estimate total teratogenic effects. However, it should be noted that the 1977 UNSCEAR estimate from animals was 5E-03 cases of structural abnormalities per R per live birth (about the same number per rad of low-LET radiation). This estimate must be viewed as a minimum one since it is based, to a large extent, on observation of grossly visible malformations. Differences in criteria for identifying malformations have compounded the problem, and questions of threshold and species differences have made risk projection to humans unwarranted.

7.6.9 Nonstochastic Effects

Nonstochastic effects, those effects that increase in severity with increasing dose and have a threshold, have been reviewed in the 1982 UNSCEAR report (UNSCEAR82). In general, acute doses of 10 rads low-LET radiation and higher are required to induce these effects. It is possible that some of the observed effects of in utero exposure are nonstochastic, e.g., the risk of embryonic loss, estimated to be 10^{-2}

per R (UNSCEAR77), following radiation exposure soon after fertilization. However, there are no data to address the question. Usually, nonstochastic effects are not expected at environmental levels of radiation exposure.

7.7 Radiation Risk - A Perspective

To provide a perspective on the risk of fatal radiogenic cancers and the hereditary damage due to radiation, we have calculated the risk from background radiation to the U.S. population using the risk coefficients presented in this chapter and the computer codes described in Appendix E. The risk resulting from background radiation is a useful perspective for the risks caused by emissions of radionuclides. Unlike cigarette smoking, auto accidents, and other measures of common risks, the risks resulting from background radiation are neither voluntary nor the result of self-induced damage. The risk caused by background radiation is largely unavoidable; therefore, it is a good benchmark for judging the estimated risks from radionuclide emissions. Moreover, to the degree that the estimated risk of radionuclides is biased, the same bias is present in the risk estimates for background radiation.

The radiation dose equivalent rate from low-LET background radiation has three major components: cosmic radiation, which averages to about 28 mrad/yr in the U.S.; terrestrial sources, such as radium in soil, which contributes an average of 26 mrad/yr (NCRP75); and the low-LET dose resulting from internal emitters. The last differs among organs, to some extent, but for soft tissues it is about 24 mrad/yr (NCRP75). Other, minor radiation sources such as fallout from nuclear weapons tests, naturally-occurring radioactive materials in buildings, and consumer products, contribute about another 10 mrad for a total low-LET whole-body dose of about 90 mrad/yr. The lung and bone receive somewhat larger doses, not included in the 90 mrad/yr estimate, due to high-LET radiations; see below. Although extremes do occur, the distribution of this background annual dose to the U.S. population is relatively narrow. A population weighted analysis indicates that 80 percent of the U.S. population would receive annual doses that are between 75 mrad/yr and 115 mrad/yr (EPA81).

As outlined in Section 7.2, the BEIR-3 linear, relative risk models yield, for lifetime exposure to low-LET radiation, an average lifetime risk of fatal radiogenic cancer of 395 per 10^6 person-rad. Note that this average is for a group having the age- and sex-specific mortality rates of the 1970 U.S. population. We can use this risk estimate to calculate the average lifetime risk due to low-LET background radiation as follows. The average duration of exposure in this group is 70.7 yr, and at 9E-02 rad/yr, the average lifetime dose is 6.36 rads. The risk of fatal cancer per person in this group is:

$$\frac{395 \text{ fatalities}}{10^6 \text{ person-rad}} \times 6.36 \text{ rem} = 2.5 \times 10^{-3}$$

or about 0.25 percent of all deaths. The vital statistics we use in our radiation risk analyses indicate that the probability of dying from cancer in the United States from all causes is about 0.16, i.e., 16 percent. Thus, the 0.25 percent result for the BEIR-3 linear dose response model indicates that about 1.6 percent of all U.S. cancer is due to low-LET background radiation. The BEIR-3 linear-quadratic model indicates that about 0.1 percent of all deaths are due to low-LET background radiation or about 0.6 percent of all cancer deaths.

Table 7-6 indicates a risk of 466 fatalities per 10^6 organ rad for alpha emitters in lung tissue. UNSCEAR estimated in "normal" areas the absorbed dose from alpha emitters, other than radon decay products, in the lungs would be about 0.51 mrad (UNSCEAR77). The individual lifetime cancer risk from this exposure is:

$$\frac{395}{280} \times \frac{460 \text{ fatalities}}{10^6 \text{ organ rad}} \times \frac{5.1 \times 10^{-4} \text{ rad}}{\text{yr}} \times 70.7 \text{ yr} = 2.4 \times 10^{-5}.$$

This is about 1/100 of the risk due to low-LET background radiation calculated by means of the BEIR-3 linear model.

The 1982 UNSCEAR report indicates that the average annual dose to the endosteal surfaces of bone due to naturally occurring, high-LET alpha radiation is about 6 mrad/yr or, based on a quality factor of 20, 120 mrem/yr (UNSCEAR82). Table 7-6 indicates that the individual lifetime risk of fatal bone cancer due to this portion of the naturally occurring radiation background is:

$$\frac{395}{280} \times \frac{19 \text{ cases}}{10^6 \text{ person-rad}} \times \frac{0.006 \text{ rad}}{\text{year}} \times 70.7 \text{ years} = 1.1 \times 10^{-5}.$$

The exposure due to naturally occurring background radon-222 progeny in the indoor environment is not well known. The 1982 UNSCEAR report lists for the U.S. an indoor concentration of about 0.004 working levels (15 Bq/m^3) (UNSCEAR82). This estimate is not based on a national survey and is known to be exceeded by as much as a factor of 10 or more in some houses. However, as pointed out in UNSCEAR82, the national collective exposure may not be too dependent on exceptions to the mean concentration. The UNSCEAR estimate for the U.S. now appears low (Ne86); the average residential exposure is probably 0.2-0.3 WLM/yr (in standard exposure units).

Assuming 0.25 WLM/yr is a reasonable estimate for indoor exposure to radon-222 progeny in the U.S., the mean lifetime exposure, indoors, is about 18 WLM. Based on the geometric mean lifetime risk coefficient from Section 7.4.4, 460 cases/10^6 WLM, a lifetime risk of 0.83 percent is estimated. For comparison, roughly 5 percent of all deaths in 1980 were

due to lung cancer. Based on these assumptions, therefore, about one out of six lung cancer deaths may be attributable to background radon exposure. This would correspond to about 4 percent of all cancer deaths and 0.8 percent of all deaths. We note that this is comparable to the 1 percent cancer fatality incidence estimated above for low-LET background radiation. The reader is cautioned, however, that this risk estimate only applies to the U.S. population taken as a whole, i.e., men and women, smokers and nonsmokers. While we believe it is a reasonable estimate for the U.S. 1980 population in which the vast majority of the lung cancer mortality occurred in male smokers, we do not believe this risk estimate can be applied indiscriminately to women or nonsmokers. As noted in Section 7.4, the risk to these groups may not be comparable.

The spontaneous incidence of serious congenital and genetic abnormalities has been estimated to be about 105,000 per 10^6 live births, about 10.5 percent of live births (NAS80, UNSCEAR82). The low-LET background radiation dose of about 90 mrad/year in soft tissue results in a genetically significant dose of 2.7 rads during the 30-year reproductive generation. Since this dose would have occurred in a large number of generations, the genetic effects of the radiation exposure are thought to be an equilibrium level of expression. Since genetic risk estimates vary by a factor of 20 or more, EPA uses a log mean of this range to obtain an average value for estimating genetic risk. Based on this average value, the background radiation causes 700 to 1,000 genetic effects per 10^6 live births, depending on whether or not the oocyte is as sensitive to radiation as the spermatogonia (see Section 7.6). This result indicates that about 0.67 to 0.95 percent of the current spontaneous incidence of serious congenital and genetic abnormalities may be due to the low-LET background radiation.

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Chapter 8: METHODOLOGY FOR THE ASSESSMENT OF HEALTH IMPACT

8.1 Introduction

A health impact analysis or risk assessment is required when developing an EPA standard. It is the primary technical submission from Agency staff to Agency management in the decision-making process. In the case of the standard for the disposal of LLW, its purpose is to answer the following important questions: (1) What are the potential health impacts of LLW disposal? (2) What are the possible technical means for limiting these impacts? and (3) Using various technical means, what reduction in health impact (benefit) can be achieved? In addition, it is informative to estimate the health impact resulting from LLW disposal as it has been conducted in the past, and as it might be conducted with and without an EPA standard.

Because the standard must be generally applicable, it will apply over the entire U.S. and not be specific to certain disposal methods or management alternatives. Therefore, the health impact assessment compares combinations of as many as 10 different disposal methods, 3 general hydrogeologic and climatic settings, 24 waste streams, 4 waste forms, and a variety of other variables. The assessment must also evaluate the potential health impacts from all important pathways, using the Agency's radiation risk methodology. Because of these complexities, the only feasible means of estimating the health impacts was through the use of a computer model. To meet these needs, the PRESTO-EPA computer code was developed jointly by EPA and Oak Ridge National Laboratory (EPA83). This model, which was completed in 1983, was expanded by EPA and Rogers and Associates Engineering Company into a family of codes used to estimate health impact from the disposal of LLW under a variety of disposal alternatives (Ro84a). These codes are described in more detail in later sections.

8.2 Health Impact Assessment Modeling: PRESTO-EPA

The evaluation of the potential health impact, consisting of cumulative population health effects and maximum annual dose to the critical population group, from shallow-land disposal of LLW is performed with the computer code PRESTO-EPA (Prediction of Radiation Effects from Shallow Trench Operations - Environmental Protection Agency). The PRESTO-EPA code models the transport of radionuclides through hydrogeologic and atmospheric pathways to the eventual ingestion and inhalation by or direct exposure of humans. The results from the environmental transport portion of the model are used in the assessment of cumulative population health effects (consisting of fatal cancers and serious genetic effects to a local population for a period of 1,000 years and to a regional basin population for 10,000 years) or to estimate the maximum annual dose to a critical population group located close to the disposal site.

Environmental transport of radionuclides away from the LLW disposal site occurs through the hydrologic pathway, including infiltration, overflow, surface runoff, and ground-water flow, and through the atmospheric pathway. Figure 8-1 presents a schematic representation of these pathways. Radiation dose to humans is estimated for internal exposure from inhalation and ingestion and for direct exposure from contaminated air and soil. For a schematic representation of the food chain and direct exposure pathways, see Figure 8-2. Exposures can be calculated for onsite intruders who may grow crops with roots into the waste or build houses over the waste.

It is important to point out that the PRESTO-EPA model was developed for a specific purpose, to estimate health impacts from various disposal methods, as an aid in developing a generally applicable LLW standard. The model is a relatively simple, generic model, consisting of a one-dimensional ground-water transport submodel and other submodels of the compartment type. The model is <u>not</u> designed to be site specific, and the results should not be interpreted as pertaining to any particular disposal facility. This analytical approach is used since it is adequate for comparison of disposal methods; avoids potential errors in the numerical approach; and is consistent with the quality of the input data currently available.

A generic model is required because EPA will not be establishing site-specific standards but generally applicable standards, which will be implemented on a site-specific basis by NRC and DOE. The potential impact of disposing of a wide variety of wastes by different disposal methods under sharply different hydrogeologic and climatic conditions dictates use of a broad-based model. For example, three separate hydrogeological environments are modeled which encompass the majority of environmental conditions in the U.S. that would be feasible for LLW disposal.

The model is modular in design to allow substitution or modification of submodels when analyzing different disposal systems and is sufficiently flexible to analyze disposal systems within a wide range of hydrogeologic and climatic conditions. A disposal system includes the wastes, the disposal method, the site geology, hydrology, and climate, and all the applicable exposure pathways. In this way, the performance of the system can or will change if changes are made to any of its parts.

The use of a simple, one-dimensional, generic model has certain limitations. The model was developed for EPA's comparison of disposal alternatives and, therefore, was not designed, and should not be used, for making site-specific estimates of health impact. In simplifying the model, certain processes had to be omitted. The model assumes that all inputs, such as population size, demographics, and food consumption, remain constant over the modeling period. Radioactive daughter in-growth is not included in the PRESTO-EPA model, although activity due to the daughters can be accounted for by direct input of the daughter source

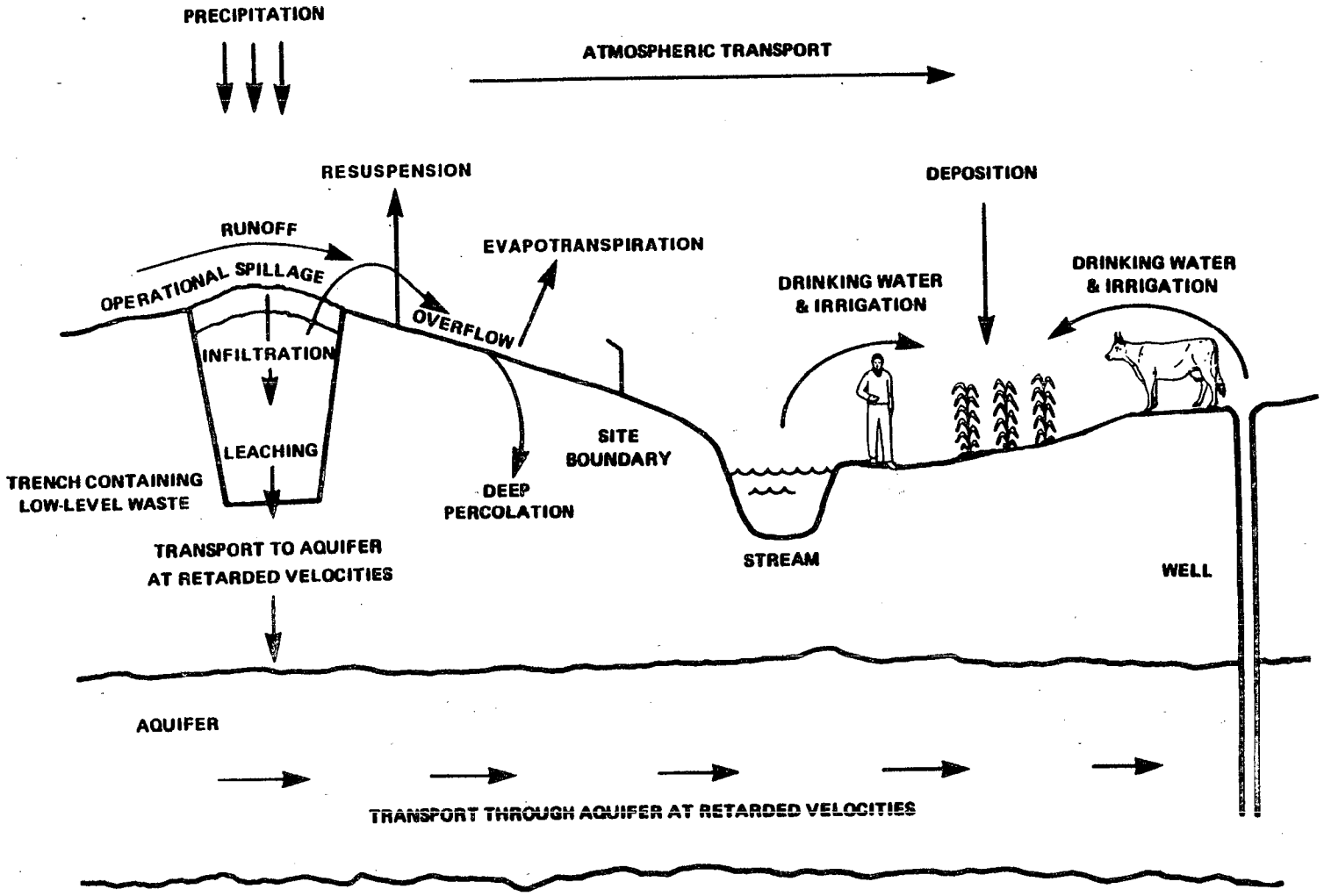


Figure 8-1. Hydrogeologic and Atmospheric Transport Pathways Used in PRESTO-EPA

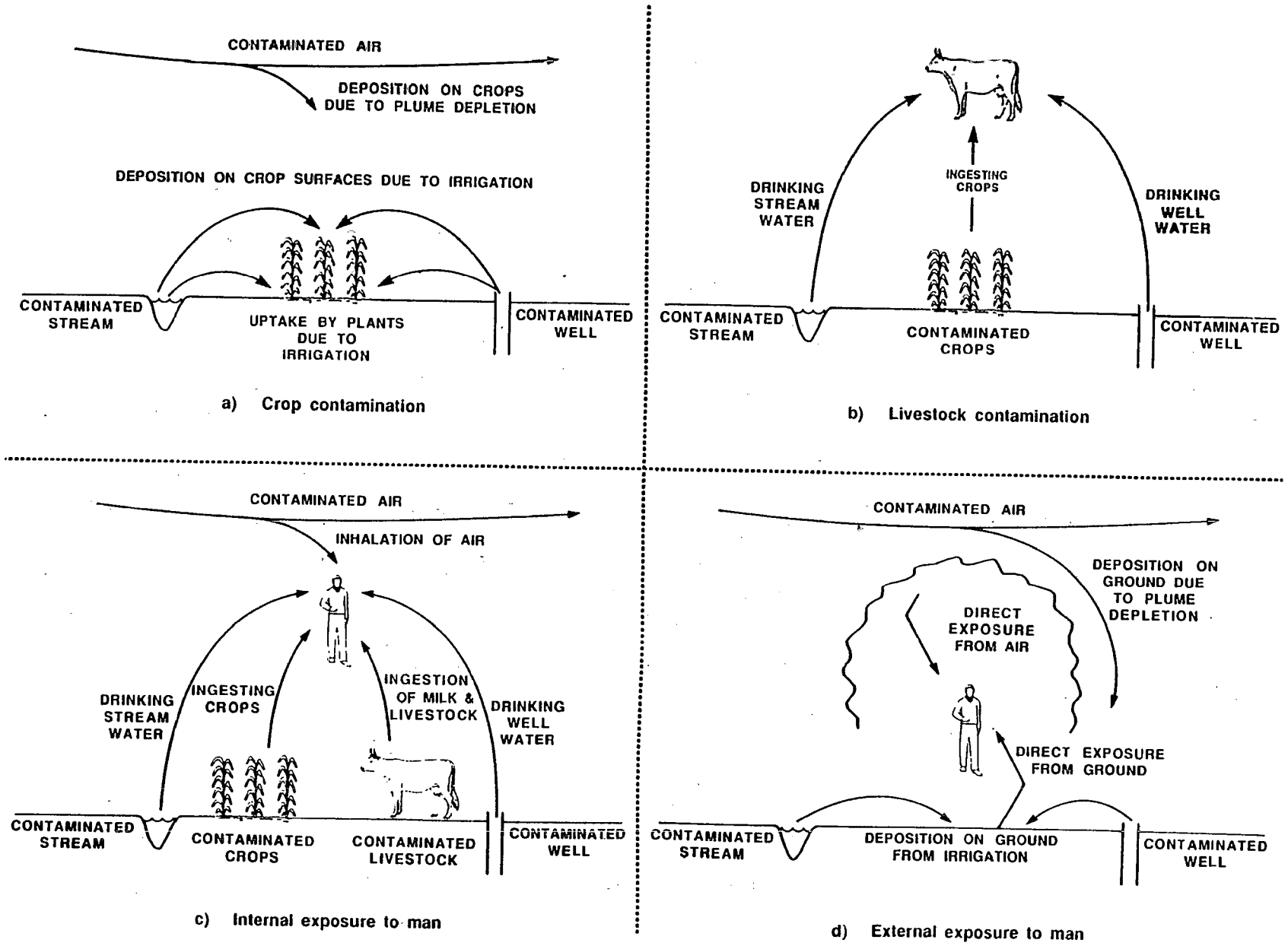


Figure 8-2. Food Chain and Direct Exposure Pathways Used in PRESTO-EPA

term. The cumulative health effects beyond 1,000 years are estimated using conversion factors that are based on the analysis over the first 1,000 years. While these limitations exist, the PRESTO-EPA model is appropriate for its intended use.

Because PRESTO-EPA was developed specifically for this standard-setting effort and is a new code, a program of code improvement and verification was conducted. This program includes:

- Quality assurance audits of all codes;
- Improvements and corrections based on extensive test and production runs;
- Peer review (PR84);
- Sensitivity testing and analysis;
- Improvements and corrections based on review and use by others, including national laboratories and universities;
- Review by EPA's Science Advisory Board (SAB85);
- Review and comparison of the geological transport portion of the code with results obtained by the U.S. Geological Survey (USGS) simulated for existing LLW disposal sites;
- Intercomparison with similar codes, such as those of the NRC; and
- Comparison of PRESTO-EPA results with data from actual LLW disposal sites.

The program to verify the PRESTO-EPA codes is described in more detail elsewhere (Me85). The sensitivity analysis portion of the program is described separately (Ba86a). A discussion of the sensitivity analysis is included in Chapter 11 and a discussion of model uncertainties in Chapter 12.

The original PRESTO-EPA code was developed to estimate cumulative population health effects from shallow-land burial of regulated LLW. Additional information was required, however, for the standard development, including: maximum annual doses to a critical population group located close by the disposal site, cumulative population health effects and maximum annual doses from deep disposal options, and cumulative population health effects and maximum annual doses from less restrictive disposal of BRC wastes. In order to provide these estimates, the original PRESTO-EPA code was expanded into a family of codes that includes:

PRESTO-EPA-POP — Estimates cumulative population health effects to local and regional basin populations from land disposal of LLW by shallow methods; long-term analyses are modeled (generally 10,000 years);

PRESTO-EPA-DEEP — Estimates cumulative population health effects to local and regional basin populations from land disposal of LLW by deep methods;

PRESTO-EPA-CPG — Estimates maximum annual dose to a critical population group from land disposal of LLW by shallow or deep methods; dose in maximum year is determined;

PRESTO-EPA-BRC — Estimates cumulative population health effects to local and regional basin populations from disposal of BRC wastes by sanitary landfill, municipal dump, and incineration methods; and

PATHRAE-EPA — Estimates annual pathway doses to a critical population group from less restrictive disposal of BRC wastes by sanitary landfill, municipal dump, and incineration methods.

These codes and how the Agency uses them have been described in detail (Hu83a, Ga84, Ro84b). Information on obtaining complete documentation and users' manuals for the PRESTO-EPA family of codes (EPA87a through EPA87g, Me81, Me84) is available from the Agency.

8.3 Health Impact Assessment Methodology - Overview

In this section the basic PRESTO-EPA code is discussed in a general manner. The specific codes that make up the PRESTO-EPA family are discussed in more detail in Sections 8.4 and 8.5.

While being a relatively simple computer model, the PRESTO-EPA code incorporates a number of different submodels that are used to analyze various pathways and scenarios and to determine health impact. Some members of the PRESTO-EPA family of codes use a unit response approach (see Section 8.3.6) and a conversion factor for long-term analyses in order to reduce computer time (see Section 8.3.5). Numerous input parameters are required for the PRESTO-EPA codes. These input parameters and their values are listed in Appendix C.

8.3.1 Infiltration/Leaching

At many sites, water is the most important medium for transport of radionuclides away from the trench. Whether the transport pathway is predominantly through the aquifer or by overland flow, the parameter that generally drives the system is the amount of water entering the trench via infiltration through the trench cap.

Infiltration is computed using a method by C.Y. Hung (Hu83b). This method simulates the infiltration of rainwater through a trench cover by modeling three separate flow systems: subsurface, overland, and atmospheric. Normal infiltration rates, calculated by the model, occur on the intact portions of the trench cap. On the failed portions, the infiltration also includes all the surface runoff that is diverted into the trench from the area of the trench cap up-slope from the failure area. Failure of the trench cap is through erosion or other processes. Erosion is determined by the model based on input parameters. However, in most cases, an actual trench cap will fail from such processes as subsidence, gully formation, or mechanical disturbance. To model these cases, the failure of the trench cap is based on assumed failure percentages occurring in user-specified years. The assumed values used are listed in Appendix C.

Water infiltrating the trench will be contaminated through contact with the waste in the trench. The amount of contamination is determined by the selected leaching methodology. Options include the use of specific distribution coefficients (K_d) for trash and absorbing waste forms, or a specified annual release fraction for activated metal and solidified waste forms. The amount of leaching can be modified by options such as solidification of the waste, use of high integrity containers (HIC), or active site maintenance. A listing of the options that were used for various scenarios is included in Appendix C.

Contaminated water may leave the trench by draining through the trench bottom or by overflowing. The model estimates the radionuclide activities and the amount of water leaving the trench on an annual basis.

8.3.2 Transport/Uptake Pathways

The PRESTO-EPA model estimates health impact from the hydrologic, atmospheric, food chain, and direct exposure pathways (Ba85). These pathways are described in the following sections. The hydrologic and atmospheric environmental transport pathways in PRESTO-EPA are shown in Figures 8-3 and 8-4.

(A) Hydrologic Pathways

In the ground-water flow model (Hu81), material that leaches from the trench is transported vertically to the underground aquifer and horizontally through the aquifer to a well. Transport velocities are calculated using saturated or unsaturated flow models, depending on sub-trench hydrological conditions. Radionuclides are transported at velocities lower than the characteristic flow velocity of the water in the aquifer. This "retardation" is due to interaction of the radionuclides with the solid materials in the aquifer.

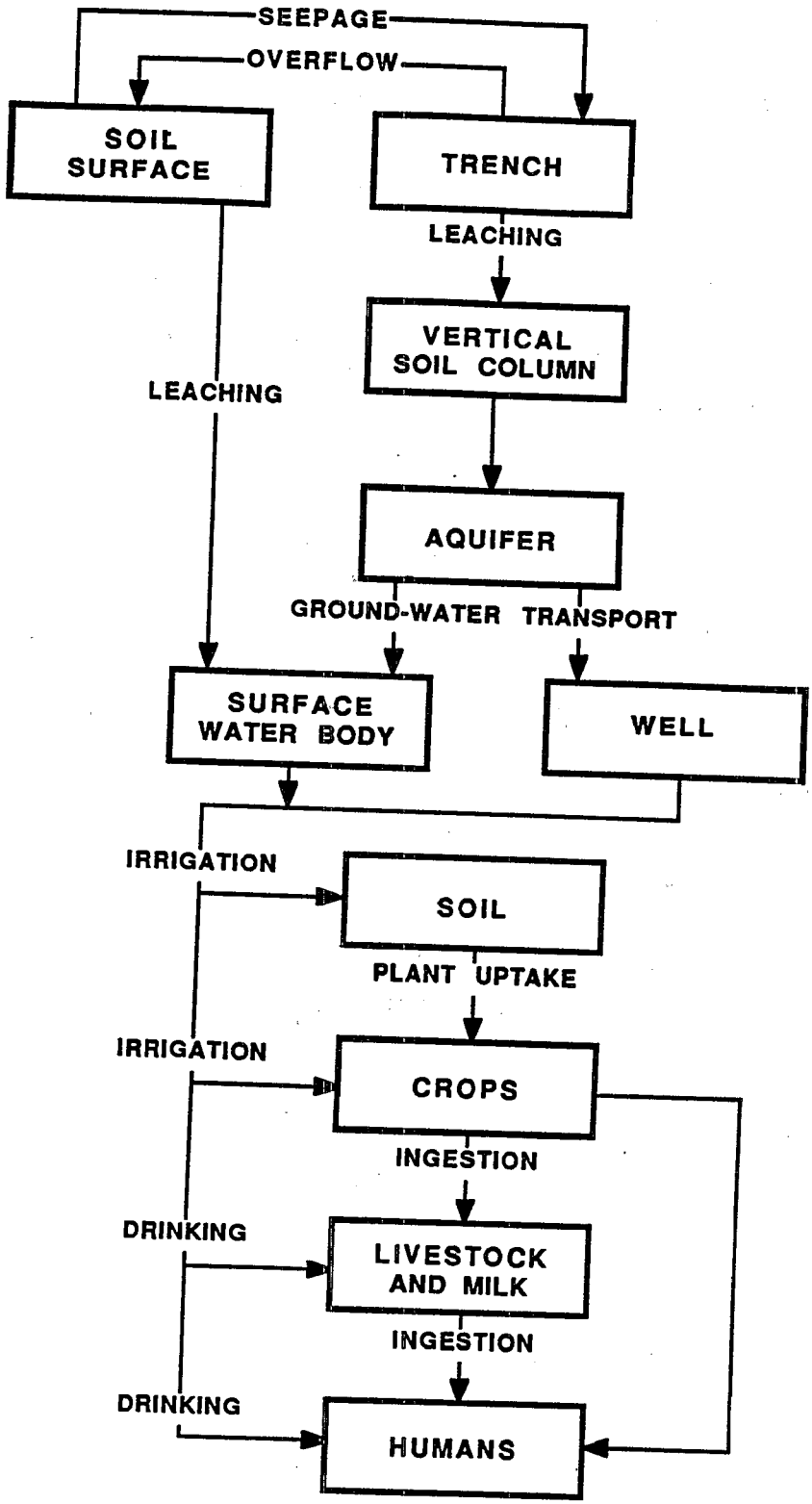


Figure 8-3. Hydrologic Environmental Transport Pathways in PRESTO-EPA

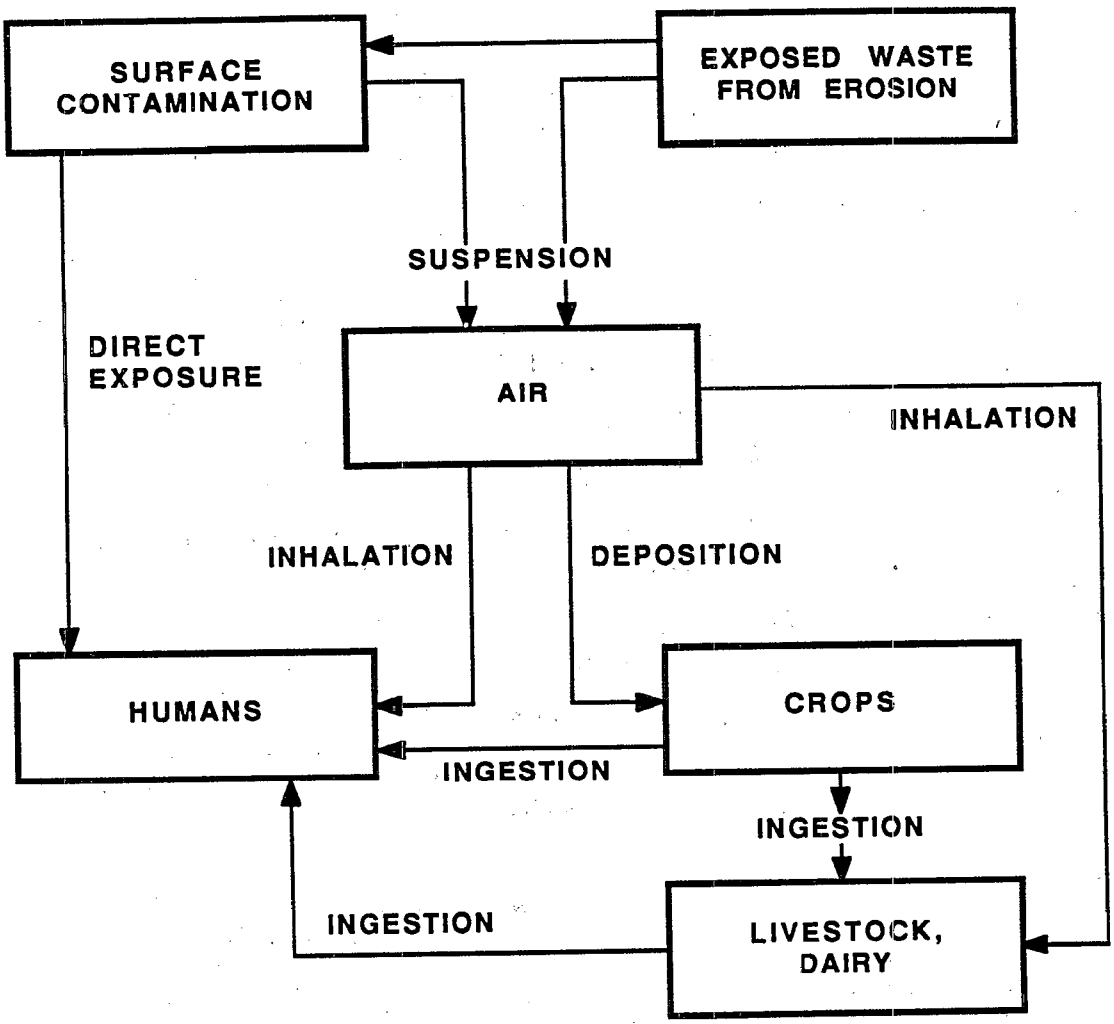


Figure 8-4. Atmospheric Environmental Transport Pathways in PRESTO-EPA

After a certain period, the contaminated aquifer water reaches a well. The activity reaching the well is diluted by the annual volume of water available at the well and used in the food chain and direct exposure pathways. In calculating cumulative population health effects, the concentrations of radionuclides in water are averaged over the length of the simulation period and used in determining exposure to humans. In calculating the maximum annual dose, the concentration in the maximum year is used.

Depending upon precipitation, infiltration rate, and hydrogeological characteristics, contaminated trench water may overflow from the trench onto the soil surface. When this occurs, radionuclides are added to the surface inventory of radionuclides from any initial operational spillage. The radionuclides on the ground surface are made up of two components, dissolved and adsorbed. The component adsorbed to soil is a source term for resuspension and subsequent atmospheric transport. The dissolved component may enter deep soil layers or nearby surface streams via overland flow. The radionuclides entering the deep soil layers will be modeled in the hydrologic pathway. The radionuclides entering the stream are divided by the annual stream flow and used in the food chain and direct exposure pathways.

(B) Atmospheric Pathways

In determining radionuclide concentrations during atmospheric transport, PRESTO-EPA includes a simple algorithm suitable for those sites where the population is concentrated into a single, small community. Because of the need to model more complex population distributions, EPA has used an optional externally computed ratio of the air concentration to source strength (chi/Q). The externally computed ratio is population weighted and can be used in cases of complex population distribution (see Section 8.3.8).

The atmospheric dispersion equation involves characterization of the source strength and calculation of the atmospheric concentration of radionuclides at the receptor location. The atmospheric emission rate (source strength) is made up of wind-driven suspension and mechanical suspension of radionuclides sorbed to the surface soil. No gaseous emissions, such as methane, carbon dioxide, or water vapor, are included in the source term. Radionuclides at the disposal site are deposited on the soil surface by spillage of wastes during operation and through overflow from the trench after site closure.

The atmospheric concentration of radionuclides at the receptor location is calculated using a Gaussian-plume atmospheric dispersion model (S168). Dispersion parameters are the standard deviation of the plume concentration in the horizontal and vertical directions. The radionuclides are transported at a height-independent wind speed to the receptor location. Plume depletion, effective plume height, and stable air layers at high altitudes are taken into account. Neutral atmospheric

stability is generally assumed. The model calculates a radionuclide concentration in air, averaged over a 22.5 degree sector for use in the inhalation, ingestion, and direct exposure pathways.

(C) Food Chain and Direct Exposure Pathways

Radionuclides in water, either well or stream, may impact humans through both internal and external radiological exposure. The internal exposure occurs from drinking contaminated water and from ingesting milk, beef, and crops contaminated through irrigation or livestock watering. External exposure results directly from exposure to crops and soil that have been irrigated with contaminated well or stream water. Radionuclides in air may also expose humans through both internal and external pathways. Direct external exposure may result from immersion in a plume of contaminated air or by exposure to soil surfaces contaminated by plume deposition. Internal exposures may result from inhalation of contaminated air or ingestion of food products contaminated by plume deposition.

To calculate the impact to humans from ingestion, inhalation, and direct exposure from radionuclides in air and water, the computer code DARTAB (see Section 8.3.4) is used as a subroutine in PRESTO-EPA. Radionuclide input data to DARTAB from the air and water pathways must be in four specific formats:

(1) average concentrations in air (Ci/m^3);

(2) average concentrations on ground surface (Ci/m^2);

(3) collective average inhalation rates (person-pCi/yr); and

(4) collective average ingestion rates (person-pCi/yr).

The mean concentrations of radionuclides in air are calculated for the period of simulation or for the year of maximum annual dose. Mean concentrations of radionuclides in air are used to calculate the direct exposure to humans from immersion in contaminated air.

The concentration of each radionuclide on the ground surface is due to both atmospheric deposition and irrigation. Atmospheric deposition results from plume depletion through the atmospheric transport pathway. Deposition from well and stream water used for irrigation is simulated in the irrigation subroutine. The concentration of radionuclides on the ground surface is used to calculate the direct external exposure to humans from contaminated ground.

The collective exposure from inhalation is calculated by multiplying the size of the population of interest by the average individual inhalation rate and by the average concentration of radionuclides in the air. The collective exposure from inhalation is used to calculate organ

doses and effective whole-body dose equivalent from breathing radionuclide-contaminated air.

The collective ingestion rate includes intake of drinking water, beef, milk, and crops. Except for drinking water, all of these media may be contaminated by either atmospheric deposition or by irrigation from a well and/or the stream. Humans may ingest water directly from either the well or the stream. The water may also be ingested by cattle or used to irrigate crops. Crops are contaminated through atmospheric deposition directly onto the plants, and/or by growing in soil contaminated through atmospheric deposition or irrigation with contaminated water. Once contaminated, the crops can be ingested by humans or by animals. Humans ingest beef and milk that have become contaminated from cattle eating and drinking contaminated forage and/or water. The annual individual ingestion rate is multiplied by the size of the population and the average concentration of radionuclides ingested to calculate the annual collective ingestion rate. In calculating population health effects, the mean annual collective ingestion rate is assumed to be constant for the population over the period of simulation. In contrast, for the maximum annual dose, the average concentration for the year in which the peak concentration occurs is used. The annual collective ingestion rate is used to calculate the internal organ doses and effective whole-body dose equivalent to humans, through ingestion of contaminated water, milk, beef, and crops.

8.3.3 Intruder Scenarios

The PRESTO-EPA code can be used to estimate exposures to an intruder. It is assumed that the intruder would enter the site after institutional control has ended. The intruder scenarios allow for onsite farming with crop roots growing directly into the waste or for building homes with basements dug into the waste.

If the intruder farms the disposal site, it will cause mechanical disturbance of the trench cap, as well as the possibility of crop roots growing into the waste. The mechanical disturbance is taken into account through the atmospheric transport pathway. The crop roots growing into the waste are considered in the food chain pathway.

The NRC's regulations (NRC82) require protection against inadvertent intrusion into the disposal areas. EPA feels that this exposure pathway is probabilistic in nature and that safeguards against inadvertent intrusion should be carried out on a site-specific basis. For these reasons, EPA has not included intrusion scenarios in its health impact assessments.

8.3.4 Health Impact Assessment

The PRESTO-EPA computer code estimates radionuclide concentrations in ground and surface water, concentrations in the air, rates of

deposition on the ground, concentrations on the ground, and the amounts of radionuclides taken into the body via inhalation of air and ingestion of water, meat, milk, and fresh produce. The amounts of radionuclides that are inhaled are calculated from these air concentrations and a knowledge of how much air is inhaled by an average person. The amounts of radionuclides ingested in the water, meat, milk, and fresh produce that people consume are estimated using daily food intake values based on data from the 1977-1978 USDA nationwide food consumption survey (Ne84).

The subprogram DARTAB combines the information on the amounts of radionuclides that are ingested or inhaled (as provided by the pathway analysis) with radiation health risk data for a unit quantity of each radionuclide (Be81). The health risk data are calculated by the code RADRISK (Su81).

The RADRISK code first computes dose rates to organs resulting from a unit quantity of a radionuclide that is ingested or inhaled. These dose rates are then used in a subroutine adaptation of the program CAIRD to estimate the risk of fatal cancers in an exposed cohort of 100,000 persons (Bu81). All persons in the cohort are assumed to be born at the same time and to be at risk of dying from competing causes (including natural background radiation). Estimates of potential health risk due to exposure to a known quantity of approximately 500 different radionuclides are tabulated and stored until needed. These risks are summarized in terms of the probability of premature death for a member of the cohort due to a unit quantity of each radionuclide that is ingested or inhaled. This information is then combined with the unit response data to determine the cumulative population health effects, which include fatal cancers and serious genetic effects, and the maximum annual doses to the CPG (see Section 8.3.6).

8.3.5 Regional Analysis - Use of Health Effect Conversion Factor (HECF)

Two population groups are used to estimate cumulative population health effects -- a local population and a regional basin population. The cumulative population health effects to the local population are determined directly by the PRESTO-EPA-POP, PRESTO-EPA-DEEP, and PRESTO-EPA-BRC models, through a number of detailed pathway analyses using iterative yearly updates. This impact is to a relatively small, nearby community for the first 1,000 years after site closure. After 1,000 years, the local community is assumed to become part of the larger regional basin community. This assumption is made to reduce the amount of computer time, but will not affect the model results, as the local population provides only a small percentage of the cumulative population health effects in comparison to the much larger regional basin population.

Because of the small size of the local population, not all contaminated water is used. All excess contaminated water, either ground water or surface stream, flows past the local community to enter a river

and then the downstream basin. Upon reaching the downstream basin, this residual activity is assessed as part of the regional basin analysis.

After the 1,000-year local analysis period has ended and the local population is considered to be a part of a larger regional basin community, all activity that leaves the disposal site, in either ground or surface water, travels directly to a regional basin river and enters the downstream basin, where the activity is assessed as part of the regional basin analysis. The residual activity from the first 1,000 years and the regional basin activity from the remaining 9,000 years are added together. This total activity, in curies, is termed the regional basin activity (see Figure 8-5) and is used to assess cumulative population health effects as part of the regional basin analysis.

Fatal cancers and serious genetic effects to the regional basin population are estimated by multiplying the regional basin activity removed from the basin river by nuclide-specific health effect conversion factors (HECF) and summing over all nuclides. These HECF values are calculated using regular PRESTO-EPA model runs, similar to procedures used by Smith (Sm82) as discussed below. The regional basin health effects are then added to the local health effects to generate the combined cumulative population health effects over the analysis period (10,000 years).

Before describing how the HECF values were determined, the following points should be noted:

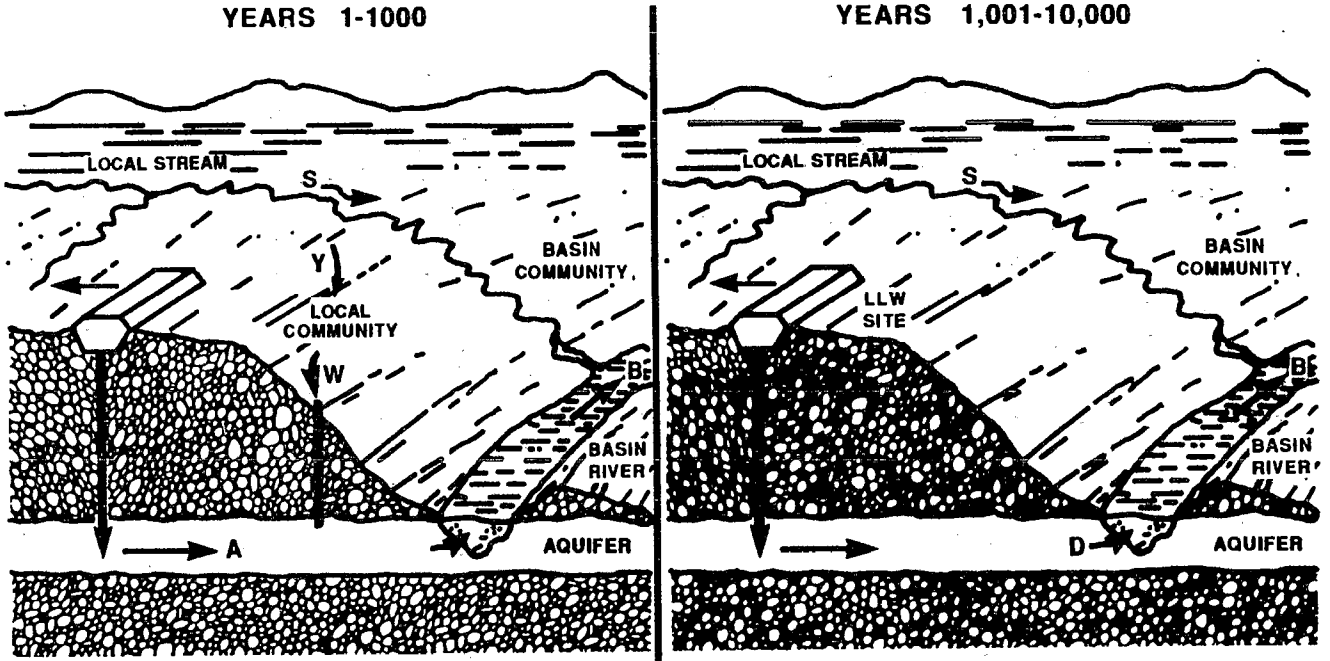
- The HECF values are nuclide specific and are in the unit of health effects per curie of radionuclide activity released to the regional basin;

- The HECF values are based on the ground-water and surface-water pathways only. The atmospheric transport pathway is not included, as analyses have shown that the long-term impact from this pathway is trivial in comparison to the water pathways;

- Health effects from the external radiation pathway, caused by deposition of nuclides on surface soil, is by water pathways only;

- Cumulative population health effects, as calculated by the PRESTO-EPA model, consist of fatal cancers and serious genetic effects, which are determined similarly by the model, although separate HECF values are used. The following discussion of the calculation of the HECF values is applicable for either;

- In the same manner as for the local population, not all activity reaching the regional basin will be removed by the basin population. Activity not removed in the regional basin is assumed to enter the ocean, which acts as a nuclide sink. Analyses have shown that because of dilution, activity reaching the ocean contributes negligible health effects to the population under consideration (EPA86);



BASIN RESIDUAL RADIOACTIVITY:	B(Ci) = (A - W) + (S - Y)	BASIN RESIDUAL RADIOACTIVITY:	B(Ci) = D + S
REGIONAL BASIN HEALTH EFFECTS:	HE = B x HECF	REGIONAL BASIN HEALTH EFFECTS:	HE = B x HECF

Figure 8-5. Regional Basin Health Effects Pathways

- The HECF values include health effects caused by the ingestion of contaminated fish. This pathway is not considered for the local population exposure, since the consumption of fish from a small, local stream would, in general, be minimal; and

- Separate HECF values are calculated for each hydrogeologic setting, based on water usage factors for that region.

(A) Calculation of HECF Values

The HECF values, which are nuclide specific, are made up of a terrestrial pathway portion ($HECF_{ti}$) and a portion based upon an aquatic pathway ($HECF_{fi}$). Separate HECF values are determined for fatal cancers and for genetic effects. The following discussion, however, is applicable to either.

(1) Terrestrial Pathway:

The nuclide-specific health effects conversion factors for the terrestrial pathway ($HECF_{ti}$) are calculated in two steps using the results from PRESTO-EPA analyses for the local population. In the first step, the health effects to the local population resulting from the withdrawal of a unit curie of a specific nuclide from the local well or stream are determined. The second step involves the calculation of the fraction of activity discharged into the basin which will be withdrawn from the regional stream. The $HECF_{ti}$ is determined by multiplying the fraction of activity withdrawn, by the health effect per curie conversion factor:

$$HECF_{ti} = (HE/Ci_i \text{ withdrawn}) \times f,$$

where:

$HECF_{ti}$ = the terrestrial HECF for radionuclide i;

HE/Ci_i withdrawn = the health effects to the local population per unit curie of radionuclide i withdrawn from the local well or stream; and

f = the fraction of activity withdrawn from the basin river per unit activity released to the regional basin.

The fraction of activity withdrawn by the regional basin population is based on the local population water usage and a standard ratio of river flow to population. The per capita water use (drinking water, cattle watering, and irrigation) of the local population is taken from appropriate PRESTO-EPA model inputs. Assuming that the per capita regional basin water use will be uniform, and using a standard ratio of

population to river flow for the regional basin, the fraction of regional basin activity that will be withdrawn by the regional basin communities can be calculated:

$$f = (U/P) \div (Q/P)$$

where:

f = fraction of activity withdrawn from the basin river per unit activity released to the regional basin;

(U/P) = per capita water usage (m^3/person-yr); and

(Q/P) = ratio of river flow to population, 3,000 m^3/person-yr (EPA85a).

Thus, when the local per capita water consumption is divided by a standard ratio of river flow to population, the fraction of the regional basin radioactivity that will be withdrawn by the regional basin community is determined.

This calculation assumes two points. The first is that local water usage is comparable to regional basin water usage. The second point is that the global ratio of river flow to population is applicable to the regional basin. As noted in the High-Level Radioactive Waste Background Information Document (EPA85a), studies show that while regional basin population and river flow vary widely, the ratio of the river flow to the population remains relatively constant and that the value we have used is within the range of similar values associated with various river basins in the U.S.

It is again noted that the fraction of nuclides that are <u>not</u> taken up by the regional basin community is assumed to enter the ocean, which acts as a nuclide sink. Health effects from the activity released to the ocean are assumed to be negligible. The percentage of activity in the basin river that reaches the ocean varies from almost zero in the arid region to about 95 percent in the humid regions. The reason that so much activity reaches the ocean in the humid regions is that, in general, very little surface water is used. This assumption is tested in the sensitivity analysis program and is discussed in Chapter 11.

(2) <u>Fish Pathway</u>:

One pathway that was viewed as negligible for the local population but is considered in calculating regional basin health effects, is that of contaminated fish ingestion. A separate $HECF_{fi}$ for fish is determined, based on the following equation:

$$HECF_{fi} = (P/Q) \times B_{fi} \times U_f \times (D/C)_i$$

where:

$HECF_{fi}$ = health effects conversion factor from consumption of fish, per curie of radionuclide i released to the regional basin;

(P/Q) = river-flow-to-population ratio (person-yr/3,000 m^3) (EPA85a);

B_{fi} = fish bioaccumulation factor (Ci/kg-fish per Ci/l of radionuclide i in water) (NAS71);

U_f = annual fish consumption rate (6.9 kg/person-yr) (Ru80); and

$(D/C)_i$ = conversion factor for health effects per curie of nuclide i ingested, obtained from PRESTO-EPA-POP calculations.

(B) Basin Health Effects Conversion Factor

The $HECF_{fi}$ is added to the $HECF_{ti}$ calculated earlier to determine the nuclide-specific regional basin $HECF_i$ values:

$$HECF_i = HECF_{ti} + HECF_{fi}.$$

(C) Combined Health Effects

The total nuclide-specific activity reaching the regional basin over 10,000 years (q_{bi}) is multiplied by the nuclide-specific HECF values ($HECF_i$), and summed over all nuclides, to determine the regional basin health effects:

$$\text{Regional Basin Health Effects} = \sum_i q_{bi} \times HECF_i.$$

These are added to the local health effects, already calculated directly by PRESTO-EPA, to estimate the total health effects over 10,000 years. Note that the above discussion relates to the calculation of either cancer deaths or serious genetic effects. These are calculated separately, using separate $HECF_i$ values, and then combined to estimate the cumulative population health effects.

Sensitivity analysis was performed on the input parameters associated with the health effects conversion factors. This analysis, as well as a general discussion on the calculation of the HECF values, is discussed in Appendix G. Also listed in this appendix are the nuclide-specific fish bioaccumulation factors.

8.3.6 Use of Unit Response

In order to evaluate 3 generic hydrogeologic disposal sites, 10 disposal methods, 24 waste streams, and 4 or 5 waste forms, a large number of computer runs would have to be performed. Therefore, a unit

response methodology was devised to reduce the number of computer runs. This methodology was used with the PRESTO-EPA-POP, PRESTO-EPA-DEEP, and the PRESTO-EPA-BRC codes. The unit response methodology is not practical for use with the PRESTO-EPA-CPG or PATHRAE-EPA codes, as an actual source term is necessary in order to determine the year in which the maximum annual dose would occur.

The unit response approach has been used in many applications and has been proved to be a valid approach. The application of the unit response approach to estimate cumulative population health effects was evaluated by comparison with a direct assessment method. The results of the evaluation showed that for the purpose of cumulative population health effects estimation, both approaches give comparable results.

The data for a unit response analysis are altered slightly from the data for a full facility analysis. In order to model a variety of waste streams with a single PRESTO-EPA run, a single curie of each radionuclide is assumed to be present in the waste inventory. The volume of the waste disposal trench is reduced to a single cubic meter of waste. The results of this unit response analysis are used as inputs to an accounting model program.

The accounting model adjusts the PRESTO-EPA unit-curie results in proportion to the number of curies of a particular nuclide, per cubic meter of a given waste stream. This results in a tabulation of health effects arising from the disposal of one cubic meter of waste from a particular waste stream. The accounting model then multiplies the impact resulting from one cubic meter of a particular waste by the number of cubic meters of that waste stream for the particular scenario being modeled. By adding together the impact from all the waste streams located at a site, the total impact from the waste disposed of at a disposal site can be estimated. This calculation is also performed by the accounting model. The accounting model is described in greater detail in a separate report (EPA87g).

8.3.7 Time Periods Analyzed

In estimating health impact from the shallow-land disposal of LLW, two main time periods are analyzed: (1) 1,000 years for impact to the local population and to the critical population group, and (2) 10,000 years for impact to a regional basin population.

The PRESTO-EPA-CPG and PATHRAE-EPA codes are used in estimating maximum annual dose to the critical population group. These two codes model a 1,000-year period, although the important output is the annual dose in the maximum year. Because the critical population group is located close to the disposal site, maximum annual doses for the mobile radionuclides generally occur soon after the assumed institutional control period has ended and almost always before 1,000 years. In some arid scenarios and with more restrictive disposal methods, annual doses

continue to rise slowly after 1,000 years, although the maximum is usually reached long after 1,000 years and is always small.

The PRESTO-EPA-POP and PRESTO-EPA-BRC codes estimate cumulative population health effects to both a local population (for a 1,000-year period) and a regional basin population (for 10,000 years). After 1,000 years, the local population is included within the larger regional basin population. A modeling period of 10,000 years is necessary since several radionuclides are hazardous for this period or longer and some disposal methods require long time periods before radionuclides reach the population. While there is a great deal of uncertainty in many parameters, especially when long time spans are used, these uncertainties are present in each of the disposal methods to the same degree. Thus, when comparing methods, the uncertainty becomes less important. This is discussed in more detail in Chapter 12.

The cumulative population health effect assessments also analyze two shorter time frames (100 years and 500 years). Both local and regional basin health effects are estimated. The results from these analyses are useful in learning more about the different disposal alternatives and how they compare in the earlier periods after disposal.

The PRESTO-EPA-DEEP code estimates cumulative population health effects to a local and regional basin population from the deep disposal of LLW. Because of the deep disposal methodology, very little radionuclide release occurs before 1,000 years. Therefore, PRESTO-EPA-DEEP estimates both local and regional basin health effects for 10,000 years.

8.3.8 Modeling Inputs

The PRESTO-EPA codes require a large number of input parameters. While some of the inputs are constant over all conditions, most vary depending upon the site characteristics, waste stream, waste form, disposal method, or radionuclide. The input parameters and values used for the various PRESTO-EPA codes are listed in Appendix C. Discussed below are the conditions upon which the inputs are dependent.

(A) Use of Generic Sites

Three sets of generic hydrogeologic disposal site characteristics are modeled. These are termed "generic" sites because the data used in the model are not specific to any actual disposal site or to any exact geographic location. To generate health impact data that are useful in the standard-setting process, the sites are typical of large areas of the United States. The three generic settings chosen are: (1) a humid site with low permeability; (2) a humid site with moderate permeability; and (3) an arid site with moderate permeability. Three sets of standard hydrogeologic input data are used to characterize the three different generic sites. Data for water usage patterns, population distribution,

and farming activities are representative of the general region of the U.S. where the site is located. More specific data requirements, such as precipitation patterns, temperature variations, and ground-water movement, require the use of actual site data. The detailed characteristics of the sites used as a source of these data are described more fully in Appendix D. All of the site-dependent input parameters are listed in Appendix C.

(B) Disposal Methods

Several of the input parameters to the PRESTO-EPA code serve to define the waste disposal method. In particular, trench width and depth, cap or cover thickness, porosity, and permeability are required. These parameters are obviously dependent on site characteristics and design. To compare the effectiveness of ten disposal methods at three generic hydrogeologic sites, a standard conceptual design was prepared for each disposal method. These conceptual designs use average dimensions and specifications and are held constant from one site to another. This generalization is consistent with the use of generic rather than actual disposal sites. The combination of the unit response analysis methodology (see Section 8.3.6), combined with generic disposal sites and standardized disposal methods, results in well-defined data sets for each site which can be easily modified to reflect different disposal methods. Detailed conceptual designs for each disposal method are presented in Chapter 4. Input values dependent upon the disposal method are listed in Appendix C.

(C) Waste Forms

Although LLW may take a variety of forms, from disposable gloves and scrap paper to large steel parts, five general waste forms are used to simplify modeling. These waste forms influence some of the physical characteristics of the trench material, such as porosity, permeability, and density. The waste form also determines the rate at which the radionuclides are released from the waste material, such as leaching rate and release fraction. The five waste forms are: trash (TR), absorbing materials (AW), solidified waste (SW), activated metal (AM), and incinerated waste which is then solidified (IS). The waste form dependent input values are listed in Appendix C.

(D) Waste Streams

The NRC identified 36 separate waste streams in supporting its regulation for LLW disposal facilities (NRC81). A more recent NRC document describes 148 waste streams, with greater emphasis on higher activity wastes and nonroutine sources (NRC86). For EPA's analysis, similar waste streams are combined to form a total of 24 waste streams. This results in fewer scenarios to be modeled without sacrificing accuracy. In addition to the 24 LLW streams, separate streams were identified for NARM wastes and BRC.

(1) NARM

To provide a basis upon which to consider the regulation of NARM wastes, EPA commissioned a study of NARM waste streams (EPA85b). Special emphasis is placed on higher activity wastes and those exhibiting characteristics analogous to LLW regulated under the AEA. The NARM waste streams included in EPA's LLW radiological source term are based on this study, and include radium sources and radium-contaminated ion exchange resins (Ba86b).

(2) BRC

To characterize waste streams appropriate for the BRC analysis, a select group of LLW waste streams was constructed. These are designated as "surrogate" BRC wastes, i.e., waste streams meant to represent the kinds of wastes that may be deregulated by other regulatory agencies under various EPA-designated BRC levels. In general, the surrogate BRC waste streams are lower activity LLW waste streams or, where enough information is available, substreams of previously defined LLW waste streams. Various wastes are represented by these surrogate BRC wastes, including those from nuclear fuel-cycle, institutional, and industrial LLW generators.

A complete description of the waste streams is found in Chapter 3. The input parameters that are dependent upon the waste streams are listed in Appendix C.

(E) Radionuclides

Each of the waste streams modeled contains various radionuclides. A total of 40 radionuclides, including 5 NARM radionuclides, are modeled.

All of the nuclides and nuclide-dependent parameters are included as input in the PRESTO-EPA models. In those models that use the unit response methodology (POP, BRC, and DEEP), one curie of each nuclide is assumed. For the models not utilizing the unit response methodology (CPG and PATHRAE), the actual source term for each radionuclide is included for each separate computer run.

The complex physiochemical interaction between the radionuclides and the solid geologic media has been grouped into a single factor, the distribution coefficient (K_d). Separate K_d values are used for soil, the mixture of soil and waste material in the trench, the subtrench soil, and the aquifer. The K_d values for each radionuclide are listed in Appendix C.

In addition to the K_d values, there are a number of other nuclide-dependent input parameters, such as radiological decay constants, soil-to-plant uptake factors, and volatilization factors. The nuclides included and their activities are discussed in Chapter 3, while the nuclide-dependent input parameters are listed in Appendix C.

(F) Atmospheric Parameters (RADE Program)

The PRESTO-EPA codes include a Gaussian plume model for atmospheric transport. However, the codes only allow one wind direction, speed, stability, distance, and population to characterize a community for each scenario modeled. In order to facilitate the assessment of health impacts for multiple communities, a utility program called RADE (Radioactive Atmospheric Dispersion and Exposure) has been designed (Ro84b). The RADE code performs standard Gaussian plume atmospheric dispersion calculations using subroutines from PRESTO-EPA. RADE produces results that are suitable for use as input parameters in the PRESTO-EPA codes, which allow for atmospheric modeling of multiple communities. The atmospheric input parameters, including those which come from the RADE program, are listed in Appendix C.

8.4 Health Impact Assessment - Regulated Disposal of LLW

The EPA LLW health impact assessment consists of two broad types of analyses: regulated and unregulated (BRC) disposal.

General characteristics of the health impact assessment for regulated disposal are discussed in this section. The second area, that of unregulated (BRC) disposal, is discussed in Section 8.5.

8.4.1 PRESTO-EPA-POP

The original PRESTO-EPA model was used as a basis for a family of PRESTO-EPA codes. The basic model is used to estimate cumulative population health effects and is called PRESTO-EPA-POP.

This model calculates the cumulative population health effects, resulting from the regulated disposal of LLW, to a local population and to the population in a regional basin in which the disposal site is located. The health impacts to the local and regional basin populations are both analyzed for a period of 1,000 years. The regional basin analysis, however, is extended for an additional 9,000 years. The values used for the PRESTO-EPA-POP input parameters are listed in Appendix C. Some characteristics specific to PRESTO-EPA-POP are discussed in the following sections.

Because of the many waste disposal scenarios considered in the PRESTO-EPA-POP analysis, the modeling was simplified. By utilizing a unit response approach, the numbers of PRESTO-EPA-POP analyses were reduced significantly. The unit response approach, which is based on the disposal of one curie of each radionuclide, is described in Section 8.3.6.

A radionuclide-specific health effect conversion factor (HECF) is used to determine health effects to the regional basin population over 10,000 years. The HECF is described in more detail in Section 8.3.5.

The starting point for the health impact assessment is assumed to be immediately after the closure of the disposal site. The radionuclide inventories of the waste in the trench are reduced to account for the radioactive decay during the operational period (assumed to be 20 years). The waste is assumed to be containerized, with the leaching of radionuclides from the waste beginning after the container fails. The length of the container integrity is a user-specified parameter and is listed in Appendix C.

Because in-growth of radiological decay products is not calculated by the model to maintain simplicity (daughter product in-growth is considered by RADRISK for internal exposures), cases where the major dose contribution is from external exposure to a short-lived progeny in equilibrium with a parent radionuclide present in the trench inventory, the progeny is included directly in the trench inventory. This is the case for the Cs-137 daughter, Ba-137m, which is included in the source term with the same activity and radiological characteristics as the parent.

Operational spillage is defined as the radionuclides spilled from waste packages and remaining on the ground surface at the close of disposal operations. These radionuclides would subsequently be transported either by the atmospheric pathway to the local population or by the surface-water pathway to the local stream and basin river.

Each member of the population is assumed to eat the same quantities of food, all grown on the same fields, and to obtain his or her drinking and crop irrigation water from the same sources (a certain percentage of which is assumed to be contaminated). This assumption simplifies the calculations and is appropriate because of the large uncertainties in predicting individual mobility, population demography, agricultural practices, and geologic and hydrologic changes that might occur during the analysis period. As input parameters, the user may specify the fraction of the drinking and irrigation water that is supplied by contaminated sources, as outlined in Appendix C.

Cumulative population health effects are calculated by characterizing the population center for each site with a single, geographically central location and the total population size. In calculating the cumulative population health effects, the population age distribution and size are held constant over the assessment period.

(A) General Characteristics of the PRESTO-EPA-POP Analyses

The results from the PRESTO-EPA-POP analyses are cumulative population health effects to both a local and a regional basin population. The specific estimates from our various computer runs are detailed in Chapters 9 and 11. In the following sections, general results for the PRESTO-EPA-POP code are discussed. Typical results from the PRESTO-EPA-POP analyses show that, in general, the local population

health effects do not dominate in any of the three regional hydrogeologic and climatic scenarios. This is due to the limited amount of water and food, contaminated by nuclides, that can be ingested and expose the relatively small local population. The majority of the cumulative population health effects are incurred by the regional basin population. The cumulative health effects to the regional basin population, and the pathways by which they occur, vary considerably over the three hydrogeologic and climatic regions (see Figure 8-6). The results are more easily reviewed by first separating them into the three general settings: humid permeable, humid impermeable, and arid permeable.

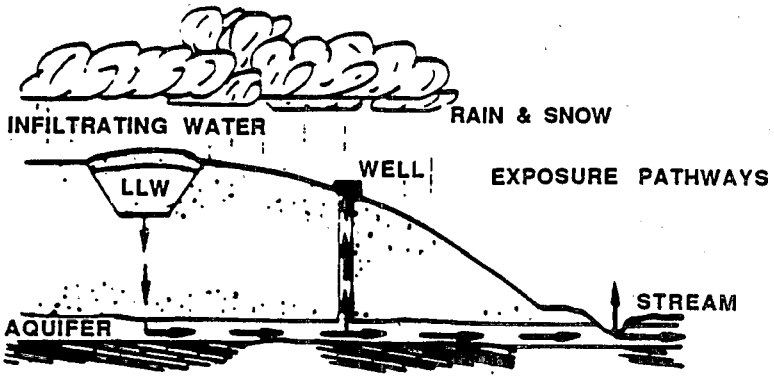
At the site characterized by relatively permeable soil and high rainfall, most of the mobile radionuclides leach out of the trench and into the aquifer during the initial 1,000-year period. The majority of the cumulative population health effects are incurred by the regional basin population through the ground-water pathway during the first 1,000 years.

At the site characterized by high rainfall and relatively low soil permeability, the trenches fill with water after the trench cap has been assumed to have failed. The activity will be leached from the wastes and will escape from the trench through overflow (bathtub effect) in a relatively short period of time. The regional basin population receives the majority of the cumulative population health effects through the surface-water pathway during the first 1,000 years.

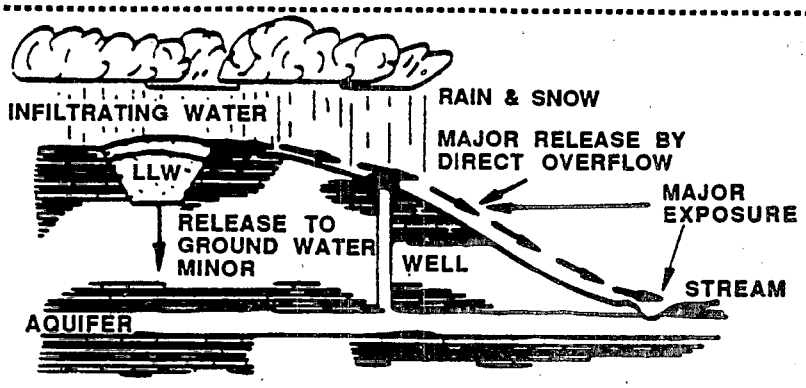
At the site characterized by relatively permeable soil but low rainfall, most activity does not reach either the local or the regional basin populations until relatively late in the modeling period. The local population incurs some minor impact in the first few years due to wind blown (atmospheric) transport of nuclides spilled onto surface soils during site operations. This health impact, while quite small, can be the only health impact during the first 1,000 years. This is due to the long travel time required for contamination to reach the aquifer and then travel to the local and regional basin populations by ground water. The overall cumulative population health effects, which are always very small, are dominated by health effects from activity reaching the regional basin population through the ground-water pathway after 1,000 years.

8.4.2 PRESTO-EPA-CPG

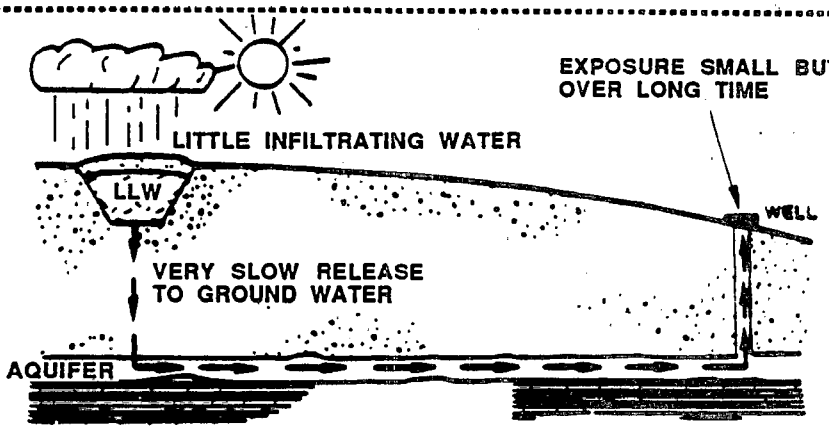
The maximum annual doses to groups of individuals located close to the disposal site are estimated using the PRESTO-EPA-CPG code. These groups are assumed to have certain characteristics and to be associated with environmental pathways where they are likely to receive a greater exposure than the average population. These individuals are called the critical population group (CPG).



a) PERMEABLE MEDIUM WITH A HUMID CLIMATE



b) IMPERMEABLE MEDIUM WITH HUMID CLIMATE



c) PERMEABLE MEDIUM WITH AN ARID CLIMATE

Figure 8-6. Environmental Pathways at a Shallow LLW Disposal Facility in the Three General Hydrogeologic and Climatic Settings.

In assessing the impact of waste disposal alternatives and site-specific characteristics on CPG dose, a methodology was employed which relied on the established PRESTO-EPA approach, with necessary and appropriate modifications. The values used for PRESTO-EPA-CPG input parameters are listed in Appendix C.

The transport of radionuclides was evaluated for the same pathways pictured in Figure 8-1, although, as shown in Figure 8-7, the location of the population of interest is changed.

Maximum annual dose rates were determined using the DARTAB subroutine in PRESTO-EPA-CPG. The RADRISK data file, used in DARTAB, contains the organ-specific dose factors for each radionuclide in the inventory; effective whole-body dose equivalent rates are generated in DARTAB using standard EPA organ-weighting factors. See Chapters 6 and 7 and Section 8.3.4 for more detailed information on DARTAB/RADRISK.

(A) Differences Between PRESTO-EPA-CPG and PRESTO-EPA-POP Methodologies

The calculation of maximum annual CPG dose relies upon the PRESTO-EPA-POP approach with appropriate modifications. Table 8-1 summarizes the major differences between the two models. The first difference is that the radionuclide-specific activity used in the CPG calculations is the best estimate of the total activity and volume of waste to be disposed of at a facility (assumed to be generally 250,000 m^3). The radionuclide-specific inventories, by waste volume, are listed in Chapter 3. This is in contrast to the normalized unit volume and unit curie inventory used in PRESTO-EPA-POP, in which the cumulative population health effects from a fully-loaded disposal facility are calculated in a separate utility program. This aspect of the PRESTO-EPA-POP code is discussed in Section 8.3.6.

Another difference lies in the treatment of waste leaching, which varies according to waste form. The total source term for the CPG analysis contains various waste streams which are grouped into one of five waste forms: (1) absorbing materials, (2) trash, (3) solidified, (4) activated metals, and (5) incinerated-solidified.

The leaching of absorbed wastes and a fraction of the trash are estimated using only a distribution factor (K_d). Leaching from the other waste forms is characterized by a radionuclide-independent annual leach fraction for each waste form, followed by K_d leaching of the leached fraction.

In PRESTO-EPA-CPG, the waste is broken down into each of the five waste form classes. In any given year, the leaching of each radionuclide is assumed to occur in two stages, based on waste class. First, the activated metals, the solidified wastes, the incinerated-solidified wastes, and that fraction of trash that is not absorbing, are assumed to leach in accordance with their respective leach fractions. The total

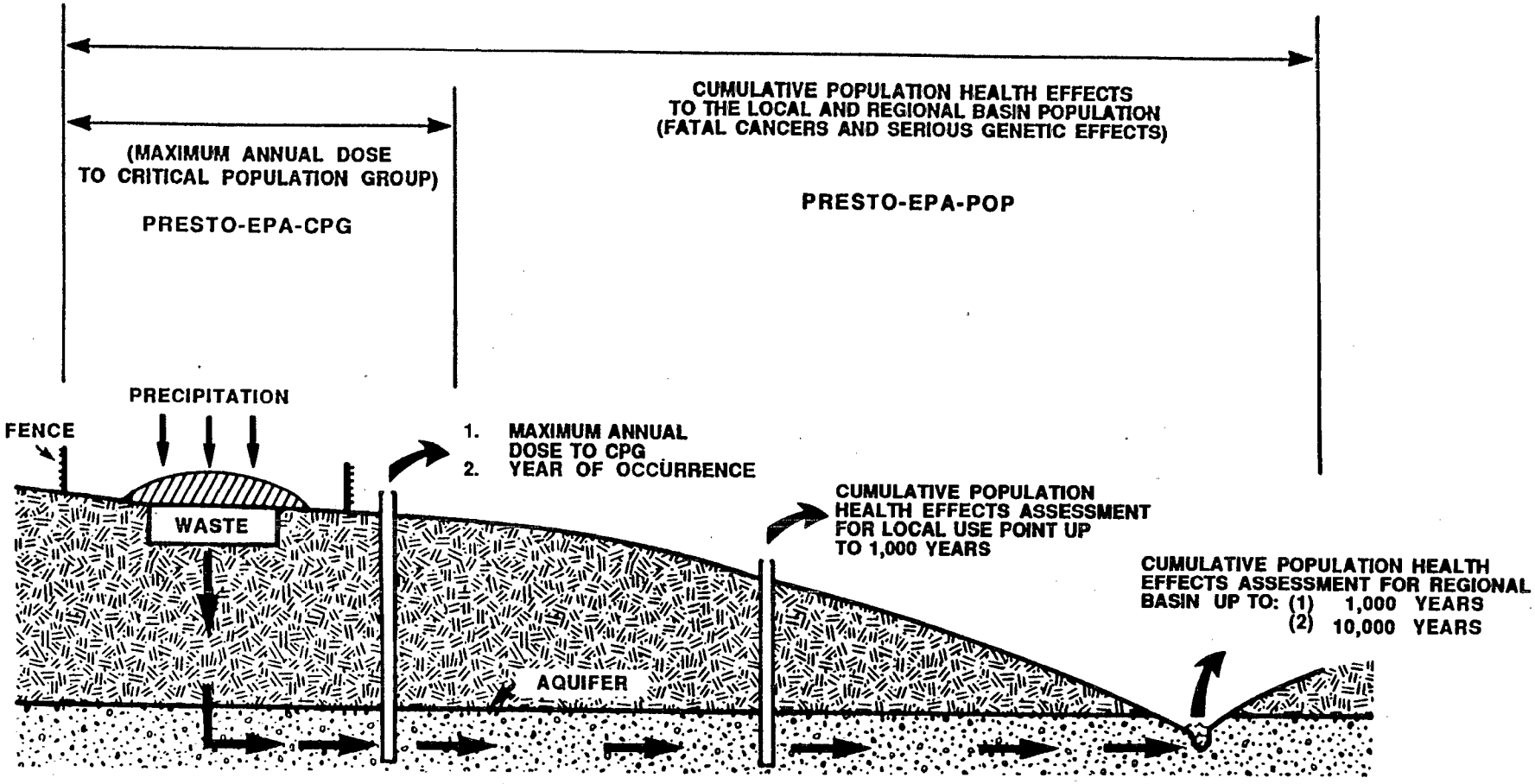


Figure 8-7. Differences in Health Impacts Estimated and Locations and Populations Evaluated for PRESTO-EPA-POP and PRESTO-EPA-CPG

Table 8-1. Main differences between PRESTO-EPA-POP and PRESTO-EPA-CPG

Characteristic	PRESTO-EPA-POP	PRESTO-EPA-CPG
Population Analyzed	Local and Regional Basin Population	Critical Population Group
Impact Analyzed	Cumulative Population Health Effects	Maximum Annual Whole-Body Dose
Modeling Period (years)	1,000 local population, 10,000 combined local and regional population	1,000 modeled, with maximum year determined
Source Term	1 curie of each nuclide (unit response approach)	Best estimate of actual 20-year disposal activities
Site Size	1 m^3 volume (unit response approach)	250,000 m^3
Waste Form/Leaching	Separate runs performed for each waste form with appropriate leaching (unit response approach)	Waste forms (five types) combined with a two-step leaching process performed

amount (curies) of each radionuclide leached in the facility water from these wastes is added to the absorbing material inventory present at the beginning of that year. This new absorbing material inventory is partitioned between the contacting water and the absorbing phase according to the distribution coefficients (K_d).

In PRESTO-EPA-CPG, the focus of attention for each disposal alternative at each regional site is on identifying the year when the dose to the CPG is maximum. Therefore, in PRESTO-EPA-CPG, at the end of the simulation when the year of maximum equivalent whole-body dose rate has been identified, radionuclide uptake information for that year is input to the DARTAB subroutine. This subroutine then calculates organ, pathway, and radionuclide whole-body dose equivalent rates for the critical population group for the maximum year.

Lifetime risk to a member of the CPG is estimated using the maximum annual dose rate, assuming it remains constant over the individual's lifetime (average of 71 years). This calculation is done outside of the PRESTO-EPA model and is based on a standard dose to risk conversion factor. The association of a definite level of individual risk with a maximum year's whole-body dose equivalent is tentative. Individuals in the CPG may experience several years at or near the maximum exposure. Where the half-life of the radionuclide is long and the maximum CPG exposure occurs relatively late after disposal site closure, the exposure may continue at close to the maximum level for many years. If the radionuclides causing the exposure have short half-lives and the maximum CPG exposure occurs soon after closure, an individual may be exposed to the maximum rate for only a few years. This leads to a possible overestimate of the risk. In most cases, however, the maximum dose remains near the maximum level for many years.

Another uncertainty is the duration of individual residence time at the CPG location. A wide range of individual risk estimates is possible depending upon the length of residence. In order to be conservative, however, we assume that the critical population group will remain in the same location throughout its lifetime.

(B) General Characteristics of the PRESTO-EPA-CPG Analyses

The results from PRESTO-EPA-CPG are maximum annual whole-body doses to a critical population group located close to the disposal site. The results of the analysis allow EPA to determine which disposal alternatives would meet various levels of the standard under certain generic circumstances. The specific estimates from our various computer runs are detailed in Chapters 9 and 11. In the following sections, general results for the PRESTO-EPA-CPG code are discussed.

Typical results show that, in general, the impacts to the critical population group and the pathways by which they occur vary considerably over the three hydrogeologic and climatic regions (see Figure 8-4).

Therefore, the analysis results are reported for each of the three general settings. At the humid permeable site, the maximum dose occurs through the ground-water pathway. The important nuclides are those with high mobility (low K_d values), such as H-3, C-14, and I-129. They reach the critical population group within 1,000 years when combined with relatively high ground-water velocities.

At the humid impermeable site, the maximum dose occurs within about 100 years of failure of the trench cap (assumed to occur in year 100) via trench overflow directly to the surface-water pathway. The important nuclides are those that are relatively mobile and have longer half-lives. An example is I-129, which reaches the critical population group soon after the trench cap fails. It leaves the trench via overflow and is transported directly to the local stream by surface water, thus bypassing the greater retardation it would have if it had moved through the ground. Nuclides with shorter half-lives, such as H-3, cause few high doses due to their decay during the period the trench cover remains intact.

At the arid permeable site, the maximum dose can occur in the first year after closure because of the atmospheric transport of less mobile nuclides, such as Co-60, Cs-137, and Ba-137m, spilled onto the surface soil during site operations. This dose is very small (much less than one mrem), however, since only a fraction of the total activity brought onto the site is assumed to have been spilled during operations and even less reaches the downwind population after dilution and dispersion by atmospheric transport. A greater dose may occur through the ground-water pathway, either late in the modeling period or even after 1,000 years. The later doses can be significantly larger, although still very small (much less than one mrem), and are dominated by mobile nuclides with relatively long half-lives, such as C-14 and I-129.

8.4.3 PRESTO-EPA-DEEP

The PRESTO-EPA-DEEP code estimates cumulative population health effects from LLW disposal by deep-well injection, hydrofracture, and deep geologic disposal. The code estimates, for up to 10,000 years following the end of LLW disposal operations, local and regional basin health effects. The maximum annual dose to a critical population group is calculated by the PRESTO-EPA-CPG code, using all of the modifications and assumptions assumed in this section, and the full source term as required for the CPG analysis. The values used for the PRESTO-EPA-DEEP input parameters are listed in Appendix C.

Water, principally from deep aquifers, is the primary transport medium for radioactivity from LLW stored in deep facilities. Water moving upward through the deep facility may ultimately enter a shallow aquifer. Radionuclides that enter the aquifer may eventually reach irrigation or drinking wells or surface streams and be consumed. The consumption of radionuclides is through the food chain pathway, modeled in the same manner as that in PRESTO-EPA-POP.

The deep disposal scenarios implemented in PRESTO-EPA-DEEP consider only the naturally occurring pathways such as natural ground-water and surface-water flows and, in some scenarios, atmospheric air transport. Intrusion scenarios, such as accidental drilling, geological faulting, and the failure of the access shaft sealing, have a probabilistic nature and are not considered.

In general, the environmental transport pathways are the same for PRESTO-EPA-POP and PRESTO-EPA-DEEP. The major modifications found in PRESTO-EPA-DEEP include extension of the period for local population analysis, modification of the ground-water transport submodel, and the bypassing of some submodels that are not applicable. These bypasses include the infiltration submodel in the case of all the deeper geological disposal alternatives and the air transport submodels in the case of deep disposal in a mined cavity.

PRESTO-EPA-DEEP considers the vertical movement of ground water from a lower confined aquifer through the waste facility and surrounding strata to an upper aquifer, as shown in Figure 8-8. Water in the upper aquifer moves horizontally to a receptor location where the water and radioactive contaminants are introduced into the food chain pathway in the same manner as that in PRESTO-EPA-POP.

In addition to the major changes made in the ground-water transport pathway, minor modifications had to be made to certain portions of the model to simulate the deep disposal scenarios. One change was to increase the simulation time frame for the local population analysis from 1,000 to 10,000 years. This change was necessary because of the long time periods required for radionuclides to travel from the deeply located facilities to the local population. The PRESTO-EPA-DEEP model is discussed in detail in the PRESTO-EPA-DEEP User's Manual (EPA87c).

8.5 Risk Assessment - Unregulated Disposal (BRC)

The methods used to estimate the health impacts resulting from disposal of certain very low activity LLW by less restrictive practices than those currently used are presented in this section. A number of surrogate waste streams and several types of facilities that would ultimately receive them for disposal are identified. The migration of radionuclides from these facilities through various pathways and the resulting human exposures are calculated. From this information, cumulative population health effects and maximum annual doses are calculated. The results of this analysis are made compatible with the results of the LLW analysis so that comparisons of disposal methods can be made for LLW and BRC waste.

The PRESTO-EPA-BRC health impact assessment model (EPA87e), a modified version of the PRESTO-EPA code, is the primary analytical tool used in the cumulative population health effects assessment. In addition, an accounting model (see Section 8.3.6) is used to facilitate

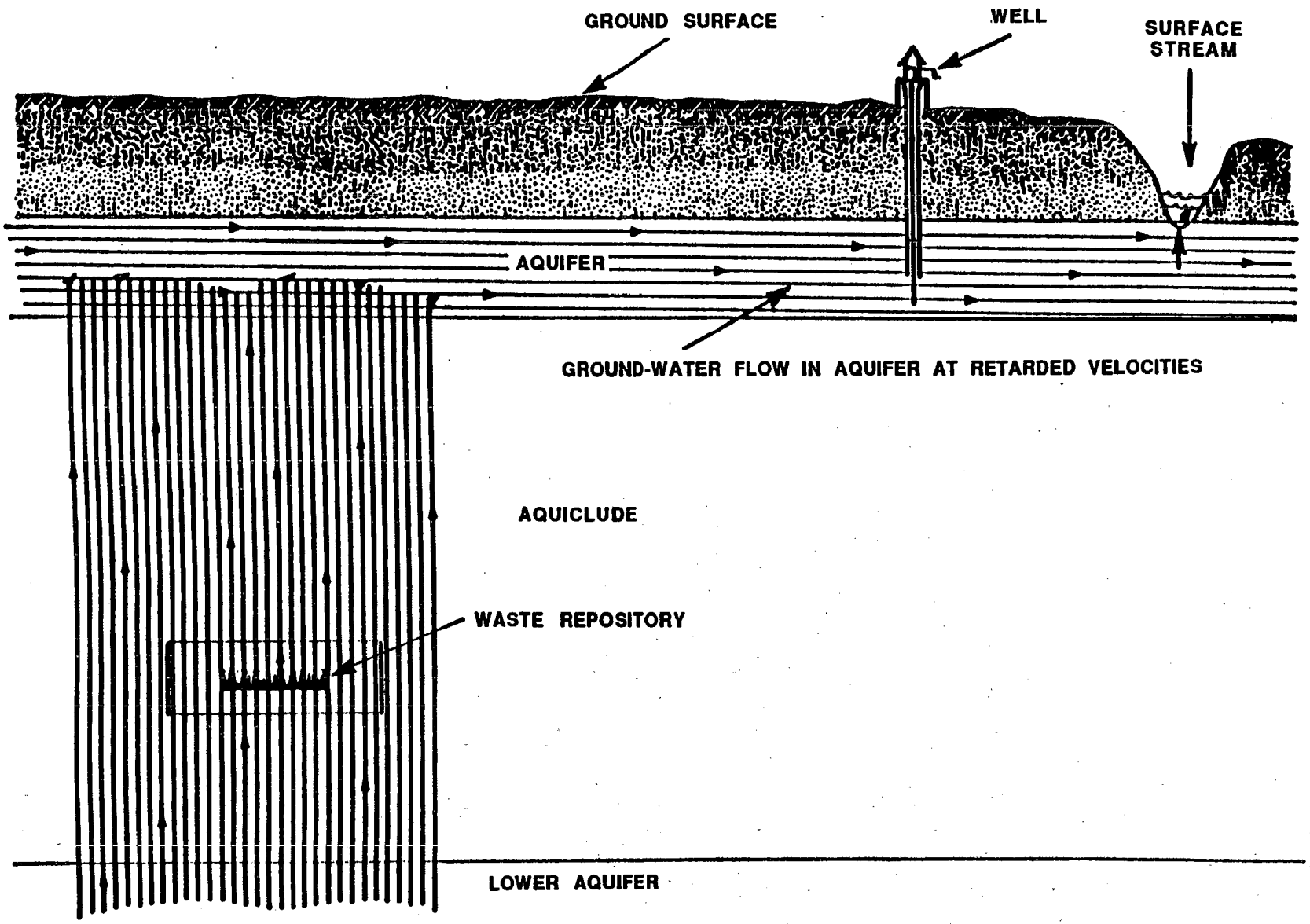


Figure 8-8. Ground-Water Model for Deep Disposal Scenarios

the use of the unit response analysis methodology. The PATHRAE-EPA code (EPA87f) is the primary tool used to estimate maximum annual doses to the CPG.

8.5.1 PRESTO-EPA-BRC

The PRESTO-EPA-BRC code, which is used to estimate cumulative population health effects from unregulated disposal of LLW, is a modified version of the PRESTO-EPA model. In addition to the standard pathways, the code includes the capability to determine health effects from onsite operations and airborne radioactivity released during incineration of the waste. The number of onsite workers during the active operation of the site is defined, and the maximum exposure level to which they are subjected is estimated. For scenarios involving incineration of the waste, a second set of air pathway parameters is required to account for exposures to onsite and offsite individuals during the period of incineration. These parameters are the fractions of time that the wind blows toward the population of interest when incineration is considered and the exposure per unit source strength. These parameters are determined in the same manner as were the general air pathway parameters, but apply only during the operational period of the incinerator. In addition, a fractional volatility factor must be specified for each radionuclide to facilitate calculation of the quantity of radioactivity being emitted by the incinerator. The input values associated with these parameters are listed in Appendix C. For more detailed information on the PRESTO-EPA-BRC code, see the User's Manual (EPA87e).

A number of scenarios were developed for the BRC health impact assessments, including a variety of deregulated disposal alternatives, i.e., sanitary landfills, municipal dumps, onsite landfills, and incineration methods situated in rural, suburban, and urban settings. For more detailed information on the disposal methods, see Chapters 4, 5, and 10.

In evaluating whether some types of LLW might potentially be BRC wastes, a number of surrogate LLW wastes were analyzed, as they have been generated at nuclear power reactors, uranium fuel fabrication and uranium process facilities, and industrial, medical, and educational facilities, as well as by consumers. Since the BRC analysis assumes that the waste will receive no special treatment or packaging, the "as is" waste form is used. For more detailed information on waste streams and waste form, see Chapter 3.

The hydrogeologic settings used were comparable to those analyzed in the LLW scenarios, although demographic characteristics were modified to model more urban settings (see Chapter 4). The input parameters and parameter values for the PRESTO-EPA-BRC code are listed in Appendix C. The radionuclides used in the BRC analysis were comparable to those used in the LLW analysis.

The results of the PRESTO-EPA-BRC analysis are cumulative population health effects, consisting of fatal cancers and serious genetic effects, to a local and a regional basin population. These results are similar to those from the PRESTO-EPA-POP analysis. The results of the PRESTO-EPA-BRC analysis are presented in Chapter 10.

8.5.2 PRESTO-EPA-BRC Pathways

The evaluation of cumulative population health effects from disposed BRC waste involves exposure during the operating period of the disposal facility. For example, workers at a sanitary landfill are not radiation workers and their doses must be considered in evaluating the cumulative population health effects. In addition, the general public has access to many facilities appropriate for BRC waste disposal. Since the primary PRESTO-EPA computer code for assessing population health effects did not consider exposure mechanisms possible with BRC waste disposal, it was necessary to develop a new computer program to accomplish this function.

The PRESTO-EPA code was modified by additional exposure pathways as discussed below. The resulting computer code, PRESTO-EPA-BRC, more completely assesses the cumulative population health effects that could result from unregulated or less restrictive disposal of BRC waste. The other pathways are the same as those used with the PRESTO-EPA-POP model. The model assumes that the radionuclide inventory is the amount of activity found in the facility at the end of the disposal operation. The wastes disposed of in the facility are assumed to be a homogeneous mixture of radionuclides and waste materials.

The exposure pathways for the BRC scenarios are listed in Table 8-2. Other combinations of pathways may be specified by changing the input parameters. We consider these to be the maximum exposures involving the actual BRC waste disposal activities, either during operations or after site closure.

The major modifications to the PRESTO-EPA code that were required for BRC analysis, involve adding the ability to calculate and accumulate exposures during the disposal facility operations (Ro84a). Since PRESTO-EPA is oriented to post-closure events, all impacts from BRC operations are summed over the 20 years of operations and assumed to occur in the first year after site closure in PRESTO-EPA-BRC. Major modifications incorporated into PRESTO-EPA-BRC include the following supplemental pathways in addition to the regular PRESTO-EPA pathways:

- Worker and site visitor dust inhalation during operations;
- Population dust inhalation from mechanical disturbances during operations;
- Population inhalation of incinerator releases during operations;

Table 8-2. Radiological exposure pathways for PRESTO-EPA-BRC scenarios

Scenario	For population
Normal	Ingests offsite water Ingests offsite foods Inhales downwind air
Farming	Ingests onsite foods Ingests offsite water
Eroded Trench	Inhales suspended material at centroid Direct exposure plume

- Worker and site visitor gamma exposure during operations; and

- Changes in the onsite farming option; includes human intrusion onsite (construction of a house, i.e., residential use) immediately after closure.

These additional exposure pathways and changes in PRESTO-EPA are described in the following sections. For our modeling we assume that during the first year following closure the maximum health impact will occur.

(A) Dust Exposures

When calculating health effects to non-radiation workers at typical municipal waste disposal facilities, such as SF's and MD's, it is necessary to consider the disposal facility workers and members of the general public who frequent the facility as non-radiation workers. Members of the general public who frequent the facility include those who themselves dispose of waste and refuse at the facility and unauthorized persons who are potential scavengers. Both workers and visitors to the site will be exposed to varying levels of potentially contaminated atmospheric dust and direct gamma radiation, depending on their locations and times spent at each location.

Dust exposures to the population occur through the following four mechanisms:

(1) Worker and Site Visitor Dust Exposure

Worker and site visitor dust exposures are assumed to be constant from year to year during operations. Thus, the cumulative dust exposure over the operational life of the facility can be determined by multiplying the annual exposure rate by the operational life of the facility. In PRESTO-EPA-BRC, this cumulative exposure is calculated and included with the other post-closure exposures that do not occur during the 20 years of BRC waste disposal operations. This cumulative exposure is then considered in the first year after operations cease, which is the first year of the PRESTO-EPA-BRC simulation.

(2) Population Dust Exposure from Mechanical Disturbances

The cumulative population dust exposure resulting from mechanical disturbances during operations is also calculated for 20 years and considered with the other events during the first year of post-closure during the PRESTO-EPA-BRC simulation.

(3) Population Exposures from Incinerator Releases

The cumulative population exposure resulting from incinerator releases during operations is also calculated for 20 years and considered with the other events in the first year of post-closure simulation.

(4) Population Exposure from Natural Resuspension

The population exposure resulting from natural resuspension is calculated during operations in the identical manner as in PRESTO-EPA after operations. As with other dust exposures during operations, this exposure is accumulated over the facility's operational life and is added in the first year of post-closure simulation. Transport and dispersion from the site are also included.

(B) Worker and Site Visitor Gamma Exposure

In calculating exposures resulting from direct gamma radiation by PRESTO-EPA-BRC, the maximum expected gamma radiation level is used for calculating gamma doses to workers and visitors. Depending on their locations, the workers and visitors are then exposed to some gamma radiation depending on time spent at various locations near the waste trenches. The equivalent full-time, full-exposure population is used to represent population exposure to workers and visitors from gamma radiation. The cumulative gamma exposure to both workers and site visitors is also accumulated over the 20-year facility lifetime and maximized in the first year after closure.

(C) Onsite Farming

In the post-operation period, i.e., the period after the site is closed and returned to unrestricted use (considered to be the first year after closure), for the farming or reclamation scenario, the onsite farmer may grow and eat his or her own vegetables, beef, and milk produced on land irrigated by the potentially contaminated onsite water, but drink offsite public supply water equal in concentration to the normal scenario population intake. The farmer also may inhale suspended, contaminated soil from the residual operational spillage. Population dose and risk calculations under the farming scenario may assume that the food products grown on the site are ingested by the general population or by the farmer and his family. (Appendix C presents food product parameters.)

PRESTO-EPA-BRC will calculate exposures resulting from consumption of vegetation grown onsite. PRESTO-EPA-BRC modeling also calculates population dust exposures from post-closure activities such as onsite farming.

8.5.3 PATHRAE-EPA

The PATHRAE-EPA code, which was originally developed by Rogers and Associates Engineering Company (Ro84a), is used to assess the exposures to the CPG from the unregulated disposal of BRC waste. While this code is not directly based on PRESTO-EPA, it was modified extensively to incorporate the analytical concepts used for the PRESTO-EPA family of codes. The PATHRAE-EPA code most closely resembles PRESTO-EPA-CPG, and

studies have been done to compare the output. The results of the studies show the output of the two codes to be comparable under the circumstances for which they are being used in the LLW standard development (Sh86).

The computer code PATHRAE-EPA is designed to assess the maximum annual CPG dose for each exposure pathway resulting from the disposal of LLW. Maximum annual doses are calculated to workers during disposal operations, to offsite personnel after site closure, and to reclaimers and inadvertent intruders after site closure. Dose conversion calculations are performed to give annual doses. The PATHRAE-EPA code is described in greater detail in the PATHRAE-EPA User's Manual (EPA87f).

The main advantages of the PATHRAE-EPA model are its ease of operation and simplicity of presentation, although with the simplicity comes some sacrifice in the accuracy of the dose assessment and the loss of the ability to assess combined effects from each pathway. It is felt that this loss is not significant, however. Site performance and facility designs for LLW disposal can be readily investigated, with relatively few parameters needed to define the problem. Some important parameter values are obtained from the results of PRESTO-EPA calculations. Results are annual doses, by radionuclide, as a function of time for each pathway, as well as total annual dose rates with time. This permits quick focusing on key pathways and parameters.

For conservatism, the entire radionuclide inventory is used as the source term for each pathway calculation, and depletion of the inventory via migration through other pathways is ignored. This, of course, provides conservative estimates while saving computer time.

The PATHRAE-EPA methodology considers both offsite and onsite pathways through which man can be exposed to radioactive waste. The onsite pathways include the ingestion of food grown onsite, direct gamma exposures to workers and intruders, and the inhalation of radioactive dust by workers and intruders. Offsite pathways are the same as for the PRESTO-EPA codes.

The PATHRAE-EPA analysis produces a set of annual doses to an individual as a function of time, nuclide, and pathway. Radionuclide concentrations in river water and well water are also given for times up to 10,000 years.

8.5.4 PATHRAE-EPA Pathway Analysis

In considering the pathways through which man can be exposed to BRC waste, both onsite and offsite, two major assumptions were made:

1. Waste materials in the trench are assumed to be a homogeneous mixture of radionuclides and other waste materials.

2. Radionuclides are transported vertically from the trench bottom to the aquifer and then horizontally through the aquifer.

(A) The Onsite Worker Pre-Closure Pathways

Pre-closure exposures to onsite workers occur through two pathways: (1) direct exposure to gamma radiation from the buried waste and (2) internal exposures from radioactive dust inhaled during operations.

These two pathways are calculated in the last year of operation for the 20-year accumulation of decayed radioactive waste and occur for all three hydrogeologic/climate settings. This is based on the assumption that the last year would provide the maximum exposure for the 20-year accumulation of waste.

(B) The Post-Closure Onsite Resident Pathways

Post-closure exposures to onsite residents occur through two pathways - ingestion of food grown onsite and biointrusion.

In the food grown onsite pathway radionuclides are brought to the surface by construction activities or burrowing animals, both of which disturb trench cover to a depth of 3 meters. The food plants grown in onsite gardens are then assumed to absorb radionuclides from the disturbed ground.

In the biointrusion pathway, the roots of food plants grown in onsite gardens are assumed to penetrate into the undisturbed waste (greater than 6 meters), and the plants are later consumed by humans.

These two pathways are calculated in the first year after closure for the 20-year accumulation of decayed radioactive waste and occur for all three hydrogeologic/climatic settings. This is based on the assumption that the first year following closure, and the site's return to unrestricted use, will provide the maximum exposure based on 20 years of accumulated BRC waste.

(C) Post-Closure Offsite Resident Pathways

Post-closure exposures to offsite residents occur mainly through water pathways: ground-water to the river, facility overflow (bathtub effect), surface erosion, and ground-water to the well.

In the surface water or ground-water to the river pathway, the contaminated leachate from the waste trenches migrates through the ground water to a major aquifer that supplies a nearby river used for irrigation, livestock, or domestic purposes.

In the facility overflow or commonly called the bathtub effect pathway, the disposal trenches accumulate water because of trench cap failure and eventually overflow to nearby surface streams.

During surface erosion, the cover and subsequently the waste itself are eroded. The radionuclides then may reach nearby surface waters by overflow.

The ground-water to well pathway is based on a nearby well used for irrigation, livestock watering, or domestic purposes, and is contaminated with leachate through ground-water migration from the waste trenches.

The river water and facility overflow pathways occur only for the humid impermeable hydrogeologic/climatic setting; the surface erosion pathway occurs only for humid climatic settings. The well water pathway occurs at all settings. The ground-water to river, ground-water to well, erosion, and overflow pathways all occur at from 100 to thousands of years after closure.

(D) Pre-Closure Offsite Pathways

Pre-closure exposures to offsite residents occur through two pathways: spillage and atmospheric inhalation.

The spillage pathway occurs during placement in the trench and the spilled material mixes with surface water and discharges to a nearby stream.

For the atmospheric inhalation pathway, dust resuspension, a trench fire, or a waste incinerator may be a source of contaminated gas and particulate matter in which the radioactive plume migrates offsite before affecting people.

For these two pathways the spillage and atmospheric inhalation occur in the last year of operation. The spillage pathway occurs only for the humid impermeable hydrogeologic/climate setting. The atmospheric inhalation pathway occurs at all settings.

8.5.5 Additional BRC Analyses

In addition to the PRESTO-EPA-BRC and PATHRAE-EPA runs, other analyses are required to evaluate BRC waste disposal scenarios. These analyses include the evaluation of transportation exposures and exposures caused by flooding of the disposal site.

A special analysis was made to evaluate direct radiation exposures to workers who would collect and transport BRC wastes from the generator to the disposal facility. For this analysis, additional short-lived nuclides (half-lives of less than 1 year) were included, as they might provide additional direct radiation doses. The results of this analysis are described in Chapter 10.

Preliminary analyses were performed to evaluate the risks from disposal site flooding (La84). The exposures from these scenarios were

found to be minimal or much less than other scenarios because of the effect of dilution from flooding. These analyses were not included in our final methodology or results.

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Chapter 9: ESTIMATED DOSES AND HEALTH EFFECTS FROM THE REGULATED DISPOSAL OF LLW

9.1 Introduction

Previous chapters provided detailed descriptions of the key elements needed to perform a risk assessment of LLW disposal. LLW has been described in terms of volumes and concentrations representing numerous waste streams (Chapter 3). Disposal methods have been identified and described (Chapter 4), along with major categories of hydrogeological and climatic settings (Chapter 5). The calculational models describing environmental transport and radiation dosimetry have also been defined (Chapters 6, 7, and 8).

This chapter presents the rationale for the selection and results of the base case assessments of health impact performed for the regulated disposal of LLW. Comparing the potential impacts from LLW disposal under a broad range of disposal alternatives and regional conditions is an important element supporting the development of EPA's generally applicable LLW standards, as these standards will apply to LLW facilities throughout the United States.

The only practical method of reducing the hazard of the land disposal of LLW is to isolate it from people and the environment until the radioactivity has decayed to very low levels. This assessment projects the capability of the disposal system for isolating the radioactivity in LLW from human populations. These results reflect the undisturbed performance of an engineered LLW disposal system without disruption by human intrusion or unlikely natural events. Such external disruptions may best be handled on a site-specific basis.

This chapter compares the undisturbed performance of engineered disposal systems located in various hydrogeologic settings in terms of two critical radiological effects: (1) the maximum annual critical population group (CPG) dose (provided in terms of a committed effective whole-body dose equivalent), and (2) cumulative population health effects (in terms of fatal cancers or serious genetic effects). The results given in this chapter can be described as the base case analysis, because the selection of disposal methods and waste forms is intended to illustrate a step-wise progression in technological sophistication rather than all possible combinations of disposal methods and waste forms. Chapter 11, sensitivity analysis, presents a complete listing of the results obtained for all combinations of disposal methods and waste forms investigated.

9.2 Input Data and Rationale for Base Case Analysis

The purpose of this chapter is to compare the health impacts from the undisturbed performance of engineered disposal systems covering a

wide range of technological sophistication. In order to evaluate health impacts, certain critical input data must be specified for each analysis. These factors include the source term, the hydrogeologic setting, the engineered disposal method and associated waste forms, and the radiological risk assessment model. These factors are described in Chapters 3, 4, 5, and 8. The following sections discuss the pertinent data, assumptions, and rationale related to each factor that is used to construct the base case analysis.

9.2.1 The Low-Level Radioactive Waste Source Term

The overall LLW source term is described in Chapter 3. Twenty-six waste streams are defined based on similarities in origin and their general physical, chemical, and radiological characteristics. Table 3-2 indicates 24 waste streams are presently regulated under AEA authority. As explained in Chapter 3, the EPA source term is based in large part upon the NRC characterization of commercial LLW (NRC86, Gr86) regulated under the AEA. In addition, EPA includes two high radionuclide-concentration source streams containing NARM wastes (PEI85).

The 26 waste streams included in EPA's overall LLW source term and their projected 20-year volumes for 1985-2004 are shown in Table 9-1. The NRC classification of these waste streams under 10 CFR 61 and the physical-chemical form assigned to each waste stream are also indicated in the table. Since the EPA analysis of regulated LLW disposal is principally directed at estimating and comparing impacts over the long term, the EPA LLW source term considers only longer-lived (i.e., half-life of more than one year) radionuclides. In addition to long half-life, certain radionuclides are included if they exhibit high radiotoxicity (e.g., I-129, Np-237) and/or are present in significant amounts in LLW (e.g., Cs-134). Table 9-2 lists 34 radionuclides considered in the EPA analysis. The inventories shown in this table reflect the total projected activities in commercial LLW and NARM from 1985 to 2004. The total volume associated with the commercial LLW inventory is about 3E+06 m^3 over the same time period.

In order to derive the source term used for the base case analysis, it is necessary to modify the overall LLW source term described above. To illustrate implementation of all phases of the EPA LLW standard, namely, a generally applicable radiation protection standard with provisions for inclusion of high specific activity NARM wastes and implementation of a BRC level, an appropriate LLW source term is constructed for the base case analysis. To illustrate implementation of BRC, seven waste streams are excluded from consideration as a regulated LLW. These waste streams are as follows: N-SSTRASH, N-SSWASTE, F-COTRASH, F-NCTRASH, U-PROCESS, F-PROCESS, and I-LQSCNVL.

Table 9-1. Overall LLW source term: Commercial LLW and NARM volumes by waste stream, 1985 - 2004 (PHB85)

Waste stream[a]	10 CFR 61 Class	Waste form[b]	Volume (m^3)
L-IXRESIN	B	AW	9.91E+04
L-CONCLIQ	A	AW	3.31E+05
L-FSLUDGE	B	AW	1.31E+05
P-FCARTRG	A	TR	1.28E+04
L-DECONRS	C	AW	2.24E+03
L-NFRCOMP	A	AM	6.45E+04
F-PROCESS	A	AW	5.95E+04
U-PROCESS	A	AW	2.14E+04
L-COTRASH	A	TR	5.98E+05
L-NCTRASH	A	TR	4.78E+05
F-COTRASH	A	TR	1.79E+05
F-NCTRASH	A	TR	3.17E+04
I-COTRASH	A	TR	2.82E+05
N-LOTRASH	A	TR	1.01E+05
N-SSTRASH	A	TR	3.59E+05
N-SSWASTE	A	TR	6.34E+04
I-LQSCNVL	A	AW	1.50E+04
I-ABSLIQD	A	AW	1.11E+04
I-BIOWAST	A	AW	7.52E+03
N-LOWASTE	A	AW	6.03E+04
N-ISOPROD	C	TR	9.97E+03
N-SOURCES	C	AM	5.82E+02
N-TRITIUM	B	TR	6.94E+03
N-TARGETS	B	AM	2.23E+02
R-RAIXRSN	C	AW	6.60E+03
R-RASOURC	C	AM	4.45E-01
		Total Volume	2.93E+06

[a]See Table 3-2 for description.
[b]As-generated waste form:
AW Absorbing Waste
AM Activated Metal
TR Trash

Table 9-2. Estimated total activity of major radionuclides in commercial LLW and NARM, 1985 - 2004

Nuclide	Activity*	Nuclide	Activity*
H-3	1.80E+06	Po-210	6.82E+02
C-14	5.88E+03	Pb-210	6.82E+02
Fe-55	3.99E+06	Bi-214	6.82E+02
Ni-59	2.66E+03	Pb-214	6.82E+02
Co-60	3.35E+06	Ra-226	7.41E+02
Ni-63	3.61E+05	U-234	8.21E+01
Sr-90	7.33E+05	U-235	2.85E+00
Nb-94	2.70E+01	Np-237	1.55E-04
Tc-99	2.49E+01	U-238	3.41E+01
Ru-106	4.00E+03	Pu-238	1.13E+03
Sb-125	5.68E+03	Pu-239	4.12E+02
I-129	6.95E+01	Pu-241	1.76E+04
Cs-134	6.62E+05	Am-241	1.68E+03
Cs-135	2.47E+01	Pu-242	8.55E-01
Cs-137	9.66E+05	Am-243	2.53E+01
Ba-137m	9.66E+05	Cm-243	2.56E+01
Eu-154	5.70E+02	Cm-244	3.33E+02

Total Activity: 1.29E+07 Ci.

*Activity is in Ci.

These particular waste streams have low concentrations of radionuclides. Table 9-3 lists the remaining waste streams and estimates the volume of each for a disposal site capacity of 2.5E+05 m^3. This reference site capacity is derived by consideration of the total LLW volumes presented in Table 9-1 divided by an assumed number of 10 to 12 disposal sites formed under the LLRWPAA of 1985. While the number of State compacts may vary, present indications are that there will be quite a few such compacts. Exclusion of the seven "designated" BRC waste streams reduces the 20-year volume of regulated LLW from about 3E+06 m^3 to about 2.2E+06 m^3 (Table 9-1). Rather than reduce site capacity accordingly, the base case analysis presumes that the 2.5E+05 m^3 site is filled to capacity with the remaining waste streams shown in Table 9-3. Chapter 11 provides additional analyses of the implications of including or excluding certain categories of LLW (e.g., BRC, NARM). In general, excluding BRC wastes from the site inventory while simultaneously allowing the corresponding volume to be filled with the remaining higher activity LLW results in slightly higher CPG doses and health effects.

9.2.2 Hydrogeologic/Climatic/Demographic Conditions

The hydrogeologic and climatic conditions at a site can directly affect and change the importance of pathways and impacts of releases from a LLW disposal facility. Because the LLW standards will be applied to LLW facilities throughout the United States, they must be applicable under a wide range of conditions. Therefore, we have conducted all of our base case health impact assessments under three very different regional hydrogeologic/climatic scenarios. These three scenarios were also used for our sensitivity analyses of the health impact assessment codes.

The three hydrogeologic/climatic scenarios used are for sites in humid permeable, humid impermeable, and arid permeable regions. Figure 8-6 illustrates the environmental pathways for the transport of water (and radionuclides) at these three hydrogeologic/climatic settings. Realistic site data, which are typical for these scenarios, were obtained from USGS studies at the LLW disposal facilities at Barnwell, West Valley, and Beatty. General population distributions and water usage patterns typical of these three climate settings were also used. These scenarios span a wide range of conditions under which a disposal facility would normally be sited in the continental United States. Chapter 5 provides more detail concerning these hydrogeologic/climatic/demographic settings.

9.2.3 Disposal Methods

Nine different disposal methods are defined for the overall analysis of the land-based disposal of LLW. Conceptual designs for these disposal methods are developed in sufficient detail to estimate disposal costs and health impacts. The disposal methods chosen for analysis represent a wide range of technological sophistication and would need little or no

Table 9-3. Input data for EPA's base case analysis: commercial LLW and NARM disposed of in a regulated disposal facility, 1985-2004

Waste stream[a]	10 CFR 61 Class	Waste form[b]	Volume (m^3)
L-IXRESIN	B	AW	1.13E+04
L-CONCLIQ	A	AW	3.75E+04
L-FSLUDGE	B	AW	1.48E+04
P-FCARTRG	A	TR	1.46E+03
L-DECONRS	C	AW	2.54E+02
L-NFRCOMP	A	AM	7.32E+03
L-COTRASH	A	TR	6.78E+04
L-NCTRASH	A	TR	5.43E+04
I-COTRASH	A	TR	3.20E+04
N-LOTRASH	A	TR	1.15E+04
I-ABSLIQD	A	AW	1.26E+03
I-BIOWAST	A	AW	8.54E+02
N-LOWASTE	A	AW	6.85E+03
N-ISOPROD	C	TR	1.13E+03
N-SOURCES	C	AM	6.61E+01
N-TRITIUM	B	TR	7.88E+02
N-TARGETS	B	AM	2.54E+01
R-RAIXRSN	(NARM)	AW	7.49E+02
R-RASOURC	(NARM)	AM	5.05E-02
		Total Site Volume	2.5E+05

[a]See Table 3-2 for description.
[b]As-generated waste form:
AW Absorbing Waste
AM Activated Metal
TR Trash

engineering development. A special tenth disposal option is also presented to illustrate current disposal practice according to the requirements of the 10 CFR 61 disposal technology. These regulations actually imply the use of a combination of two of the nine basic disposal methods listed in Table 9-4, as described below.

Disposal alternatives chosen to represent the results of the base case analysis include those that can handle all 26 LLW streams, represent a wide range of technological sophistication, and typify to some degree past, present, and potential disposal techniques. They include:

1. Regulated Sanitary Landfill;
2. Shallow Land Disposal;
3. Improved Shallow Land Disposal;
4. Current Disposal Practice: 10 CFR 61;
5. Intermediate Depth Disposal; and
6. Concrete Canister Disposal.

A regulated SLF is the simplest land disposal technology analyzed. This would essentially be a landfill operating in accordance with EPA regulations, 40 CFR 241 to 257, but with the additional features required for a facility authorized to receive, handle, and dispose of radioactive materials. SLD represents LLW disposal as practiced between 1963-1980 in the United States. ISD incorporates all of the requirements of 10 CFR 61, but places all classes of LLW in narrower, deeper trenches than those used by SLD. Current disposal practice according to 10 CFR 61 disposal technology is actually a combination of the SLD and ISD methods. Class A and Class B wastes are disposed of in separate disposal trenches typical of SLD, while Class C wastes are disposed of in trenches typical of ISD. IDD meets the requirements of 10 CFR 61 disposal technology, but places the waste even deeper than ISD (i.e., 15 meters vs 8 meters below ground level). Finally, the CC method is analyzed. This method has been designed and engineered within the last few years and emphasizes concrete as a barrier to limit radioactivity releases to the environment. All LLW received at the site is re-packaged into standardized hexagonal concrete containers, or canisters. Void spaces are filled with grout, creating a solid hexagonal container.

Four other disposal methods are analyzed but not included in the base case results. They are:

1. Hydrofracture;
2. Deep-Well Injection;
3. Deep Geological Disposal; and
4. Earth-Mounded Concrete Bunker.

The results for these disposal methods are listed in Chapter 11. The first three methods are deemed appropriate only for selected waste streams. The EMCB technique is used in France. More details on all of these disposal methods are presented in Chapter 4.

Table 9-4. Input data for EPA's LLW risk assessments: Disposal options and waste form

Disposal option	Acronym
Disposal	
Regulated Sanitary Landfill	SLF
Shallow Land Disposal (as practiced from 1963-1982)	SLD
Improved Shallow Land Disposal	ISD
Current Disposal Practice (Combination of SLD and ISD)	10 CFR 61
Intermediate Depth Disposal	IDD
Hydrofracture	HF
Deep-Well Injection	DWI
Deep Geologic Disposal	DGD
Concrete Canister	CC
Earth-Mounded Concrete Bunker	EMCB
Pretreatment	
Waste form option	Acronym
Packaged As Generated	AG
Solidified	S
Incinerated, Then Solidified	I/S
Packaged in a High Integrity Container	HIC

In addition to the various disposal methods, the form of the waste being disposed of may also vary for a given disposal method. Table 3-7 illustrates the basic waste forms analyzed and the 10 CFR 61 disposal technology waste class for each of the 26 waste streams. The base case analysis relates two categories of waste form to the base case disposal methods shown in Table 9-5. The "as-generated" waste form designation actually encompasses any one of three basic waste forms (trash, activated metal, and absorbing waste) for a given waste stream, reflecting minimal treatment and packaging. Use of the 10 CFR 61 disposal technology waste classes in Table 9-5 simplifies the description of matching waste streams to disposal methods as well. For example, according to Table 9-5, all Class A waste streams are in the "as-generated" waste form for a regulated SLF. Referring back to Table 3-7, the Class A waste streams are identified, along with the basic "as-generated" waste form for each Class A waste stream. Appendix B presents the NRC's waste classification system as set forth in 10 CFR 61.55. Chapter 11 (Sensitivity Analysis) presents the results for other combinations of waste forms and disposal methods.

9.2.4 Health Impact Assessment Codes

The original PRESTO-EPA model was developed jointly by EPA and Oak Ridge National Laboratory for use in the LLW standard development effort (EPA83). This model, which was completed in 1983, was expanded by EPA and Rogers and Associates Engineering Company into a family of health impact assessment codes in order to estimate such impacts from the disposal of regulated LLW under a wider range of conditions. The following derivatives of PRESTO-EPA have been developed:

PRESTO-EPA-POP — Cumulative health effects to local and regional populations from land disposal of LLW by shallow methods; long-term analyses are modeled (10,000 years).

PRESTO-EPA-CPG — Maximum annual committed effective whole-body dose equivalent to a critical population group (CPG) from land disposal of LLW by shallow or deep methods; dose in maximum year is estimated. Maximum annual CPG dose can be converted to an annual or lifetime risk using appropriate conversion factors.

Chapter 8 provides detailed descriptions of the above health impact assessment codes.

9.3 Summary of Base Case Analysis

As indicated above, the purpose of the base case analysis is to estimate and compare the health impacts from the undisturbed performance of engineered LLW disposal systems. The disposal systems examined include disposal methods requiring little or no further engineering

Table 9-5. Base case analyses of LLW disposal

10 CFR 61 waste class	Waste forms assigned to disposal methods						
	SLF	SLD	SLD	SLD (10 CFR 61)*	ISD	IDD	CC
Class A	AG	AG	AG	AG	AG	AG	S
Class B	AG	AG	S	S	S	S	S
Class C, NARM	AG	AG	S	S	S	S	S

Notes:

1. Abbreviations Used:

   SLF = Sanitary Landfill
   SLD = Shallow Land Disposal
   ISD = Improved Shallow Land Disposal
   IDD = Intermediate Depth Disposal
   CC = Concrete Canister Disposal
   AG = "As-Generated" Waste Form
   S = Solidified

2. See Table 3-7 for the relationship between "waste class" and specific waste streams.

*10 CFR 61 disposal technology incorporates practices from both SLD and ISD.

development, methods applicable for all types of LLW, and methods illustrative of a wide range of technological sophistication. They are SLF, SLD, ISD, 10 CFR 61, IDD, and CC. The health impact assessments presume a disposal site with a capacity of 2.5E+05 m^3 of LLW.

The base case analysis source term is derived from the overall LLW source term presented in Chapter 3. Modifications include the elimination of seven lower activity LLW streams from those presently regulated under AEA authority to reflect implementation of a BRC criterion, and the inclusion of two high concentration NARM waste streams to reflect inclusion of these NARM wastes as regulated wastes. Table 9-3 indicates the volume of each waste stream contributing to the 2.5E+05 m^3 disposal capacity of the model site.

Having defined the waste streams associated with the model site and the disposal methods designed to accept such wastes, it is also necessary to define the waste form associated with each disposal method. Table 9-5 relates the waste form (according to the 10 CFR 61 waste classification) to the disposal method. (Note that Tables 9-1 and 9-3 provide a listing of the LLW streams and their corresponding 10 CFR 61 waste class.) Selection of waste form was first made for the 10 CFR 61 disposal method using the waste form requirements therein. Disposal methods that are less sophisticated generally are assigned the simpler waste forms. Likewise, more sophisticated disposal methods are assigned waste forms comparable to, or more engineered than, those employed by 10 CFR 61 disposal technology requirements.

The base case health impact assessments are then carried out for the matrix of disposal method/waste form combinations shown in Table 9-5. Each disposal method/waste form combination is evaluated for all three settings. Thus, each disposal method/waste form/hydrogeologic setting combination is evaluated in terms of maximum annual CPG dose and cumulative population health effects using the appropriate health impact assessment computer code (i.e., PRESTO-EPA-CPG or PRESTO-EPA-POP). For example, Table 9-5 indicates that the base case source term, Table 9-3, is to be evaluated in the "as-generated" waste form for all classes of LLW (as defined by 10 CFR 61) disposed of in a regulated SLF. By varying the hydrogeologic input data, maximum annual CPG doses are calculated for a regulated SLF located in each of the three different hydrogeologic settings. Similarly, cumulative population health effects are also evaluated for an SLF located in each of the three hydrogeologic settings. Corresponding health impact assessments are then carried out for the remaining disposal method/waste form combinations in Table 9-5. The results of this base case analysis are described in the following sections.

9.4 Results of Base Case Health Impact Analyses

The purpose of the EPA assessment of LLW disposal is to compare potential risks from LLW disposal methods. In so doing, EPA has taken

great care to use input data as realistic as possible and health impact assessment computer codes that are "state-of-the-art." None of the predicted population health effects or CPG doses described below should be taken to be predicted impacts from any existing or future site. Site-specific predictions would require site-specific assessment code(s) and site-specific engineering, hydrogeologic/climatic, and waste stream data. The results presented below reflect the undisturbed performance of natural and engineered barriers.

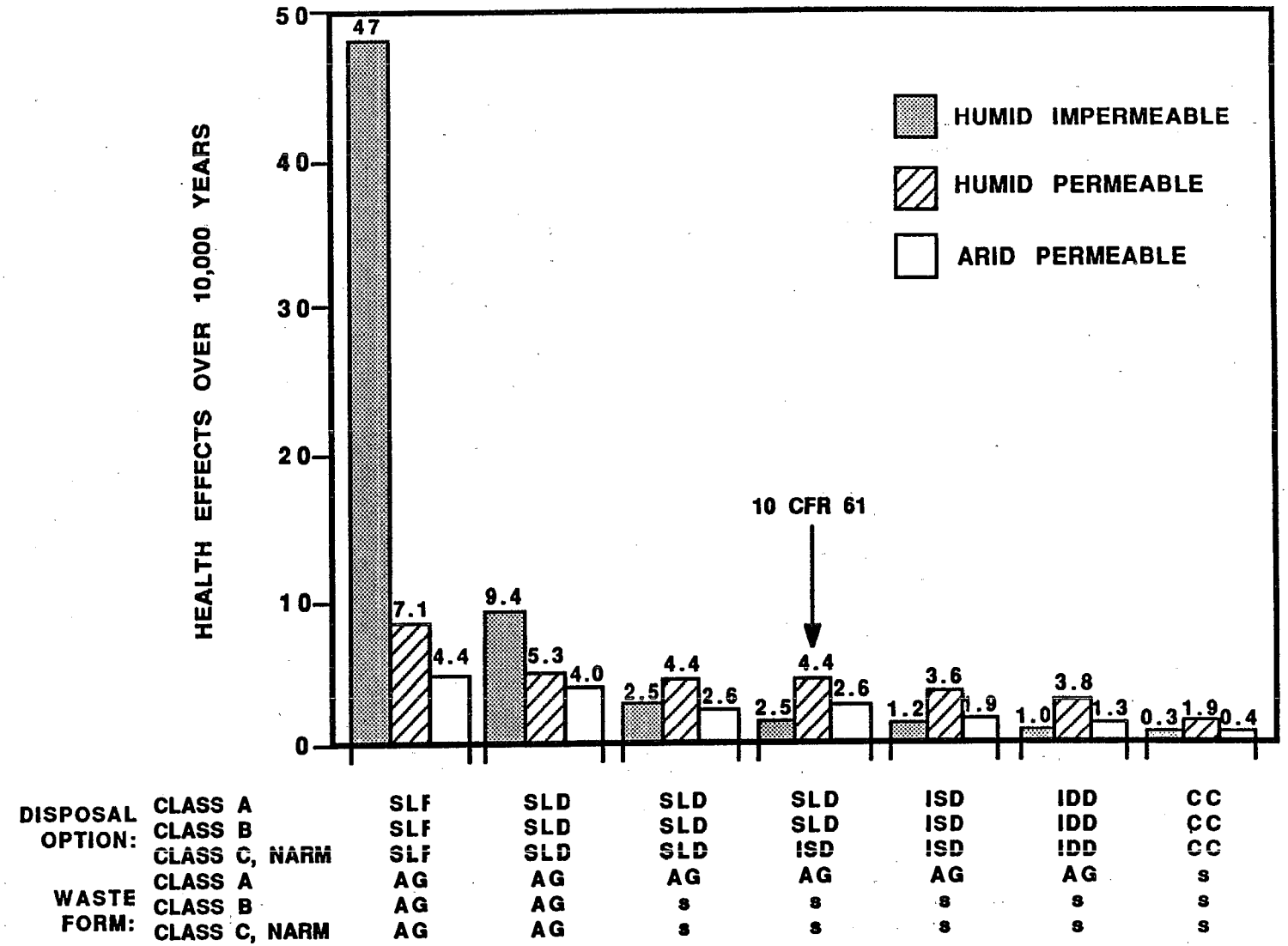
9.4.1 Health Effects to the General Population

Figure 9-1 summarizes the potential health effects incurred by a general population from disposing of 2.5E+05 m^3 of regulated LLW by six different disposal methods in three different hydrogeologic/climatic settings. These disposal methods include SLF, SLD, 10 CFR 61, ISD, IDD, and disposal using CC. As shown in Figure 9-1, these disposal methods incorporate more sophisticated waste forms as the sophistication of the disposal method increases. Two different waste form cases are considered for SLD. Current disposal practice under 10 CFR 61 is modeled as a combination of the SLD and ISD methods.

In the humid permeable setting, estimated total health effects (fatal cancers and genetic effects) ranged from 7.1 for SLF, the least stringent disposal method, to 1.9 for CC disposal, the most highly engineered method, with 4.4 health effects for 10 CFR 61 disposal. These health effects were incurred primarily by the regional basin population via the ground-water pathway, and usually occurred within the first 500 years.

In the humid impermeable setting, estimated total health effects ranged from 47 for SLF to 0.3 for CC disposal, with 2.5 health effects for 10 CFR 61 disposal. Again, the majority of the health effects were incurred by the regional basin population. However, the major release pathway shifted from the ground-water pathway to surface water by direct overflow onto the land surface because of the "bathtub" effect (Me76). In the direct overflow case, because of the impermeable disposal medium, both mobile and less mobile radionuclides were released more quickly than would have been expected by the ground-water pathway.

In an arid permeable setting, estimated total health effects ranged from 4.4 for SLF to 0.4 health effects for CC disposal, with 2.6 health effects for the 10 CFR 61 disposal technology. As in the case of the humid permeable settings, essentially all of the health effects were incurred by the regional basin population via the ground-water pathway. In this case, however, they occurred during the second thousand years rather than the first 500 years because of the much lower rainfall and therefore the much smaller flux of water entering the trench to leach the wastes. This, in turn, caused a much smaller flux of water to leave the trench. A much thicker unsaturated zone between the trench bottom and the aquifer also provides additional delay. Once the contaminant reached the aquifer, however, it moved at the same rate as the ground water, less any retardation from exchange with geologic media.



*REFER TO TABLE 9-3 FOR DEFINITION OF ACRONYMS.

Figure 9-1. Comparison of Population Health Effects over 10,000 Years by Disposal Options for a Reference Disposal Facility Containing 250,000 m^3 of Regulated LLW

Both disposal method and hydrogeologic setting influence the magnitude of predicted population health effects. Analysis of Figure 9-1 shows that for simpler disposal methods such as SLF, the humid impermeable setting results in the greatest number of health effects. This method provides for minimal containment of all radionuclides that are able to reach numerous surface water pathways in a relatively short time because of trench overflow, or the "bathtub effect." However, with enhanced waste form technology, such as solidification, radionuclide containment is improved. When both waste form and disposal method are improved, such as for 10 CFR 61 disposal technology or more advanced methods, radionuclide containment at the humid impermeable site is much improved. In such cases, the humid permeable setting results in a greater number of health effects.

It is also of interest to examine the distribution of cumulative population health effects over time for the whole United States. Nationwide impacts may be calculated by weighting the results of the three hydrogeologic settings by the percentage of LLW expected to be disposed of in each setting. Considering only the 10 CFR 61 waste technology disposal option, the following illustrates the proportion of nationwide health effects projected to occur over the indicated time periods:

Time period (yrs)	% of total health effects
0 - 100	5.7
101 - 500	37.5
501 - 1,000	9.8
1,001 - 10,000	47.0
Total: 0 - 10,000	100.

For less sophisticated disposal methods, total health effects are greater, and a much larger fraction of total health effects occurs in the first 500 years. For more advanced disposal methods and waste forms, the bulk of total health effects is slightly less and occurs later, in the 1,001- to 10,000-year time frame. Such advanced technology provides for greater containment, allowing decay of short-lived radionuclides and very slow release of the remaining long-lived radionuclides.

Table 9-6 shows the critical radionuclides and their relative contribution to total health effects for the 3 different hydrogeologic settings using 10 CFR 61 disposal technology and waste form requirements. Carbon-14 is the dominant radionuclide at all the settings. Only at the humid impermeable setting does any other radionuclide make a substantial contribution, namely, Am-241. The humid impermeable setting allows radionuclides to reach surface water pathways (due to the "bathtub effect"). Thus, radionuclides that would normally be retarded in their movement through the soil reach the surface via

Table 9-6. Critical radionuclides at a model LLW site

Hydrogeologic setting	Nuclide	Percent of total impact	
		CPG (exposure)	Population (health effects)*
Humid Permeable	I-129	93%	-
	C-14	7%	90%
	Other	-	10%
Humid Impermeable	I-129	78%	-
	C-14	22%	70%
	Am-241	-	20%
	Other	-	10%
Arid	C-14	100%	95%
	Other	-	5%

Note: Model site assumes 250,000 m^3 of regulated LLW using 10 CFR 61 disposal technology methods.

*Approximate values.

overflow and become more readily available to pathways affecting human populations.

9.4.2 Exposure of Critical Population Groups

Figure 9-2 summarizes the estimated maximum annual effective whole-body doses to a CPG living within a few tens of meters of a standard reference disposal facility containing 2.5E+05 m^3 of regulated LLW. Estimated exposures are calculated for the same 6 disposal methods and waste form combinations as for the preceding health effects assessments. Analyses are terminated at 1,000 years for all of the standard CPG assessments, but in some sensitivity runs, analyses are extended to 10,000 years to look for possible significant exposures beyond 1,000 years. This is discussed in more detail in Chapter 11.

In the humid permeable setting, estimated maximum CPG doses range from 62 mrem/yr for SLF to 1.3 mrem/yr for CC disposal, with 9.2 mrem/yr for 10 CFR 61 disposal. These maximum doses occur at 60 years for SLF, 780 years for 10 CFR 61 disposal, and at 1,000 years for CC disposal. As shown in Table 9-6, C-14 and I-129 are the principal radionuclides contributing to CPG dose for 10 CFR 61 disposal technology. Less sophisticated disposal methods allow short-lived radionuclides to be released sooner, along with long-lived radionuclides, resulting in larger maximum CPG doses at earlier times. For example, use of SLD method without solidification of any wastes results in a maximum CPG dose of about 35 mrem/yr at 30 years. In this case, the peak dose is caused primarily by H-3 and C-14. Over time, other radionuclides are released, but they result in annual CPG doses lower by about a factor of 4. These long-term CPG doses are attributed to the slow release of long-lived radionuclides, particularly I-129. Chapter 11 provides a graphical representation of CPG dose over the first 1,000 years for SLD in all 3 hydrogeologic settings. The benefits of solidification are apparent by comparing the two SLD cases depicted in Figure 9-2. On the other hand, disposal methods more sophisticated than the 10 CFR 61 disposal method modeled do not appreciably reduce maximum CPG dose, except for the CC disposal.

In the humid impermeable setting, estimated maximum CPG doses are all less than 1 mrem/yr, ranging from 0.8 to 0.001 mrem/yr for SLF and CC disposal, respectively, with 0.03 mrem/yr for 10 CFR 61 disposal. Exposures are due to releases to land surface and subsequently to surface waters, again due to the "bathtub" effect caused by the low permeability disposal medium. The year of maximum exposure occurs at 24 years for SLF, 190 years for 10 CFR 61 disposal technology, and 250 years for CC disposal. These 190- to 250-year periods for 10 CFR 61 and CC disposal include 100 years of institutional control, during which it is assumed that the trench covers are maintained intact, with the balance of the time for the caps to fail and the trenches to fill and overflow. No active maintenance on the cover is assumed for SLF. The critical radionuclides for the humid impermeable setting are C-14 and I-129. Though the peak CPG dose is short-lived and low (i.e., less than

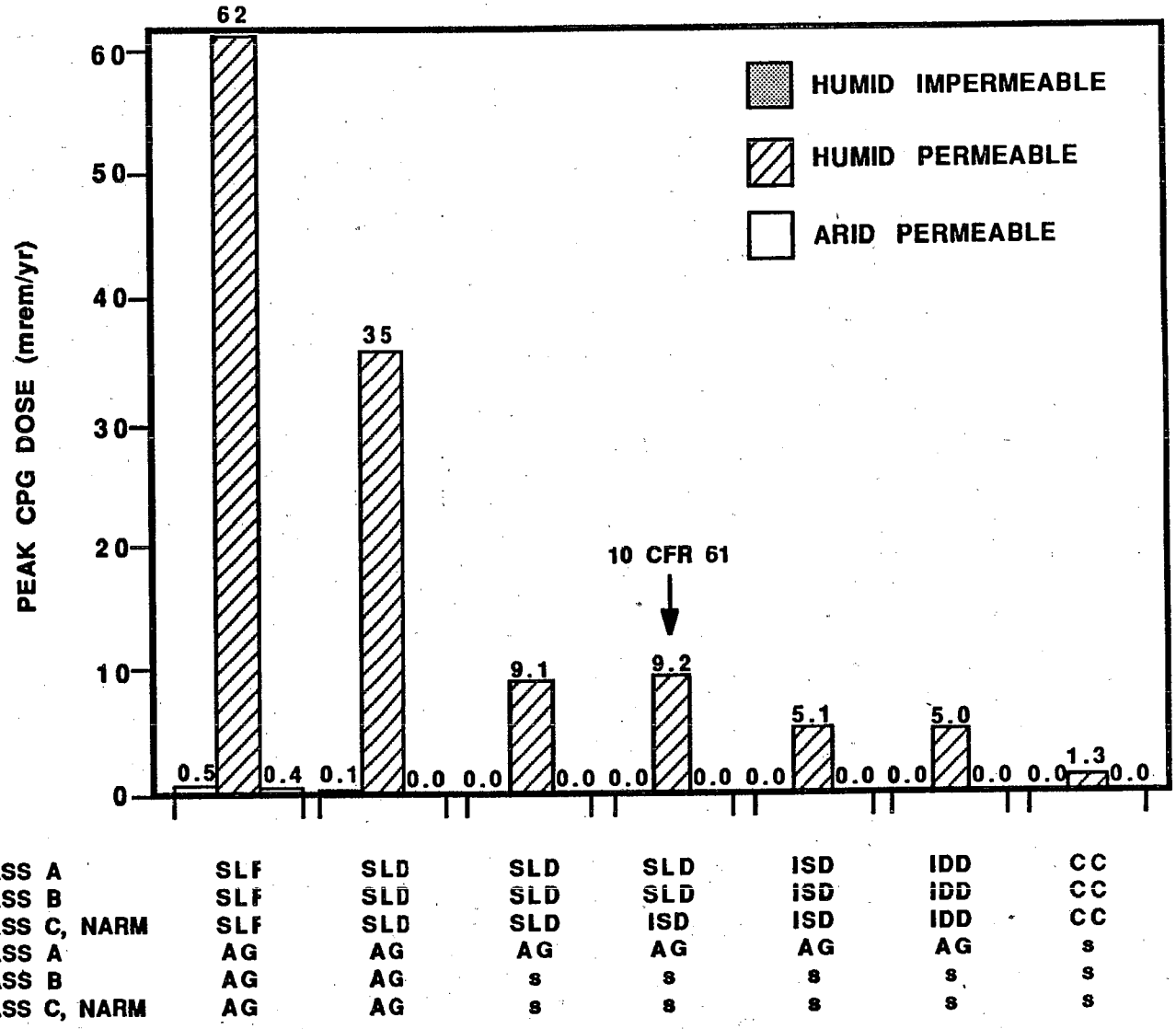


FIGURE 9-2. Comparison of Effective Whole-Body Dose to Critical Population Group by Disposal Options for a Reference Disposal Facility Containing 250,000 m^3 of Regulated LLW

0.1 mrem/yr), the expected chronic CPG dose over the remaining time period is expected to be relatively constant. For SLD, the chronic CPG dose is expected to remain at about 10 percent of the peak CPG dose for many hundreds of years.

The estimated maximum exposures were all less than 1 mrem/yr for the arid permeable setting. They ranged from 0.4 mrem/yr for SLF to almost zero for CC disposal (calculations terminated at 1,000 years) with 0.007 mrem/yr for 10 CFR 61 disposal technology. In the arid setting, the principal radionuclide contributing to CPG dose is C-14, as shown in Table 9-6. Peak doses occur much later in the arid settings. For example, for SLD without waste solidification, the peak dose is very low (less than 0.1 mrem/yr) and occurs at about 950 years. More sophisticated disposal methods result in maximum CPG doses via ground water occurring even later.

As indicated for population health effects estimates, these results should not be considered as applying to any specific facility, either present or future. They are only applicable to generic comparisons of methods. Also, the absolute values have considerable uncertainty associated with them even within the context of a generic analysis, and a factor of 2 variability in the scale would not be unreasonable. Chapter 12 discusses uncertainty in more detail.

It is interesting to observe the relationship between maximum CPG dose and estimated health effects for the disposal methods and waste forms analyzed for the base case (see Figures 9-1 and 9-2). For those methods where waste is not solidified (i.e., the as-generated waste form), the maximum CPG dose occurs in a <u>different</u> hydrogeologic setting than that corresponding to the maximum health effects. Typically, the maximum CPG dose occurs in the humid permeable setting, whereas maximum health effects occur in the humid impermeable setting. For these less sophisticated methods, greater amounts of short-lived radionuclides are released. For the humid permeable setting, the bulk of such a release is permitted to flow into the ground water and reach a well, creating a relatively high concentration in ground water (and well water), resulting in a large CPG dose. However, the population using such a well is relatively small compared to the overall population at risk, and thus total health effects also remain relatively small. In the humid impermeable setting, however, virtually no radionuclides migrate via the ground water to well pathway because of the low permeability soil. Instead, many more radionuclides are forced to the surface via trench overflow, dispersing among numerous surface water and airborne pathways. Dilution afforded by these numerous surface pathways ensures a lower CPG dose, but the short-lived and wider variety of radionuclides forced to the surface result in the exposure of much larger populations, thus producing greater health effects.

For disposal methods where some or all of the waste is solidified, the maximum CPG dose occurs in the <u>same</u> hydrogeologic setting as for maximum health effects (humid permeable). As the sophistication of the disposal method increases, however, the reduction in maximum CPG dose and health effects is generally modest.

Note that the results provided above for CPG exposures and general population health effects represent the base case analysis of the impacts from LLW disposal. Many additional analyses have been performed to investigate the effects of varying key input or assumptions. These special analyses are presented in Chapter 11 and examine other disposal methods, site size, special waste treatment options, the effects of regional compact waste volumes and characteristics, and many other factors that may have a bearing on the base case analysis presented here.

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Chapter 10: THE ESTIMATED HEALTH IMPACT ASSESSMENT OF DISPOSAL OF BRC WASTES

10.1 Introduction

The health impact assessment associated with the nonregulated disposal of those wastes considered "Below Regulatory Concern" (BRC) is a major factor in developing proposed disposal criteria for use by other regulatory agencies. The risk assessments carried out are intended to be generic in nature. To begin to determine the health impacts resulting from BRC waste disposal, we developed several scenarios (described in Chapter 4) for modeling based on surrogate waste streams (described in Chapter 3), disposal methods (described in Chapter 4), and various hydrogeologic/climatic settings (described in Chapter 5).

The candidate or surrogate waste streams were chosen to represent a postulated set of BRC waste types. These surrogate waste types originate from a variety of waste generators (power reactors, institutional, industrial facilities, etc.). For the sake of analysis, these surrogate wastes are declared BRC and therefore qualify for less restrictive disposal practices. In order to scope the range of cumulative impacts from many surrogate BRC waste types, numerous realistic scenarios or combinations of BRC waste types and disposal methods were constructed (see Chapter 4, Section 4.4). For each scenario, the radionuclide source term was defined, as well as the numerous parameters necessary to define the potential pathways of human exposure to radiation.

These data serve as the input to the PRESTO-EPA-BRC and PATHRAE-EPA methodologies developed specifically to calculate the health impacts consisting of cumulative population health effects and maximum CPG risk (see Chapter 8).

10.1.1 Wastes

To determine if it was feasible to allow some types of LLW to be BRC wastes, a number of waste streams were identified that had very low radioactivity, were reasonably well characterized, and had potentially large volumes to provide a cost savings. We chose 18 waste streams (see Table 3-10) generated at nuclear power reactors, uranium fuel fabrication and uranium process facilities, and industrial, medical, and educational facilities, as well as by consumers (see Chapter 3). To make our work comparable with that of others, EPA's BRC waste sources and volumes are based on waste characterizations done by NRC, the AIF, and others (NRC81b, NRC86, NRC84, Oz84, AIF78). Two of the surrogate consumer waste streams, smoke detectors and timepieces, were chosen because they presently are not regulated, and they help to provide a comparison and perspective for our analysis (NRC80). Another special deregulated waste stream (BIOMED), modeled after the upper limits of the NRC biomedical rule (NRC81a), was also chosen for specific incineration scenarios for further comparison [Chapter 3, Section 3.3.3(A)(3)].

10.1.2 Disposal Methods

Once a group of surrogate BRC waste streams was selected, it was necessary to determine reasonable disposal methods for such wastes. Consideration of the numerous generators represented by the surrogate BRC waste types indicated the very real possibility that a given waste disposal site might receive BRC wastes from more than one generator. To account for this, several realistic scenarios were constructed involving the disposal of various BRC waste streams from different generators. The choice of both surrogate BRC waste streams and disposal scenarios was made for the purposes of the assessment. EPA is not implying that these are the only streams or the only disposal scenarios available, but rather that they are the most likely, considering the types of generators and their locations (see Chapter 4, Section 4.4). Each scenario combined a generic BRC waste disposal method with selected groups of surrogate BRC waste types. Generic BRC waste disposal methods included a variety of options, i.e., sanitary landfills, municipal dumps, onsite landfills, and incineration methods situated in rural, suburban, and urban demographic settings. Chapter 4 discussed the selection of these methods and their varying parameters. Table 10-1 shows the major characteristics of the disposal methods and associated demographic settings.

10.1.3 Hydrogeologic/Climatic Settings

In developing the scenarios to model the disposal methods, three hydrogeologic/climatic settings were used that we believe cover the expected range of values for parameters affecting radionuclide retention and site performance anywhere in the United States. The settings include: (a) an arid zone site with permeable disposal medium (water infiltrating through the waste trench into the ground and radionuclides moving very slowly to ground water); (b) a humid zone site with permeable disposal medium (water infiltrating through the waste trench into the ground and radionuclides moving more rapidly to ground water); and (c) a humid zone site with impermeable disposal medium (water infiltrating into the waste trench and radionuclides potentially overflowing to surface waters rather than moving to the ground water).

10.2 Selection of Health Impact Assessments

Health impacts were estimated for (a) the general population and (b) the critical population group (CPG) (including both onsite and offsite workers). This impact consists of cumulative population health effects and a maximum CPG whole body effective annual dose equivalent. The assessment of impacts from the disposal of BRC wastes involves the simulation of the transport of radionuclides through geological, atmospheric, and ecological systems, and the evaluation of human organ doses and fatal cancer risks after ingestion and inhalation of radionuclides.

10.3 Cumulative Population Health Effects Assessment

Cumulative population health effects were estimated as both fatal cancers and serious genetic effects for both the local and regional basin general populations over 100, 500, 1,000, and 10,000 years.

Table 10-1. Major characteristics of BRC waste disposal methods

1. MD - Municipal Dump (pop. served = 60,000)
   capacity = 2.1 million m^3
   size = 35 ha

2. SF - Suburban Sanitary Landfill (pop. served = 175,000)
   capacity = 6.0 million m^3
   size = 100 ha

3. UF - Urban Sanitary Landfill (pop. served = 1,000,000)
   capacity = 34.7 million m^3
   size = 576 ha

4. LURO - Large University/Medical Center with Onsite Landfill and Onsite Incineration (pop. served = 175,000)
   capacity = 0.17 million m^3
   incinerator at disposal site
   size = 2.8 ha

5. SI - Suburban Sanitary Landfill with Onsite Incineration (pop. served = 175,000)
   capacity = 1.0 million m^3
   incinerator at disposal site [aggregate VRF = 6.0]
   size = 16 ha

6. UI - Urban Sanitary Landfill with Onsite Incineration (pop. served = 1,000,000)
   capacity = 5.78 million m^3
   incinerator at disposal site [aggregate VRF = 6.01]
   size = 96 ha

Fifteen scenarios were defined and used to assess the consequences to the general population of deregulated or less restrictive disposal (i.e., BRC) of some radioactive wastes. The scenarios consist of combinations of radioactive wastes, disposal methods, and diverse demographic, climatic, and hydrogeologic settings. The health effects to the general population resulting from disposal of radioactive waste streams without regard to radioactivity have been estimated using the PRESTO-EPA-BRC computer model (Ro84, EPA87a), as described in Chapter 8.

The cumulative population doses and resulting health effects are separated into projections for a local population during the first 1,000 years of analysis, and projections for a regional basin population during the entire 10,000-year analytical period. After the first 1,000 years, the local population is assumed to become part of the larger regional basin population. The magnitude of local versus regional basin health effects is highly variable and either may predominate, depending on the waste streams and radionuclides present, local and regional water uses, and site-specific hydrogeology, climate, and demographics. (See Chapter 8, Section 8.3.5 for more detail on local and regional analysis.)

Local health effects occur through various pathways, including ingestion, inhalation, immersion, and surface (gamma radiation) exposure. In most cases, ingestion of radionuclides by their presence in contaminated ground and surface waters, either directly or through ingestion of contaminated food, appears to be the predominant pathway and accounts for greater than 90 percent of the total projected health effects. Ground surface exposures can predominate in certain situations. For example, in the case of the arid southwest climate with permeable soil, it is hypothesized that lack of rainfall infiltrating the trenches reduces pollutant transport rates and the importance of the ground- and surface-water pathways, and enhances the air (inhalation) and ground surface exposure pathways. The air immersion pathway is responsible for less than one percent of predicted health effects in all cases.

Health effects include the effect of onsite exposures of workers and visitors to the surrogate BRC waste streams through surface (gamma) exposure and inhalation. These effects are only a small fraction of total predicted effects and are probably due to the relatively brief operational period of the disposal site (20 years), compared to the lengthy period used in the post-closure analysis (10,000 years).

10.4 Maximum Annual Dose Estimates from BRC Wastes to a Critical Population Group (CPG)

Eleven of the 15 localized disposal scenarios described in Sections 4.4.1 through 4.4.11 have been used to assess the consequences of less restrictive disposal (i.e., BRC) of some radioactive wastes in terms of a maximum annual dose to a Critical Population Group. The four reference disposal scenarios (Chapter 4, Sections 4.4.12 through 4.4.15) are not relevant to this analysis for regulatory considerations and were only used for comparison purposes. The maximum whole-body dose equivalent rates to a CPG, located at or near the BRC disposal sites, are estimated using the PATHRAE-EPA computer modeling program (EPA87b, Ro84).

The individual exposures were calculated as maximum annual radiation dose and year of occurrence over 10,000 years for the CPG. For those offsite individuals living close to the disposal site, the major pathway is via water from a well or stream a few tens of meters from the site boundary. Other individual exposures were also estimated for incinerator disposal operations, garbage collectors who might collect the wastes, onsite workers during routine disposal operations, reclaimers, and offsite personnel from other exposure pathways besides water. In the case of BRC waste disposal, the onsite worker is considered because exposure of these personnel cannot be construed as occupational (i.e., radiation workers). For the PATHRAE-EPA model, it was assumed that the only major significant human exposure pathways available are those listed in Table 10-2. In its review (SAB85), the SAB commented that "in general believes, the [BRC] scenarios discussed...to be sufficient."

A special CPG pathway analysis was made to evaluate direct radiation exposures to transportation workers who would collect and transport BRC wastes from generators to the various less restrictive disposal facilities. The primary exposure pathway to the transportation workers will be gamma exposures (PEI85, Ro86). The methodology used was based upon NRC's de minimis methodology (Oz84). Figure 10-1 shows the exposure pathways evaluated for our analyses.

10.5 Health Effects Results to the General Population

Two different assessments were made concerning health effects to the general population. The first was based on the localized scenarios (described in Chapter 4, Section 4.4) where EPA felt these to be realistic cases involving the disposal of various surrogate BRC waste streams. The localized scenarios also provide results concerning the disposal of multiple BRC waste streams at one site.

The second assessment was done to examine a total BRC waste disposal impact on a national basis by individual waste stream; i.e., the 20-year total of each waste stream for both commercial and DOE facilities was disposed of at one site and the resulting national health effects calculated and summed (see also EIA, Chapter 7).

10.5.1 Population Health Effects by Scenario

The general population health effects analyses for the BRC wastes using the 11 localized scenarios (the 4 reference scenarios were not included for regulatory considerations) were based on the 16 surrogate waste streams from nuclear fuel-cycle, industrial, and institutional sources. Two of the scenarios do include the consumer wastes with the other BRC wastes to ascertain if the multiple waste streams plus consumer wastes have any effect.

The BRC waste disposal scenarios are described in Chapter 4, Section 4.4. The six BRC disposal methods are described in Section 4.3, while the disposal site hydrogeologic/climatic settings are discussed in Chapter 5.

Table 10-2. PATHRAE-EPA CPG pathways considered by which exposure may reach humans from the less restrictive disposal of BRC wastes

1.	Ground-water migration with discharge to a river.
2.	Ground-water migration with discharge to a well.
3.	Surface erosion of the cover material.
4.	Spillage of the waste.
5.	Saturation of the waste and surface-water contamination by the bathtub effect.
6.	Food grown on land.
7.	Biointrusion by plant roots.
8.	Direct gamma exposure.
9.	Atmospheric inhalation of radioactive airborne contaminants from dust resuspension, incinerator, or trench fire.
10.	Inhalation of radioactive dust stirred up by workers.

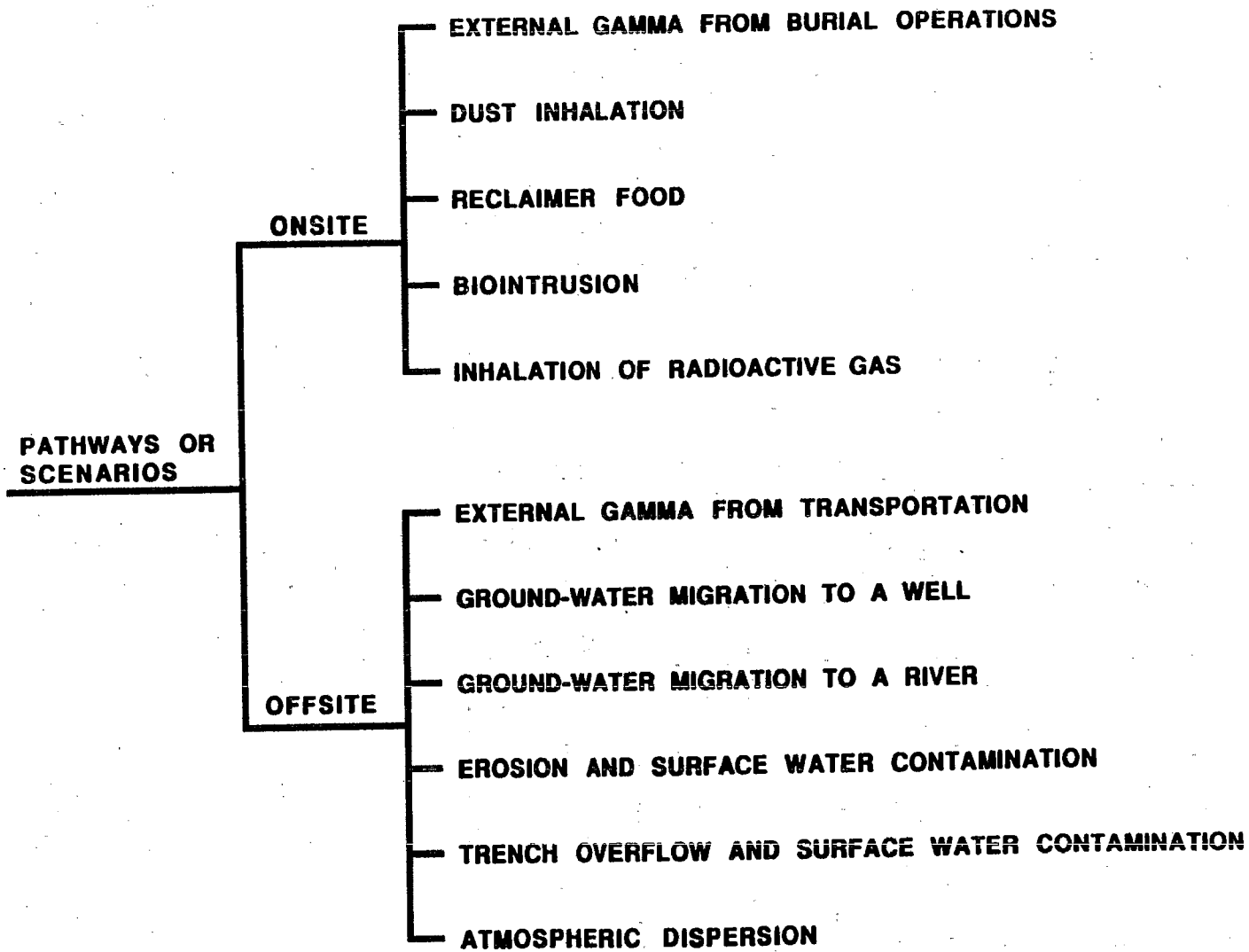

Figure 10-1. Pathways Included in the EPA Analysis

The combination of surrogate waste streams and the disposal site to which they would be shipped is based in part on regional considerations and actual situations currently known to exist. The volumes of the surrogate waste streams are based on probable routine quantities generated over 20 years. The various scenario waste volumes do not represent the quantity of that waste stream generated on a national basis.

The two surrogate consumer waste streams as described in Chapter 3 (smoke detectors and timepieces) were chosen because they presently are not regulated and provide a reference comparison and perspective on our analysis. Another deregulated waste stream, modeled after the upper limits of the NRC biomedical rule (NRC81a), was also selected for a comparison and perspective on our risk analysis. These are incineration scenarios where, in one case, 100 percent of the waste is incinerated and in the other case 50 percent of the waste is incinerated.

The estimated excess health effects (total cancers plus serious genetic effects) from the surrogate BRC waste disposal over 10,000 years are listed in Table 10-3 for the 11 BRC scenarios and 4 reference scenarios.

10.5.2 Population Health Effects on a Total Nationwide Basis

A population health effects assessment was performed on each of the surrogate BRC waste streams for the 20-year total U.S. inventory of surrogate BRC waste (the BRC wastes are all assumed to be Class A wastes as defined by the NRC). A comparison was made between BRC disposal and the same waste being disposed of in a regulated LLW disposal facility (referred to as SLD -- see Chapter 4). Table 10-4 lists the excess population health effects over 10,000 years from the nationwide disposal of the surrogate commercial and DOE BRC waste streams for a 20-year accumulation of waste.

The methodology behind the compilation of the estimated nationwide population health effects for the BRC waste streams is presented in detail in the EIS Volume 2 -- Economic Impact Assessment (Chapters 3 and 7). Briefly, the incremental or excess health effects are calculated from the difference between the health effects from the current regulated disposal practice (SLD) and the health effects for each unregulated BRC waste stream. The health effects are determined for each of the disposal options across the three hydrogeologic/climatic regions, and the total health effects are added together for all three regions. To take into consideration the five unregulated disposal options, a weighted average is used.

10.6 Results of the Maximum CPG Dose Assessments

The BRC waste disposal scenarios discussed in Section 10.5 (and described in Chapter 4, Section 4.4) were also used for the CPG exposures. The results of the CPG exposure assessments are in terms of millirem per year. The life span currently used is 70.76 years and the annual risk from a 1-mrem low-LET exposure (gamma and beta) is 3.95E-07. Therefore, the CPG can be converted from millirem per year to lifetime risk by using the factor 2.8E-05.

Table 10-3. Excess population health effects over 10,000 years from BRC waste disposal for various scenarios, disposal sites, and hydrogeologic/climatic settings

	Disposal scenario	Hydrogeologic/climatic setting		
	Description	Humid permeable	Humid impermeable	Arid permeable
1.	3-Unit pressurized water power reactor complex - municipal dump	9E-02	6.6E-02	1.7E-02
2.	2-Unit boiling water power reactor complex - municipal dump	1.3E-01	1.5E-01	2.8E-02
3.	University and medical center complex - urban sanitary landfill	3E+01	2.9E-01	2.1E+01
4.	Metro area with fuel-cycle facility - suburban sanitary landfill	1E+01	2.4E-01	5.9E-02
5.	Metro area with fuel-cycle facility - suburban sanitary landfill with incineration	2.6E+00	7.3E-02	2.6E-02
6.	2-Unit power reactor, institutional, and industrial - municipal dump	1E+00	9E-02	1.6E-02
7.	Uranium hexafluoride facility - municipal dump	2.3E-04	2.2E-03	5.6E-04
8.	Uranium foundry - municipal dump	7.1E-05	6.8E-04	1.6E-04
9.	Large university/medical center; volatilization of 90% H-3 and 75% C-14; onsite landfill with onsite incineration	1.5E+00	4.5E-02	2.3E-02
10.	Large metropolitan area with consumer wastes - suburban sanitary landfill with incineration	2.1E+00	8.8E-02	3.7E-02
11.	Large metropolitan area with consumer wastes - urban sanitary landfill with incineration	1.1E+01	2.1E-01	7.5E+00
*12.	Consumer product wastes - suburban sanitary landfill	1E-02	2.5E-03	1.9E-05
*13.	Consumer product wastes - urban sanitary landfill	1.4E-01	3.4E-02	2.9E-04
*14.	Large university/medical center; 100% volatilization of H-3 and C-14; onsite landfill with onsite incineration	2.8E-01	7.5E-03	4.8E-03
*15.	Large university/medical center; 50% volatilization of H-3 and C-14; onsite landfill with onsite incineration	2.5E+01	6.2E-01	3.7E-01

*Indicates those reference scenarios where the waste streams are already deregulated.
NOTE: Analysis is based on 20 years of waste accumulation.

Table 10-4. Excess population health effects over 10,000 years from unregulated disposal of commercial plus DOE BRC waste streams versus regulated LLW disposal on a nationwide total

Surrogate BRC waste stream	Excess population health effects
P - COTRASH	3.8
P - CONDRSN	0.0009
L - WASTOIL	0.0001
B - COTRASH	2.6
I - COTRASH	371.4
I - BIOWAST	18.7
I - ABSLIQD	22.4
I - LQSCNVL	0.93
N - SSTRASH	0.0011
N - SSWASTE	0.0037
N - LOTRASH	32.8
N - LOWASTE	10.7
F - PROCESS	0.0035
U - PROCESS	0.0011
F - COTRASH	0.0006
F - NCTRASH	0.0001
	Total 463.3411
C - SMOKDET*	1.1
C - TIMEPCS*	7.3

*Waste streams already deregulated.

NOTE: Analysis is based on a 20-year accumulation of waste. The reason there are so many significant numbers is that each waste stream is considered on a separate basis.

Appendix F provides tables for all 15 localized scenarios listing the maximum CPG exposure, the radionuclide providing the major exposure, and the year in which the maximum CPG exposure occurs for each of the 10 major pathways (Table 10-2) at each of the 3 hydrogeologic/climatic settings. The transportation pathway is not included in these analyses.

Table 10-5 lists each of the BRC disposal scenarios, indicating the predominant pathway, the associated maximum CPG exposure, and the disposal setting. Table 10-6 lists the four scenarios analyzed for reference and comparison purposes containing the consumer and the BIOMED wastes, showing the pathway through which the maximum CPG exposure is delivered and the disposal setting. Neither of these analyses include the transportation pathway.

10.6.1 Results of the Transportation CPG Dose Assessment

Based upon the NRC methodology (Oz84) and values for parameters determined by EPA (Ro86, PEI85), external gamma doses to transportation workers handling BRC waste streams were estimated. The primary exposure pathway to the transportation workers is gamma exposure. Additional short-lived nuclides (half-lives ranging from 2 days to 1 year) were included to ensure consideration of any important nuclides exhibiting gamma exposure during transportation (Table 3-12 lists these additional nuclides). The estimated average annual radiation exposures for the transportation worker exposed to BRC wastes and assuming a 30-day storage time prior to transporting the wastes are presented in Table 10-7.

The transport analyses are centered around the BRC disposal scenarios, except that the transport scenarios assume a single transport worker hauls all the waste volume (based on volumes in Chapter 4, Section 4.4) associated with a group of waste streams from the generator's site to the landfill or dump. As shown in Table 10-7, all the surrogate BRC waste streams are covered by the analyses, although not all BRC disposal scenarios are included. It was assumed that these transport scenarios would represent the maximum for each group of wastes (i.e., reactor, institutional, foundry, etc.) where a combination of waste groups occur, the data may be extrapolated to obtain any other transport or disposal scenario desired.

10.7 Discussion of the Health Impacts from BRC Waste Disposal

This section will examine and discuss the health impacts from the disposal of BRC waste. These impacts consist of cumulative population health effects and maximum CPG doses.

10.7.1 Cumulative Population Health Effects

The predicted population health effects for the localized scenarios ranged from extremely small fractions, 0.00007, to about 30 excess health effects over 10,000 years for 20 years of waste accumulation, where excess health effects is defined as both fatal cancers and serious genetic effects. The genetic effects range from 5 to 24 percent of the total health effects.

Table 10-5. The maximum CPG annual doses of BRC waste disposal by scenario, setting, and pathway for 20 years of accumulated waste

Scenario (disposal setting)	Humid Impermeable		Humid Permeable		Arid Permeable	
	Dose (mrem)	Pathway	Dose (mrem)	Pathway	Dose (mrem)	Pathway
1 PWR-MD	12	Gamma	12	Gamma	12	Gamma
	1.1	Bioint.				
2 BWR-MD	11	Gamma	11	Gamma	11	Gamma
	2.7	Bioint.	1.6	Bioint.	2.2	Bioint.
3 LUMC-UF	0.18	Gamma	0.18	Gamma	0.18	Gamma
4 MAFC-SF	0.89	Gamma	1.1	Well	0.89	Gamma
5 MAFC-SI	5.4	Gamma	5.4	Gamma	5.4	Gamma
			1.5	Well		
6 PWRHU-MD	8.8	Gamma	8.8	Gamma	8.8	Gamma
7 UHX-MD	0.13	Dust	0.13	Dust	0.13	Dust
8 UF-MD	0.039	Dust	0.039	Dust	0.039	Dust
9 LURO3-ON	0.54	Well	2.4	Well	0.16	Gamma
10 LMACW-SI	21	Gamma	21	Gamma	21	Gamma
			1.1	Well		
11 LMACW-UI	4.4	Gamma	4.4	Gamma	4.4	Gamma

Notes: The transportation pathway was not considered in this analysis.

Key for Disposal Settings (see also Chapter 4, Section 4.4)

MD = Municipal Dump
SF = Suburban Sanitary Landfill
UF = Urban Sanitary Landfill
SI = Suburban Sanitary Landfill with Incineration
UI = Urban Sanitary Landfill with Incineration
ON = Onsite Disposal with Incineration

Key for Pathways (see also Table 10-2)

Bioint. = Biointrusion by plant roots to onsite resident
Dust = Onsite worker dust inhalation
Gamma = Direct gamma radiation to onsite worker
Well = Exposure to offsite resident from drinking contaminated well water

Table 10-6. The maximum CPG annual doses from already deregulated waste streams for 20 years of accumulated waste for 4 specific reference scenarios

Scenario (disposal setting)	Humid Impermeable		Humid Permeable		Arid Permeable	
	Dose (mrem)	Pathway	Dose (mrem)	Pathway	Dose (mrem)	Pathway
12 CW-SF	0.0018	Dust	0.043	Well	0.0018	Dust
13 CW-UF	0.0017	Dust	0.017	Well	0.0017	Dust
14 LURO1-ON*	0.00031	Atmos.	0.00076	Atmos.	0.00032	Atmos.
15 LURO2-ON**	9.1	Well	40	Well	0.0022	Well

* 100% incineration of wastes.

** 50% incineration of wastes.

Notes: The transportation pathway was not considered in these scenarios.

Key for Disposal Settings

See Notes in Table 10-5 and Chapter 4, Section 4.4.

Key for Pathways

See Notes in Table 10-5 and Table 10-2.

Atmos. = Exposure to offsite residents from atmospheric inhalation of radioactive airborne contaminants.

Table 10-7. Transportation worker exposures to BRC wastes with a 30-day storage time (Ro86)

Scenarios* and waste streams	Dose (mrem/yr)	Major nuclide(s)	Half-life
(1)** PWR (P-COTRASH, P-CONDRSN, L-WASTOIL)	160	Cs-134	2.05 yr
	87	Co-60	5.26 yr
	10	Co-58	71.3 da
	3.5	Cs-137	30 yr
	1.7	I-131	8.05 da
	1.5	Cs-136	13.7 da
(2)** BWR (B-COTRASH, L-WASTOIL)	410	Cs-134	2.05 yr
	69	Co-60	5.26 yr
	9.1	Cs-137	30 yr
	8.0	Co-58	71.3 da
(3)** Institutional (I-COTRASH, I-ABSLIQD, I-BIOWAST, I-LIQSCVL)	11	Co-60	5.26 yr
	0.93	Cs-137	30 yr
(7)** Uranium Hexafluoride (U-PROCESS)	0.0014	U-235	7.1E+08 yr
(8)** Foundry (N-SSTRASH, N-SSWASTE)	0.00025	U-235	7.1E+08 yr
Fuel Cycle Wastes (F-PROCESS, F-COTRASH, F-NCTRASH)	0.0026	U-235	7.1E+08 yr
Industrial Wastes (N-LOTRASH, N-LOWASTE)	1.8	Co-60	5.26 yr
(12)** Consumer Wastes*** (C-TIMEPCS, C-SMOKDET)	0.0001	Am-241	458 yr

* Scenarios do not necessarily reflect the same BRC waste disposal scenarios as listed in Chapter 4, Section 4.4.

** These scenarios are the same as the BRC waste scenarios listed in Chapter 4, Section 4.4.

***Indicates a reference scenario where the waste streams are already deregulated.

(A) Health Effects Versus Demographic Setting

As expected from a cumulative population health effects analysis, the demographics will play an important part in how the health effects are distributed, i.e., rural populations will generally incur the least number of health effects, suburban populations the next largest number, and the urban population the greatest number of health effects. As shown in Table 10-1, the populations served for each of the disposal method settings are:

rural	60,000
suburban and onsite	175,000
urban	1,000,000

(B) Health Effects Versus Disposal Method Using Incineration

The analysis showed that where the waste was disposed of in conjunction with incineration, the number of health effects was usually reduced by a factor of two.

The reason behind the reduction in health effects is that the incineration transforms the majority of the radionuclides from the water ingestion pathway to the air inhalation pathway. In the air pathway, the radionuclides are diluted considerably and the body response from inhalation is generally much less than when a material is ingested.

(C) Health Effects Versus Hydrogeologic/Climatic Setting

The health effects results for the three hydrogeologic/climatic settings -- humid permeable, humid impermeable, and arid permeable -- showed several trends.

- In the arid and humid permeable scenarios, health effects to the local populations dominated in the first 1,000 years. In these regions the limited amount of ground-water dilution is the major factor, along with the larger defined populations. In the humid impermeable scenarios, regional basin populations dominated in the first 1,000 years. In this region the larger dilution becomes a factor, along with the much larger regional basin population.

- The number of estimated excess health effects was greatest in the humid permeable region, next greatest in the arid permeable region, and least in the humid impermeable region. These comparisons are in relation to one another and, as such, in the humid impermeable region, the greater ground-water dilution becomes a factor in the amount of health effects observed. In the arid region, there is limited ground-water dilution; thus, more health effects are observed.

- The majority (greater than 95 percent) of the health effects were incurred in the first 1,000 years in all the hydrogeologic/climatic settings. This is because it is assumed that water infiltration through the trench cap will be greater; thus, increasing the movement of radionuclides through the ground will be greater for landfills than for regulated LLW sites.

(D) Health Effects Versus Waste Stream

On a national basis the surrogate BRC waste streams causing the most health effects for the cumulative population analysis are as follows (in descending order): I-COTRASH, N-LOTRASH, I-BIOWAST, and I-ABSLIQD. This can also be seen in Table 10-4.

(E) Health Effects Versus Radionuclide

The dominant radionuclide causing the most population health effects in the four surrogate BRC waste streams mentioned above is carbon-14.

10.7.2 Critical Population Group (CPG) Exposures

The individual exposures were calculated for 10 pathways (shown in Table 10-2 and Figure 10-1) as a maximum annual radiation dose (effective whole-body dose equivalent) and year of occurrence over 10,000 years for the CPG. For the overall time span, the maximum individual in any given year may be one of three persons: onsite worker, onsite resident, or offsite resident.

In the case of BRC waste disposal, the onsite worker is employed at a BRC waste disposal facility and is not regulated for radiological protection. The onsite visitor is also considered in this context (see Chapter 4, Section 4.3). Both the worker and visitor are considered members of the general public. (The offsite transportation worker is discussed in Section 10.8.) The onsite resident is any member of the general public building a house and living on the BRC waste disposal site after closure and growing crops for human consumption. The offsite resident is any member of the general public who lives away from the BRC waste disposal site, but is subjected to the various pathways capable of exposing radionuclides to the human population (see Chapter 8, Section 8.5.4).

There are three time periods involved in the CPG analysis. In all cases we are assuming that the disposal site has a full 20-year inventory of BRC waste, with radioactive decay taken into consideration. The first time period is 0 year or the last year before closure or (pre-closure). In the 0 year, the maximum individual is either the onsite worker involved with direct gamma and dust inhalation or an offsite resident exposed via the atmospheric inhalation or spillage pathway.

Second, the year 1 is considered to be the first year of the post-closure phase with the maximum exposure from the full site inventory. In the year 1, the onsite resident is the individual most likely to be exposed via the food pathways. Finally, there are the variable years, i.e., greater than the first year in which the offsite resident is most likely to be exposed via the major water pathways.

Three of the 10 pathways examined for CPG exposures -- the ground water to the river, the spillage of waste on the surface and subsequent discharge to surface waters, and the saturation of waste in the trench with overflow to surface waters (bathtub effect) -- are applicable only to the humid impermeable hydrogeologic/climatic setting and not to the other two settings. This is because this setting deals only with surface water flow, while the other two settings deal with ground-water migration to water sources.

In the erosion pathway, the arid permeable setting has no results, mainly because it is estimated that erosion will not uncover the waste for over 13,000 years. This is due to the minimal rainfall for the arid setting.

For the urban demographic disposal settings, it was assumed that there would be no food grown onsite after site closure.

Appendix F shows the detailed exposures per scenario per pathway, the critical radionuclide, and the year of maximum CPG dose.

A general overview of the CPG doses indicated the following:

- the maximum annual dose was less than 4 mrem (1.6E-06 annual risk or a lifetime risk of 1.1E-04) in roughly one-half of the principal localized scenarios with presently regulated wastes (see Table 10-5);
- the dominant radionuclides were cobalt-60 through direct exposure of workers, cesium-137 through biointrusion, and carbon-14 through well water usage;
- the maximum annual dose occurs within the first year in most scenarios for non-ground-water pathways; and
- in all regions, the dominant pathways providing the maximum annual doses that exceed 4 mrem were external gamma radiation, biointrusion, and ground water.

(A) Exposure Versus Hydrogeologic/Climatic Setting

The maximum annual doses for a given set of waste streams generally show little variability between disposal method, demographic, or hydrogeologic/climatic settings. The only exceptions are the ground water-to-well and erosion pathways for the hydrogeologic/climatic settings. In the ground water-to-well pathway, the maximum annual doses were greatest in the humid permeable, next greatest in the humid impermeable, and least in the arid permeable.

(B) Exposure Versus Specific Humid Impermeable Site Pathways

Three CPG pathways apply only to the humid impermeable hydrogeologic/climatic setting and affect only offsite residents. They are the ground water-to-river, spillage, and the bathtub effect (saturation of the waste and surface water contamination by trench overflow) pathways.

In all cases, the spillage pathway occurs in the last year of site operation or year 0; the bathtub effect occurs in the year 100 after site closure, and the ground-water migration with discharge to a river occurs beyond 2000 years in most scenarios. In some cases, it occurs beyond tens of thousands to hundreds of thousands of years. Table 10-8 presents the CPG results for each of the 11 localized scenarios for the three humid impermeable pathways only. (See Table 10-5 for acronyms of disposal scenarios.)

(C) Exposure to the Onsite Worker or Visitor

For the onsite worker there are two pathways: direct gamma exposure and inhalation of radioactive dust. These onsite worker pathways are independent of the hydrogeologic/climatic setting. Where cobalt-60 is present in the waste, it becomes the dominant radionuclide for that pathway and the maximum exposure takes place during operations (in the year 0). In two scenarios, UHX-MD and UF-MD, there is only uranium in the wastes, and although there is direct gamma exposure to the onsite worker (less than 1.0E-08 mrem/yr), the maximum gamma exposure is not to a worker and does not take place until erosion removes the trench cover, in the humid areas beyond 3,000 years and beyond 10,000 years in the arid areas. Table 10-9 shows these data.

The high doses from the direct gamma exposure is the result of Co-60 in the waste streams. The dust inhalation always occurs to the onsite worker during operations (in the year 0).

(D) Exposure to Offsite Residents

For the offsite residents there are six pathways. Three of the pathways only occur in the humid impermeable setting and were discussed previously in Section 10.7.2 (B). The remaining three pathways are: (1) ground-water migration to a well; (2) surface erosion and deposition to a nearby water source; and (3) inhalation from atmospheric contamination from dust resuspension, incineration, or trench fire. The atmospheric pathway exposure occurs during site operations (in the year 0), while the erosion occurs in the humid areas beyond 3,000 years and in the arid areas beyond 10,000 years.

In most cases, releases through the ground water-to-well pathway occur beyond several hundred years. The longest time is in the humid impermeable (greater than 2,000 years), next longest (greater than 200 years) in the arid, and least (greater than 16 years) in the humid permeable hydrogeologic/climatic setting.

Table 10-10 presents exposure data versus the scenario for the ground water-to-well pathway. In all scenarios, the humid permeable hydrogeologic/climatic setting has the highest maximum annual exposure; the next highest is in the humid impermeable setting, with the arid setting having the least exposure.

Table 10-11 presents exposure data versus scenarios for the erosion pathway at the humid permeable and humid impermeable sites. Table 10-12 presents the doses delivered through the atmospheric inhalation pathway to all three hydrogeologic/climatic settings.

Table 10-8. CPG exposures for humid impermeable settings affecting offsite residents

Scenario	Maximum exposure, mrem/year Pathway		
	Ground Water to River	Spillage	Bathtub
1	7.5E-07	2.1E-03	2.1E-04
2	1.7E-06	3.5E-03	4.7E-04
3	7.3E-07	5.4E-04	6.3E-04
4	1.4E-06	5.2E-04	5.7E-04
5	8.0E-07	2.8E-04	1.5E-04
6	5.0E-07	1.6E-03	2.5E-04
7	9.0E-09	5.9E-05	1.5E-05
8	4.2E-09	1.8E-05	4.7E-06
9	1.1E-06	1.4E-05	8.9E-05
10	7.1E-07	6.9E-04	2.4E-04
11	5.5E-07	9.0E-04	3.5E-04

Table 10-9. CPG exposures for direct gamma and dust inhalation pathways to onsite workers and visitors

Scenario	Maximum exposure, mrem/year Pathway	
	Direct Gamma	Dust Inhalation
1	1.2E+01	3.2E-02
2	1.1E+01	1.1E-02
3	1.8E-01	1.6E-04
4	8.9E-01	5.3E-02
5	5.4E+00	2.1E-01
6	8.8E+00	2.1E-02
7	2.4E-02	1.3E-01
8	4.9E-03	3.9E-02
9	1.6E-01	7.6E-03
10	2.1E+01	3.9E-02
11	4.4E+00	1.3E-02

Table 10-10. CPG exposures for the ground water-to-well pathway to offsite residents

Scenario	Maximum exposure, mrem/year Ground water-to-well pathway		
	Humid Impermeable Site	Humid Permeable Site	Arid Permeable Site
1	3.1E-02	1.5E-01	7.3E-04
2	7.1E-02	3.3E-01	1.7E-03
3	1.5E-03	1.2E-01	6.4E-05
4	1.8E-02	1.1E+00	1.4E-04
5	6.6E-02	1.5E+00	9.4E-05
6	2.0E-02	6.7E-01	4.9E-04
7	3.7E-04	1.2E-03	4.7E-05
8	1.7E-04	5.7E-04	2.2E-05
9	5.4E-01	2.4E+00	1.3E-04
10	6.1E-02	1.1E+00	7.3E-04
11	7.5E-03	4.9E-01	3.0E-04

Table 10-11. CPG exposures for erosion pathway to offsite residents

Scenario	Maximum exposure, mrem/year Erosion pathway	
	Humid Impermeable Site	Humid Permeable Site
1	2.8E-06	3.3E-03
2	1.8E-06	1.9E-03
3	1.1E-06	1.7E-03
4	4.1E-06	5.6E-03
5	3.4E-06	4.5E-03
6	2.0E-06	2.5E-03
7	2.7E-06	3.6E-03
8	8.5E-07	1.1E-03
9	6.4E-08	1.0E-04
10	2.1E-06	2.5E-03
11	2.5E-06	3.1E-03

Table 10-12. CPG exposures for atmospheric inhalation pathway to offsite residents

Scenario	Maximum exposure, mrem/year Atmospheric inhalation pathway		
	Humid Impermeable Site	Humid Permeable Site	Arid Permeable Site
1	2.1E-06	4.4E-06	5.0E-06
2	1.8E-06	3.8E-06	4.3E-06
3	1.1E-09	2.3E-09	2.6E-09
4	8.0E-06	1.7E-05	2.0E-05
5	3.6E-02	6.0E-02	3.7E-02
6	1.2E-06	2.6E-06	2.9E-06
7	6.7E-06	1.5E-05	1.6E-05
8	1.1E-05	2.4E-05	2.7E-05
9	1.3E-03	3.2E-03	1.3E-03
10	1.8E-03	3.0E-03	1.8E-03
11	5.8E-04	9.4E-04	6.0E-04

(E) Exposure to Onsite Resident

After the disposal facility is closed, it is assumed that the land area will be available for certain functions. There are two pathways assumed for an onsite resident, the biointrusion by plant roots to the undisturbed waste giving a dose to people from eating the plants and a reclamation pathway, where the land is disturbed by excavating for a basement and the waste is brought to the surface and mixed with the surface soil and food grown within this mixed soil/waste. The maximum exposure from these two pathways occurs in the first year after closure. Only the rural and suburban disposal settings are assumed to have food grown on the land. Cesium-137 is the dominant radionuclide in those scenarios handling several waste streams and uranium-234/238 in the two uranium disposal scenarios, UHX-MD and UF-MD.

Tables 10-13 and 10-14 present maximum annual doses versus scenarios for these two onsite resident pathways.

(F) CPG Dominant BRC Waste Streams

The surrogate BRC waste streams causing the highest CPG exposures are as follows (in descending order): P-COTRASH, B-COTRASH, I-COTRASH, N-LOTRASH, and I-ABSLIQD.

(G) CPG Dominant BRC Waste Stream Radionuclides

The radionuclides dominating the CPG analyses for the various pathways involved in the BRC waste disposal scenarios are listed in Table 10-15.

10.8 Discussion of Transportation CPG Results

The primary exposure pathway to the transportation workers will be gamma exposures. The study (see Section 10.4) showed that the major exposure to the transportation worker appears to occur from the cobalt-60/58 and the cesium-134/137 radionuclides (see Table 10-7). These radionuclides primarily occur in the following waste streams: P-COTRASH, B-COTRASH, I-COTRASH, I-ABSLIQD, and N-LOTRASH.

10.9 Discussion of CPG Versus Population Results

Table 10-16 presents a combination of the cumulative population health effects versus a range of BRC standards using CPG exposures and the surrogate waste streams considered.

As shown in Table 10-16 and Sections 10.7.2 (F) and (G), certain radionuclides and specific waste streams are the major contributors causing the excess health effects and CPG exposure. It is therefore possible to vary both the population health effects and the CPG doses by selecting appropriate waste streams with and without specific radionuclides to be declared BRC. The risks of any alternative BRC scenario can be examined by constructing a scenario which eliminates those waste streams contributing the highest dose. Also restricting any of the waste streams will influence the amount of BRC waste as a percent of the total volume.

Table 10-13. CPG exposures for biointrusion pathway to onsite residents

Scenario	Maximum exposure, mrem/year Biointrusion pathway Humid Impermeable Site	Humid Permeable Site	Arid Permeable Site
1	1.1E+01	6.4E-01	9.0E-01
2	2.7E+00	1.6E+00	2.2E+00
3	N/A	N/A	N/A
4	7.6E-02	4.6E-02	6.4E-02
5	3.0E-01	1.8E-01	2.6E-01
6	8.1E-01	4.8E-01	6.8E-01
7	4.2E-04	3.8E-04	4.4E-04
8	1.3E-04	1.2E-04	1.4E-04
9	N/A	N/A	N/A
10	6.8E-01	4.1E-01	5.7E-01
11	N/A	N/A	N/A

Note: NA - Not applicable.

Table 10-14. CPG exposure for the food grown onsite pathway

Scenario	Maximum exposure, mrem/year Food grown onsite pathway Humid Impermeable Site	Humid Permeable Site	Arid Permeable Site
1	3.2E-01	1.9E-01	2.7E-01
2	8.1E-01	4.8E-01	6.7E-01
3	N/A	N/A	N/A
4	2.3E-02	1.4E-02	1.9E-02
5	9.0E-02	5.5E-02	7.7E-02
6	2.4E-01	1.5E-01	2.0E-01
7	1.2E-04	1.1E-04	1.3E-04
8	3.8E-05	3.5E-05	4.1E-05
9	N/A	N/A	N/A
10	2.1E-01	1.2E-01	1.7E-01
11	N/A	N/A	N/A

Note: N/A - Not applicable.

Table 10-15. Dominant radionuclides for the CPG pathways

Pathway	Dominant radionuclides
Ground Water-to-River	C-14 I-129 U-234/238
Ground Water-to-Well	C-14 I-129 U-234/238
Spillage	Cs-137 Co-60 U-234/238
Erosion	Pu-239 U-234/238 C-14 I-129
Bathtub	C-14 I-129 U-234/238
Food Grown on Site	Cs-137 U-234/238
Biointrusion	Cs-137 U-234/238
Direct Gamma	Co-60 U-235
Dust Inhalation	Am-241 U-234/238 Cs-60
Atmospheric	Am-241 U-234/238 H-3 Co-60

Table 10-16. Excess health effects over 10,000 years nationwide from disposal of 20 years of accumulated DOE and commercial wastes versus regulated LLW disposal

Alternative BRC levels (maximum CPG dose) mrem/yr	Lifetime risk	BRC waste percent of total volume	Additional health effects versus current practice[a]	BRC surrogate waste stream rejected[b]
15	4.2E-04	43%	457	P-COTRASH and B-COTRASH
4	1.1E-04	34%	85	I-COTRASH and above
1	2.8E-05	30%	30	N-LOTRASH, I-ABSLIQD, and above
0.4	1.1E-05	28%	20	N-LOWASTE and above
0.1	2.8E-06	25%	1	F-PROCESS, U-PROCESS, I-BIOWAST, P-CONDRSN, and above

[a]Under current practice, commercial LLW is treated as Class A and disposed of at a SLD; consumer wastes are unregulated. DOE waste is disposed of in the as-generated form in an SLD site.
[b]Table 10-5 presents a complete listing of the BRC surrogate waste streams used in the assessment.

10.10 Discussion of the Reference Scenarios

Four localized scenarios using presently deregulated waste streams were chosen as a reference for comparison and give a perspective for our health impacts analysis. The deregulated waste streams used were consumer wastes and those wastes associated with the NRC biomedical rule. Sections 3.3.3, 4.4, and 10.6 describe and discuss the waste streams and the four scenarios.

10.10.1 Cumulative Population Health Effects

Table 10-3 shows the four BRC reference scenarios and their excess population health effects over 10,000 years. As shown in the table, the deregulated consumer wastes presented less than 0.1 health effects over the 10,000 years for urban, suburban, and all hydrogeologic/climatic settings. For the deregulated BIOMED waste stream containing only institutional wastes with just carbon-14 and tritium (H-3) and disposed of in a suburban setting, the 100 percent volatilization of the two radionuclides for this incineration scenario (LURO-1) causes less than 0.3 health effect over 10,000 years for all hydrogeologic/climatic settings. For the same incineration scenario (LURO-2), except that there is only 50 percent volatilization of the two radionuclides, the humid permeable hydrogeologic/climatic setting indicates the possibility of greater than 20 health effects over 10,000 years, while the other two hydrogeologic/climatic settings cause less than one health effect over 10,000 years.

As mentioned in Section 10.7.1 (B), the use of incineration can have the effect of reducing population health effects in relation to direct burial. In this case, the 50 percent volatilization incineration scenario allows enough of the wastes' radionuclides (C-14 and H-3) to be disposed of by burial, thus allowing a larger source to reach the population through the water pathway in the humid permeable setting. During the 100 percent volatilization incineration scenario, the source terms are diluted and the inhalation pathway provides a much lower body response, as compared to the water ingestion pathway.

10.10.2 CPG Exposures

Table 10-6 shows the BRC reference scenarios and their maximum CPG doses. As shown, releases from the deregulated consumer wastes presented less than 0.02 mrem/yr for the urban and suburban areas, and for all the hydrogeologic/climatic regions. For the deregulated BIOMED institutional wastes and the 100 percent volatilization incineration scenario (LURO-1), the CPG doses are less than 0.0008 for all three hydrogeologic/climatic regions. However, the 50 percent volatilization incineration scenario (LURO-2) does show higher doses in the ground water-to-well pathway (at year 16 after closure for humid permeable and at 2,320 years for humid impermeable), where carbon-14 is the critical radionuclide.

As indicated in the previous section, the total incineration of the wastes provides for lower health impacts due to the fact that the inhalation pathway affords less of a risk to the whole body than does the ingestion pathway.

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Chapter 11: SENSITIVITY ANALYSIS OF THE PRESTO-EPA MODELS

11.1 Introduction

This chapter describes the program of sensitivity analysis conducted on the PRESTO-EPA models. The analysis methodology is described and the results summarized. The rationale for the analyses, as well as their limitations, is presented.

11.1.1 Background

In developing the LLW Standard, it was necessary for EPA to assess the impacts from the disposal of LLW using a variety of disposal methods, site locations, and other variables. These assessments were performed using the PRESTO-EPA computer models.

The PRESTO-EPA model was developed jointly by EPA and Oak Ridge National Laboratory (EPA83). The model, completed in 1983, was expanded by EPA and Rogers and Associates Engineering Company into a family of health impact assessment codes (Ro85). These codes are described in more detail in Chapter 8.

Because PRESTO-EPA was developed specifically for the LLW standard-setting effort and is a new code, a program of code improvement and verification was conducted. This program included: quality assurance audits of all codes, extensive test runs, peer review, and review by EPA's Science Advisory Board. Another important aspect of this program, sensitivity analysis, is discussed in this chapter.

11.1.2 Description of Sensitivity Analysis Program

Sensitivity analysis can be defined as changing the values of specified input parameters, either individually or as a group, in order to assess the change in the model output. The output from the test runs is compared to the output from standard runs, where all the input parameters remain constant. In this way the results can be quantified and a relative measure of the model's sensitivity to changes in various input parameters is determined.

In our sensitivity analysis program we conducted two broad types of analyses. The first, called single parameter sensitivity analysis, consisted of varying only a single parameter (in some cases, a few parameters) at a time. Examples of this type of analysis would be increasing or decreasing the aquifer flow rate or the permeability of the trench cap. The single parameter sensitivity analysis is summarized in this chapter and the analysis results and conclusion given. A more detailed discussion of the analysis and results is contained in a separate EPA technical report (EPA88).

The second type, which we called scenario sensitivity analysis, consisted of varying a group of input parameters associated with a specific scenario variable, in order to modify the scenario associated with one of the base case analyses. Examples of this type of analysis would be changing the waste form, the size of the site, or the disposal methods. We termed as scenario sensitivity analyses any assessments we performed other than the base case analyses outlined in Chapters 9 and 10.

11.1.3 Rationale for Conducting Sensitivity Analyses

Since the PRESTO-EPA models were new codes, it was important to test them as extensively as possible. The single parameter sensitivity analyses were an important part of this test program, as they allowed us to identify the most sensitive input parameters. The identification of the sensitive input parameters prior to the final production runs allowed for more efficient use of limited resources in better characterizing those parameters which would most affect model output. Also, sensitive input parameters were flagged for more thorough review when checking input lists for accuracy, prior to production runs.

In addition to identifying the sensitive parameters, the program of single parameter sensitivity analyses allowed us to:

- Reconfirm code logic and reliability;
- Test the effects of parameters with wide ranges of values or large degrees of uncertainty; and
- Evaluate controversial input parameter values.

We also carefully reviewed the results of each of the sensitivity tests, which provided us with a great deal of knowledge about how the codes responded to changes in individual input parameter values and how various aspects of the output were affected.

The scenario sensitivity analyses allowed us to analyze the results of scenarios different from those chosen as our standard base-case analyses for the LLW standard-setting effort (discussed in Chapters 9 and 10). In this way we were able to test how scenario assumptions that were made about the base cases affected the output results.

The scenario sensitivity analyses differed from the single parameter analyses in that, in general, a group of input parameters related to a specific scenario variable was varied. The purpose of the analyses was not necessarily to determine what would happen when certain input parameter values were changed, but to see what would happen when the scenario variables were changed. In choosing a set of standard scenarios to analyze for our standard-setting effort, it was necessary to make certain assumptions about the scenarios, such as the volume of waste disposed of, the form of the waste, or the disposal methods that would be used. The scenario analyses allowed us to determine how sensitive the results were to these assumptions.

11.1.4 Limitations of the Sensitivity Analyses

During the testing and verification of the PRESTO-EPA code, the question of the uncertainty of the results was raised. A Monte Carlo analysis technique was suggested as a method for quantifying the uncertainty in the risk assessments. In this technique, the PRESTO-EPA deterministic risk assessment models would be combined with random input parameter sampling and statistical analysis submodels to form a probabilistic risk assessment model. A Monte Carlo analysis of a large, complex risk assessment model such as PRESTO-EPA would require a large number of computer calculations. This was not feasible from the standpoint of either the funds or time available, especially considering that the main purpose of our generic LLW standard setting analyses was to conduct a relative comparison of various control methods, for the purpose of setting a standard, rather than obtaining absolute values from site-specific disposal situations.

A combination of single parameter and scenario sensitivity analyses was selected as a less rigorous but acceptable method of determining some of the uncertainties associated with the PRESTO-EPA results. In addition, this type of analysis is a very useful method for determining the sensitivity associated with the various PRESTO-EPA input parameters and disposal scenarios, since sensitivity analysis provides the relative sensitivity of model results due to changes in the parameters tested. It also helps in re-verifying the PRESTO-EPA code and in discerning differences in overall disposal system performance due to changing scenario assumptions, such as the use of a buffer zone or high-integrity containers.

The type of sensitivity analysis that we performed, however, has shortcomings and limitations in that the relative importance of each parameter could be affected by the values of the other parameters. We addressed this problem to a certain degree by performing a large number of sensitivity runs under many different scenarios. This helped to identify parameters and scenario variables that were sensitive under various assumptions, but did not eliminate the overall problem of having determined the sensitivity based upon some assumed set of input parameters. The problem of choosing an assumed set of input parameters was dealt with by using "standard input data sets," which were the values used for our base case analyses. In this way we were able to determine which parameters and scenario variables were most sensitive under the base case scenarios and which were the most important, since the results of these scenarios were what would be used as a basis to develop the LLW Standard. If the base case scenarios were changed, the sensitivity of certain input parameters might change, but characterizing this was felt to be beyond the scope of this study and of lesser importance.

Another limitation of our analyses was that not every input parameter or scenario was tested. Because of the large number of input parameters and possible scenarios, it was impractical, if not impossible, to test each one. For these analyses, we tested those parameters and scenarios which we felt would be most sensitive, based on the extensive test runs and code

review we had performed prior to the sensitivity analysis program, as well as on good engineering judgment. In addition, we tested parameters and scenarios if we felt the results to be of particular interest or if there was some uncertainty or controversy over the value or assumptions we were using for the base case.

Finally, because of the inherent limitations of the single parameter and scenario sensitivity analyses, no direct measure of model uncertainty could be made. In order to provide some direct measure of the uncertainty of the model output, an analysis of uncertainty was conducted. This analysis and the results are described in Chapter 12.

11.2 Single Parameter Sensitivity Analysis

The single parameter sensitivity analyses, as conducted by EPA, consisted of systematically varying the values of specific input parameters, to quantify their effects on code output. To determine the relative sensitivity of each of the parameters tested, we developed a quantitative sensitivity index, as well as an associated qualitative rating of sensitivity. For these analyses, we chose specific standard data sets, with input parameter values equivalent to those used in the base case runs described in Chapters 9 and 10, and known output against which to compare the output from the sensitivity runs. Table 11-1 outlines the important features of our "standard" input data sets.

The majority of the parameters tested were related to infiltration, nuclide retention and release, transport, and exposure mechanisms. The health risk factors used in the codes were not tested. The health risk factors are calculated by the EPA RADRISK code and are used as inputs to the program DARTAB. The DARTAB code, which is used as a subroutine by PRESTO-EPA, combines radionuclide uptakes with the RADRISK health risk factors to determine health impacts. The DARTAB portion of the PRESTO-EPA code was not included in the sensitivity analyses, as it received extensive review during development of the AIRDOS-EPA code, for which it was originally developed (Be81, Mo79). We did, however, perform sensitivity analyses on the health effect conversion factors (HECF), which are used to assess long-term health effects in the regional basin population, as described in Chapter 8.

The various PRESTO-EPA codes, while basically similar in design and function, have differences that are important to understanding the analysis results. The PRESTO-EPA-POP code is used to estimate the cumulative health effects, consisting of fatal cancers and serious genetic effects, to both local and regional basin populations over 10,000 years. Local population health effects are calculated through a number of detailed pathway analyses using iterative yearly updates for a period of 1,000 years. Health effects for the regional basin population are calculated using a HECF for both an initial 1,000-year period and for an additional 9,000 years (during which the local population is included within the regional basin) for a total of 10,000 years. The local and regional basin populations are assumed to live at distances from the disposal site similar to what one might encounter today in those geographic regions. Because the PRESTO-EPA-POP code was the basis for the other codes, it was tested extensively, with a total of 54 test runs performed on 30 input parameters (see Table 11-2).

Table 11-1. Important features of "standard" data sets used in the single parameter sensitivity analyses

Characteristics	PRESTO-EPA-POP	PRESTO-EPA-CPG	PRESTO-EPA-DEEP	PRESTO-EPA-BRC
Sites Evaluated	Humid Permeable Arid Permeable Humid Impermeable	Humid Permeable Arid Permeable Humid Impermeable	Humid Permeable Arid Permeable Humid Impermeable	Humid Permeable Arid Permeable **
Disposal Method	Conventional Shallow Disposal	Conventional Shallow Disposal	Deep Geological Hydrofracture Deep Well Injection	Urban Sanitary Landfill Municipal Dump Urban Sanitory Landfill w/Incineration
Waste Type	Absorbed Waste	Trash Waste Absorbed Waste Solidified Waste Activated Metal Incinerated/ Solidified Waste	Absorbed Waste	Absorbed Waste
Site Capacity	250,000 m^3*	250,000 m^3	Variable	Variable
Modeling Period	10,000 years	1,000 years	10,000 years	10,000 years
Population Analyzed	Local and Regional Populations	Critical Population Group	Local and Regional Populations	Local and Regional Populations
Impact Analyzed	Cumulative Health Effects	Maximum Annual Whole-Body Dose	Cumulative Health Effects	Cumulative Health Effects

* Individual nuclide activity is used based on 250,000 m^3 site, but since only 10 of the 40 nuclides are evaluated, the actual source term used is less than 250,000 m^3.

**The humid impermeable site was not evaluated in the BRC sensitivity analysis, since the results for the parameters tested would be the same as for the other sites tested.

Table 11-2. Summary of input parameters analyzed and tests performed

Code	Total Input Parameters	Input Parameters Evaluated	Sensitivity Tests Performed
PRESTO-EPA-POP	146	30	54*
PRESTO-EPA-CPG	151	56	121
PRESTO-EPA-DEEP	154	20	41
PRESTO-EPA-BRC	151	12	22

*An additional 10 test runs were performed using the PRESTO-EPA-POP code in testing the health effect conversion factor (HECF).

The PRESTO-EPA-CPG code is used to estimate the maximum annual dose and the year in which the maximum dose occurs for a critical population group. This group is assumed to live adjacent to the disposal site and obtain its water from a well or stream located at the boundary fence. Only the first 1,000 years are evaluated and impacts to the regional basin population are not determined, as discussed in Chapter 8. Figure 11-1 illustrates the differences between the population groups and their locations, as modeled by these two codes. The PRESTO-EPA-CPG code is quite different from the PRESTO-EPA-POP code, in the impacts that are assessed and in how the source term is modeled. Therefore, this code was also tested extensively, with a total of 121 test runs performed on 56 input parameters (see Table 11-2).

The PRESTO-EPA-DEEP code is used to estimate the cumulative population health effects to both local and regional basin populations for 10,000 years from deep disposal options. As with the PRESTO-EPA-POP code, regional basin health effects are determined using a conversion factor. The results of the PRESTO-EPA-DEEP code are not used as extensively in our standard development effort as are those from the other codes; however, since the pathways are different, the code was tested fairly extensively. A total of 41 test runs were performed on 20 input parameters (see Table 11-2).

The PRESTO-EPA-BRC code is used to estimate the cumulative population health effects to both local and regional basin populations for 10,000 years, in the same manner as the PRESTO-EPA-POP code. The major difference between the two codes is that PRESTO-EPA-BRC also determines health effects to onsite workers from unregulated disposal of BRC wastes through incineration, dust inhalation, and direct exposure pathways. Because the two codes are so similar otherwise, the major portions of the PRESTO-EPA-BRC code that were tested were those having to do with the pathways for the onsite workers. Therefore, only 22 test runs were performed on 12 input parameters (see Table 11-2).

The PATHRAE-EPA code, which estimates the annual pathway doses to a critical population group and to onsite workers, is not based on the PRESTO-EPA codes, although it is compatible with these codes (Sh86). Because PATHRAE-EPA is a different code, it was tested in a separate study by Rogers and Associates Engineering Company (Sh87a). The results of this analysis are summarized in section 11.2.2 (G).

The codes described briefly above are discussed in detail in Chapter 8. The number of input parameters associated with each code, how many were tested, and the number of test runs performed, are outlined in Table 11-2.

In addition to the evaluations of the codes, a set of tests was performed to evaluate the HECF. The PRESTO-EPA-POP code was used for this purpose, although the results apply equally to the other codes using the HECF. A total of 10 tests were conducted on the HECF.

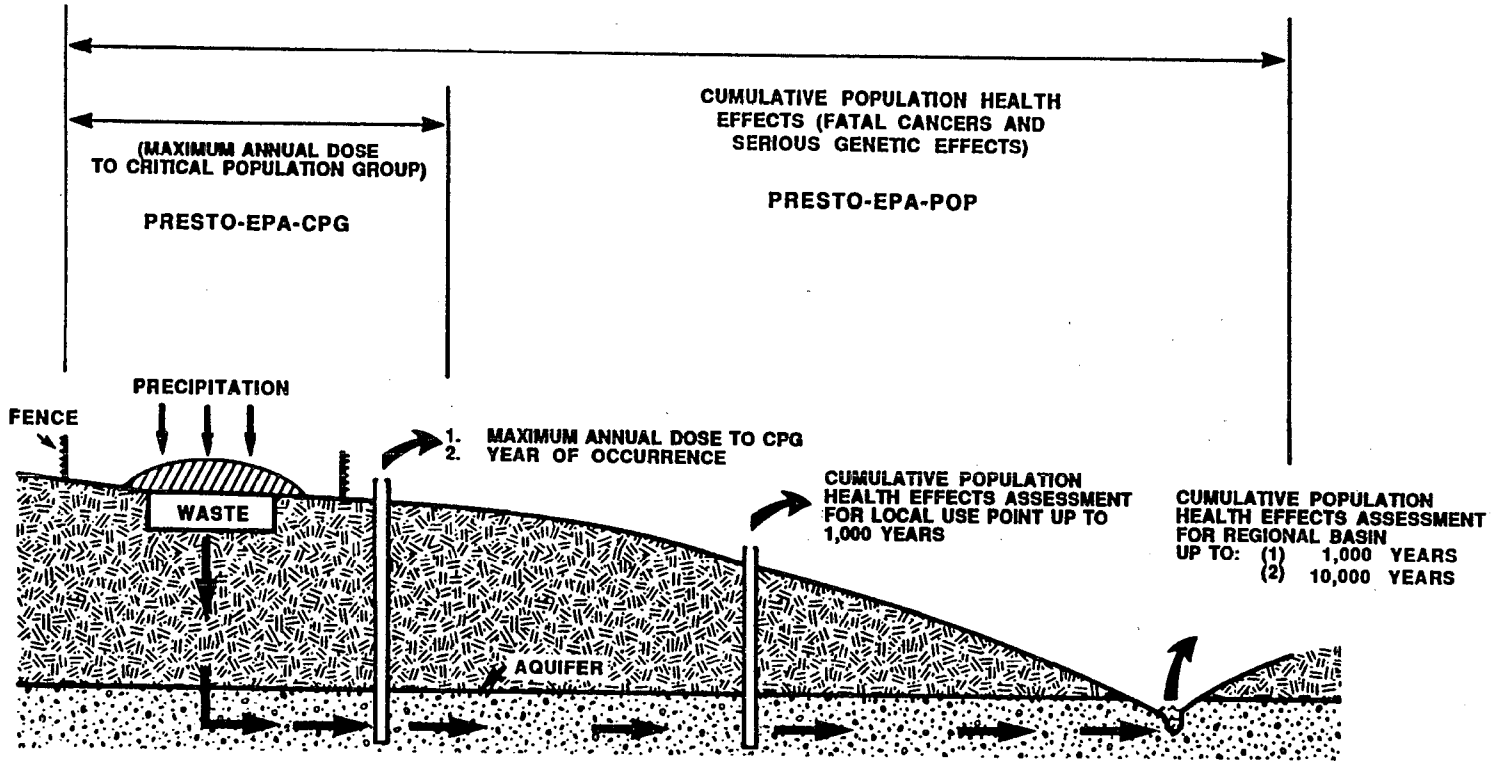

Figure 11-1. Differences in the Health Impacts Estimated and the Locations and Populations Evaluated for the PRESTO-EPA-POP and PRESTO-EPA-CPG Analyses

11.2.1 Methodology of Single Parameter Sensitivity Analysis

In conducting the single parameter sensitivity analyses, a number of separate steps were required. Before performing the actual sensitivity runs the "standard" scenarios had to be chosen, the input parameters to be tested had to be identified, and input parameter test values had to be determined. After these initial steps were completed, the sensitivity runs were performed and the results processed so that they could be more easily analyzed. The analysis of the results consisted of comparisons of the test run to standard run results; calculation of test run and standard run output and input ratios; and the determination of quantitative sensitivity indices and a qualitative sensitivity rating for each parameter tested. Each of these steps is discussed briefly in the following sections, with a more detailed discussion contained in an EPA technical report on the single parameter sensitivity analyses (EPA88).

(A) Choosing Standard Scenarios

The standard data sets and their output serve as a comparison against the test data sets and output. In conducting the sensitivity analyses we typically ran the code using a standard data set and recorded the results. We then changed one input parameter value and ran the code again using this test data set, again recording the results. Finally, we compared the results from the standard data set to the results from the test data set to see how much of a change in output had occurred due to the input change. This process, therefore, required a set of standard data sets.

In choosing our standard data sets for the sensitivity analyses, it was decided to use the data sets associated with our "base case" scenarios described in Chapters 9 and 10, since these data sets were used extensively in performing the LLW standard setting effort. Although there were 7 base cases analyzed for regulated disposal and 15 for BRC disposal, because of the large number of sensitivity analyses we would be performing we could not test all of the base case scenarios in detail. Instead, we elected to more carefully analyze only a few scenarios to keep the analysis from becoming unwieldy. The scenarios that were used are summarized in Table 11-1 and are discussed in more detail in the EPA technical report on sensitivity analysis (EPA88).

(B) Choosing Input Parameters to Test

The PRESTO-EPA codes have approximately 150 input parameters associated with each of them. Because of the large number of input parameters, it was not practical to test every one. We had, however, performed a large number of test and production runs prior to our sensitivity analysis program and were able to choose, based on the results of these runs and good engineering judgment, those input parameters which we felt were most important to analyze. We chose parameters which we felt would be most sensitive, which had a large degree of uncertainty or a wide range of possible values associated with them, or which were controversial for some reason.

Using these criteria, we chose 30 PRESTO-EPA-POP, 56 PRESTO-EPA-CPG, 20 PRESTO-EPA-DEEP, and 12 PRESTO-EPA-BRC input parameters to analyze. In addition, we tested a number of variables associated with the HECF. A listing of those parameters which were tested is included in a separate EPA technical report (EPA88).

(C) Determining Test Input Values

Once the input parameters to be analyzed were chosen, we had to determine what test values to use for the input parameter. We generally chose values at either one or both ends of a reasonable range around the standard value or, if the parameter was very uncertain, one or two orders of magnitude above or below the standard value. In all cases we tried to select test values that were realistic and would give us meaningful results.

Based on these criteria, 54 separate PRESTO-EPA-POP test runs were performed. A total of 121 were done for PRESTO-EPA-CPG, 41 for PRESTO-EPA-DEEP, and 22 for PRESTO-EPA-BRC. Ten test runs were performed on the HECF. Each of the sensitivity tests performed, along with the test and standard values used are listed in a separate EPA technical report (EPA88).

(D) Processing of Output

In order to analyze the results of the test runs, a convenient measure of output was required so that comparisons could be made to the standard runs. Both the standard and test run output was processed to result in a single measure of impact, i.e., maximum annual whole-body dose for PRESTO-EPA-CPG and 10,000 year cumulative population health effects for PRESTO-EPA-POP, PRESTO-EPA-DEEP, and PRESTO-EPA-BRC. A more detailed discussion of how the output was processed is contained in the EPA technical report (EPA88). How these measures of impact are used is discussed in the next section.

(E) Comparing Test to Standard Runs

Most sensitivity tests consisted of changing only one specific input parameter value, although in some cases it was more efficient or made more sense to change a set of input values as a group. Once the input parameter value(s) was changed from the standard to the test value, the code was run and the output results recorded.

The output results from the test runs were compared to the output results from the standard runs. This was done not only to measure the sensitivity of the input parameter, but also to help re-evaluate the code logic and reliability and gain some knowledge of how changing a certain input parameter would affect the model output. The output was evaluated in two ways: using ratios of standard values to test values and using summaries of the test output, such as the test result summary form developed for most of the PRESTO-EPA test runs.

The use of ratios in evaluating the sensitivity tests provided a general measure of an input parameter's sensitivity and of the reasonableness of the results. Ratios were calculated for the input values by simply dividing the input parameter test value by the input parameter standard value. The output ratios were based on a convenient measure of health impact, the total number of cancers over 10,000 years in the case of PRESTO-EPA-POP, PRESTO-EPA-DEEP, and PRESTO-EPA-BRC, and maximum annual dose to the CPG in the case of PRESTO-EPA-CPG. The output ratios were calculated by dividing the test run impact by the standard run impact for each sensitivity test. The sensitivity run could then be evaluated by comparing the input and output ratios.

A results summary was also completed for each sensitivity test. This summary contains a description of the input parameters, how they vary from the standard, a listing of how the output varies from the standard, and the input and output ratios. Input and output ratios and results summaries for each test are included in a separate EPA technical report (EPA88).

(F) Determination of Sensitivity

In order to determine which were the most sensitive input parameters, input and output ratios were used to determine a quantitative sensitivity index for each input parameter tested. The quantitative sensitivity index was then used to assign each input parameter a qualitative sensitivity rating of none, low, medium, or high. A listing of input parameters with sensitivity ratings of medium or high is contained in Table 11-3.

A more detailed discussion of the quantitative sensitivity index and the qualitative sensitivily rating, including a listing of the sensitivity index and qualitative rating associated with each input parameter tested, is included in a separate EPA technical report (EPA88).

11.2.2 Results and Discussion of Single Parameter Sensitivity Analyses

In performing the sensitivity analyses and completing the test results summary forms, basic information was learned about the codes and how they respond to changes in input values. This is discussed below under the separate model headings. Results from tests on specific parameters also provided useful information for later production runs, and our final runs incorporated many modifications based upon information learned during the sensitivity analysis program. The major emphasis of the sensitivity analysis, however, was to identify those parameters which were most sensitive.

The identification of the most sensitive parameters was felt to be particularly important since small differences in the value used for an input parameter might affect the output results to a large degree. By identifying the sensitive input parameters, limited resources (both time and money) could be used to characterize those parameters which would

Table 11-3. PRESTO-EPA input parameters identified as exhibiting relatively medium or high sensitivity under the conditions of this sensitivity analysis

PRESTO-EPA-POP

- Percent of Trench Cap Failure
- Release Fraction for Solidified Waste

PRESTO-EPA-CPG

- Percent of Trench Cap Failure
- Trench Cover Permeability and Porosity
- Waste Release Fraction and Distribution Coefficients
- Waste Container Related Parameters
- Trench and Sub-Trench Porosity and Residual Saturation
- Distance from Trench to Well and Trench to Aquifer
- Aquifer Porosity and Thickness
- Ground and Surface Water Velocity
- Mobile Nuclide Source Term
- Spillage Fraction for Arid Sites
- Atmospheric Pathway Parameters for Arid Sites
- Duration of Institutional Control (Active Site Maintenance)
- Amount of Water Uptake by Humans

PRESTO-EPA-DEEP

- Vertical Water and Groundwater Velocity
- Density of the Confining Stratum
- Distribution Coefficient (K_d) for the Vertical Zone
- Waste Release Fraction

PRESTO-EPA-BRC

- Volatilization Factor for Incinerated Radionuclides
- Fraction of Surface Spillage for Arid Sites

HEALTH EFFECT CONVERSION FACTOR

- Fish Bioaccumulation Factors
- Fish Consumption Rates
- Human Water Consumption Rates
- River-flow-to-population Ratio

have the greatest effect on the results. Once a parameter was identified as sensitive, it could be flagged for more detailed consideration or at least noted as a parameter that should be reviewed carefully when performing later production runs.

Each parameter tested was given a qualitative sensitivity rating ranging from none to high. In general, these ratings were based on the input and output ratios. If an input ratio was large (i.e., a relatively large difference between the input values used in two runs) and the output ratio relatively small, the qualitative rating would be low or none. If the input and output ratios were approximately equal, the sensitivity rating was labeled as medium. A qualitative sensitivity rating of high meant that the output ratio was relatively high compared to the input ratio. A listing of the quantitative sensitivity ratings for each of the parameters tested is given in a separate EPA technical report (EPA88). Those parameters identified as having "medium" or "high" sensitivity are summarized in Table 11-3.

In reviewing Table 11-3, one notices that the number of sensitive parameters varies depending upon the code. The reason for this is based on the differences between the codes and on what health impact they are evaluating. The following sections discuss each of the codes and how their characteristics affect the sensitivity analysis results. The results from the analysis on the HECF is also discussed.

(A) PRESTO-EPA-POP

In order to understand the sensitivity analysis results, it is necessary to have a basic idea of the results of the base case analyses, as discussed in Chapters 9 and 10. The general results from the PRESTO-EPA-POP base case analyses show that the local population health effects do not dominate in any of the three regional hydrogeologic and climatic scenarios. This is due to the limited amount of contaminated nuclides which the relatively small local population can take in. The majority of health effects are incurred by the much larger regional basin population. The health effects to the regional basin population, and the pathways by which they occur, vary considerably over the three hydrogeologic and climatic regions. The general trends in the results are more easily seen by reviewing separately the three general settings: humid permeable, humid impermeable, and arid permeable.

At the site characterized by relatively permeable soil and high rainfall, most of the mobile radionuclides leach out of the trench and into the aquifer during the initial 1,000-year period. The majority of the total health effects are incurred by the regional basin population through the groundwater pathway during the first 1,000 years.

At the site characterized by high rainfall and soil with relatively low permeability, the trenches fill with water after a portion of the trench cap has failed, and much of the activity will be leached from the waste and will escape from the trench through overflow (bathtub effect) in a relatively short period of time. Because of transport through the surface

water pathway, less mobile nuclides can reach the populations more quickly than they would have through the groundwater pathway. The regional basin population receives the majority of the total health effects through the surface water pathway during the first 1,000 years.

At the site characterized by relatively permeable soil but low rainfall, most activity does not reach either the local or regional basin populations until relatively late in the modeling period. The local population incurs a few health effects in the first few years after site closure due to windblown (atmospheric) transport of nuclides spilled onto surface soils during site operations. These health effects, while quite small, are the only health effects until late in the modeling period. This is due to the long travel time required for contamination to reach the aquifer and then travel to the local and regional basin populations by groundwater. The overall impact is dominated by health effects from activity reaching the basin population through the groundwater pathway between 500 and 10,000 years, depending upon the disposal method.

Based on the above general results, we can identify those parameters which are most sensitive in causing changes in the PRESTO-EPA-POP output. First of all, there are a number of input parameters that affect the health effects to the local population during the first 1,000 years. However, since the health effects to the local population are only a small portion of the total health effects, these changes will not usually affect overall results significantly. Second, since regional basin health effects are incurred only from the surface water and groundwater pathways, only input parameters that ultimately affect these pathways will cause significant changes in the total health effects.

At the humid permeable site, the health effects will be most sensitive to parameters that change the release and transport of nuclides from the trench to the groundwater system, such as the integrity of the trench cap.

At a humid impermeable site, the health effects will be most sensitive to parameters that increase or decrease the rate of transport of nuclides from the trench into the surface water system, such as the integrity of the trench cap and the solidified waste release fraction.

At the arid permeable site, the very short-term health effects will be most sensitive to parameters that change the amount of surface spillage or downwind nuclide concentrations. The overall health effects, however, will be most sensitive to parameters that alter groundwater concentrations, the same as for the humid permeable site. Because of the long travel time required for groundwater transport of even the mobile nuclides at the arid permeable site, radiological decay of the longer-lived nuclides becomes more important. Parameters that modify the time required for these nuclides to reach the local and regional basin populations such as trench cap failure and, to a lesser degree, trench-to-aquifer distances, aquifer flow rates, or distances to the population, could lead to significant changes in total health effects.

In summary, the PRESTO-EPA-POP code exhibits relatively low sensitivity to changes in input parameter values for the impacts that were assessed--cumulative population health effects. This is because the impact that is assessed, long-term cumulative health effects, is buffered by the long time period analyzed and the cumulative nature of the output. Changing an input parameter may have a short-term effect, but over the long term, the results will tend to change very little. Input parameters that were found to be relatively sensitive were the parameters which affected infiltration through the trench cap and leaching out of the trench. In addition, other input parameters were found to be sensitive when evaluating local population health effects or short-term impacts.

(B) PRESTO-EPA-CPG

In a similar manner to the PRESTO-EPA-POP analysis, the sensitivity analysis results for PRESTO-EPA-CPG are more easily understood if the general base case analysis results are first reviewed. Unlike the PRESTO-EPA-POP code, the health impact that is assessed by the PRESTO-EPA-CPG code is not cumulative population health effects, but the maximum annual dose to a critical population group located close to the disposal site. The peak doses to the CPG and the pathways by which they occur vary considerably over the three hydrogeologic and climatic regions. Therefore, the base case results for PRESTO-EPA-CPG are also broken down into the three general settings.

At the humid permeable site, the maximum dose rate occurs relatively quickly from the groundwater pathway. The important nuclides are those with high mobility (low K_d values), such as H-3, C-14, and I-129. They can reach the CPG very quickly when combined with permeable soil characteristics and relatively high groundwater velocities.

At the humid impermeable site, the maximum dose rate occurs soon after failure of the trench cap (assumed to occur in year 100 for our "standard" scenario) via trench overflow directly to the surface water pathway. The important nuclides are those that are relatively mobile and have longer half-lives. An example is I-129, which reaches the CPG soon after the trench cap fails. It leaves the trench via overflow and is transported directly to the local stream by surface water, thus bypassing the greater retardation its higher K_d might afford if it had moved through groundwater. Nuclides with shorter half-lives, such as H-3, will not contribute high doses due to their decay during the period the trench cover remains intact.

At the arid permeable site, an initial peak in the CPG dose rate occurs in the first year after site closure due to atmospheric transport of less-mobile, high-dose nuclides, such as Co-60 and Cs-137, spilled onto the surface soil during site operations. This peak is relatively small, however, since only a fraction of the total activity brought onto the site is assumed to have been spilled during operations and even less reaches the downwind population after dilution and dispersion by atmospheric transport.

A much greater peak can occur through the groundwater pathway, although not until much later in the analysis and even after the 1,000 year modeling period for many scenarios. This later peak, if it occurs, would be significantly larger and would be dominated by mobile nuclides with relatively long half-lives, such as C-14 and I-129.

Based on the above general results, we can identify those parameters which are most sensitive in causing the largest changes in PRESTO-EPA-CPG results. In general, the parameters found to be sensitive in the PRESTO-EPA-POP code were also sensitive parameters in the PRESTO-EPA-CPG code.

At the humid permeable site, the maximum dose rate to the CPG is most sensitive to parameters that have an effect on: the amount of water infiltrating into the trench, such as the percentage of trench cap failure and the trench cover permeability and porosity; the rate at which radionuclide contaminated leachate leaves the waste matrix and then the trench, such as waste container related parameters, duration of institutional control, and nuclide specific release fractions and distribution coefficients; and radionuclide transit time in groundwater, such as the distance from the trench to the aquifer and the well.

At the humid impermeable site, the maximum dose rate to the CPG is most sensitive to parameters that affect the release to the surface water system and transit time of mobile and relatively long-lived nuclides, such as the percentage of trench cap failure, waste container related parameters, and the nuclide specific release fractions.

At the arid permeable site, the maximum dose to the CPG is most sensitive to parameters that modify groundwater transport characteristics, such as increasing the amount of trench cap failure, decreasing the trench-to-aquifer distance, or increasing the aquifer flow rate. In addition, the spillage fraction and atmospheric pathway parameters are very sensitive for the scenarios where short-term, atmospheric pathway doses dominate.

In summary, the PRESTO-EPA-CPG code exhibits greater relative sensitivity to changes in input parameter values than does the PRESTO-EPA-POP code. This is because the impact that is assessed, maximum annual dose to the CPG, is sensitive to small changes due to the model's assessing peak doses over short time periods to individuals close to the disposal site. Because the model is evaluating maximum doses relatively soon after disposal, sensitive parameters are those that affect leaching and transport of highly mobile, short-lived nuclides, such as H-3. In addition, the maximum dose will be very sensitive to the source term and release of the mobile radionuclides. In general, in a similar manner to the PRESTO-EPA-POP code, the most sensitive parameters will be those affecting infiltration through the trench cap, leaching out of the trench, and transport to the CPG.

(D) PRESTO-EPA-DEEP

The PRESTO-EPA-DEEP model is based on and is very similar to the PRESTO-EPA-POP code. The impact that is assessed, cumulative population health effects, is the same and, in general, the sensitivity results are similar. The major exceptions are that the PRESTO-EPA-DEEP code models a pathway of vertical water movement from a lower aquifer, through the waste, to an upper aquifer. This pathway is very significant, so input parameters associated with this pathway, such as vertical water velocity, density of confining stratum, and distribution coefficients (K_d) for the vertical zone were found to be sensitive. In addition, since deep disposal assumes solidified waste, the assumed waste release fraction is a very sensitive input parameter. Finally, unlike the shallow disposal options, infiltration through the trench cap is not applicable. Otherwise, input parameter sensitivity is the same as for the PRESTO-EPA-POP code.

(E) PRESTO-EPA-BRC

The PRESTO-EPA-BRC model is also based on and very similar to the PRESTO-EPA-POP code. The major difference is that exposure to on-site workers and visitors from direct gamma exposure and dust inhalation is included, as well as an incineration pathway to the general public. Because the number of on-site workers and visitors is small compared to the total number of persons affected over 10,000 years, parameters affecting this exposure pathway are not sensitive in changing overall impact. When assessing exposures from the incineration pathway, however, H-3 and C-14 become large contributors and the assumed volatilization fraction for these nuclides is found to be very sensitive. Also, because of the importance of the atmospheric pathway for the arid sites, the fraction of surface spillage is sensitive at these sites. Otherwise, the sensitivity of the input parameters are similar to those found for the PRESTO-EPA-POP model.

(F) Health Effect Conversion Factor (HECF)

The health effect conversion factor is used to determine cumulative population health effects to the regional basin popultion in the PRESTO-EPA-POP, PRESTO-EPA-DEEP, and PRESTO-EPA-BRC models. In many cases, the health effects to the regional basin population dominate. Therefore, it was felt to be important to test this parameter in some detail. This section summarizes the sensitivity analysis and results on the HECF. For a more detailed discussion, see the separate EPA technical report on sensitivity analysis (EPA88).

A number of assumptions associated with the calculation of the HECF values were tested, including: the assumed fraction of water useage, the amount of food and water consumption, the river-flow-to-population-ratio, and fish consumption and bioaccumulation rates. How these values are used in the calculation of the HECF is discussed in some detail in Chapter 8.

The results of the analysis on the HECF show that the sensitivity varies depending upon the nuclide, as the HECF values are nuclide dependent. Since the majority of population health effects are attributable to C-14 and I-129, the sensitivity results for these nuclides are the most important. A few parameters, however, such as the river-flow-to-population ratio, will affect the HECF independently of specific nuclides.

Because PRESTO-EPA output values are used to determine the HECF values, changing PRESTO-EPA input parameters can affect the values of the HECF. The basic parameters being determined from the PRESTO-EPA output is the number of health effects per unit of activity removed from a well or stream and the amount of water used by the local community. Therefore, the input parameters which will most affect the HECF values are those related to water usage. The input parameter UWAT, which is the assumed consumption of drinking water, is the most sensitive of the PRESTO-EPA input parameters in regard to the HECF values. Even this parameter, however, does not cause a large change in the HECF values. In general, changing PRESTO-EPA input parameter values will not affect the HECF values greatly.

The calculation of the HECF values also requires some parameters which are independent of PRESTO-EPA. These inlcude the river-flow-to-population ratio and, for the fish pathway, fish bioaccumlation factors and fish consumption rates. These parameters, which affect the HECF values directly, are generally more sensitive to changes than are those which are used indirectly through the PRESTO-EPA code. In fact, the HECF values are very sensitive to the river-flow-to-population ratio, the fish consumption rate, and the fish bioaccumulation factor, with changes in these parameters causing proportional changes in the HECF values for many nuclides.

The river-flow-to-population ratio (3000 m^3/person-yr) is used to calculate both the health effects from water usage and the health effects from fish consumption (see Chapter 8). The value used will, therefore, affect all components of the HECF calcuation and will affect them directly. The fish consumption rate and bioaccumulation factors will affect only the fish component of the HECF calculation. The importance of the fish pathway in the HECF values varies by nuclide, but for the most important nuclide for cumulative health effects, which is C-14, the fish pathway contributes over 95% of the total health effects. This shows that the HECF values for the most important nuclides and, therefore, the population health effects in general, will be very sensitive to the values used for the river-flow-to-population ratio, the annual consumption rate for fish, and the fish bioaccumulation factor for C-14.

In summary, the HECF values are calculated using PRESTO-EPA related parameters and direct input parameters. The PRESTO-EPA related parameters will not, in general, affect HECF values greatly, although of the PRESTO-EPA input parameters, the most sensitive will be the human water consumption rate. The direct input parameters, river-flow-to-population ratio, fish consumption rate, and fish bioaccumulation factors (C-14 especially) are very sensitive in affecting the HECF values and, therefore, the cumulative population health effects. It should be noted, however, that the HECF is not used in calculating CPG dose, so these conclusions are not applicable to the PRESTO-EPA-CPG model or to CPG doses.

(G) PATHRAE-EPA

The PATHRAE-EPA code was analyzed separately (Sh87a). The results of this analysis showed that sensitivity was comparable to that of the PRESTO-EPA-CPG code with which PATHRAE-EPA is very similar. The sensitivity rankings for the PATHRAE-EPA input parameters tested indicate that a number of the parameters exhibit moderate to high sensitivity. However, the overall importance of the majority of these parameters was judged small owing to the fact that the dose projections of the affected exposure pathways were negligible to all parameter values tested. The parameters judged to be significant pertained to hydrogeologic characteristics at the humid permeable site and disposal facility characteristics at all three sites, including facility area, waste and cover thickness, and operational period.

11.2.3 Summary and Conclusions of Single Parameter Sensitivity Analysis

Single parameter sensitivity analyses were performed on each of the PRESTO-EPA codes. The testing concentrated on the PRESTO-EPA-POP and PRESTO-EPA-CPG codes, as these were the main codes used in the LLW analysis and were the basis for the other codes. Because of the importance of the HECF values in calculating cumulative population health effects, parameters which affected the HECF were also analyzed. The PRESTO-EPA-DEEP and PRESTO-EPA-BRC codes were evaluated to a lesser degree. The PATHRAE-EPA code was evaluated in a separate analysis (Sh87a).

In conducting the sensitivity analysis program, over 100 input parameters were evaluated and over 200 separate sensitivity tests performed. The input parameters tested were related to waste form and composition, hydrogeologic conditions at the disposal site, and engineering barriers. PRESTO-EPA parameters related to the calculation of the HECF values were evaluated, as well as parameters independent of PRESTO-EPA which were used directly in calculating the HECF values. All three hydrogeologic/climatic sites were included in the analyses.

The main conclusions from the single parameter sensitivity analyses were:

- Single parameter sensitivity analysis allows for the identification of those parameters which have the greatest impact on model results; in addition, it is useful for checking that the code performs in a logical and consistent manner.

- The identification of sensitive input parameters allows for more efficient use of limited resources in better characterizing those parameters which would most affect model output. In addition, sensitive input parameters can be flagged for more thorough review when checking input lists for accuracy prior to production runs. Also, identification of the most sensitive input parameters allows for the evaluation of some degree of the uncertainty in model output, based upon knowledge of the uncertainty associated with the most sensitive input parameters.

- In general, the PRESTO-EPA-CPG and PATHRAE-EPA codes showed much greater sensitivity than the PRESTO-EPA-POP, PRESTO-EPA-DEEP, and PRESTO-EPA-BRC codes to an equivalent change in PRESTO-EPA input values. This is because the PRESTO-EPA-CPG and PATHRAE-EPA codes estimate the maximum annual dose to a nearby population group, whereas the other codes evaluate long-term cumulative population health effects to local and regional basin populations, with intakes and exposures averaged over the entire period of interest.

- Based on quantitative measures of sensitivity, a qualitative sensitivity rating was given to each parameter tested and the most sensitive input parameters associated with the various codes were identified. These parameters are listed in Table 11-3. The other parameters tested were found to have either low or no sensitivity.

- The sensitivity results showed, in general, that model results were most sensitive to PRESTO-EPA input parameters which affected the infiltration through the trench cover and the leaching out of the trench. In addition, the parameters associated with the PRESTO-EPA-CPG groundwater movement and uptake, the PRESTO-EPA-DEEP vertical water movement, and the PRESTO-EPA-BRC volitalization factors were very sensitive.

- The results of the single parameter sensitivity analysis seem to imply that for the PRESTO-EPA model, the most effort should be placed in better characterizing input parameter values and transport processes having to do with infiltration through the trench cap and leaching out of the trench. Furthermore, this implication might be carried on to the actual disposal sites, as suggesting that reducing infiltration through the trench cap and leaching out of the trench would be an effective means of reducing health impact from the disposal of LLW.

- Based on the analysis of the HECF calculations, it was determined that PRESTO-EPA related parameters will not, in general, affect HECF values greatly. The input parameters which are used directly to calculate the HECF values, river-flow-to-population ratio, annual fish consumption rate, and river-to-fish bioaccumulation factors (especially for C-14), are very sensitive in affecting the HECF values and, therefore, cumulative population health effects. In determining cumulative population health effects, these input parameter values should be evaluated very carefully. It should be noted, however, that HECF values are not used in calculating the maximum annual CPG dose.

- Although the population health effects analyses were carried out to 10,000 years, the majority of the impacts in most scenarios occur before year 1,000. Therefore, a program of single parameter sensitivity analyses can be used to evaluate changes to the "disposal system," which can reduce impacts in the first several hundred years. These evaluations, along with the results from the PRESTO-EPA-CPG analyses, may be useful in deciding among several management or disposal alternatives.

11.3 Scenario Sensitivity Analyses

In developing the LLW Standard, it was necessary to assess the health impacts that would result from the disposal of LLW under various assumed scenarios. These scenarios, which reflect a broad range of disposal options, include assumptions on the disposal sites, disposal methods, waste form, waste volume, and regional waste mix. Because there was such a large number of possible scenario combinations that could have been assessed, we felt that it was necessary to choose a limited number for our basic analyses. These basic analyses, which are called the base case scenarios, are discussed in Chapters 9 and 10.

Additional analyses were also performed where the basic scenarios were changed to see how different assumptions would affect the results. These tests were conducted as part of our sensitivity analysis program as scenario sensitivity analyses and are described in this section.

11.3.1 Methodology of Scenario Sensitivity Analyses

The two basic codes that we tested (by varying input parameters associated with a particular scenario variable we wanted to evaluate the sensitivity of) were PRESTO-EPA-POP, which is used to estimate long-term population health effects, and PRESTO-EPA-CPG, which is used to estimate maximum annual doses to a nearby CPG. Since the other codes are based on and are very similar to these two, scenario sensitivity analyses were not performed on the other PRESTO-EPA codes.

As part of the basic economic and cost-benefit analysis, a total of 93 runs were performed with the PRESTO-EPA-POP and PRESTO-EPA-CPG codes. These runs were evenly distributed over the three standard site locations. Among these runs were the 21 base case scenarios (seven for each site location) described in Chapter 9. The additional runs were assessed as part of the scenario sensitivity analyses. Tables 11-4, 11-5, and 11-6 list all of the 93 runs performed, by site location, with the base case scenarios identified. These tables list each run by a scenario number and include information on the disposal method, waste form, and waste volume. Acronyms were used and are described in a key. Also included are the PRESTO-EPA-CPG and PRESTO-EPA-POP results, in terms of maximum annual dose and long-term population health effects, respectively. Additional runs, not listed in Tables 11-4, 11-5, and 11-6, were performed to test specific scenario assumptions. These scenario sensitivity analyses and their results will be described in later sections.

Certain scenario variables, such as site location, waste form, regional waste mix, or site size, can affect the output results. We felt it was important to determine how sensitive the results from the base case scenarios were to changes to the variables associated with the assumed scenarios. In the following sections, the analyses associated with certain variables and the results of those analyses are described.

Table 11-4. Listing of PRESTO-EPA runs performed for a humid permeable site, with associated maximum annual dose and cumulative population health effects

SCENARIO NUMBER	Disposal method			Waste forms			As gen volume ($1000\ m^3$)	CPG dose in peak year (mrem/yr)	Population Health Effects (cancer deaths)*	Comments
	Class A	Class B	Class C,D,N	Class A	Class B	Class C,D,N				
1●	SLD	SLD	SLD	AS IS	AS IS	AS IS	250	35	4.7	Base Case Run
4●	SLD	SLD	ISD	AS IS	SOL	SOL	250	9.2	3.9	Base Case Run
7●	SLF	SLF	SLF	AS IS	AS IS	AS IS	250	62	6.2	Base Case Run
10●	ISD	ISD	ISD	AS IS	SOL	SOL	250	5.1	3.4	Base Case Run
13●	IDD	IDD	IDD	AS IS	SOL	SOL	250	5.0	3.4	Base Case Run
16	EM	CB	CB	GR	SOL	SOL	250	2.0	2.0	EMCB Disposal
19●	CC	CC	CC	SOL	SOL	SOL	250	1.3	1.7	Base Case Run
23	DWI	DWI	DWI	AS IS	AS IS	AS IS	6.72	7.3	0.036	Deep Disposal
26	SLD	SLD	SLD	INCIN/SOL	INCIN/SOL	INCIN/SOL	250	13	3.1	Incineration
29	SLD	SLD	ISD	INCIN/SOL	INCIN/SOL	INCIN/SOL	250	13	3.1	Incineration
32	SLD	SLD	ISD	AS IS	HIC	SOL	250	82	4.6	Use of HIC
35	SLD	SLD	ISD	HIC	HIC	HIC	250	40	4.5	Use of HIC
38●	SLD	SLD	SLD	AS IS	SOL	SOL	250	9.1	3.9	Base Case Run
41	SLD	ISD	ISD	AS IS	AS IS	AS IS	250	44	4.6	Waste as is
44	SLD	SLD	ISD	AS IS	AS IS	AS IS	250	35	4.7	Waste as is
47	SLD	SLD	ISD	AS IS	SOL	SOL	373	8.6	4.4	Larger Waste Volume
50	SLD	SLD	ISD	AS IS	SOL	SOL	250	7.0	2.9	LLW + NARM only
53	SLD	SLD	ISD	AS IS	SOL	SOL	170	5.7	2.0	Smaller Waste Volume
56	SLD	SLD	ISD	AS IS	SOL	SOL	366	8.6	4.3	Larger Waste Volume
59	SLD	SLD	ISD	AS IS	SOL	SOL	250	7.0	3.0	LLW Only
62	SLD	SLD	ISD	AS IS	SOL	SOL	170	5.7	2.0	Smaller Waste Volume
65	SLD	SLD	SLD	AS IS	AS IS	AS IS	249	35	4.7	LLW-BRC+NARM-Class D
68	SLD	SLD	ISD	AS IS	SOL	SOL	249	9.1	3.9	LLW-BRC+NARM-Class D
71	SLD	SLD	SLD	AS IS	AS IS	AS IS	500	48	9.4	Larger Waste Volume
74	SLD	SLD	SLD	AS IS	AS IS	AS IS	100	22	1.9	Smaller Waste Volume
77	SLD	SLD	ISD	AS IS	SOL	SOL	500	13	7.8	Larger Waste Volume
80	SLD	SLD	ISD	AS IS	SOL	SOL	100	5.6	1.6	Smaller Waste Volume
95	SLD	SLD	SLD	AS IS	AS IS	AS IS	250	37	2.8	Regional Compact
98	SLD	SLD	ISD	AS IS	SOL	SOL	250	9.7	2.4	Regional Compact
101	SLD	SLD	ISD	AS IS	SOL	SOL	590	15	5.6	Regional Compact
103	SLD	SLD	ISD	AS IS	SOL	SOL	250	9.2	3.9	10,000 Year CPG Run

● Base case scenarios (see Chapter 9, Figures 9-2 and 9-3).

*Cumulative Population Health Effects Over 10,000 years (cancer deaths only).

Table 11-5. Listing of PRESTO-EPA runs performed for a humid impermeable site, with associated maximum annual dose and cumulative population health effects

SCENARIO NUMBER	Disposal method			Waste forms			As gen volume ($1000 m^3$)	CPG dose in peak year (mrem/yr)	Population Health Effects (cancer deaths)*	Comments
	Class A	Class B	Class C,D,N	Class A	Class B	Class C,D,N				
2●	SLD	SLD	SLD	AS IS	AS IS	AS IS	250	0.13	8.1	Base Case Run
5●	SLD	SLD	ISD	AS IS	SOL	SOL	250	0.03	2.2	Base Case Run
8●	SLF	SLF	SLF	AS IS	AS IS	AS IS	250	0.77	39	Base Case Run
11●	ISD	ISD	ISD	AS IS	SOL	SOL	250	0.012	1.1	Base Case Run
14●	IDD	IDD	IDD	AS IS	SOL	SOL	250	9.3E-03	0.85	Base Case Run
17	EM	CB	CB	GR	SOL	SOL	250	1.9E-03	0.64	EMCB Disposal
20●	CC	CC	CC	SOL	SOL	SOL	250	1.4E-03	0.26	Base Case Run
22	HF	HF	HF	SOL	SOL	SOL	15.2	1.7E-04	0.007	Deep Disposal
27	SLD	SLD	SLD	INCIN/SOL	INCIN/SOL	INCIN/SOL	250	0.03	6.4	Incineration
30	SLD	SLD	ISD	INCIN/SOL	INCIN/SOL	INCIN/SOL	250	0.025	3.4	Incineration
33	SLD	SLD	ISD	AS IS	HIC	SOL	250	0.12	3.8	Use of HIC
36	SLD	SLD	ISD	HIC	HIC	HIC	250	0.13	6.1	Use of HIC
39●	SLD	SLD	SLD	AS IS	SOL	SOL	250	0.03	2.2	Base Case Run
42	SLD	ISD	ISD	AS IS	AS IS	AS IS	250	0.048	3.9	Waste as is
45	SLD	SLD	ISD	AS IS	AS IS	AS IS	250	0.12	4.6	Waste as is
48	SLD	SLD	ISD	AS IS	SOL	SOL	373	0.034	2.5	Larger Waste Volume
51	SLD	SLD	ISD	AS IS	SOL	SOL	250	0.023	1.7	LLW+NARM only
54	SLD	SLD	ISD	AS IS	SOL	SOL	170	0.015	1.1	Smaller Waste Volume
57	SLD	SLD	ISD	AS IS	SOL	SOL	366	0.033	2.5	Larger Waste Volume
60	SLD	SLD	ISD	AS IS	SOL	SOL	250	0.023	1.7	LLW only
63	SLD	SLD	ISD	AS IS	SOL	SOL	170	0.015	1.2	Smaller Waste Volume
66	SLD	SLD	SLD	AS IS	AS IS	AS IS	249	0.12	3.6	LLW-BRC+NARM-Class D
69	SLD	SLD	ISD	AS IS	SOL	SOL	249	0.030	2.2	LLW-BRC+NARM-Class D
72	SLD	SLD	SLD	AS IS	AS IS	AS IS	500	0.25	16	Larger Waste Volume
75	SLD	SLD	SLD	AS IS	AS IS	AS IS	100	0.05	3.2	Smaller Waste Volume
78	SLD	SLD	ISD	AS IS	SOL	SOL	500	0.060	4.5	Larger Waste Volume
81	SLD	SLD	ISD	AS IS	SOL	SOL	100	0.012	0.9	Smaller Waste Volume
96	SLD	SLD	SLD	AS IS	AS IS	AS IS	250	0.15	12	Regional Compact
99	SLD	SLD	ISD	AS IS	SOL	SOL	250	0.031	3.8	Regional Compact
102	SLD	SLD	ISD	AS IS	SOL	SOL	470	0.058	7.1	Regional Compact
104	SLD	SLD	ISD	AS IS	SOL	SOL	250	0.032	2.2	10,000 year CPG Run

● Base case scenarios (see Chapter 9, Figures 9-2 and 9-3).

*Cumulative Population Health Effects over 10,000 years (cancer deaths only).

Table 11-6. Listing of PRESTO-EPA runs performed for an arid permeable site, with associated maximum annual dose and cumulative population health effects

SCENARIO NUMBER	Disposal method: Class A	Disposal method: Class B	Disposal method: Class C,D,N	Waste forms: Class A	Waste forms: Class B	Waste forms: Class C,D,N	As gen volume (1000 m^3)	CPG dose in peak year (mrem/yr)	Population Health Effects (cancer deaths)*	Comments
3●	SLD	SLD	SLD	AS IS	AS IS	AS IS	250	2.2E-03	3.6	Base Case Run
6●	SLD	SLD	ISD	AS IS	SOL	SOL	250	9.2E-04	2.3	Base Case Run
9●	SLF	SLF	SLF	AS IS	AS IS	AS IS	250	0.41	3.9	Base Case Run
12●	ISD	ISD	ISD	AS IS	SOL	SOL	250	4.4E-05	1.7	Base Case Run
15●	IDD	IDD	IDD	AS IS	SOL	SOL	250	3.7E-05	1.2	Base Case Run
18	EM	CB	CB	GR	SOL	SOL	250	3.1E-06	0.69	EMCB Disposal
21●	CC	CC	CC	SOL	SOL	SOL	250	0	0.31	Base Case Run
24	DGD	DGD	DGD	SOL	SOL	SOL	118.5	0	0.047	Deep Disposal
25	DGD	DGD	DGD	SOL	SOL	SOL	250	0	0.023	Deep Disposal
28	SLD	SLD	SLD	INCIN/SOL	INCIN/SOL	INCIN/SOL	250	3.0E-04	1.4	Incineration
31	SLD	SLD	ISD	INCIN/SOL	INCIN/SOL	INCIN/SOL	250	3.0E-04	1.4	Incineration
34	SLD	SLD	ISD	AS IS	HIC	SOL	250	3.9E-03	3.5	Use of HIC
37	SLD	SLD	ISD	HIC	HIC	HIC	250	1.9E-03	3.5	Use of HIC
40●	SLD	SLD	SLD	AS IS	SOL	SOL	250	9.2E-04	2.3	Base Case Run
43	SLD	ISD	ISD	AS IS	AS IS	AS IS	250	1.4E-03	3.3	Waste as is
46	SLD	SLD	ISD	AS IS	AS IS	AS IS	250	2.2E-03	3.6	Waste as is
49	SLD	SLD	ISD	AS IS	SOL	SOL	373	8.4E-04	2.6	Larger Waste Volume
52	SLD	SLD	ISD	AS IS	SOL	SOL	250	6.8E-04	1.7	LLW+NARM only
55	SLD	SLD	ISD	AS IS	SOL	SOL	170	5.5E-04	1.2	Smaller Waste Volume
58	SLD	SLD	ISD	AS IS	SOL	SOL	366	8.4E-04	2.5	Larger Waste Volume
61	SLD	SLD	ISD	AS IS	SOL	SOL	250	6.8E-04	1.7	LLW only
64	SLD	SLD	ISD	AS IS	SOL	SOL	170	5.5E-04	1.2	Smaller Waste Volume
67	SLD	SLD	SLD	AS IS	AS IS	AS IS	249	2.2E-03	3.5	LLW-BRC+NARM-Class D
70	SLD	SLD	ISD	AS IS	SOL	SOL	249	9.2E-04	2.3	LLW-BRC+NARM-Class D
73	SLD	SLD	SLD	AS IS	AS IS	AS IS	500	3.1E-03	7.1	Larger Waste Volume
76	SLD	SLD	SLD	AS IS	AS IS	AS IS	100	1.3E-03	1.4	Smaller Waste Volume
79	SLD	SLD	ISD	AS IS	SOL	SOL	500	1.3E-03	4.6	Larger Waste Volume
82	SLD	SLD	ISD	AS IS	SOL	SOL	100	5.6E-04	0.92	Smaller Waste Volume
97	SLD	SLD	SLD	AS IS	AS IS	AS IS	250	1.5E-03	2.5	Regional Compact
100	SLD	SLD	ISD	AS IS	SOL	SOL	250	1.0E-03	2.1	Regional Compact
105	SLD	SLD	ISD	AS IS	SOL	SOL	250	1.7E-02	2.3	10,000 Year CPG run

● Base case scenarios (see Chapter 9, Figures 9-2 and 9-3).

*Cumulative Population Health Effects over 10,000 years (cancer deaths only).

Key to Tables 11-4, 11-5, and 11-6

Disposal Method

SLF	-	sanitary landfill
SLD	-	conventional shallow land disposal**
ISD	-	improved shallow land disposal**
IDD	-	intermediate depth disposal
EM	-	earth-mounded disposal***
CB	-	concrete bunker disposal***
CC	-	concrete canister disposal
DWI	-	deep well injection disposal
HF	-	hydrofracture disposal
DGD	-	deep geologic disposal

Waste Form

As Is	-	no treatment, disposed of as generated
SOL	-	solidified
INCIN/SOL	-	incinerated and then the ash is solidified
HIC	-	placed in a high-integrity container
GR	-	supercompacted and grouted

Waste Class

A	-	NRC definition (10 CFR 61)
B	-	NRC definition (10 CFR 61)
C	-	NRC definition (10 CFR 61)
D	-	greater than Class C AEA waste (An EPA designation for the purpose of this analysis)
N	-	naturally occurring and accelerator-produced waste (NARM)
•	-	base case scenarios (see Chapter 9)

* Cumulative population health effect values are for cancers only and do not include the serious genetic effects. The addition of those effects will, in general, increase the total health estimates by about 10%.

** SLD-ISD - combination is equivalent to 10 CFR 61 disposal (see Chapter 4)

***EMCB - combination is earth-mounded concrete bunker disposal (see Chapter 4)

(A) Site Location

As noted in Chapter 5, three generic locations (humid permeable, humid impermeable, and arid permeable) are used to reflect the range of disposal sites in the United States. Since the disposal site is considered a critical protection factor, all base case assessments were made for each of the three locations. In addition, with the exception of the three deep disposal methods and the eight regional compact scenarios, the scenarios listed in Tables 11-4, 11-5, and 11-6 are spread evenly over the three sites. The deep disposal methods not assessed for all three sites are hydrofracture, deep well injection, and deep geological disposal. These methods are costly and not necessarily suitable for all wastes or locations. The regional compact scenarios preclude certain site characteristics, i.e., some regions have no arid sites.

The humid impermeable location is evaluated using 31 scenarios, including one scenario for the hydrofracture deep disposal method and three scenarios to evaluate a northeast compact waste mix. The humid permeable location is evaluated using 31 scenarios, including one scenario for the deep well injection deep disposal method and three scenarios to evaluate a southeast compact waste mix. The arid permeable location is evaluated using 31 scenarios, including two scenarios for deep geological disposal and two scenarios to evaluate a Rocky Mountain compact waste mix.

In the scenarios described above, it was assumed that the humid permeable site was located in the southeast and the humid impermeable in the northeast. A set of scenario sensitivity analyses were performed to evaluate a humid permeable site located in the northeast and a humid impermeable site located in the southeast. In addition, scenario sensitivity analyses were performed to evaluate an arid site which is impermeable. The results of these analyses are discussed in a later section.

(B) Disposal Methods

As discussed in Chapter 4, nine disposal methods were analyzed (SLF, SLD, ISD, IDD, EMCB, CC, DWI, HF, and DGD (see Key to Tables 11-4, 11-5, and 11-6 for explanation of these acronyms). In the scenario analyses there is at least one assessment for each disposal method. Each disposal method includes all of the waste that can be suitably disposed of by that method. Some of the assessments, however, involve combinations of different disposal methods. For example, the disposal method (Scenarios 4, 5, and 6) that most closely resembles existing NRC rules requires shallow land disposal (SLD) for Class A and B wastes and improved shallow land disposal (ISD) for Class C waste.

In most of the scenarios, Class A waste and Class B waste are disposed of by the SLD method, with Class C waste and NARM waste disposed of by the ISD method. This is consistent with existing NRC rules (10 CFR Part 61) and reflects the policy that greater isolation should be required for the more hazardous wastes.

The sanitary landfill (SLF), earth-mounded concrete bunker (EMCB), intermediate depth disposal (IDD), and concrete canister (CC) methods have three assessments each, one for each site. The deep geological disposal (DGD) method has two assessments, both for the arid permeable location, one for a normal volume and one for a smaller volume. The hydrofracture (HF) and deep well injection (DWI) methods have one assessment each, for humid impermeable and humid permeable sites, respectively.

(C) Waste Forms

The form the waste is in when placed in the disposal site is a variable which was analyzed. As part of the scenario sensitivity analyses, waste forms which were analyzed included, "as is", solidified, placed within high-integrity containers, and special. The "as is" form means that the waste is left as generated or unsolidified and includes trash, adsorbing waste, and activated metal. The special form includes wastes that are suitable for disposal by the hydrofracture, deep well injection, or deep geological disposal methods.

Waste that is placed in high integrity containers is done so in the "as is" form. For six scenarios, all of the waste is incinerated and then the ash is solidified.

The waste form parallels the waste disposal method in most cases, conforming to the requirements of 10 CFR Part 61 that the more hazardous waste should receive greater isolation from the environment. Thus, most scenarios place the Class A waste (the least hazardous) in the "as is" form, while Class B, Class C, and NARM waste are generally in the solidified form.

(D) Waste Mixes (Regional Compacts)

The waste mix reflects the relative volume of each waste stream which makes up the total waste volume that is disposed of at a site. The United States average waste mix is used in the base case analyses and the majority of the other scenarios. There are, however, eight scenarios which reflect the waste mixes in three geographical regions: Southeast, Northeast, and Rocky Mountain. There are three Southeast waste mix scenarios, three Northeast, and two Rocky Mountain. Waste forms and disposal methods are adjusted in six scenarios. Waste volumes are varied in two scenarios.

These scenarios are used to estimate the potential effects that the Low-Level Waste Policy Act will have on LLW disposal. In particular, waste volumes can be expected to vary significantly by region. For more detail on the differences between the U.S. average and the regional waste mixes, see Appendix A of the EIA.

(E) Waste Groups

For this analysis, wastes were grouped into five broad classes. The classes represent: AEA-regulated low-level waste (LLW class), the waste

that would be considered a candidate for less restrictive disposal (BRC class), LLW not covered under the AEA (NARM class), wastes whose activities are greater than the NRC's Class C (designated by EPA as Class D for the purpose of this study), and special waste. NARM wastes are represented by the R-RASOURC and R-RAIXRSN waste streams. Class D wastes are represented by a select set of Class C wastes (N-SOURCES, R-RASOURC, and L-DECONRS) for scenarios 65-70 only (See Tables 11-4, 11-5, 11-6). These particular waste streams are, on average, Class C wastes. However, they contain one or more substreams whose activity concentrations are greater than the NRC's Class C specifications in 10 CFR Part 61. Special wastes include those suitable for disposal by hydrofracturing, deep well injection, or deep geological methods. These waste groups are described in more detail in Chapter 3.

The base case analyses, as well as the majority of the other analyses use the "LLW-BRC+NARM" waste group. Additional scenarios that were analyzed include: LLW+NARM, LLW only, LLW-BRC+NARM-Class D, and special.

By comparing scenarios with and without the different groups, it is possible to determine the impacts that any one group contributes to the total. For example, the impact from disposal of NARM wastes can be obtained by subtracting the LLW impact from the LLW+NARM impact.

(F) Waste Volumes

A 250,000 cubic meter site is used for the base case scenarios. Other waste volumes are analyzed to assess the effect of varying the waste volume on disposal site impact, to address the lower volume special wastes, to address the larger volumes for two waste compacts, and to address the larger and smaller volumes when NARM, BRC, and Class D wastes are removed from the total. In addition, in order to evaluate the consequences of volume reduction, analyses were performed where the total activity disposed of was kept constant, while the site size was reduced.

(G) Time Horizon

The analysis period used for the base case and most of the other CPG assessments is 1,000 years, although 10,000 years was used for three scenarios. The additional three runs were performed in order to identify any peak CPG doses that might occur after 1,000 years.

In order to provide additional information on how the CPG doses vary over time, the annual CPG dose is plotted over the 1,000 year analysis period. This information is provided from PRESTO-EPA-CPG analyses using conventional shallow land disposal technology and 10 CFR Part 61 disposal technology. Each of the three hydrogeological sites is evaluated.

Cumulative population health effects, estimated using the PRESTO-EPA-POP model, are assessed for 10,000 years. In order to evaluate how these health effects accumulate over time, the model estimates cumulative health effects for shorter time periods, including 100, 500, and 1,000 years. This information is provided for a site using 10 CFR Part 61 disposal technology in all three hydrogeologic environments.

(H) Radionuclides

Although specific scenarios were not developed to analyze the sensitivity of various radionuclides, the unit response methodology employed for our analyses lends itself to such an assessment. Therefore, we evaluated the sensitivity of the disposal site impact to the various radionuclides included within the disposal site.

(I) Additional Analyses

A number of additional analyses were performed to answer specific questions about the scenarios. These included tests to evaluate the importance of a buffer zone on CPG dose by varying the distance from the trench to the well. Because global health effects were not assessed in the PRESTO-EPA analyses, a separate analysis was performed to evaluate the health effects to the global population associated with releases into the ocean. The results of these analyses are discussed in following sections.

Because of the importance of the volatility fraction on incineration doses, a set of analyses were performed to test this variable. These analyses are discussed, however, in Chapter 10, which relates to the BRC incineration scenarios.

11.3.2 Results and Discussion of Scenario Sensitivity Analyses

The results from the scenario assessments, including those listed in Tables 11-4, 11-5, and 11-6, can be organized into various groupings to evaluate the sensitivity of certain scenario assumptions. Test runs are described according to their scenario number, if applicable, and results are generally characterized by both maximum annual dose to the CPG and long-term population health effects. The results of the scenario sensitivity analyses are described in the following sections.

(A) Sensitivity to Disposal Site

The sensitivity of the results to the hydrogeology and climate of the disposal site can be observed by reviewing Tables 11-4, 11-5, and 11-6. The CPG dose estimates indicate significant differences in impact among the three sites. Maximum CPG dose comparisons for the different site locations are all evaluated assuming a well or stream located 100 meters from the edge of the waste disposal area (different buffer zone distances are evaluated as well, see Section 11.3.2(I)). The differences are best observed by comparing the scenarios for conventional shallow disposal (1, 2, 3), the 10 CFR Part 61 combination of conventional shallow and improved shallow land disposal specified by current NRC rules (4, 5, 6), and regulated sanitary landfill (7, 8, 9). The maximum annual doses at the humid impermeable location are approximately 100 times lower than at the humid permeable location. The maximum annual doses at the arid permeable location are approximately 10,000 times lower than at the humid permeable site for the SLD and 10 CFR Part 61 disposal technology scenarios and approximately 100 times lower than at the humid permeable site for the sanitary landfill scenario. This pattern generally holds for all comparable scenarios for the CPG assessments, and is due to both hydrogeology and climate.

At the humid permeable site, leachate travels via the groundwater pathway to a well, while at the humid impermeable site the major pathway is via surface water flow to a river. Maximum doses at the humid impermeable site are less than at the humid permeable due to the dilution that a river provides compared to consumption of contaminated groundwaters. Maximum doses are much less at the arid site due to the much smaller amount of water infiltrating the trench and the extremely long travel times in the arid environment.

The long-term, cumulative population health effects estimates do not demonstrate as specific a pattern regarding site location as do the maximum CPG doses. For SLF disposal (scenarios 7, 8, 9) and SLD disposal (1, 2, 3), the humid impermeable site has the greatest number of health effects, while for 10 CFR Part 61 disposal technology (4, 5, 6) and IDD disposal (13, 14, 15) the humid permeable site has the greatest number.

The reason for these results has to do with both the hydrogeology of the site and the form in which the waste is disposed. With the less sophisticated disposal methods, such as SLF and SLD, the waste is assumed to be disposed of in the as-generated form. This method provides minimal containment for the nuclides, so that when trenches overflow at the humid impermeable site due to the "bathtub effect," larger releases of mobile and non-mobile nuclides occur than would at the permeable site through groundwater transport processes. With 10 CFR Part 61 disposal technology and other greater confinement type disposal, the waste is assumed to be solidified or in high-integrity containers and trench covers do not fail as readily. Therefore, overflow does not take place and transport through impermeable soil leads to lower health effects than at the site with permeable soil.

While site characteristics and disposal technology causes the relationship between the maximum number of health effects for a particular scenario to vary between the two humid sites, long travel times cause the arid site to have consistently lower health effects estimates. However, in contrast to the maximum CPG doses, health effects estimates vary by a much smaller degree over the three sites, generally within a range of about 10 for the various scenarios.

Because of concerns related to the scenario assumption that a humid permeable site would be located in the Southeast and a humid impermeable site in the Northeast, an additional set of analyses were conducted to evaluate the hydrogeological differences between the humid permeable and humid impermeable site, separate from regional climatic and lifestyle differences. Also evaluated was an arid site with impermeable hydrogeology. All analyses were conducted using 10 CFR Part 61 disposal technology and are summarized in Table 11-7. The results are discussed in the following paragraphs and in more detail in reference Sh87b. When reviewing the results, it should be understood that there are many complex, interrelated input parameters associated with the hydrogeology of the sites and that these are limited analyses, with results that are informative but not conclusive.

Table 11-7. Summary of Varying Site Characteristics Using 10 CFR Part 61 Disposal Technology

Scenario[1]	Maximum CPG Dose (mrem/yr)	Year of max. Dose
Southeast (HP) Site[2]		
Standard HP Scenario	9.2	777
Soil Characteristics from HI Site[3]	8.6	390
Water usage from HI Site [4]	3.6	777
Soil and Water Usage from HI Site	12.0	196
Northeast (HI) Site[5]		
Standard HI Scenario	3.0E-2	191
Soil Characteristics from HP Site[6]	2.0E-6	1
Southwest (AP)Site[7]		
Standard AP Site	9.2E-4	1000
Soil Characteristics from HI Site[3]	1.7E-5	2

1. For a more complete description of scenarios and analysis, see Sh87b.

2. Humid Permeable (HP) Site is assumed to be located in Southeast.

3. For the following soil characteristic input parameters, values were used from the standard humid impermeable (HI) site: trench, trench cap, and sub-trench permeability; trench, sub-trench, and aquifer porosity; density of waste material; local soil infiltration rate; fraction of residual saturation; bulk density of soil; runoff fraction; porosity in gravity and pellicular zone; upward diffusivity; upward hydraulic conductivity; and pellicular and gravity infiltration capacity.

4. Water usage characteristics (i.e., fractional usage of stream and well water) values taken from humid impermeable (HI) site input parameters.

5. Humid Impermeable (HI) site is assumed to be located in Northeast.

6. Soil characteristic input parameter values (as listed in 3) of the standard humid permeable (HP) site were used.

7. Arid permeable (AP) site assumes an arid site located in the Southwest.

With standard Southeast site, humid permeable scenario input parameters, the maximum dose (9.2 mrem/yr) is due to ground water transport to a well. When soil characteristic parameters (see Table 11-7) from the humid impermeable site are used, the maximum dose goes down but occurs earlier. This seems somewhat contradictory, but is due to competing effects. The impermeable soil characteristics cause water to pool in the trench, leading to increased concentrations of nuclides in the leachate due to the increased contact time. Even though the leachate has greater nuclide concentrations, the impermeable soil causes a slower rate of travel to the well. With the normal scenario, C-14 reaches the well early in the analysis, but with a small peak, while I-129 takes longer to arrive at the well, but with the larger peak dose. In the test scenario, the greater concentration of the leachate causes the peaks to increase, but the slower rate of travel causes them to occur later. Therefore, the dose due to C-14 goes up due to its greater concentration in the leachate, but the dose from I-129 no longer shows up due to its arrival at the well after 1,000 years. In other words, doses from individual nuclides go up, but occur later. The nuclide contributing the maximum dose changes from I-129 in year 777, to C-14 in year 390.

When humid impermeable (HI) site water usage characteristics (i.e., fractional usage of well water decreases and stream water increases) are used the dose goes down due to the greater dilution afforded by the use of a surface water stream over an aquifer. However, when HI water usage is combined with HI soil characteristics, doses go up and occur earlier due to trench overflow and surface water usage combined with some groundwater flow and eventual discharge to the stream of mobile nuclides. It should be noted, however, that all doses are very similar for these scenarios.

When evaluating the Northeast HI site, doses are greatly reduced when humid permeable soil characteristics (see Table 11-7) are assumed. This is because the permeable soil does not allow trench overflow to occur, but groundwater travel times are still too slow to allow contamination to reach the well within 1,000 years. Therefore, the maximum dose occurs in year one, due to atmospheric transport of surface spillage.

Maximum doses are reduced at the arid permeable site when the HI soil characteristics (see Table 11-7) are used. This is due to groundwater transport being sufficiently slowed so that nuclides will not reach the well until after 1,000 years. This causes the maximum dose to occur in year two, due to atmospheric transport of surface spillage.

(B) Sensitivity to Disposal Method

The health impact from the disposal of LLW varies depending upon the disposal method used. In Figures 9-2 and 9-3 the sensitivity of health impact to a number of different disposal methods are shown and the results for these methods are discussed in Chapter 9. The analysis in Chapter 9 is based on specific base case disposal methods which do not include all of the disposal methods that were assessed. In our broader analysis, a total of nine disposal methods were considered, including SLF, SLD, ISD, IDD, EMCB, CC, HF, DWI, and DGD. The HF and DWI methods only apply to certain waste streams, however.

The results of the analysis of the nine disposal methods are displayed in Table 11-8, grouped by site location. These results show the sensitivity of using the various disposal methods at a single site. Three deep disposal assessments (22, 23, 24) consider smaller volumes and limited waste streams, as noted. Because of these differences, these three scenarios cannot be directly compared with the others.

The maximum annual CPG dose and long-term health effects are greatest from the sanitary landfill method at all sites. The concrete canister method leads to the lowest CPG dose and health effects at all sites, although the more rigorous disposal methods offer essentially an equivalent level of protection.

The difference in maximum annual CPG doses among the disposal methods varies depending on location. The range between the least protective and the most protective is about 50 times more protective for the humid permeable location, about 500 times more protective for the humid impermeable location, and at least five orders of magnitude for the arid permeable location.

The difference in long-term health effects among the disposal methods also varies depending on location, but to a lesser degree. The range between the least protective and the most protective methods is about four for the humid permeable location, about 150 times more protective for the humid impermeable location, and about 200 times more protective for the arid permeable location.

The above discussion is based on disposal methods where it is assumed that as the methods change the waste form changes as well. For example, when waste is disposed of under SLF methodology all waste is assumed to be as generated, while for IDD disposal it is assumed that only class A waste is disposed of as generated and Class B and C are solidified. To gain an idea of how CPG dose and population health effects vary when only the disposal method is changed, a review of scenarios 8, 2, 45 and 42 from the humid impermeable site are shown in Table 11-9, where all waste is disposed of as generated.

From these scenarios it can be seen that as the disposal methods become more rigorous, while the waste form remains the same, maximum CPG dose and population health effects are reduced. This is not as true, however, once the waste is in a solidified form, as can be seen in scenarios 11 and 14, where ISD disposal is compared to IDD disposal, both with Class B and C wastes solidified. Under those conditions the reduction in health impact is much less.

Table 11-8. Summary of maximum annual CPG doses and long-term health effects from nine disposal methods at different site locations

Scenario	Disposal method	CPG dose in peak year (mrem/yr)	Health effects over 10,000 years (fatal cancers only)
Humid Impermeable Location			
8	SLF	0.77	39
2	SLD	0.13	8.1
11	ISD	1.2E-02	1.1
14	IDD	9.3E-03	0.85
17	EMCB	1.9E-03	0.64
20	CC	1.4E-03	0.26
22*	HF	1.7E-04	0.007
Humid Permeable Location			
7	SLF	62	6.2
1	SLD	35	4.7
10	ISD	5.1	3.4
13	IDD	5.0	3.4
16	EMCB	2.0	2.0
19	CC	1.3	1.7
23*	DWI	7.3	0.036
Arid Permeable Location			
9	SLF	4.1E-01	3.9
3	SLD	2.2E-03	3.6
12	ISD	4.4E-05	1.7
15	IDD	3.7E-05	1.2
18	EMCB	3.1E-06	0.69
21	CC	0	0.31
25	DGD	0	0.023
24*	DGD	0	0.047

* 22 includes L-CONCLIQ, I-ABSLIQD, L-DECONRS, L-FSLUDGE, L-IXRESIN, and RAIXRSN; 23 includes L-CONCLIQ, I-ABSLIQD, and L-DECONRS; 24 includes L-IXRESIN and Class C waste.

Table 11-9. Summary of maximum annual CPG dose and long-term health effects when disposal methods vary but the waste form remains constant

Scenario[1]	Disposal Method[2] A	B	C	CPG Dose in Peak Year (mrem/yr)	Health Effects Over 10,000 years (fatal cancers only)
Waste "As Is"[3]					
8	SLF	SLF	SLF	0.77	39
2	SLD	SLD	SLD	0.13	8.1
45	SLD	SLD	ISD	0.12	4.6
42	SLD	ISD	ISD	0.04	3.9
Waste Solidified[4]					
11	ISD	ISD	ISD	0.012	1.1
14	IDD	IDD	IDD	0.009	0.9

1. All waste is disposed of at the humid impermeable site.

2. Class A, B, and C waste are disposed of as listed. See Key to Tables 11-4, 11-5, and 11-6, for a description of waste classes and disposal methods.

3. All waste disposed of in the "as generated" form.

4. Class A waste disposed of "as generated," while class B and C are disposed of in a solidified waste form.

(C) Sensitivity to Waste Forms

The sensitivity of the results to the waste forms analyzed is presented in Tables 11-10 and 11-11, grouped by site characteristics. Table 11-10 lists tests in which the SLD disposal method is used (past practice shallow land disposal). Table 11-11 lists tests in which the disposal method is according to 10 CFR Part 61 disposal technology, which require shallow land disposal (SLD) for Class A and B wastes and improved shallow disposal (ISD) for Class C wastes. Any NARM wastes are assigned the same disposal method as Class C wastes.

Regarding the sensitivity of the output to the waste form, a major result is that solidification appears to provide four or five times the protection that unsolidified waste forms provide. A secondary finding is that high-integrity waste canisters provide no additional population health effects protection when they are filled with wastes in the "as is" form, due to the long time periods analyzed. In addition, when evaluating CPG doses, maximum doses actually may go up about 10 percent since releases occur quickly once containers fail causing a larger peak dose. Incineration of the waste prior to solidification causes little additional reduction in health impact.

(D) Sensitivity to Regional Waste Mixes

The impact from three different waste mixes is compared to the base case scenarios (1, 2, 3, and 4, 5, 6) in Table 11-12. There appears to be little significant difference in impact from regional waste mixes. The maximum range of CPG doses is less than 50 percent, with the greatest being for the comparison of scenarios 3 and 97. The maximum range for long-term health effects is about 70 percent for three of the scenario comparisons. This is due to the fact that the differences between the U.S. average waste mix and the various compact's waste mix are not significant for the nuclides and waste streams causing the majority of the health impacts.

Two additional assessments were made for the Northeast and Southeast regional compact areas. The volumes of waste are 470,000 m^3 for the Northeast scenario (102) and 590,000 m^3 for the Southeast scenario (101). The impacts from these scenarios are compared to risks from scenarios 78 and 77, which use a 500,000 m^3 volume of the U.S. average waste mix. The maximum CPG doses are essentially the same. The long-term health effects are within a factor of 1.6.

(E) Sensitivity to Waste Groups

The sensitivity of the results to the various combinations of wastes is presented in Table 11-13. The wastes are grouped into LLW-BRC+NARM, LLW+NARM, LLW only, and LLW-BRC+NARM-CLASS D. The maximum annual CPG doses fall within 30 percent of each other at any location, as do the long-term health effects.

Table 11-10. Summary of long-term health effects and maximum annual CPG doses from selected waste forms at different site locations using the SLD disposal method

Scenario	Waste form				CPG dose in peak year (mrem/yr)	Health effects over 10,000 years (fatal cancers only)
	Class A	Class B	Class C	NARM		
Humid Impermeable Location						
2	AS IS	AS IS	AS IS	AS IS	0.13	8.1
27	— Incinerated/Solidified —				0.03	6.4
39	AS IS	SOL	SOL	SOL	0.03	2.2
Humid Permeable Location						
1	AS IS	AS IS	AS IS	AS IS	35	4.7
26	— Incinerated/Solidified —				13	3.1
38	AS IS	SOL	SOL	SOL	9.1	3.9
Arid Permeable Location						
3	AS IS	AS IS	AS IS	AS IS	2.2E-03	3.6
28	— Incinerated/Solidified —				3.0E-04	1.4
40	AS IS	SOL	SOL	SOL	9.2E-04	2.3

Table 11-11. Summary of long-term health effects and maximum annual CPG doses from selected waste forms at different locations using 10 CFR Part 61 disposal technology

Scenario	Waste form				CPG dose in peak year (mrem/yr)	Health effects over 10,000 years (fatal cancers only)
	Class A	Class B	Class C	NARM		
Humid Impermeable Location						
5	AS IS	SOL	SOL	SOL	0.03	2.2
30	— Incinerated/Solidified —				0.025	3.4
33	AS IS	HIC	SOL	SOL	0.12	3.8
36	HIC	HIC	HIC	HIC	0.13	6.1
45	AS IS	AS IS	AS IS	AS IS	0.12	4.6
Humid Permeable Location						
4	AS IS	SOL	SOL	SOL	9.2	3.9
29	— Incinerated/Solidified —				13	3.1
32	AS IS	HIC	SOL	SOL	82	4.6
35	HIC	HIC	HIC	HIC	40	4.5
44	AS IS	AS IS	AS IS	AS IS	35	4.7
Arid Permeable Location						
6	AS IS	SOL	SOL	SOL	9.2E-04	2.3
31	— Incinerated/Solidified —				3.0E-04	1.4
34	AS IS	HIC	SOL	SOL	3.9E-03	3.5
37	HIC	HIC	HIC	HIC	1.9E-03	3.5
46	AS IS	AS IS	AS IS	AS IS	2.2E-03	3.6

Table 11-12. Summary of long-term health effects and maximum annual CPG doses from various waste mixes at different site locations

Scenario	Waste mix	Disposal method	CPG dose in peak year (mrem/yr)	Health effects over 10,000 years (fatal cancers only)
Humid Impermeable Location				
2	U.S. average	SLD	0.13	8.1
96	N.E. compact	SLD	0.15	12
5	U.S. average	SLD/ISD	0.03	2.2
99	N.E. compact	SLD/ISD	0.03	3.8
Humid Permeable Location				
1	U.S. average	SLD	35	4.7
95	S.E. compact	SLD	37	2.8
4	U.S. average	SLD/ISD	9.2	3.9
98	S.E. compact	SLD/ISD	9.7	2.4
Arid Permeable Location				
3	U.S. average	SLD	2.2E-03	3.6
97	Rocky Mtn. compact	SLD	1.5E-03	2.5
6	U.S. average	SLD/ISD	9.2E-04	2.3
100	Rocky Mtn. compact	SLD/ISD	1.0E-03	2.1

Table 11-13. Summary of long-term health effects and maximum annual CPG doses from four waste groups at different site locations using 10 CFR Part 61 disposal technology

Scenario	Waste group	CPG dose in peak year (mrem/yr)	Health effects over 10,000 years (fatal cancers only)
Humid Impermeable Location			
5	LLW-BRC+NARM	3.0E-02	2.2
51	LLW+NARM	2.3E-02	1.7
60	LLW	2.3E-02	1.7
69	LLW-BRC+NARM -CLASS D	3.0E-02	2.2
Humid Permeable Location			
4	LLW-BRC+NARM	9.2	3.9
50	LLW+NARM	7.0	2.9
59	LLW	7.0	3.0
68	LLW-BRC+NARM -CLASS D	9.1	3.9
Arid Permeable Location			
6	LLW-BRC+NARM	9.2E-04	2.3
52	LLW+NARM	6.8E-04	1.7
61	LLW	6.8E-04	1.7
70	LLW-BRC+NARM -CLASS D	9.2E-04	2.3

An additional test of waste groups compares scenarios 1, 2, 3 against scenarios 65, 66, 67, from which Class D wastes are removed. In these scenarios all wastes are disposed of by the SLD method. This comparison indicates that removal of the Class D waste, because it is solidified, has very little effect on estimated impact. This is due to the fact that the Class D waste, which is always solidified, results in little release to the environment, thus very little health impact.

Removing the BRC component from the wastes (4,5,6) causes the impacts to increase. This is because the size of the disposal site is kept constant. Therefore, when lower activity BRC wastes are removed, they are replaced with higher activity LLW, which causes impacts to increase. The reason that removing NARM or Class D components from the wastes causes little change in impact is due to the disposal method that was analyzed (SLD/ISD). In this method the NARM and Class D wastes are solidified. This causes the activity to be released very slowly, thus not reaching the population until long after the time period analyzed has been exceeded. Therefore, since these nuclides do not reach the population, removing them from the waste has no effect on impacts.

It is worth noting that the NARM wastes included in this analysis are not regulated at the Federal level. Many of these NARM wastes are presently being stored. For the purpose of our analyses, however, we assume that when these NARM wastes are disposed of, they are disposed of in ISD facilities. The total U.S. health effects from their disposal, using this assumption, is less than one health effect. This is from 20 years of waste, evaluated for 10,000 years. If these wastes were disposed of in a sanitary landfill, the health effects would increase to 71. The difference in health effects is due to the solidification of the waste when using the ISD method and the very low releases which result.

(F) Sensitivity to Waste Volumes

The sensitivity of the results to various waste volumes at a site is presented in Table 11-14 for two different disposal methods. The long-term health effects vary directly with the volume of waste at all locations, i.e., when the volume of waste at a site is doubled, the number of fatal cancers doubles. The maximum CPG dose for the humid impermeable site also varies directly with the volume of waste. However, the dose estimates for the other two locations do not demonstrate a similar relationship. For the humid permeable site, the CPG dose increases 30 to 40 percent when the volume is doubled, while for the arid permeable site, the CPG dose increases 40 to 50 percent when the volume is doubled, as explained below.

Additional comparisons of waste volume effects can be made by evaluating variations in volumes caused by changes in the waste groups (LLW, BRC, NARM combinations). These volume tests are presented in Table 11-15 using 10 CFR Part 61 disposal technology (SLD/ISD). The same observation applies to these tests as to the tests made specifically to observe the effect of volume changes. The long-term health effects vary directly with waste volume at all sites and for all waste group combinations. The maximum annual CPG dose varies directly with volume for the humid impermeable site, but varies less than directly with volume at the two permeable sites, as explained below.

Table 11-14. Summary of long-term health effects and maximum annual CPG doses from different volumes of wastes

Scenario	Waste volume ($1000 m^3$)	Disposal method	CPG dose in peak year (mrem/yr)	Health effects over 10,000 years (fatal cancers only)
Humid Impermeable Location				
75	100	SLD	0.05	3.2
2	250	SLD	0.13	8.1
72	500	SLD	0.25	16
81	100	SLD/ISD	0.01	0.9
5	250	SLD/ISD	0.03	2.2
78	500	SLD/ISD	0.06	4.5
Humid Permeable Location				
74	100	SLD	22	1.9
1	250	SLD	35	4.7
71	500	SLD	48	9.4
80	100	SLD/ISD	5.6	1.6
4	250	SLD/ISD	9.2	3.9
77	500	SLD/ISD	13	7.8
Arid Permeable Location				
76	100	SLD	1.3E-03	1.4
3	250	SLD	2.2E-03	3.6
73	500	SLD	3.1E-03	7.1
82	100	SLD/ISD	5.6E-04	0.92
6	250	SLD/ISD	9.2E-04	2.3
79	500	SLD/ISD	1.3E-03	4.6

Table 11-15. Summary of long-term health effects and maximum annual CPG doses from different volumes of waste using 10 CFR Part 61 disposal technology

Scenario	Waste Volume (1000 m^3)	Waste group	CPG dose in peak year (mrem/yr)	Health Effects over 10,000 years (fatal cancers only)
Humid Impermeable Location				
48	373	LLW+NARM	3.4E-02	2.5
51	250	LLW+NARM	2.3E-02	1.7
54	170	LLW+NARM	1.5E-02	1.1
57	366	LLW	3.3E-02	2.5
60	250	LLW	2.3E-02	1.7
63	170	LLW	1.5E-02	1.2
Humid Permeable Location				
47	373	LLW+NARM	8.6	4.4
50	250	LLW+NARM	7.0	2.9
53	170	LLW+NARM	5.7	2.0
56	366	LLW	8.6	4.3
59	250	LLW	7.0	3.0
62	170	LLW	5.7	2.0
Arid Permeable Location				
49	373	LLW+NARM	8.4E-04	2.6
52	250	LLW+NARM	6.8E-04	1.7
55	170	LLW+NARM	5.5E-04	1.2
58	366	LLW	8.4E-04	2.5
61	250	LLW	6.8E-04	1.7
64	170	LLW	5.5E-04	1.2

The sensitivity of maximum annual CPG doses to site size is shown in Figure 11-2. The changes in CPG dose from disposing of 100,000, 170,000, 250,000, 373,000, and 500,000 m^3 of the U.S. average mix of LLW by the 10 CFR Part 61 disposal technology under three different hydrogeologic/-climatic settings are shown. Based on this figure and Table 11-15, it can be seen that with a linear increase in inventory: (1) the long-term health effects increase linearly and (2) the CPG dose increases (a) linearly with the inventory in a humid, low-permeability (overflow) setting and (b) at a less than linear rate for humid and arid permeable (contaminant movement to ground water) settings. This less than linear increase for the permeable sites is due to areal dilution of ground water caused by the larger site area required for the source term. This areal dilution does not occur at the impermeable site, since the surface water pathway predominates, for which the site area is less important in calculating water concentrations. It is worth noting, however, that the CPG dose did not exceed 15 mrem/yr for a 500,000 m^3 capacity site in a humid permeable setting and was always less than 0.1 mrem/yr for the other settings.

An additional set of analyses relating to site volume were performed to evaluate a scenario where the waste was volume reduced, such as by compaction, but the total activity remained the same. This would result in a smaller site size, but with the same radionuclides and activity. This analysis was performed at humid permeable and humid impermeable sites, using 10 CFR Part 61 disposal technology (consisting of SLD for Class A and B waste and ISD for Class C waste). Two separate sets of analyses were performed; the first with the well located the same distance from the center of the disposal site (which results in a larger buffer zone, since the site area will be smaller) and the second with the well moved closer to the site (resulting in the same buffer zone distance as in the base case analyses). The results are shown in Table 11-16.

Peak dose projections for the humid permeable disposal site increase as the disposal site area decreases. The increase in dose is due to the fact that, as the site area is decreased, the concentration of the site leachate increases since the volume of water into which the contaminants released from the site are diluted declines. The PRESTO-EPA-CPG code calculates an aquifer dilution volume based on the disposal site width, the aquifer thickness, and the aquifer dispersivity, as described in the PRESTO-EPA Methodology Manual (EPA87). As the site area is reduced, the groundwater flow rate drops and, in light of constant rate of radionuclide releases from the waste, contaminant concentrations rise. Radionuclide releases are constant despite the fact that contaminant concentrations in the trench are higher. While the higher concentrations result in higher leachate concentrations, the volume of water draining from the trench is reduced. These effects balance one another such that total curie releases remain unchanged.

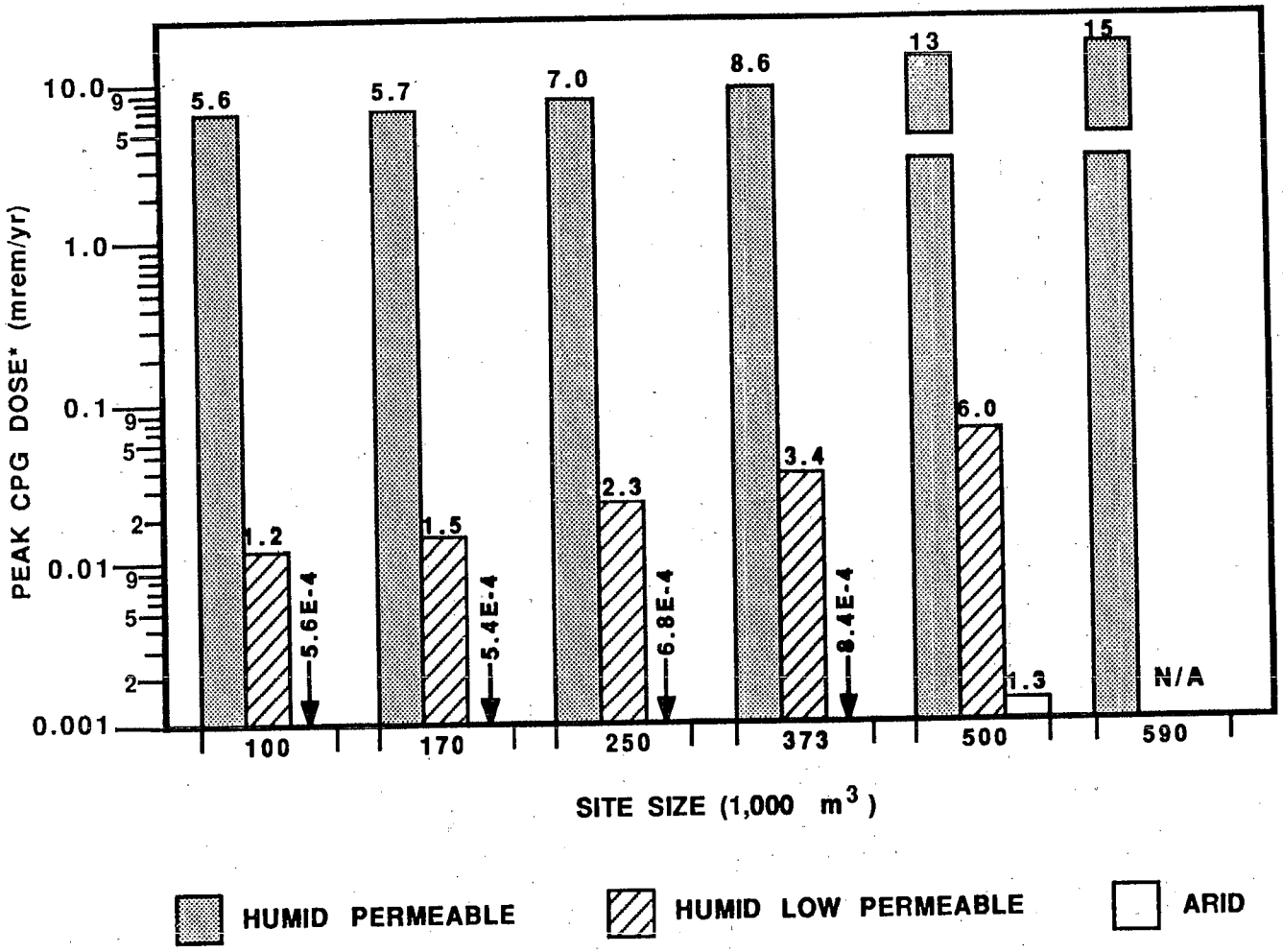

Figure 11-2. Sensitivity of CPG Dose to Site Size (Waste Volume) for 10 CFR Part 61 Disposal Technology

Table 11-16. Results summary of reducing waste volume with total activity constant

Scenario[1]	CPG Dose in Peak Year (mrem/yr)
Humid Permeable Site	
Standard Scenario	9.2
Volume Decreased by 50%, larger buffer zone	12
Volume Decreased by 75%, larger buffer zone	15
Volume Decreased by 50%, same buffer zone	13
Volume Decreased by 75%, same buffer zone	17
Humid Impermeable Site	
Standard Scenario	0.03
Volume Decreased by 50%, larger buffer zone	0.03
Volume Decreased by 75%, larger buffer zone	0.03
Volume Decreased by 50%, same buffer zone	0.03
Volume Decreased by 75%, same buffer zone	0.03

1. All scenarios are for disposal in a site using 10 CFR Part 61 disposal technology. Volume is reduced as listed, with buffer zone distances either increasing or staying the same, as discussed in the text.

The elevated doses seen for the smaller sites are greater still when the distance from the site boundary to the well (buffer zone) is held constant. With a decline in the distance to the well (from site center) the degree of dispersion seen in the aquifer prior to arrival at the well is minimized, thereby reducing the overall aquifer dilution volume. This reduction results in higher aquifer nuclide concentrations and, consequently, higher doses from the use of the contaminated water.

In contrast to the humid permeable site, reductions in disposal site area has no affect on peak dose projections at the humid impermeable site. The peak dose at this site arises from the overflow of the waste trenches and the subsequent transport of released contaminants to a stream. While the concentrations of nuclides in the overflow water are higher due to waste volume reduction, the volume of water exiting the trenches is reduced as it is based on trench surface area. The end result is that total curie releases remain constant.

These releases are diluted in the same amount of water as the dilution volume of the receiving stream in either case and, therefore, are unaffected by the site area. Consequently, contaminant concentrations are unchanged from the base case simulations and peak doses remain constant, regardless of the size of the buffer zone.

From these results it can be seen that at permeable sites, if volume reduction takes place such that the outcome is a smaller site with the same total activity, peak CPG doses will increase. The increase will be even greater if the size of the buffer zone, (distance from trench boundary to well) remains constant, resulting in the well moving closer to the site center. It should be noted, however, that even with a 50% decrease in site size, the peak CPG dose is still not above 15 mrem/yr and that it can be reduced to 12 mrem/yr by using the smaller site size as a means of increasing the size of the buffer zone.

(G) Sensitivity to the Time Horizon

The maximum annual CPG doses for a 1,000- and a 10,000-year time horizon are shown in Table 11-17, where 10 CFR Part 61 disposal technology is analyzed at all three hydrogeologic/climatic sites. The peak CPG doses for the two humid locations are the same for both periods. This finding reflects the fact that the movement of the high-dose, mobile radionuclides to the CPG at humid locations occurs relatively quickly, so that extending the analysis past 1,000 years has no effect on peak CPG dose

At the arid site, however, extending the analysis period to 10,000 years causes an increase in the peak dose from about 0.0009 to about 0.02 mrem/yr. This increase is due to the nature of the arid site, where radionuclide transport through the groundwater pathway is very slow for even the more mobile nuclides. In the 1,000 year analysis, C-14 reaches the CPG and causes the peak dose, with other less mobile nuclides remaining in transit. Extending the analysis period to 10,000 years allows some of the less mobile nuclides, such as I-129, to reach the CPG, causing a greater peak dose. It should be noted, however, that all doses, whether from C-14 or I-129, are very small at the arid site.

Table 11-17 Summary of maximum annual CPG doses from LLW using different time periods and 10 CFR Part 61 disposal technology

Scenario	Period Analyzed (yr)	CPG dose in peak year (mrem/yr)
Humid Impermeable Location		
5	1,000	3.0E-02
104	10,000	3.2E-02
Humid Permeable Location		
4	1,000	9.2
103	10,000	9.2
Arid Permeable Location		
6	1,000	9.2E-04
105	10,000	1.7E-02

In Figures 11-3, 11-4, and 11-5 the maximum annual CPG dose is plotted over time (with time zero corresponding to site closure) for conventional shallow land disposal and 10 CFR Part 61 disposal technology. For conventional shallow land disposal, all waste is in the as-generated waste form, while for 10 CFR Part 61 disposal technology, much of the waste is solidified (Class B and C). As can be seen from the three figures, the combination of more rigorous disposal practices and the solidified waste form for the 10 CFR Part 61 disposal technology leads to similar behavior of CPG dose versus time as for conventional disposal for the radionuclide releases, but, in general, significantly smaller dose rates.

Figure 11-3, shows the relationship of dose rate over time at the humid permeable site. The annual dose rate rises quickly from the very mobile radionuclides H-3 and C-14. As this dose diminishes, the Tc-99 and then the I-129 reach the receptor. The maximum annual dose rate is from I-129, around year 700. After about 900 years the dose rates start to level off. Less mobile radionuclides will continue to reach the CPG after 900 years, but the doses will remain below that of I-129.

Figure 11-4 shows the relationship of dose rate over time at the humid impermeable site. There is very little dose until the trench cover fails in year 100. Failure of the trench cover allows a quick release of many nuclides and a large peak in dose rate soon after, due to overflow of the trench and transport via surface water. The peak dose is mainly due to I-129 in about year 200. The peak drops quickly, leveling off after about year 300.

The arid permeable site is depicted in Figure 11-5. The small dose rate in the first few years is due to windblown transport of nuclides spilled during operation. The most mobile nuclide, C-14, reaches the CPG about year 900. As discussed earlier, extending the analysis past 1,000 years results in a larger peak dose from I-129, although this peak dose is still very small. In general, the plot of CPG dose versus time for the arid site will look similar to that of the humid permeable site, except that the doses will be much lower and the time at which the peaks occur will be shifted to the right.

Cumulative population health effects are assessed for 10,000 years, but occur at different rates over that period. This can be seen in Table 11-18, where health effects are broken down by time period. Looking at the U.S. totals, as opposed to the three specific sites, and assuming 10 CFR Part 61 disposal technology, we see that 43 percent of the population health effects occur in the first 500 years. An additional 10 percent occur over the following 500 years, with 47 percent occurring during the last 9000 years of the analysis.

It should be noted that these results are estimates of the total potential U.S. health effects, weighted for the waste in each of the three hydrogeologic regions. These results would be less for each specific region and would vary depending on the region.

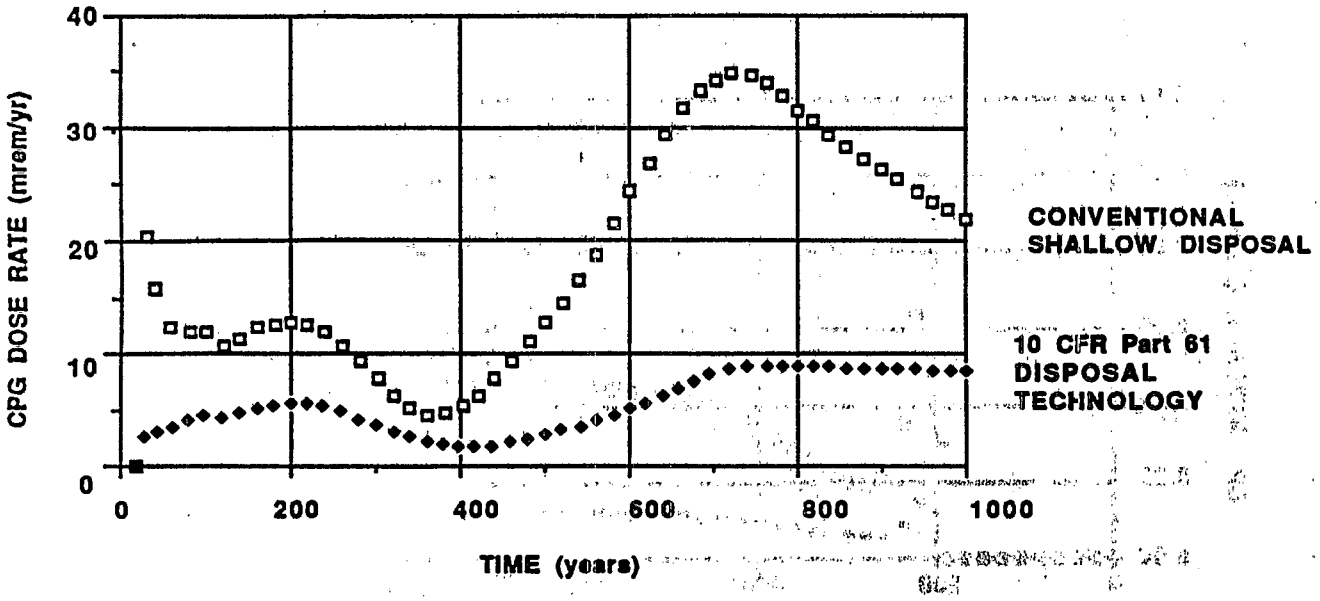

Figure 11-3. CPG Dose Versus Time for the Humid Permeable Hydrogeologic Site

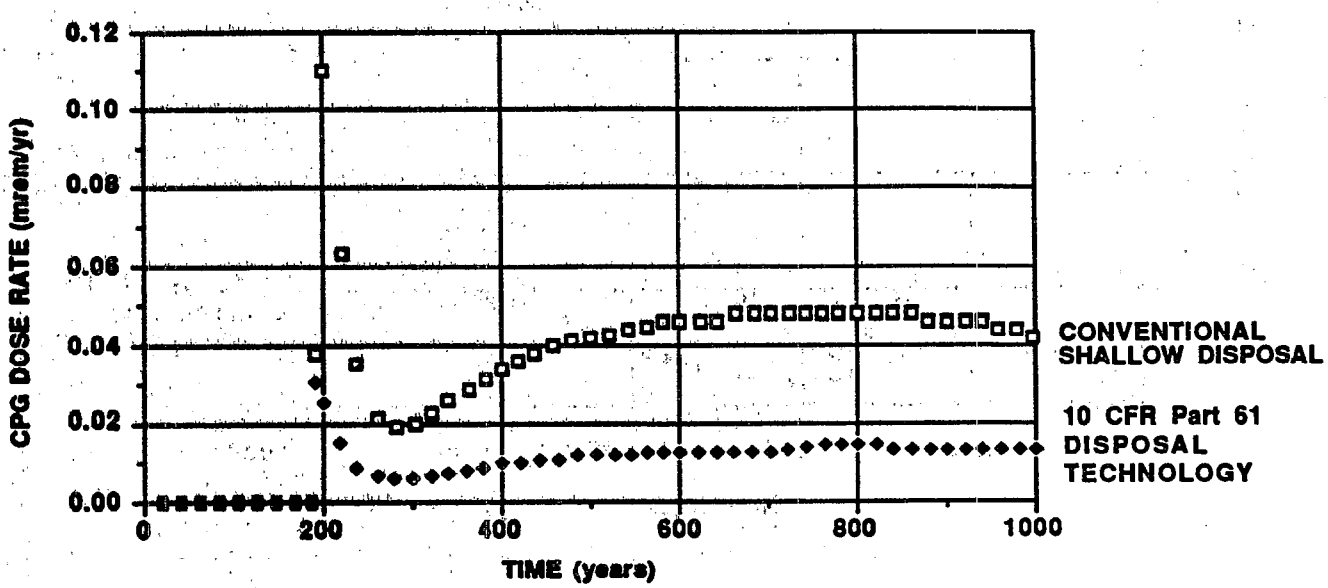

Figure 11-4. CPG Dose Versus Time for the Humid Impermeable Hydrogeologic Site

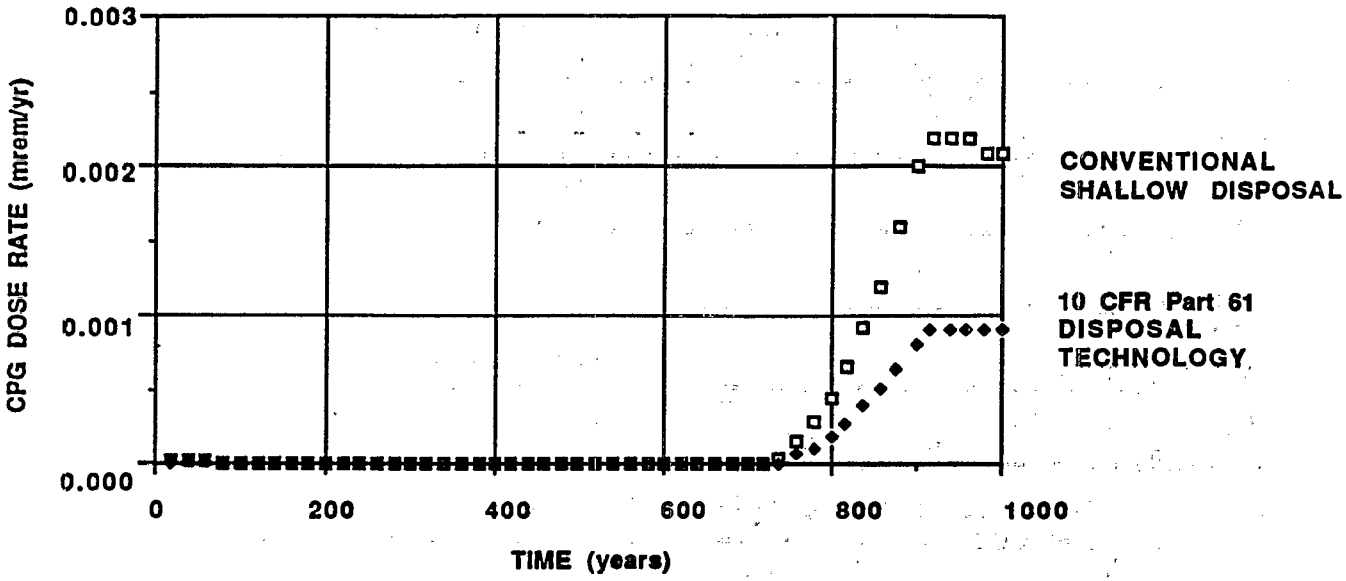

Figure 11-5. CPG Dose Versus Time for the Arid Permeable Hydrogeologic Site

Table 11-18. U.S. total health effects over time, from disposal of 20 year U.S. waste volume using 10 CFR Part 61 disposal technology

Time Period (yr)	Cumulative Health Effects[1]	% of Total
0 - 100	1.6	5.7
101 - 500	10.6	37.5
501 - 1,000	2.8	9.8
1,001 - 10,000	13.3	47.0
Total (0 - 10,000)	28.3	100

1. Health effects are for fatal cancers and serious genetic effects.

If the assessment that was described above was done on a regional basis, the population health effects for the humid sites would be very similar to the results shown. About 50% of the health effects occur before 1,000 years and the other half after 1,000 years. For less restrictive disposal methods, such as conventional shallow, or with less waste solidified, the proportion of health effects occuring before 1,000 years would increase. For the arid sites, a greater percentage of the health effects occur in later years due to the slower rate of groundwater transport. In general, numbers of health effects occuring over the hydrogeologic/climatic regions or between various disposal methods are fairly constant, but with the health effects occuring later at the arid sites and with more restrictive disposal methods or solidification of the waste.

In assessing long-term, cumulative population health effects over 10,000 years, a question is raised as to what health effects could potentially occur due to radionuclides which do not reach the population within 10,000 years. There is some concern, of course, that any analysis over 10,000 years contains so much inherent uncertainty that estimates at extremely long time periods are ludicrous. Some rough estimate may be instructive, however.

In Table 11-19, the radionuclides which reach the population (breakthrough) after 10,000 years at the humid permeable site are listed, along with their half-life and the percentage of their original activity which would remain at the time of breakthrough. As can be seen from this table, many of these less-mobile nuclides will have decayed completely prior to breakthrough. For a few, however, a significant percentage of their original activity remains.

For the nuclides with a significant fraction of their original inventory remaining at time of breakthrough, a rough estimate of potential health effects can be made as outlined in Table 11-20. The percent of inventory remaining (i.e., not yet decayed) at breakthrough can be multiplied by the original inventory to determine the inventory remaining at breakthrough. This inventory is very conservative, however, as much of it could still be in the trench or in transit at the time some reaches the population. Using this conservative inventory, however, an estimate of potential health effects can be made by multiplying the inventory by the radionuclide specific HECF values for the humid permeable site. The HECF values, which are described in detail in Chapter 8, provides an estimate of the health effects to the population from releases of activity to the basin, assuming the same water useage patterns for the basin population as for the local population.

Table 11-19. Radionuclides which reach the population after 10,000 years and the percent of their original activity remaining at time of breakthrough

Nuclide	Half-Life (yr)	Breakthrough Time to Population* (yr)	% of Original Inventory Remaining at Time of Breakthrough
Fe-55	2.7E+0	7.6E+5	0
Ni-59	8.0E+4	1.9E+4	85
Ni-65	1.1E+2	1.9E+4	0
Sr-90	2.9E+1	1.4E+4	0
Nb-94	2.0E+4	4.5E+4	21
Ru-106	1.0E+0	2.8E+4	0
Cs-134	2.1E+0	1.1E+5	0
Cs-135	2.3E+6	1.1E+5	97
Cs-137	3.0E+1	1.1E+5	0
Ba-137m	3.0E+1	1.1E+5	0
Eu-154	8.5E+0	5.1E+5	0
Po-210	3.8E-1	2.8E+4	0
Pb-210	2.0E+1	2.8E+4	0
Bi-214	8.6E-1	2.8E+4	0
Pb-214	5.1E-5	2.8E+4	0
Ra-226	1.6E+3	2.8E+4	0
U-234	2.5E+5	9.6E+4	77
U-235	7.0E+8	9.6E+4	100
U-238	4.5E+9	9.6E+4	100
Pu-238	8.8E+1	4.5E+5	0
Pu-239	2.4E+4	4.5E+5	0
Pu-241	1.3E+1	4.5E+5	0
Am-241	4.6E+2	1.0E+7	0
Pu-242	3.8E+5	4.5E+5	44
Am-243	7.4E+3	1.0E+7	0
Cm-243	3.2E+1	4.2E+5	0
Cm-244	1.8E+1	4.2E+5	0

*Breakthrough time is the time at which nuclides reach the population and is based on the humid permeable site.

Table 11-20. Potential population health effects from nuclides which reach the population after 10,000 years

Nuclide	% of Original Inventory Remaining at Time of Breakthrough[1]	Original Inventory (Ci)[2]	HECF[3]	Potential Health Effects[4]
Ni-59	85	3.0E+2	2.0E-5	5.1E-3
Nb-94	21	3.1	5.2E-2	3.4E-2
Cs-135	97	2.8	5.9E-3	1.6E-2
U-234	77	3.8	3.1E-4	9.0E-4
U-235	100	7.5E-2	3.5E-4	2.6E-5
U-238	100	7.2E-1	3.0E-4	2.2E-4
Pu-242	44	9.7E-2	6.2E-3	2.6E-4
Total Potential Health Effects				6.2E-2

1. Breakthrough time is the time at which nuclides reach the population and is based on the humid permeable site.

2. Original inventory is the total activity for each nuclide used in the base case runs.

3. HECF values are the health effect conversion factors from the humid permeable site used to calculate health effects from radionuclides released to the regional basin.

4. Potential health effects are calculated by multiplying the amount of inventory remaining at time of breakthrough by the appropriate HECF value.

Using the methodology described above, it can be seen that potential health effects from radionuclides which reach the population after 10,000 years would be very low. Using conservative assumptions, total potential health effects from these nuclides are much less than one health effect. The results of this analysis would not vary significantly for the other hydrogeologic sites. An additional point can be raised as to what the contribution would be from radionuclides which reach the well prior to 10,000 years, but are not completely consumed in the regional basin. This is evaluated in Section (I), as part of the global analysis.

(H) Sensitivity to Radionuclides

An additional consideration regarding health impacts from LLW disposal is identifying the radionuclides that contribute most to the health impacts under different disposal situations. Since a unit curie and unit volume methodology was used to calculate population health effects (see Chapter 8), the results from these separate analyses can be used to illustrate which radionuclides could be most sensitive under various disposal scenarios. The analyses and results are described in the following sections for population health effects with a separate discussion of CPG peak doses.

The "unit curie" methodology, wherein the population health effects from disposing of one curie of each radionuclide of interest by a specific combination of disposal methods, waste forms, and hydrogeologic/climatic conditions is modeled (see chapter 8), is used to identify potentially important radionuclides. The relative importance of radionuclides change, depending on hydrogeologic/climatic setting, the form of the waste, and the critical release, transport, and exposure pathways.

Where ground water pathways are important, such as at the humid permeable and arid permeable sites, the majority of the population health effects are contributed by long-lived, mobile radionuclides with high risk factors, such as C-14, Tc-99, I-129, and Np-237 . Less mobile radionuclides, such as Nb-94, Cs-135, and Cm-243, become important in cases where the trenches overflow, such as at the humid impermeable site. When the trenches overflow, these radionuclides are discharged directly onto the land surface and subsequently into surface waters, after retardation by the soils. For ground surface exposure pathways, such as in the first years after site closure when atmospheric transport of nuclides spilled during operation are important, gamma-emitting radionuclides such as Co-60, Cs-134, and Cs-137 can dominate. Figure 11-6 shows the relative importance, on a per curie basis, of selected radionuclides in each hydrogeologic region as derived from a "unit curie" analysis.

The contribution to health impact of individual radionuclides depends on the concentration of the nuclide in the waste. The "unit volume" analysis (see Chapter 8), on a waste stream by waste stream basis, is useful for identifying potentially important waste streams and radionuclides. In the "unit volume" calculation, one cubic meter of waste is loaded with the scaled concentration of activity for each radionuclide in that waste stream. The population health effects are then calculated based on the unit volume being disposed of by specific disposal methods, waste forms, and hydrogeologic and climatic combinations.

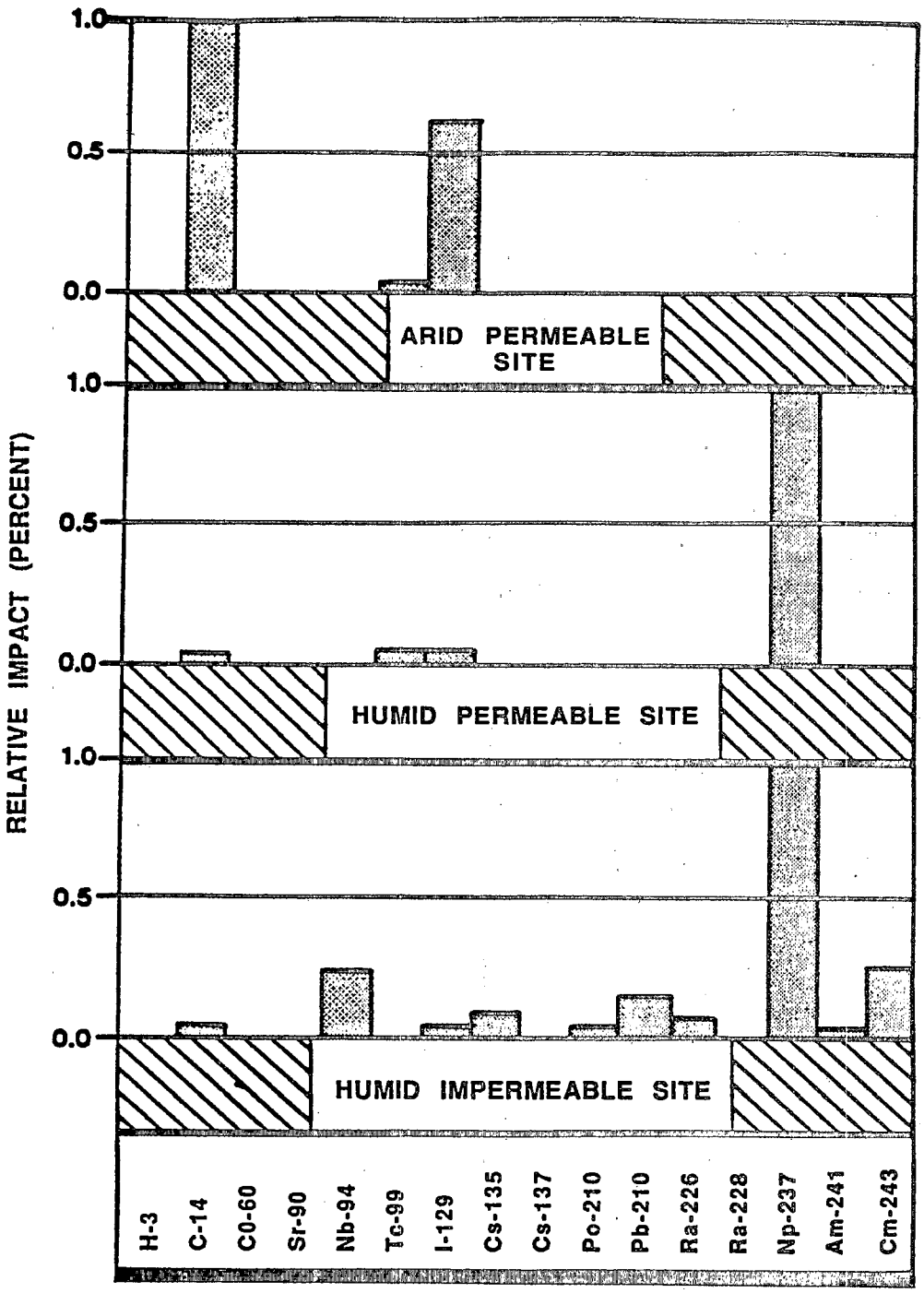

Figure 11-6. Relative Impact from the Disposal of a Unit Curie of Various Radionuclides by Hydrogeologic Setting

Figure 11-7 identifies the important radionuclides in each waste stream for three different hydrogeologic/climatic settings, based on the "unit volume" analysis. This allows identification of important radionuclides on a waste stream basis; for example, how specific radionuclides can shift in importance, depending on the hydrogeologic/climatic setting. As can be seen, in the permeable sites, where groundwater transport is the major pathway, C-14 is the most significant nuclide due to its long half-life, mobility, and high risk factor. Note however, that this is based on a unit-volume approach, with one cubic meter of each waste stream. The results can change with a full inventory. At the impermeable site, the significant radionuclides vary due to the predominance of the overflow and surface water pathway, which allows less mobiles nuclides, such as Am-241 to dominate certain waste streams.

If Figure 11-6 is compared with Figure 11-7, it can be seen how radionuclides which were important on a per curie basis are no longer important on a waste stream basis. Neptunium-237 is a good example. In the "unit curie" analysis, Np-237 is identified as potentially the most critical radionuclide because of its high mobility in water pathways, its high toxicity, and its long half-life. However, since only negligible concentrations of Np-237 are in commercial LLW, Np-237 is usually not important, as can be seen from its absence in Figure 11-7.

Finally, we can evaluate a fully loaded, 250,000 m^3 site to determine which nuclides contribute the greatest number of population health effects for the base case 10 CFR Part 61 disposal technology scenario. Table 11-21 shows the radionuclides that contribute the greatest percentage of the population health effects at a site using 10 CFR Part 61 disposal technology at the three site locations. As can be seen from Table 11-21, C-14 is the critical radionuclide, contributing 90 percent of the population health effects at the humid permeable site and 95 percent at the arid permeable site. At the humid impermeable site, with its overflow pathway, C-14 contributes only 70 percent of the population health effects, with Am-241 contributing most of the remainder. The importance of C-14 is due to its high mobility, long half-life, and, to a lesser degree, its relatively large source term and high dose conversion factor.

The unit curie and unit volume methodology, which was used for calculating population health effects, was not used in calculating peak doses to the CPG. The impact that is determined for the CPG, maximum annual dose rate, is not amenable to that type of a methodology, as nuclide concentration over time is required. Instead, analyses are done directly with a full source term as described in Chapter 8.

The nuclides which are most critical to the CPG peak dose at each of the three sites, using a complete source term and the 10 CFR Part 61 disposal technology, are shown in Table 11-21. At the humid sites, I-129 is the major nuclide, while at the arid site, C-14 dominates. These results can be most easily explained by reviewing Figures 11-3, 11-4, and 11-5.

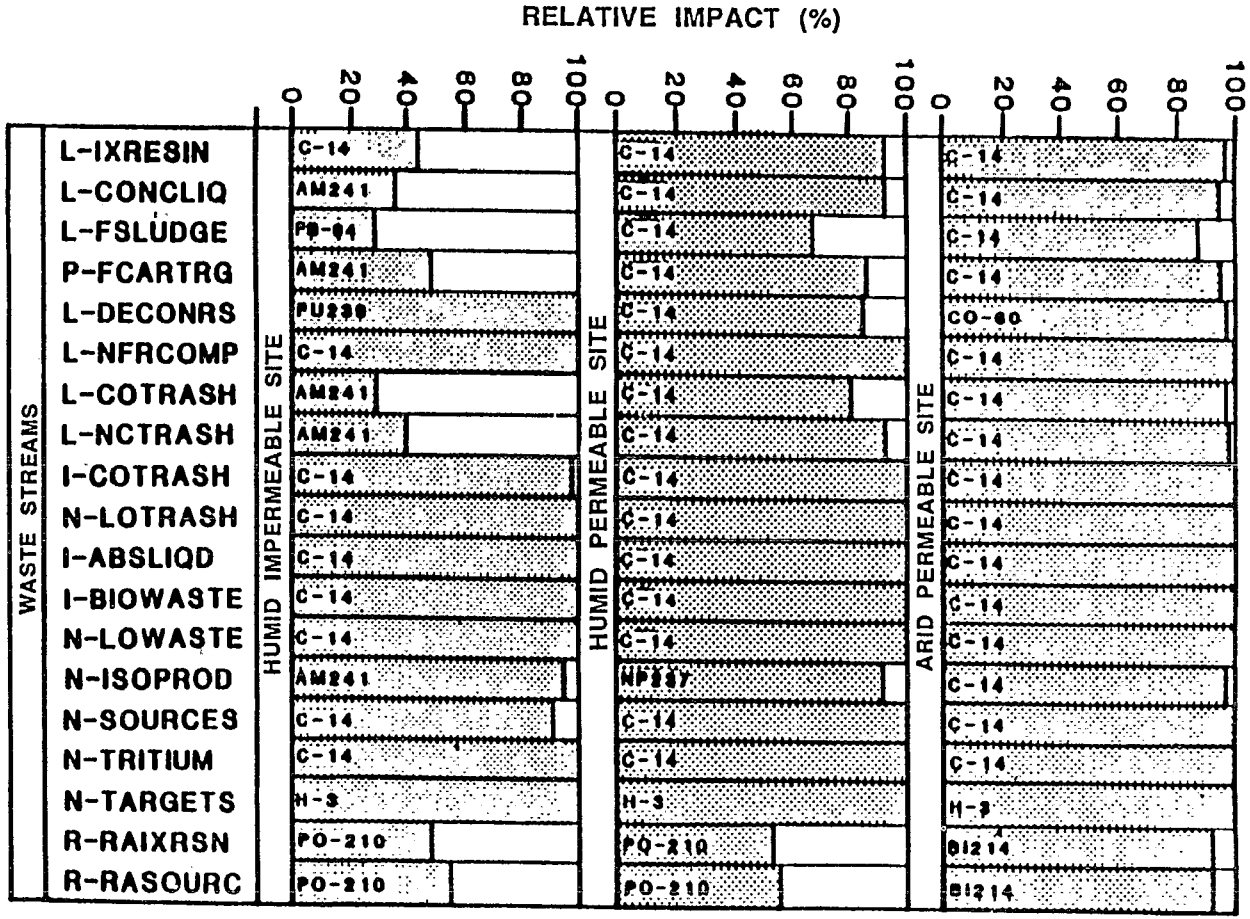

Figure 11-7. Dominant Radionuclides When Unit Volumes of Various Waste Streams are Disposed of in Three Different Hydrogeologic Regions

Table 11-21. Critical radionuclides at model LLW site [1]

Scenario	Critical Radionuclides	Percent of total health impact caused by critical radionuclides: Peak CPG Dose	Percent of total health impact caused by critical radionuclides: Population Health Effects[2]
Humid Permeable	I-129	93%	-
	C-14	7%	90%
	Other	-	10%
Humid Impermeable	I-129	78%	-
	C-14	22%	70%
	Am-241	-	20%
	Other	-	10%
Arid Permeable	C-14	100%	95%
	Other	-	5%

1. Model site assumes 250,000 m^3 of U.S. average waste mix, using 10 CFR Part 61 disposal technology.

2. Approximate values.

At the humid permeable site (Figure 11-3), where over 90 percent of the peak CPG dose is due to I-129, one sees that C-14 contributes a small peak early in the analysis (about year 200), but that at the time of the main peak (about year 800), I-129 dominates. This is contrasted with the humid impermeable site (Figure 11-4), where I-129 contributes 78 percent of the peak CPG dose and C-14 contributes 22 percent. The reason that both nuclides contribute toward the peak CPG dose is due to the overflow pathway that predominates at the humid impermeable site, which allows both of the nuclide peaks to occur at about the same time. While both peaks occur at approximately the same time (about year 200), the contribution from I-129 is higher.

At the arid permeable (Figure 11-5), the critical radionuclide is C-14. As discussed earlier in this chapter, this is due to the long travel times at the arid site, which causes I-129 to reach the CPG after the 1,000 year analysis is over. If the analysis was extended to 10,000 years, the percentages would be similar to those of the humid permeable site, although the peak CPG dose would still be very small.

(I) Additional Analyses

Additional analyses were performed to answer specific questions related to the scenarios about effects of a buffer zone, the assessment of global health effects, and the importance of the volatility fraction on incineration doses. Because the incineration pathway relates to the BRC scenarios, the sensitivity analyses of volatility fraction are discussed in Chapter 10. The buffer zone and global analyses are discussed below.

A set of analyses were performed to see what the effect of reducing the size of the buffer zone (the distance from the trench boundry to the nearest well) around a disposal site would have on peak CPG doses. These analyses were performed at a humid permeable and an arid permeable site using 10 CFR Part 61 disposal technology. The results are shown in Table 11-22.

The results show that for both sites the CPG peak doses will increase as the buffer zone is reduced. This effect is mainly due to a decrease in the amount of dispersion that takes place as the nuclides move through the aquifer to the well. The dispersion is reduced since a smaller buffer zone results in the nearest well being closer to the disposal site. In addition, because the well is closer, nuclides reach the well sooner, which results in somewhat less decay. This effect is minor, however, since the peak dose is due to long-lived nuclides occuring in later years. As can be seen above, the year of peak dose does not vary at the arid permeable site. This is due to the peak not having been reached by the end of the 1000 year analysis, although the later peak will remain very small.

Table 11-22. Results of varying size of buffer zone on peak CPG dose

Scenario[1]	Peak CPG Dose (mrem/yr)	Year of Peak CPG Dose
Humid Permeable Site		
Standard Scenario[2]	9.2	777
Buffer Zone Decreased by 50%	9.5	754
Buffer Zone Decreased by 75%	9.7	742
No Buffer Zone	9.9	730
Arid Permeable Site		
Buffer Zone Increased by 100%	8.6E-4	1000
Standard Scenario[2]	9.2E-4	1000
Buffer Zone Decreased by 50%	9.6E-4	1000
Buffer Zone Decreased by 75%	9.8E-4	1000
No Buffer Zone	9.9E-4	1000

1. All scenarios assume disposal using 10 CFR Part 61 disposal technology.

2. Standard scenario assumes a 100 meter buffer zone, which is the distance from the trench boundry to the nearest well.

In calculating population health effects, the HECF is used to estimate health effects to the regional basin population. The HECF methodology assumes that water useage patterns for the basin population are the same as the local population. As discussed in Chapter 8, this leads to radionuclides reaching the downstream basin prior to 10,000 years, but since the contaminated water is not completely consumed by the downstream population, radionuclies leave the regional basin and enter the ocean. The nuclides reaching the ocean are ignored in the analysis, with no health effects resulting from their potential consumption. This was considered reasonable, since it was felt that global population health effects due to eventual consumption of radionuclides entering the ocean would be minimal. To further evaluate this assumption, a rough global assessment was performed.

The analysis was performed by calculating nuclide-specific, health effect conversion factors for the consumption of ocean fish and seafood, based on nuclide transfer factors (water-to-fish and water-to-seafood), annual average consumption rates for fish and seafood, and world population, as outlined below. These were compared to the health effect conversion factors for fish consumption from the basin river (the source of the majority of the cumulative basin health effects), the calculation of which is described in Chapter 8.

The health effects conversion factor for ocean fish and seafood consumption is given by the following equation:

$$HEF_i = (P/R)\ (H/C))_i\ [B_{fi}U_f + B_{si}U_s]/\lambda_{Ri}$$

Where:

HEF_i = cancer deaths from ocean fish and seafood consumption per curie of nuclide i entering the ocean

P = world population consuming the ocean fish and seafood

R = volume of upper 75 m of ocean (L)

B_{fi} = nuclide transfer factor, water-to-fish $\left(\frac{\text{pCi/kg fish}}{\text{pCi/L water}}\right)$

B_{si} = nuclide transfer factor, water-to-seafood $\left(\frac{\text{pCi/kg seafood}}{\text{pCi/L water}}\right)$

U_f = annual ocean fish consumption rate per person (kg)

U_s = Annual seafood consumption rate per person (kg)

$(H/C)_i$ = health risk per curie of nuclide i ingested by the population

λ_{Ri} = effective fraction of nuclide removal rate from top 75m of ocean (yr^{-1}).

The value of P used is 10 billion people worldwide. R has a value of 2.7E+19 litres. The parameters U_f and U_s equals 6.9 and 1.0 kg/person-yr, respectively (NRC77). The values of B_{fi} and B_{si} are given in columns 2 and 3 of Table 11-23 for each nuclide. These values are obtained from NRC77. The conversion factor $(H/C)_i$ is calculated from the DARTAB subroutine of PRESTO-EPA. The effective nuclide removal rate is around 0.2 yr^{-1} for long-lived nuclides, based upon comparisons of the present calculations to ratios given in EPA82.

The resulting ocean fish and seafood health effect conversion factors (HEF_i) for the ocean scenario were divided by the river fish health effects factors for the river scenario, as listed in Ro87, to obtain the ratios given in column 4 of Table 11-23. These ratios demonstrate that health effects over 10,000 years from nuclides entering the oceans and ingested in ocean fish and seafood are all less than one percent of the health effects of nuclides entering the rivers and ingested in river fish, except for americium, which is less than four percent of the corresponding health effects from the river scenario. Based on the results of this analysis, it appears reasonable to ignore the contribution of global health effects in our analysis of population health effects.

11.3.3 Summary and Conclusions of Scenario Sensitivity Analyses

Based on the results of the scenario sensitivity analyses, a number of conclusions can be made on the sensitivity of the model output to various scenario assumptions. These conclusions are presented in the following section. Also presented is a summary of the scenario sensitivity analysis results, in the order in which they were presented in the previous sections. It should be noted, however, that as the models are generic and not site specific, the results are as well.

The sensitivity of the results to the choice of disposal site is very pronounced when determining maximum annual doses to a nearby CPG. With other things equal, an impermeable site provides more protection than a permeable one, while arid regions provide more protection than one which is humid. In determining long-term population health effects, the results are much less sensitive to site location with long-term, population health effects fairly constant over all sites. This is due to the fact that over long time periods, the cumulative amount of radionuclides that reach the population does not vary a great deal.

The sensitivity of the results to the type of disposal method used is very similar to the results shown for the site location. Placing waste into a more stringent disposal system which provides greater isolation is similar to disposing of waste in less permeable or less humid sites. Again, the effects are more sensitive when assessing the CPG dose than when assessing long-term, population health effects.

Concerning the waste form, the results are very sensitive to solidification. Both CPG dose and cumulative population health effects are reduced by solidification, due to a reduction in the rate at which radionuclides leave the disposal trench. A secondary point is that high-integrity containers affect the cumulative population health effects

Table 11-23. Comparison of ocean health effects to river health effects

Nuclide	B_f (L/kg)	B_s (L/kg)	Ratio of Ocean to River Health Effects
H-3	9.0E-1	9.3E-1	1.3E-5
C-14	4.5E+3	3.5E+3	1.2E-5
Mn-54	1.0E+2	7.2E+1	1.2E-4
Fe-55	1.0E+2	6.7E+2	2.2E-4
Ni-59	1.0E+2	2.5E+2	9.8E-3
Co-60	2.0E+1	2.0E+2	2.7E-4
Ni-63	1.0E+2	2.5E+2	1.5E-4
Sr-90	1.1E+1	1.1E+2	2.7E-4
Nb-94	3.0E+4	1.0E+2	5.1E-3
Tc-99	4.3E+1	2.1E+2	8.7E-3
Ru-106	1.0E+1	3.3E+3	5.2E-3
Sb-125	1.0	1.0E-1	1.1E-4
I-129	3.3E+1	1.7E+2	8.9E-3
Cs-134	1.3#+3	8.1E+2	1.2E-4
Cs-135	1.3E+3	8.1E+2	4.1E-3
Cs-137	1.3E+3	8.1E+2	8.1E-4
Ce-144	2.5E+1	2.5E+1	1.3E-4
Eu-154	2.5E+1	2.5E+1	1.3E-4
Ra-226	5.0E+1	1.0E+2	4.7E-3
U-234	1.0E+1	1.0E+1	1.3E-3
U-235	1.0E+1	1.0E+1	1.3E-3
Np-237	5.0E+2	5.0E+2	4.1E-3
U-238	1.0E+1	1.0E+1	1.3E-3
Pu-238	8.0	5.3	1.0E-3
Pu-239	8.0	5.3	1.9E-3
Pu-241	8.0	5.3	9.8E-4
Am-241	8.1E+1	1.0E+3	1.7E-2
Pu-242	8.0	5.3	1.9E-3
Am-243	8.1E+1	1 0 3	3.3E-3
Cm-243	2.5E+1	1.0E+3	6.0E-3
Cm-244	2.5E+1	1.0E+3	6.0E-3

very little, since the containers are assumed to fail relatively quickly. In addition, high integrity containers can actually cause an increase in maximum CPG dose of about 10 percent, due to their failure and release of radionuclides over a shorter time period, causing a larger peak in CPG dose.

Separating the waste into waste mixes indicative of various regional compacts has little effect on impacts from disposal. The waste mix used does not vary greatly over the regional compacts for the critical nuclides. The waste groups used in the analyses have little effect on the results, since the major impacts are due to general LLW, which was not varied, and higher activity wastes generally being solidified, such that their inclusion or removal does not affect the results significantly.

Increasing or decreasing the site volume (waste volume and activity) causes a linear increase or decrease in long-term health effects and in CPG dose at impermeable sites. Permeable sites show a less than linear effect, due to areal dilution of larger source terms. In general, site size or waste volume affects health impact in a linear manner.

The sensitivity of the results to the time period used in the analysis is generally not important, as long as a minimum period of about 1,000 years is used for the CPG analyses. Increasing the time period in the CPG analyses at humid sites has no effect, as the CPG peak dose occurs prior to 1,000 years. Since the peak doses can occur as late as 800 years, however, decreasing the time period to much less than 1,000 years could cause the peak dose to be missed. For arid sites and some humid scenarios, the dose rate continues to rise after 1,000 years, due to the long travel time required for radionuclides to reach the CPG or due to greater isolation disposal methods. In these cases, however, the peak annual dose to the CPG is always small. For the long-term, population health effects, nuclides which had not reached the regional basin in 10,000 years, such as nuclides of uranium or plutonium, would require extremely long time periods to do so and would not contribute significantly to the total health effects. In general, it was felt that continuing the analysis period for greater than 10,000 years incorporated too much uncertainty to be useful.

Our analyses show that the inventories of various radionuclides are sensitive, depending upon the site location and the predominant pathways. In general, the mobile nuclides with longer half-lives, such as C-14 and I-129, predominate, although in cases of atmospheric or direct exposure pathways the critical radionuclides may change.

Additional analyses on the size of the buffer zone show this variable to be unimportant in affecting CPG dose or population health effects. Analyses of global health effects show them to be minor in comparison to the river basin health effects.

Based on the scenario sensitivity analyses, the following general conclusions can be made (the conclusions of the single parameter sensitivity analyses are in section 11.2.3.):

- The assumptions concerning the scenario variables, when analyzing the impacts from LLW disposal, cause a much greater change in the output when assessing the maximum dose to the CPG than when assessing the long-term, population health effects. This is due to the fact that CPG dose is based on only the peak release, whereas long-term health effects are based on cumulative releases.

- The health impacts from LLW disposal are very sensitive to methods that provide greater isolation, such as disposal in less permeable or less humid sites, disposal using more rigorous disposal technologies, or solidifying the waste prior to disposal.

- In terms of waste form or waste treatment, the most sensitive method for reducing health impacts is solidification of the waste.

- The health impacts from LLW disposal are dominated by the longer-lived, mobile radionuclides with high dose conversion factors, such as C-14 and I-129, although in cases of atmospheric or direct exposure pathways the critical radionuclides may change.

- In analyzing the health impacts from the disposal of LLW, an analysis period of 1,000 years is usually sufficient to assess any significant peak dose. For long-term, population health effects, 10,000 years is long enough to assess the health impacts from all but the most immobile radionuclides, which would require an extremely long analysis period to assess and are minor contributors to total health effects.

- In determining population health effects, the contribution from nuclides released to the ocean and taken up through the consumption of ocean fish or seafood will be minor.

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Chapter 12: UNCERTAINTY OF CUMULATIVE POPULATION HEALTH EFFECTS AND MAXIMUM CPG DOSE ANALYSES

12.1 Introduction

Deterministic models were used for the assessments of maximum CPG dose and cumulative population health effects conducted in support of the proposed LLW standards. This chapter presents the results of uncertainty analyses, specifically the upper bound of the maximum CPG dose and cumulative population health effects assessments described in Chapter 9, and discusses the methodology used in determining the uncertainty.

Because of the complexity of the assessment, uncertainty analysis is divided into several components. Every effort is made to quantify the upper bound of the uncertainty associated with each component. However, the quantification of uncertainty is limited to certain components because the analyses include extensive field and laboratory data as well as actual uncertainties.

Several uncertainties are not included in this analysis because of time and budget constraints. These include the uncertainties inherited from the approximation of the basic equations used in the PRESTO-EPA model and those resulting from variations of site location input parameters.

12.1.1 Uncertainty Analysis

Ideally, the dose and health effects assessments should be performed using a probabilistic model that expresses the results in the form of random variables (value versus its probability density function). The results of the analysis could then fully inform readers of the absolute value of the dose or health effects and their correspondent probability of occurrence.

Since this type of analysis would require large numbers of calculations, from a practical standpoint this probabilistic model should not be applied to complex assessment models such as the PRESTO-EPA models.

An alternative method is to use a deterministic model with sets of deterministic input parameters to calculate the most probable results of the risk assessment, and to use a separate model to estimate the probable variation of the assessed values (known as "standard deviation"). The estimated probable variation of the assessed values is commonly known as the uncertainty of the results; this analysis therefore intends to evaluate the uncertainty of the results of risk assessments used to support the LLW standards.

12.1.2 Complexity of the Uncertainty Analysis

Maximum CPG dose and cumulative population health effects assessments are extremely complex analyses involving the analysis of many

multidisciplinary areas of expertise, including the processes of atmospheric, hydrogeological, and biological transports and the bioeffects resulting from radionuclide exposure.

Since the dose and health effects analyses supporting the LLW standards are generic, i.e., nonsite specific, the discussion of the uncertainty of the results obtained from the assessments is far more complex than if these same assessments had been performed for a specific site. This is because a site-specific analysis would deal with relatively well-defined site parameter values, while a nonsite-specific analysis would be concerned with the site parameter values having both variation of parameter values for a specific site and variations of several disposal sites within a region. Therefore, the uncertainty caused by the site parameter variations was not included in this analysis, because the results of the risk assessment used for supporting the LLW standards did not intend to cover the entire spectrum of possible variations of disposal sites. When the uncertainty due to the variation of sites is considered, the results of the uncertainty analysis are expected to be dominated overwhelmingly by this uncertainty.

12.1.3 Components of Overall Uncertainties

Because of the complexity of the processes in the risk assessment, the discussion of the overall uncertainties may be simplified by dividing the uncertainties into several components.

For the purpose of this discussion, the overall uncertainties are divided into five components: (1) source term radionuclide concentration, (2) radionuclide geosphere transport, (3) radionuclide food chain transport, (4) human organ dosimetry, and (5) health effects conversion factors. The interactions of the parameters between components are presented in Figures 12-1 and 12-2 for the cumulative health effects analysis and maximum CPG dose analysis, respectively.

12.1.4 Significance of Sensitivity Analysis

A sensitivity analysis, as discussed in Chapter 11, quantifies the sensitivity of model outputs to a change in a specific parameter or group of input parameters under a set of presumed input parameter values. If the input parameter values are altered, a change in the sensitivity of that particular parameter may subsequently follow. Furthermore, the sensitivity analysis results do not provide information on the probability of occurrence associated with the particular input parameter value and the value of the analyzed results. Therefore, if a parameter is found to be very sensitive within a certain range and insensitive in another range it does not necessarily mean that the parameter is important, since the importance of that parameter should be determined by both sensitivity and its associated probability of occurrence. Using an extreme case as an example, if the probability of occurrence for the above case is found to be zero within the range value of the parameter

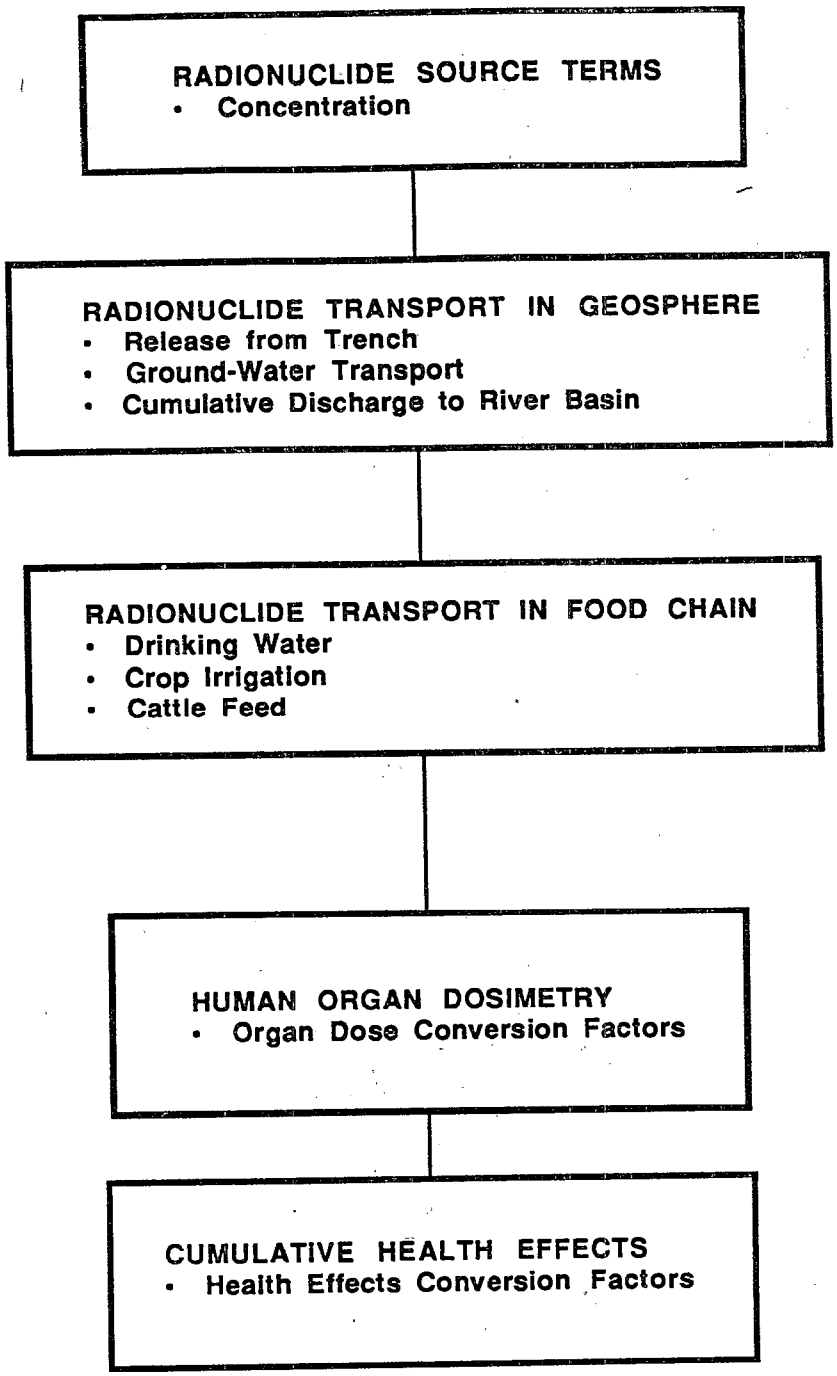

Figure 12-1. Major Components of Uncertainty Analysis: Cumulative Population Health Effects Analysis

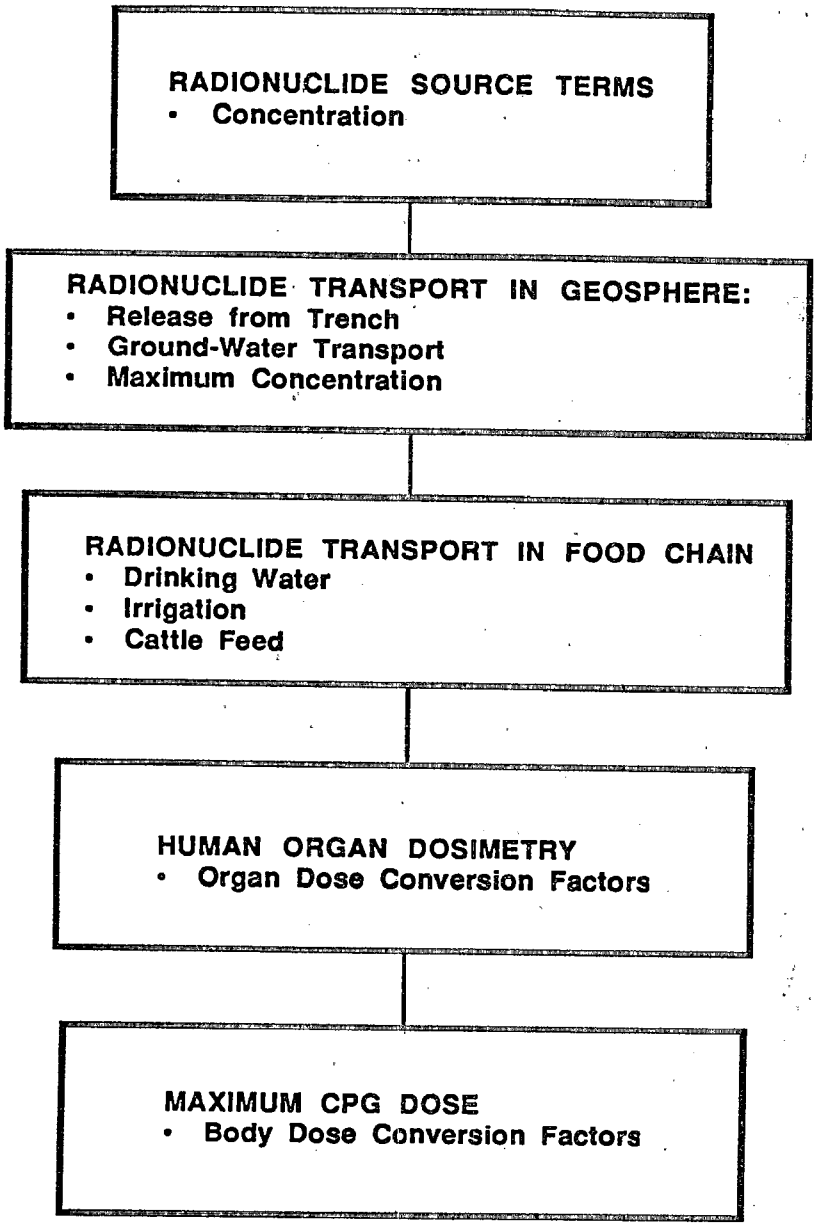

Figure 12-2. Major Components of Uncertainty Analysis: Maximum CPG Dose Analysis

which is found to be sensitive, then the parameter should be judged as unimportant. Even if a sensitivity analysis provides some information concerning the uncertainty of the overall analysis, it can never be used as a substitute for the uncertainty analysis.

On the other hand, the sensitivity analysis has its own merits; it can be performed with much less effort and cost when the deterministic model is already established. Because the quantification of the overall uncertainty of the results obtained from the PRESTO-EPA model is not possible at this stage, the results of the sensitivity analysis, as discussed in Chapter 11, play an important role by providing some information regarding the uncertainty of the assessment result.

12.1.5 Approach to the Uncertainty Analysis

As discussed in Chapters 8 and 11, PRESTO-EPA is a dynamic model simulating the radionuclide transport in the geosphere and food chain pathways, with the use of the organ doses and health effects conversion factors to calculate organ doses and health effects to be incurred from the disposal of LLW. The organ dose and the health effects conversion factors are the input data for PRESTO-EPA models and are obtained from RADRISK, another human organ dosimetry and health effects model.

All of these components of calculation are conducted in series, and the parameters transferred from one component of calculation to another are clearly defined and evaluated in PRESTO-EPA. The uncertainty of each component of calculation can be evaluated separately as a part of the overall uncertainty of the analysis. Thus, the discussion of the uncertainty is divided into the five components displayed in Figures 12-1 and 12-2.

After determining the uncertainty for each component, the uncertainty of the overall dose and health effects assessments can be estimated. Although a quantitative analysis of the uncertainty for each of the components is desirable, such an analysis is limited to the radionuclide transport in the geosphere only, because analyses for the other components are limited by the time and human resources available.

In order to quantify the uncertainty of the analysis for the radionuclide transport in the geosphere, a simplified model from PRESTO-EPA is developed. The model simplifications include:

- Using a quasi-steady-state approximation;
- Converting a numerical model to an analytical model; and
- Considering the humid permeable hydrogeological settings only.

In a quasi-steady-state simplification, the model assumes that the trench cap failure will reach its maximum at the time when the analysis is started, and that the same level of cap failure is maintained for the entire period of analysis. This approximation is essential for

converting the numerical approach to an analytical approach, which greatly reduces the complexity of simulation. This approximation will not introduce serious errors into the results of the risk assessment, and is adequate for the purposes of an uncertainty analysis. The analysis is conducted for a humid permeable site only, because the maximum CPG dose from disposal activities was found to be greatest at this site among the three sites investigated. In any case, it is speculated that the same degree of uncertainty can also be expected for the other two sites.

12.2 Uncertainty Due to Radionuclide Source Term

In order to perform a radiological risk assessment supporting a generally applicable environmental radiation protection standard for the land disposal of LLW, it was necessary to develop a radiological source term representative of LLW to be disposed of in the foreseeable future. The radiological source term consists of best estimates of the radionuclide concentrations and projected volumes of the various categories of LLW. EPA has relied on the best information available to construct the source term used in its radiological risk assessment of LLW disposal and this source term is presented in Chapter 3.

Considering the enormous amount of detail associated with the data base used to construct the EPA source term, it is not possible to perform a rigorous mathematical evaluation of the uncertainty in the EPA LLW source term within the constraints of available resources. However, one can evaluate such an uncertainty in a more qualitative manner by taking into consideration the limitations associated with the data base supporting the LLW source term, the results of the risk analysis (Chapter 9), and the results of the sensitivity analysis (Chapter 11).

12.2.1 Origin of the EPA Source Term for LLW

Extremely diverse radioactive wastes fall under the definition of LLW. EPA has derived its radiological source term for LLW from the vast LLW data base developed by the NRC in conjunction with the development of NRC technical requirements for near-surface LLW disposal facilities, 10 CFR Part 61 (NRC81, NRC82a,b, NRC86). The NRC data base defines numerous waste categories, or "waste streams," each of which consists of a consolidation of groups of wastes having common sources and similar physical, chemical, and radiological characteristics. Earlier NRC assessments (NRC81, NRC82a) for its draft and final EIS examined 36 and 37 waste streams, respectively. Table 3-2 compares the waste streams defined by NRC for its 10 CFR Part 61 rulemaking and the condensed listing of these wastes used in the EPA analysis.

The more compact EPA set of waste streams was achieved either by combining waste volumes from large and small facilities into one waste stream or by combining similar wastes from similar generators into one waste stream. This latter simplification was achieved by weighting

radionuclide concentrations from each contributing waste stream by its proportional contribution of volume to the overall waste stream volume. For example, the EPA waste stream LWR Ion Exchange Resins volume-weighted the radionuclide concentrations of two NRC waste streams, PWR Ion Exchange Resins and BWR Ion Exchange Resins.

NRC's "Update of Part 61 Impacts Analysis Methodology" report (NRC86) further revised, updated, and supplemented NRC's LLW source term characterization. This "updated" report provided more detail on the higher specific activity waste streams and nonroutine or unusual low-level waste streams. Updated information on routine LLW streams was used to provide revised radionuclide concentrations for those affected EPA waste streams as shown in Chapter 3.

Although they are discussed in Chapter 3, nonroutine or unusual low-level waste streams were not included in the EPA source term because there is much uncertainty over the timing and characteristics of such wastes. Moreover, such wastes do not comprise a significant fraction of projected LLW volumes or activities. In summary, the EPA radiological source term for LLW is based upon the most complete and recent information available for routine sources of commercial LLW. DOE has indicated that its LLW are similar to commercial LLW, but a breakdown comparable to that provided by the NRC for commercial wastes is not available (DOE86).

EPA has made one minor addition to the basic LLW source term for commercial wastes. The EPA source term includes two Naturally Occurring and Accelerator-produced Radioactive Materials (NARM) waste streams: radium sources and radium-contaminated water treatment ion exchange resins. These two waste streams comprise less than one percent of the commercial LLW volume and an even smaller percentage of the total activity. They have been included to reflect EPA's intention to regulate high specific activity, low volume NARM wastes under the TSCA.

12.2.2 Uncertainties Associated with Data Bases

Since EPA has relied heavily upon the NRC characterization of LLW (NRC81, NRC82, NRC86), uncertainties introduced into the NRC source term for LLW would also apply to the EPA source term. The major sources of information supporting the development of the NRC and EPA characterizations of LLW include:

- Computer-assisted calculations;
- Surveys of waste generators;
- Disposal site records; and
- Radiochemical analysis.

The following discussion will focus on the limitations of each of these information sources as they relate to the characterization of LLW.

Computer-assisted calculations are typically used to estimate the radionuclide composition and quantities generated by "burn-up" of nuclear fuels. Such models are based on numerous parametric values for a given power reactor design. As such, they are reasonably well-suited to identifying important radionuclides that are produced in the nuclear fuel and activated in the surrounding structural materials. Since virtually every reactor design is different, computer-assisted calculations performed for one or several plant designs would introduce some error when attempting to project LLW characteristics over all reactors. Use of one or more of the remaining information sources in conjunction with such computer calculations could reduce the uncertainty associated with any LLW predictions.

Past surveys of waste generators or disposal facility site records may have certain limitations. In practice, radionuclide distributions listed on such records frequently were calculated by applying predetermined radionuclide distributions to the total gross radioactivities obtained during screening measurements made at the time of shipment. Such measurements were probably conservative in terms of the total radioactivity measured since less sophisticated measurement techniques have been applied in the past, and because the radioactivity contribution of short-lived isotopes was included in the total activity reading. When predetermined radionuclide distributions are used, changes in actual radionuclide concentrations on a day-to-day basis may have been missed as well.

The sensitivities (minimum detection limits) of the analytical procedures for the various radionuclides are not identical, especially with respect to "hard-to-measure" radionuclides (e.g., C-14, I-129). Such radionuclides are more likely to be "scaled" from previous specialized measurements, using a "scaling factor" associated with a more abundant, easy-to-measure nuclide. This scaling factor would then be applied on a routine basis as a calculational tool, with the possibility that day-to-day variations in the actual radionuclide concentration would be overlooked.

In order to minimize the uncertainties associated with these information sources, NRC has updated its LLW source term for those LLW streams contributing large volumes, such as power reactor LLW, or those waste streams possessing relatively high specific activities, such as certain industrial LLW (NRC86). More recent LLW shipment records, more detailed surveys of certain waste generators, and, in general, more up-to-date waste volume generation rates all contributed to the revised NRC characterization of commercial LLW. These improvements were subsequently incorporated into the EPA LLW source term.

12.2.3 Estimated Uncertainty

As the discussion above indicates, numerous information sources have contributed to the development of the EPA characterization of LLW. For

each of the four categories of information sources cited above, numerous individual data bases have contributed to the characterization of LLW. To characterize the uncertainty associated with each individual source of data would be a monumental task well beyond the scope of this effort.

A more manageable approach is to develop a qualitative characterization of the uncertainty associated with the EPA LLW source term. The results of the health impacts assessment, discussed in Chapter 9, indicated that the two most important radionuclides, considering both CPG dose and population health effects, are C-14 and I-129. Other radionuclides contribute virtually nothing to the CPG dose and on the order of 20 percent or less to population health effects. Therefore, the remaining discussion of uncertainty in the EPA LLW source term will focus on C-14 and I-129.

Characterization of source terms for C-14 in LLW has received particular attention over the last few years. Such attention is well-deserved considering that C-14 is found in many categories of LLW and has a long half-life (5,700 years), high mobility via water pathways, and a relatively high dose conversion factor. The NRC has continually updated its characterization of C-14 in LLW (NRC81, NRC82, NRC86). The most recent update greatly improved the characterization of C-14 occurring in many higher activity waste streams. A recent EPA study of C-14 in LLW reviewed the various source terms (Gr86).

Each of the three major categories of LLW streams (nuclear fuel cycle, institutional, and industrial) was reviewed by comparing the EPA source term to independent estimates of LLW containing C-14. In all three cases, the EPA source term was found to be a "reasonable representation" of the C-14 in all three categories of LLW (Gr86). A recent NRC document reports on the results of the analyses of hundreds of process and waste samples from power reactors in an effort to establish useful correlation factors between "easy" and "difficult" to measure radionuclides (Cl85). Analysis of the data suggested that an empirical scaling factor for C-14 with Co-60 would be most useful. The scaling factors derived separately for BWRs and PWRs showed statistical uncertainties in the range of 27 to 45 percent. This implies that for any given reactor design, the concentration of C-14 in a given waste may vary up to plus or minus 50 percent (rounded). For institutional wastes containing C-14, average concentrations of C-14 in the EPA source term were compared with reported average concentrations. The average C-14 concentration in institutional wastes was reported by EPA as 5.1 E-3 Ci/m^3 for 1982. At the same time, the CRCPD reported an average concentration of 1.3 E-2 Ci/m^3 for 1982. For the year 1983, the National Institutes of Health cited an average concentration of 2.4 E-3 Ci/m^3 for C-14 in its LLW (Gr86).

For institutional wastes, these estimates represent a variation of about a factor of 2 from the EPA source term. More limited data are available for industrial sources of C-14, although manufacturers of

chemical compounds labeled with C-14 (and H-3), the major source of C-14, appear to be reasonably well characterized by detailed generator surveys (NRC86, Ke85). Considering the above evaluations, a qualitative uncertainty of a factor of 2 up or down is assigned to the EPA source term characterization of the C-14 concentrations in LLW.

Like C-14, I-129 possesses a long half-life (17 million years), high mobility via water pathways, and a relatively high dose conversion factor. I-129 occurs in fewer LLW streams, however. Other than power reactor waste streams, I-129 is found only in wastes from industrial radioisotope manufacturers. The contribution of I-129 from industrial radioisotope manufacturing waste is very small because of the extremely low I-129 concentrations and relatively small volumes of such wastes (see Chapter 3). Therefore, this discussion will concentrate on I-129 in power reactor LLW. The NRC and EPA characterization of I-129 in LLW relies upon previous work that attempted to derive a "scaling factor" for I-129, as related to the measured concentration of Cs-137 in the same sample. Since there were so few samples in which both were measured, statistical averaging was not possible (NRC81). A more recent investigation also attempted to develop scaling factors for I-129 in LLW (Cl85). In this study I-129 was compared with Cs-137, since both are fission products, have similar transport properties in reactor systems, and release mechanisms from reactor fuel. In this case, however, only a small percentage of BWR samples (16 out of 191) and PWR samples (22 out of 259) contained both I-129 and Cs-137. Most of the time, I-129 was at or near its detectable limit. Thus, the scaling factors used to estimate the I-129 activities in the waste contain large uncertainties, typically ranging from 50 to 90 percent. These results suggest that I-129 concentrations in LLW are highly variable. On the other hand, since I-129 was detected so seldom, it is likely that I-129 does not occur very often in LLW at levels comparable with detectable limits. Based on the measured data, therefore, a factor of 2 up or down may be reasonably assigned to the uncertainty of the I-129 concentration in LLW. However, it is felt that the EPA characterization of I-129 in LLW is probably very conservative (i.e., too high).

In summary, the results of EPA's risk analyses (see Chapter 9) have identified two radionuclides, C-14 and I-129, as predominant in producing exposures to the critical population group (CPG) and causing total population health effects. Information concerning the occurrence of these two radionuclides in LLW indicates that the concentration of each, as used in the EPA source term, has an uncertainty of approximately a factor of 2 up or down; that is, the likely concentration of each radionuclide may be as large as twice the EPA concentration or as small as one-half the EPA concentration. Data with respect to I-129 indicate that the EPA source term is probably conservative, however.

12.3 Uncertainty Due to Radionuclide Geosphere Transport for Cumulative Health Effects Analysis

Since the output requirements for the two analyses, assessment of maximum CPG dose and assessment of the cumulative population health

effects, are different, the uncertainties of these analyses are treated separately. This section discusses the uncertainty for the cumulative population health effects analysis due to geosphere transport. The key difference in the two uncertainty analyses for the radionuclide transport in the geosphere is that the cumulative health effects analysis requires an analysis of the cumulative radionuclides being discharged into the regional river basin, while the maximum CPG dose analysis requires the analysis of the maximum annual average concentration of radionuclides at the nearest accessible environment.

12.3.1 Method of Analysis

In order to evaluate the cumulative radionuclide release, a leaching-release model simplified from the PRESTO-EPA model is employed. After imposing the simplifications discussed in Section 12.1.5, the unsteady-state leaching-release model used in PRESTO-EPA was converted into a steady-state solute transport system model for which an analytical solution is obtainable. The rate of the radionuclides being discharged into the regional river basin is established using Hung's ground-water transport model, which is the same transport model used in PRESTO-EPA (Hu86).

For the humid permeable site, the total health effects incurred from LLW disposal are dominated by the effects of the residual radionuclides being discharged into a regional river basin for 10,000 years of analysis. The health effects incurred from the local community (made up of a few farmhouses) for 10,000 years of analysis are therefore combined into the downstream river basin effects. Based on this simplification, the cumulative radionuclides being discharged into the regional river basin can be obtained by integrating the discharged activity over the time frame of 10,000 years. The result is expressed by:

$$Q_T = \eta \frac{I_o E}{\varepsilon VR} \frac{1}{\lambda_d + E/\varepsilon\ VR} \left[\mathrm{Exp}(-\lambda_d t_L) - \mathrm{Exp}\left\{ -(\lambda_d + E/\varepsilon VR) \cdot 10{,}000 + \frac{E t_L}{\varepsilon\ VR} \right\} \right] \qquad (12\text{-}1)$$

where:

Q_T = the cumulative radionuclide being discharged into the regional river basin;

η = Hung's ground-water transport correction factor;

I_o = the initial radionuclide inventory;

E = the equivalent rate of infiltration through the trench cap (a constant);

ε = the porosity of the waste material;

V = the volume of waste;

R = the radionuclide retardation factor;

λ_d = the radionuclide decay constant; and

t_L = the sum of radionuclide transit time through the host soil (between waste trench and aquifer) and aquifer (between disposal site and the ground-water discharging point).

This equation brings together the major parameters that will significantly affect the cumulative radionuclide release and, subsequently, the total health effects. Equation 12-1 also shows that the cumulative radionuclide release can be expressed as a simple mathematical function of major parameters, and therefore the uncertainty of the cumulative radionuclide release can be calculated by the analytical method proposed by Hung (Hu87), instead of using a time-consuming Monte Carlo or other simulation method. Hung's method calculates the joint probability density distribution for two successive random variables based on Equation 12-1. The overall joint probability density distribution is the uncertainty of the cumulative amount of radionuclide being discharged into the regional river basin.

12.3.2 Postulated Probability Density Distribution of Parameters

Because limited data are available for analyzing the probability density distribution of the major parameters appearing in Equation 12-1, values for the arbitrary distribution of probability density distribution are estimated for this analysis on the basis of engineering judgment. The input parameters selected for the analysis are the radionuclide distribution coefficients for the trench material, the host soil, and the aquifer; the degree of trench cap failure; the distance from trench bottom to aquifer; the distance from the disposal site to the regional river basin; the ground-water velocity in the aquifer; and the percolation velocity in the host soil. The assumed probability density distribution for each parameter is normalized and presented in Figures 12-3 through 12-7.

Probability density distributions have not been assigned to some of the input parameters because (1) the probable standard deviation is so small that it can be considered a deterministic variable, or (2) the probability density distribution is programmed to be calculated from other random input parameters.

12.3.3 Results of Uncertainty Analysis

A computer program was developed based on the methodology described in Section 12.3.1 for the uncertainty analysis of the cumulative radionuclide being discharged into the regional river basin. The

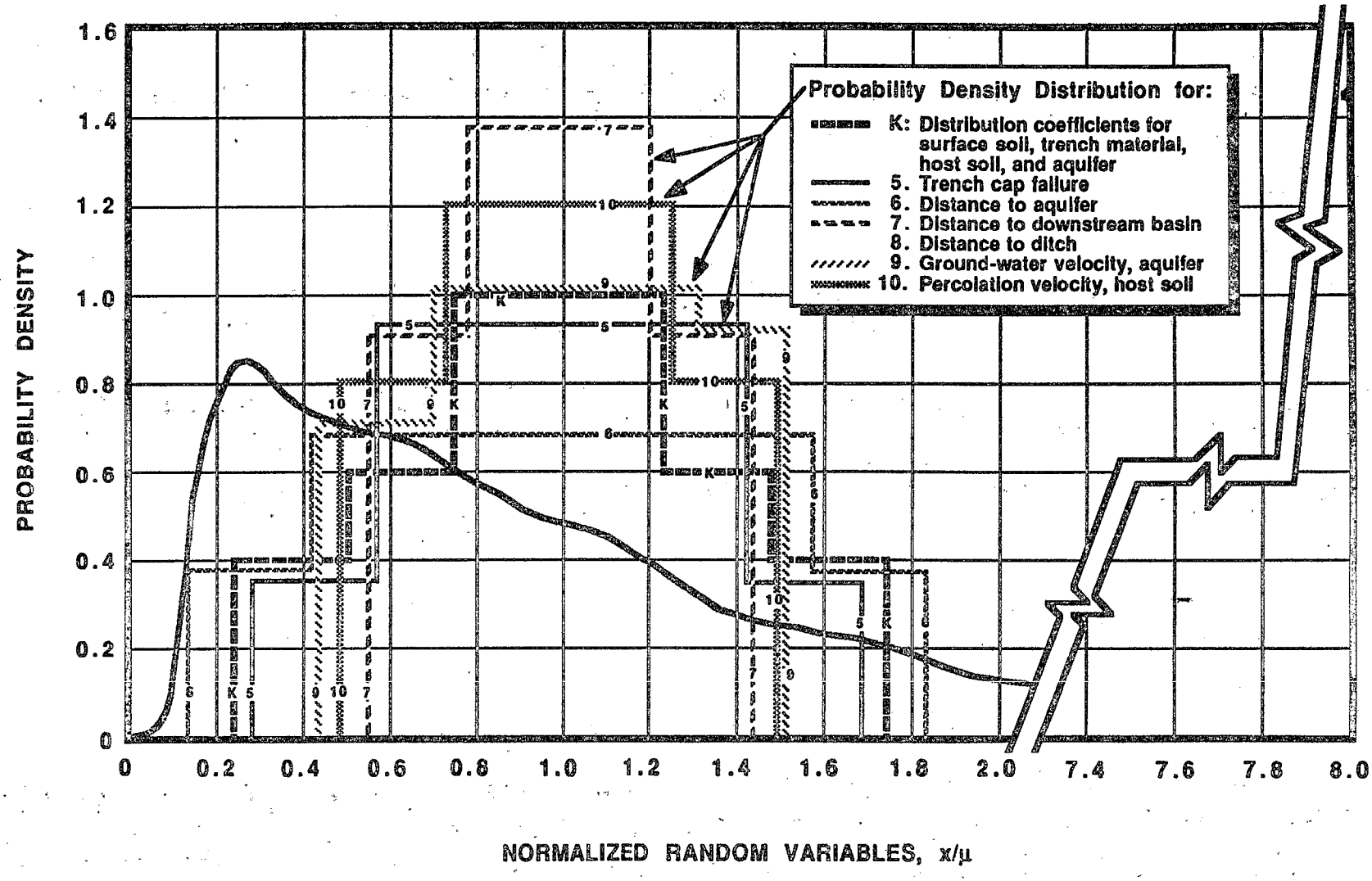

Figure 12-3. Results of Uncertainty Analysis for H-3

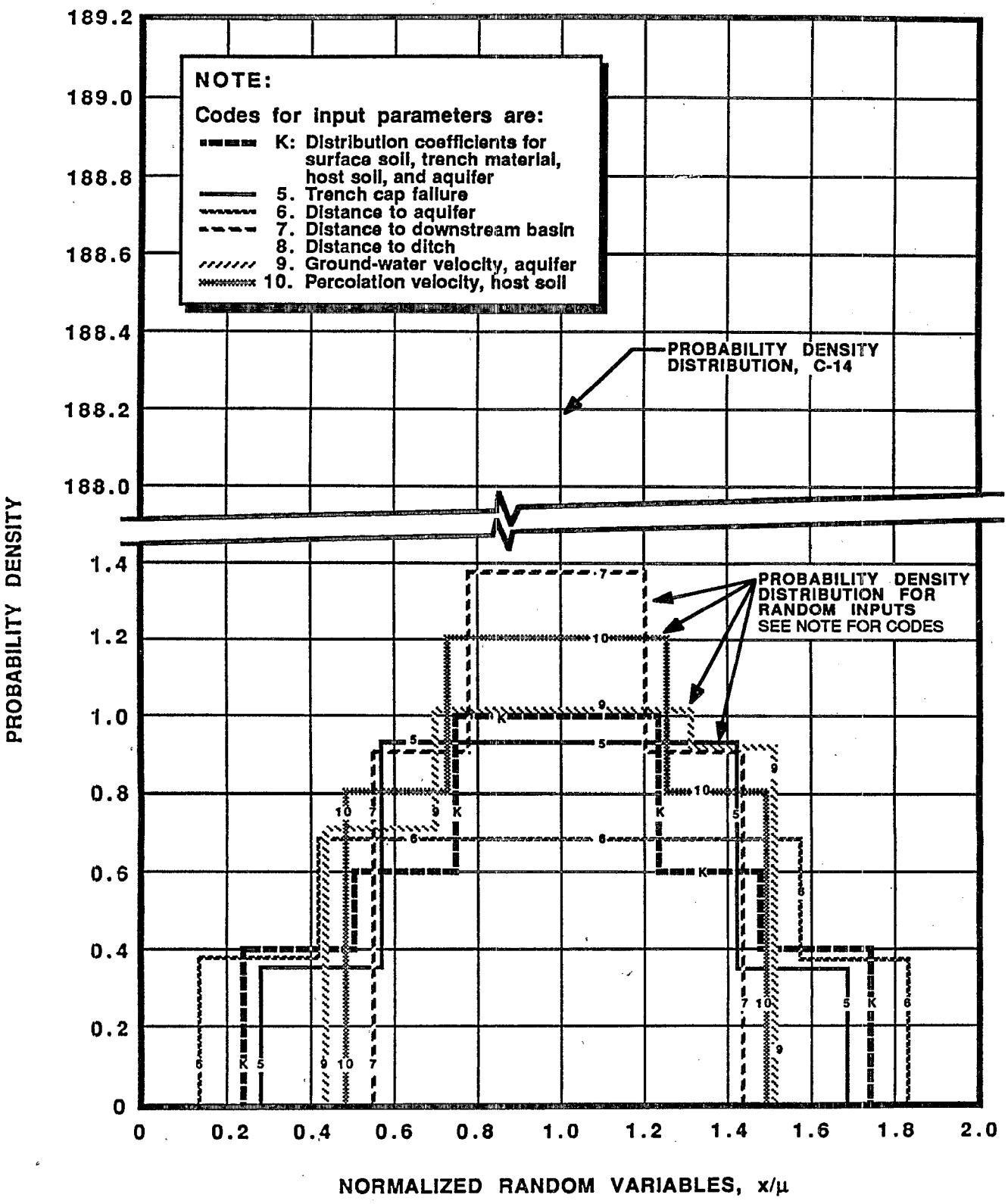

Figure 12-4. Results of Uncertainty Analysis for C-14

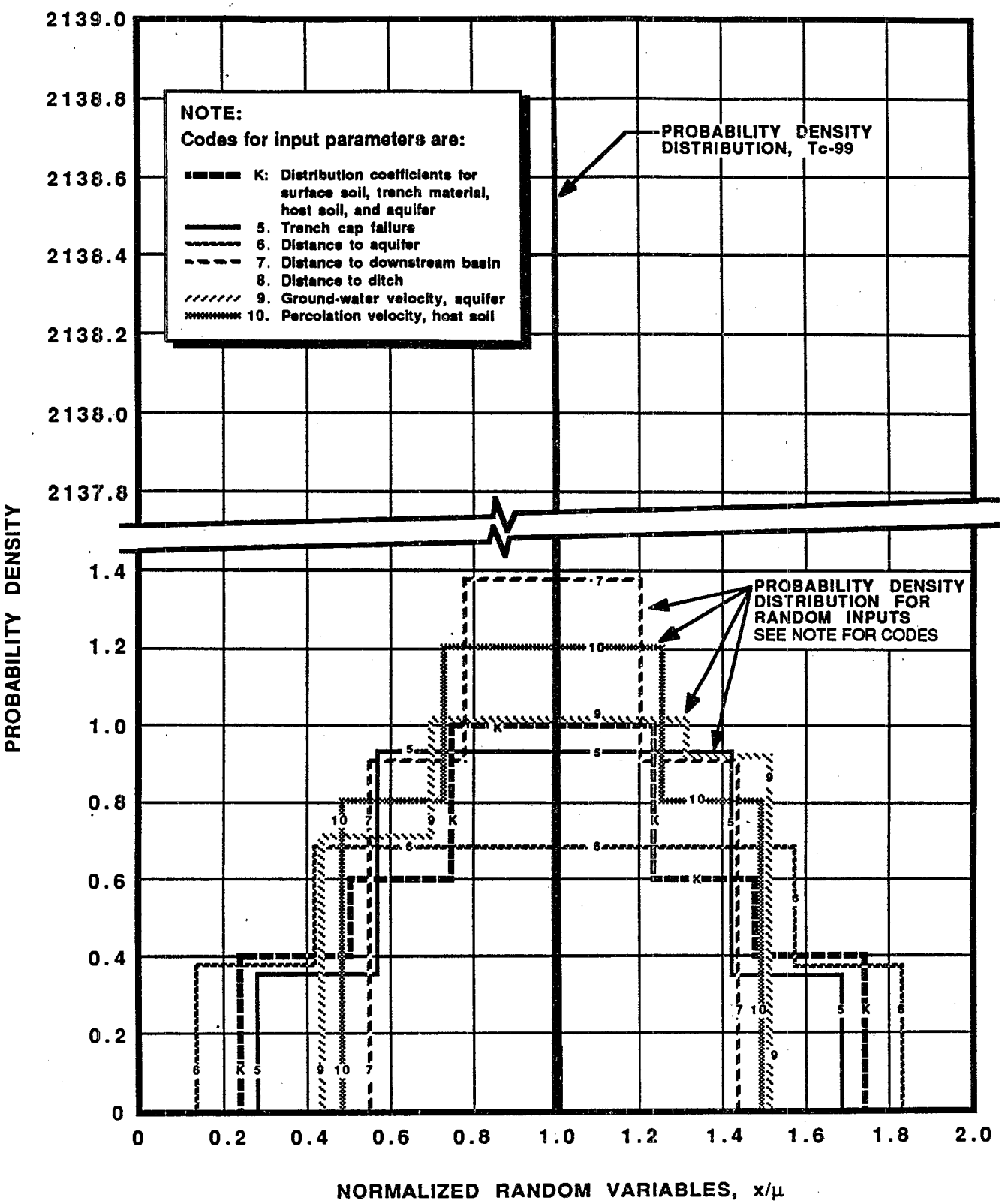

Figure 12-5. Results of Uncertainty Analysis for Tc-99

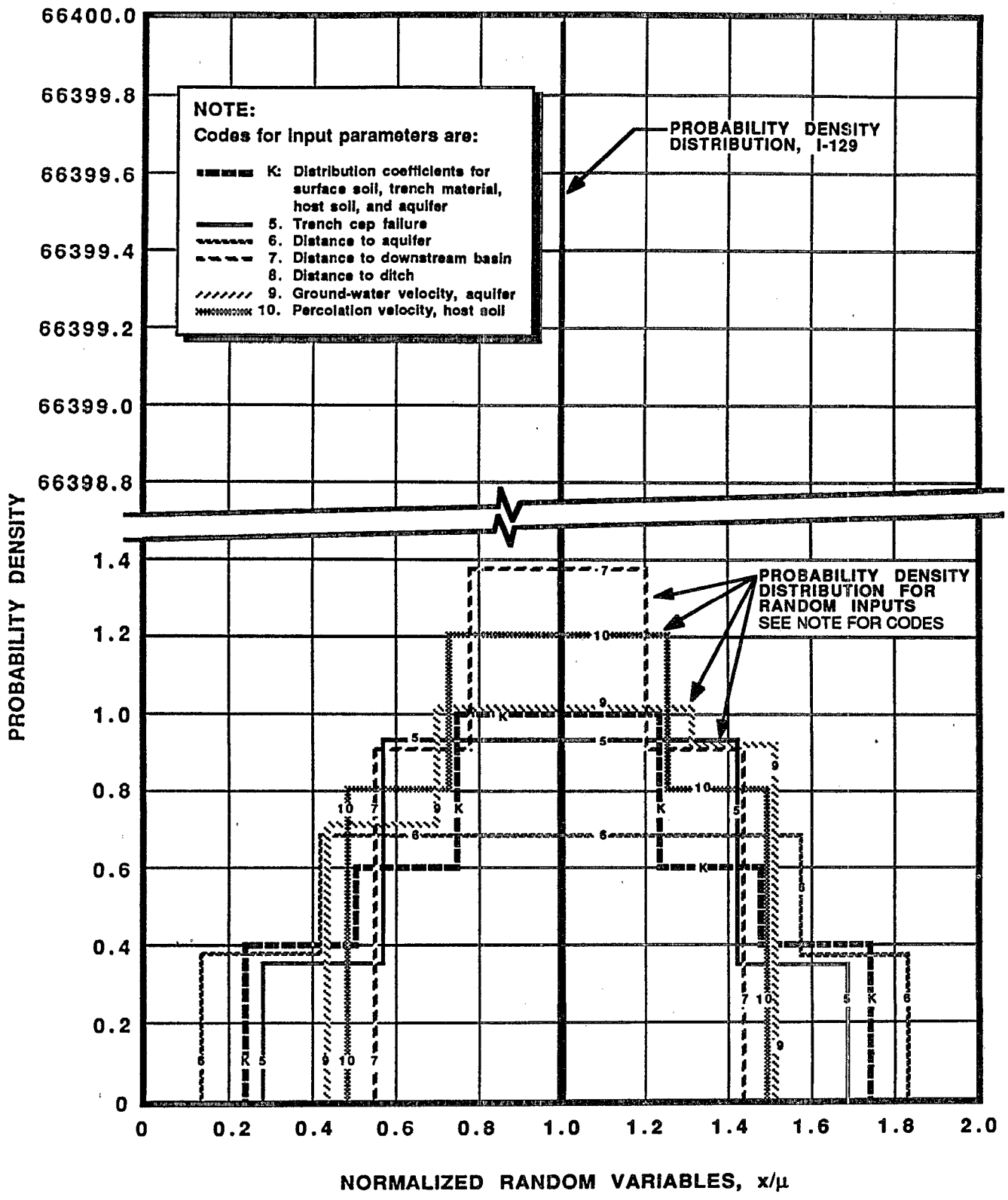

Figure 12-6. Results of Uncertainty Analysis for I-129

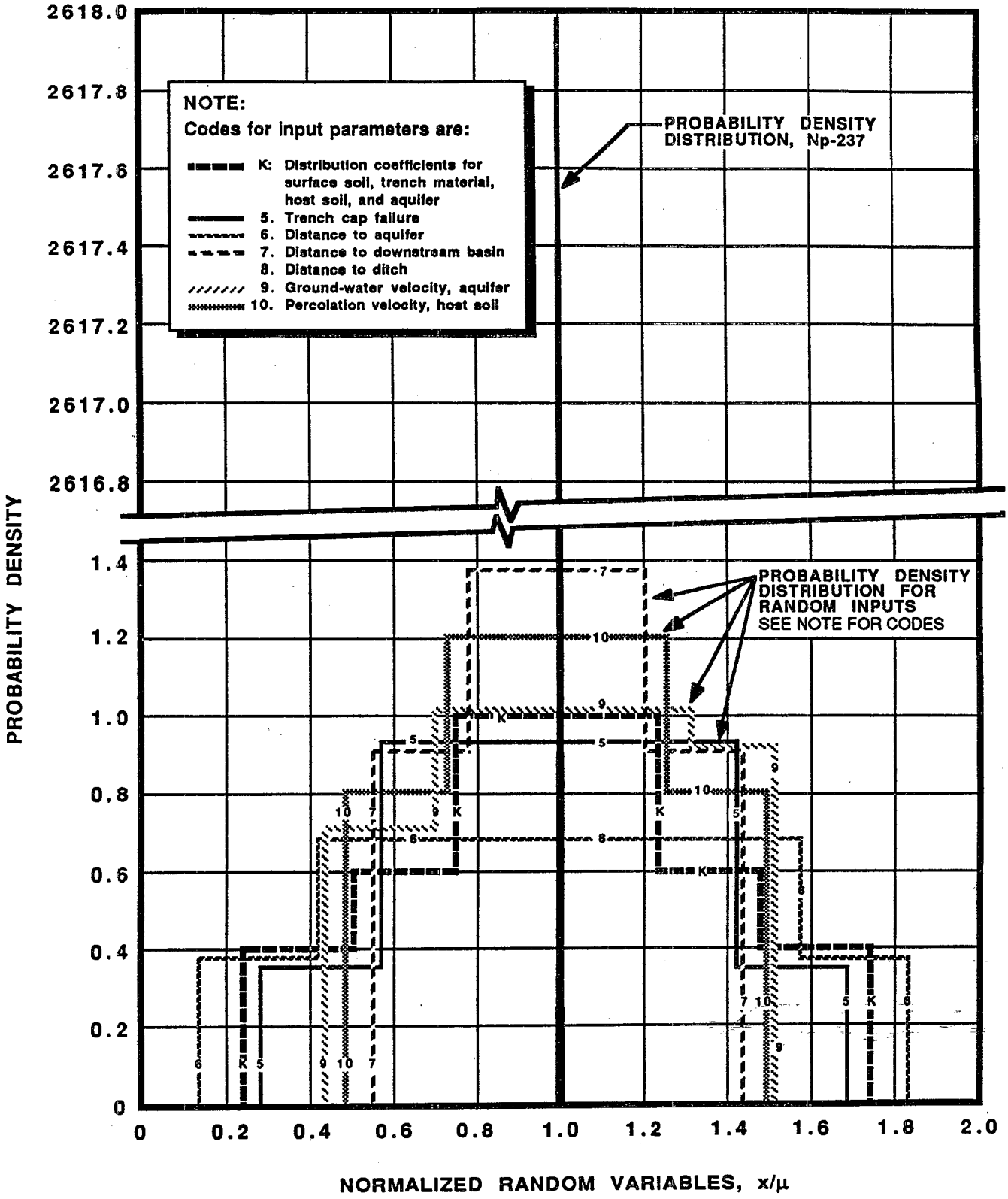

Figure 12-7. Results of Uncertainty Analysis for Np-237

computer program was implemented on a personal computer AT or a compatible. By using the estimated probability density distribution for the input parameters described in Section 12.3.2, the uncertainty of the cumulative radionuclide being discharged to the regional river basin is analyzed.

The normalized results of the uncertainty analysis are presented in Figures 12-3 through 12-7, respectively, for H-3, C-14, Tc-99, I-129, and Np-237 (for a humid permeable site only). The analysis is conducted solely for these radionuclides because the other relatively immobile radionuclides are retained either in the waste trench or in the aquifer at the end of analysis and did not contribute to the total health effects.

The results indicate that the uncertainty or the standard deviation of the cumulative activities being discharged into the regional river basin for each radionuclide is practically zero except for H-3. This is due to the fact that when the radionuclide transit time through the geosphere is prolonged because of a higher distribution coefficient, there would be additional loss from radioactive decay, and vice versa. Since the half-lives of these radionuclides, other than H-3, are relatively long, the change in the radioactive decay loss due to the change in the transit time is negligibly small and thus the uncertainty of the results is small also. On the other hand, H-3 has a relatively short half-life, so that there is a significant effect on the cumulative activity of H-3, which can be transported to the regional river basin due to the change in radionuclide transit time.

The above results are expected to remain unchanged even if there are slight changes in the probability density distributions of input parameters.

12.3.4 Summary

Since C-14 is the critical radionuclide contributing a major portion of the health effects for all three generic sites analyzed (see Chapters 9 and 10), and since the uncertainty of the analysis for the cumulative activity of C-14 being discharged into the regional river basin is near zero, one may logically conclude that the uncertainty for the cumulative health effects assessment due to geosphere transport is near zero.

The results of analyses for Tc-99, I-129, and Np-237 indicated that they have characteristics that are similar to those of C-14 (Figures 12-5 through 12-7); that is, their uncertainties may also be considered to be zero. Figure 12-3 showed that the uncertainty for H-3 is analyzed to be approximately 65 percent.

The uncertainty of the cumulative population health effects analysis for the humid impermeable site was not analyzed, because the total population health effects assessments for this region were much smaller

than that for a humid permeable region and thus will not play as important a role as that for the humid permeable region. Furthermore, the radionuclide release pathway--trench overflow pathway--analyzed for a humid impermeable region could possibly be avoided in future designs by using an improved engineering disposal method.

12.4 Uncertainty Due to Geosphere Transport for Maximum Dose Analysis

As discussed in Section 12.3, the maximum annual average concentration of radionuclides being released to the nearest accessible environment is the output parameter generated from the geosphere transport analysis for the maximum CPG dose analysis. This information is transmitted to the food chain calculation (see Figures 12-1 and 12-2). The risk assessment conducted in support of the development of EPA's LLW standards assumed that a farmhouse well is located right on the fence line and that the well continues to operate at the same location even after institutional control is lifted. The fence line is assumed to be 100 m from the edge of the trench area.

12.4.1 Method of Analysis

The basic equation used to calculate the maximum annual average concentration of radionuclides in the accessible environment combines the simplified leaching-release model and the ground-water transport model used in PRESTO-EPA-CPG. The same simplifications presented in Section 12.3 are also used for the leaching-release model.

Because the critical radionuclides that contributed over 90 percent of the maximum CPG dose were long half-life radionuclides (I-129 or C-14), the maximum concentration of a radionuclide is found to occur at the time when the contribution of the radionuclide from the far end of the trench area reaches the well. The time required to reach its maximum concentration is known as the time of arrival. Knowing the time of arrival, the maximum concentration of a specific radionuclide is obtained by integrating the contributions from each subdivided segment over the entire trench area at the time of arrival. The analytical solution for this integration is (HU87):

$$C_{max} = \frac{\eta I_o}{A\, T_{aq}\, \epsilon_{aq}\, R_H} [Exp\{-\lambda_d (t_V + t_L + t_w\}]$$

$$[1-Exp(-\xi \frac{E}{\epsilon \underline{V} R} t_w) \qquad \text{and}$$

$$t_V = \frac{R_v d}{V_v},\ t_L = \frac{R_h L}{V_h},\ t_w = \frac{R_h X_w}{V_h} \qquad (12\text{-}2)$$

where:

C_{max} = the maximum concentration of radionuclide at the well;

A = the total surface area of the disposal trenches;

T_{aq} = the thickness of the aquifer;

ϵ_{aq} = the porosity of the aquifer;

R = the retardation factor for the specific radionuclide;

d = the distance between the trench bottom and the top of the aquifer;

V = the interstitial ground-water velocity;

L = the length of the disposal site without including a buffer zone and measured in the ground-water flow direction;

X_w = the distance from the near edge of the disposal site (excluding the buffer zone) to the well;

ξ = the leaching rate correction factor;

E = the equivalent rate of infiltration;

$\underline{V}$ = the volume of waste material (including the backfill material); and

subscripts, h, v, and w designate aquifer, host soil, and waste material, respectively.

Equation 12-2 demonstrates that the parameters appearing in the equation will contribute significantly to the maximum annual average radionuclide concentration and are in a simple mathematical relationship with the model output. It should be noted that those parameters which do not have any significant effect on the radionuclide concentration are discarded from Equation 12-2. Equation 12-2 implies that the uncertainty of the maximum annual average concentration of a radionuclide at the well can be calculated by the analytical method proposed by Hung (Hu87).

12.4.2 Estimated Probability Density Distribution of Input Paramaters

For the same reasons as were stated in Section 12.3.2, for this analysis the distributions of probability density for each random input parameter are estimated through engineering judgment. The predominant random input parameters selected for the analysis are: (1) the distribution coefficients for trench material, host soil, and the aquifer; (2) the degree of trench cap failure; (3) the distance from the trench bottom to the top of the aquifer; (4) the length of the disposal site in the direction of ground-water flow; (5) the ground-water velocity in the host soil; and (6) the ground-water velocity in the aquifer.

Their distributions are shown in Figures 12-8 and 12-9. The remainder of the input parameters appearing in Equation 12-2 are considered to be deterministic numbers, and thus the same numbers as are used in PRESTO-EPA-CPG for the humid permeable site are assigned.

12.4.3 Results of Uncertainty Analysis

A separate computer program was also developed for the uncertainty analysis of the maximum annual average radionuclide concentration based on Equation 12-2. Using the probability density distribution of the input parameters discussed previously, the uncertainty of the maximum concentration in the well is analyzed. The results of the uncertainty analysis for radionuclides C-14 and I-129 are presented in Figures 12-8 and 12-9 in a form of a normalized probability density distribution. Analyses are conducted only for C-14 and I-129 because they acted as the predominant radionuclides, i.e., those that contribute the major portion of the maximum CPG dose, for all scenarios analyzed for the humid permeable site. One should notice that the contributions of C-14 and I-129 to the maximum body dose would occur in different time frames because of the difference in retardation factors.

The results indicated that the uncertainty or the standard deviation of the maximum annual average concentration for C-14 and I-129 in the well (occurring at different time frames) are approximately 8 percent and 26 percent of the mean values, respectively. Compared to the standard deviation for the input parameters, which have values ranging from 12 to 25 percent, the output parameters are considered to be converging from the uncertainties of input parameters for C-14 and diverging very slowly from the uncertainties of input parameters for I-129.

The analysis also indicated that the uncertainty of the maximum radionuclide concentration does not increase in proportion to the increase in the uncertainty of the input parameters. This is due to the fact that when the distribution coefficient increases, the initial release rate will decrease and the time of arrival of the peak concentration at the well will increase. These two effects, known as primary effects, tend to minimize the peak concentration at the well. In addition, when the rate of the initial leaching rate decreases, the rate of the radionuclide inventory depletion rate will decrease as well. This secondary effect will tend to increase the subsequent radionuclide-leaching rate and to slow down the decrease of the leaching rate. This secondary effect is also known as buffer action.

In the same manner, when the distribution coefficient decreases, the initial radionuclide release rate will increase and the time of peak concentration arrival will decrease. These two primary effects will tend to maximize the peak concentration at the well. Conversely, when the rate of the initial leaching rate increases, the rate of the radionuclide depletion rate also increases, which will tend to decrease the subsequent leaching rate. This secondary effect will also tend to slow down the increase of the leaching rate.

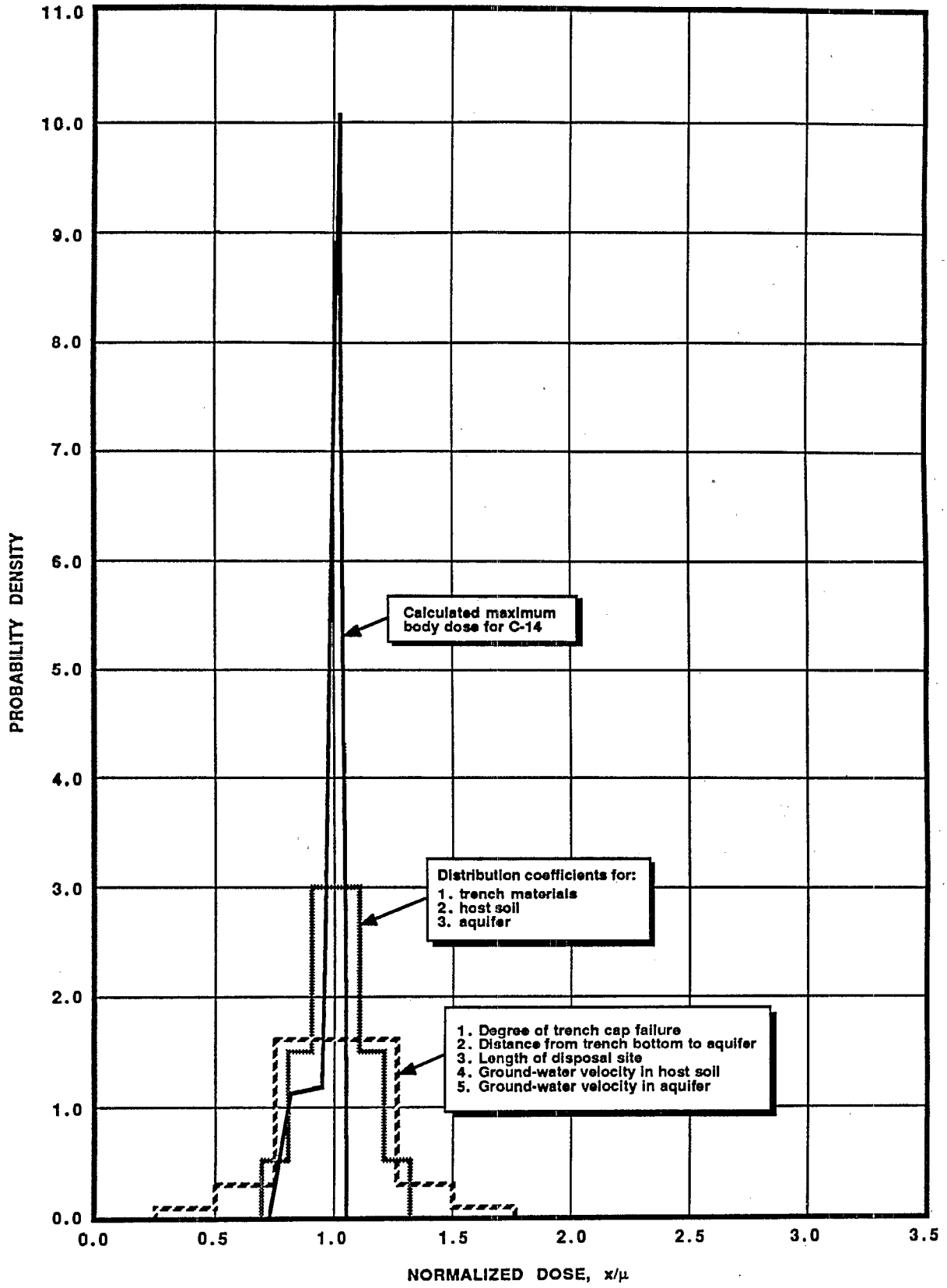

Figure 12-8. Results of Uncertainty Analysis for C-14, Standard Deviation = 8%

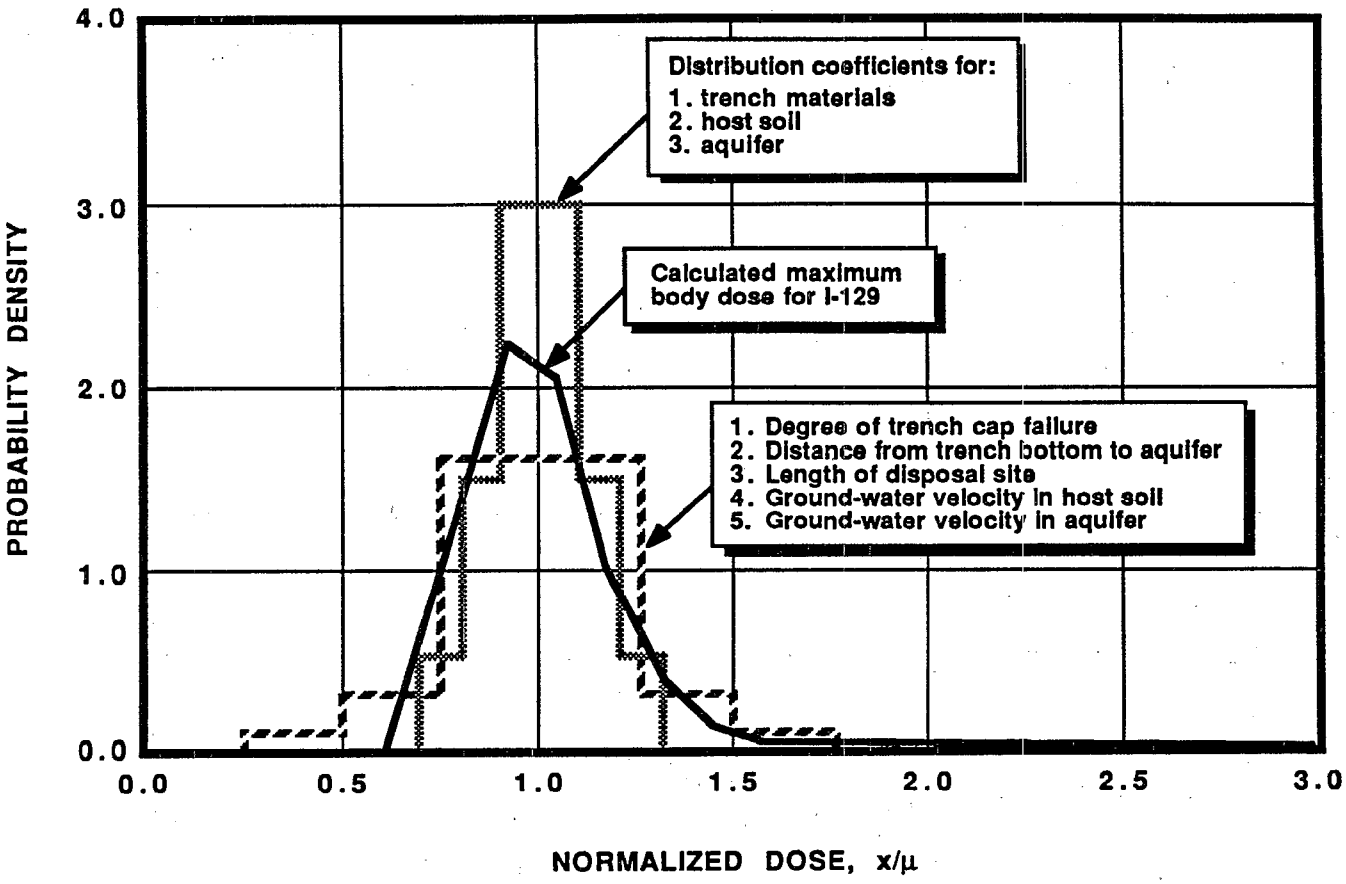

Figure 12-9. Results of Uncertainty Analysis for I-129, Standard Deviation = 26%

For the mobile radionuclides, the secondary effect is less sensitive than the primary effects. The sensitivity of the secondary effect relative to primary effects decreases with the increase in the radionuclide distribution coefficients. This phenomena can be seen from the results of the uncertainty analysis. The uncertainty of the maximum concentration analysis for C-14 (having the most probable distribution coefficient of 0.01 ml/g) is less than that for I-129 (having the most probable distribution coefficient of 3.0 ml/g). To demonstrate this tendency, an uncertainty analysis for Ra-226 was also conducted, and the results are presented in Figure 12-10. The results indicate that the standard deviation for the analysis is 65 percent, which is larger than the uncertainty for I-129 and far greater than the uncertainty for C-14. This is because Ra-226 is a relatively long half-life radionuclide (similar to C-14 and I-129) and has a medium capacity of desorption (distribution coefficient = 220 ml/g), which is much greater than the distribution coefficients for C-14 and I-129.

The half-life of the radionuclide will also significantly influence the uncertainty of the maximum concentration analysis because of the secondary effect from the retardation of the radionuclide in the aquifer. To demonstrate this tendency, an uncertainty analysis was conducted for H-3 to represent radionuclides with high mobility and a relatively short half-life. The results are presented in Figure 12-11. The results show that despite the mobile nature of H-3, an uncertainty of 47 percent could be expected, which is much greater than the uncertainty for C-14 having similar mobility. The difference is primarily due to the difference in half-lives.

12.4.4 Summary

As indicated in Section 12.4.3, the predominant radionuclides that contribute the major portion of the maximum CPG dose are C-14 and I-129; the uncertainties of the results of the maximum CPG dose analyses for C-14 and I-129 are 8 percent and 26 percent, respectively. By considering the additional uncertainty in the estimation of the probability density distribution of each input parameter, one may claim that the upper bound of the uncertainties of the results of maximum CPG dose analyses due to the geosphere transport is on the order of 10 percent of its mean value for C-14 and 40 percent of its mean value for I-129.

12.5 Uncertainty Due to Transport in the Food Chain Pathway

Given a model for predicting the concentration of radionuclides in foods, the effect of uncertainties in the parameters can be determined. There are some important limitations to this approach, however. First, any empirical model is, at best, descriptive of observations, but is neither exact nor complete. To the extent that significant processes are not included in the model, their contributions to the uncertainty cannot be determined. Second, the appropriate distributions, and inter-dependence of the model parameters, are seldom well known. While one can

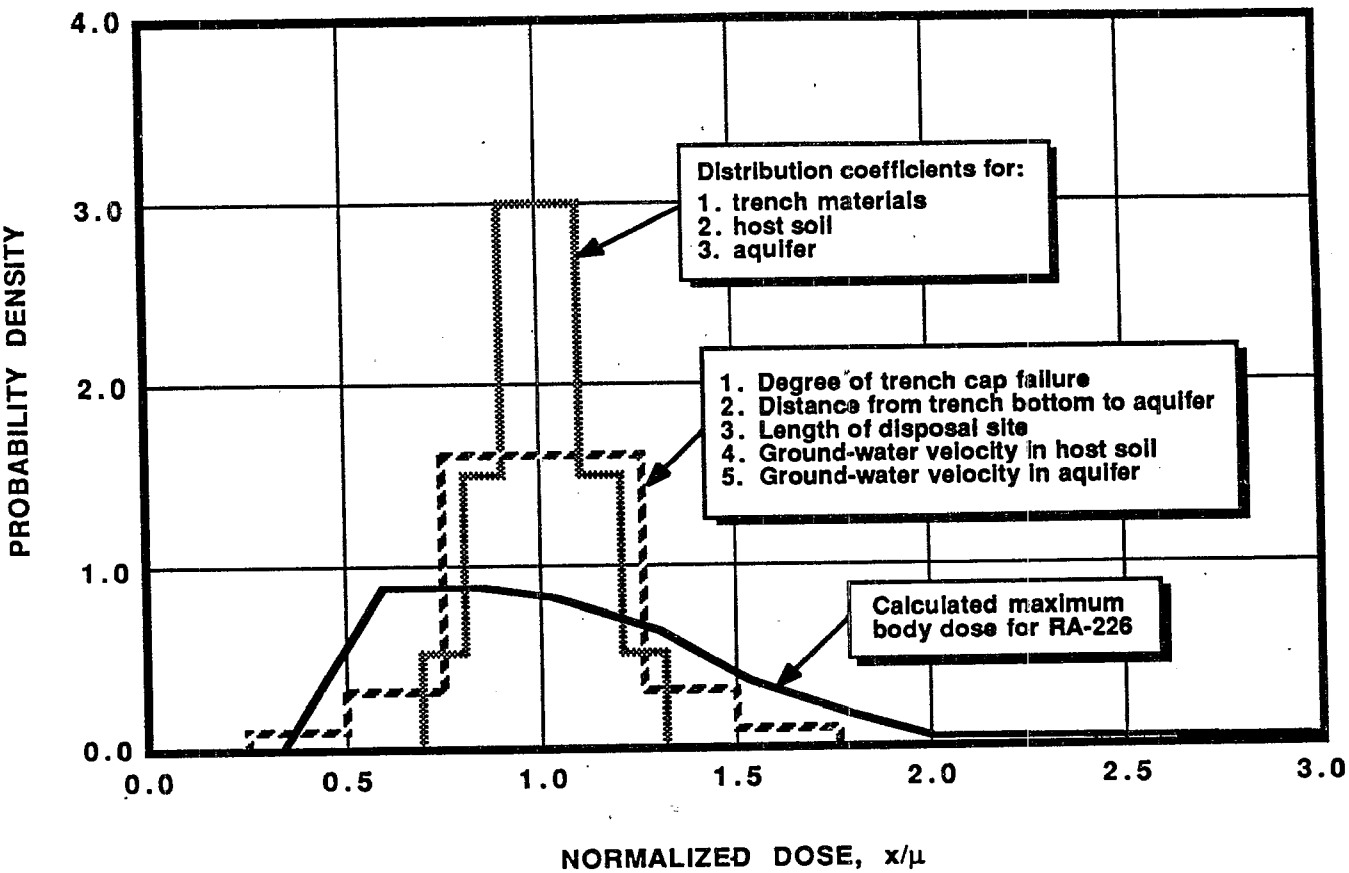

Figure 12-10. Results of Uncertainty Analysis for Ra-226, Standard Deviation = 65%

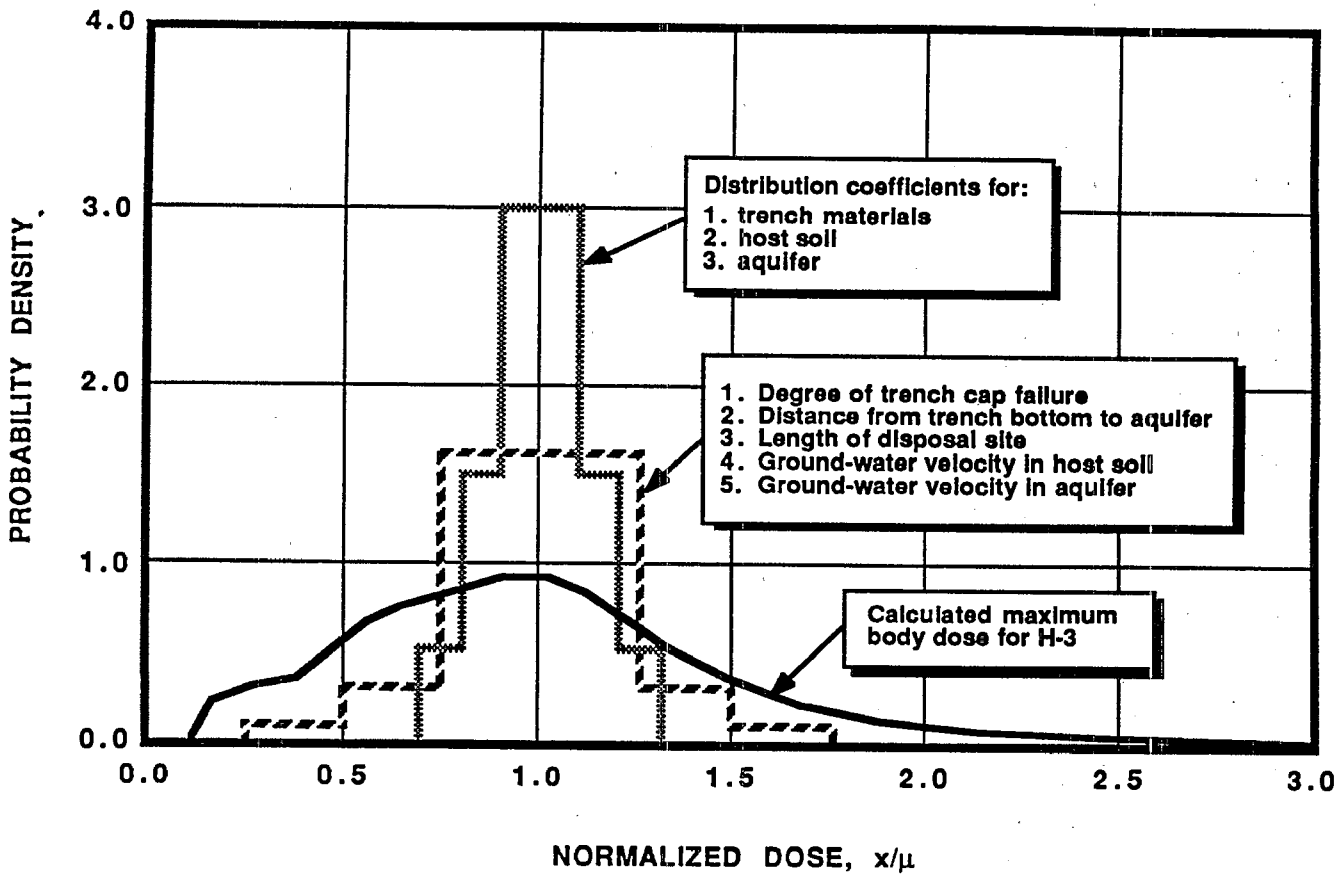

Figure 12-11. Results of Uncertainty Analysis for H-3, Standard Deviation = 47%

obtain insight into the uncertainties, a quantitative estimate of overall uncertainty may be more representative of opinion than of objective fact. The following discussion will consider uncertainties associated with two models, the deposited activity model for radionuclides and the specific activity model for C-14. The discussion in this section is based primarily on material from a National Council on Radiation Protection and Measurements (NCRP) publication (NCRP84).

12.5.1 Interception

The deposited activity model presumes that a fraction, f_r, of the depositing flux is intercepted by vegetation and incorporated into the associated crop. The remainder is considered to deposit on the soil, where it may subsequently be available for uptake by the root system. For forage or leafy vegetable crops, the interception fraction is strongly dependent on the areal density, Y_v, of the crop. In this case, the quantity f_r/Y_v can reasonably be considered as lognormally distributed with a median (geometric mean or gm) of 1.8 m^2/kg (dry weight) and a geometric standard deviation (gsd) of 1.6. For other crops, the relationship between f_r and Y_v is more case specific. While a default value of f_r, such as 0.2 or 0.25, is frequently used for irrigation spray or particulate deposition on these crops, there is no consensus as to what the distribution of f_r should be; however, a gsd of 2 might be considered reasonable.

12.5.2 Crop Yield

Strictly speaking, Y_v is the total areal density of the above-ground portion of a crop. For many crops, the edible portion may be only about one-third of this quantity. A typical value for Y_v is about 2 kg/m^2. Again, a gsd of 2 might be considered reasonable for estimating the uncertainty in this parameter.

12.5.3 Weathering Half-Life

The weathering half-life, t_w, varies with the chemical form of the depositing radionuclide, crop type, stage of development, and the processes affecting removal. As customarily measured, it also includes the effect of dilution caused by plant growth. A nominal value of 14 days with a gsd of 1.6 is representative. For radionuclides with half-lives substantially longer than 14 days, weathering is the principal loss mechanism.

12.5.4 Other Parameters

The time of exposure during the growing season generally is significantly longer than the weathering half-life and therefore its uncertainty is not an important factor in the model. Similarly, the uncertainties in the bulk and surface density of soil do not make substantial contributions to the overall uncertainty.

12.5.5 Uptake from Soil

Generally, direct deposition onto plant surfaces is a much more important contaminating mechanism than uptake from soil. If trench rather than spray irrigation is used, however, uptake from soil would be important because there would be no direct deposition on vegetation. Soil-to-plant transfer factors show a wide range--a gsd of 4 is typical. Since soil-to-plant transfer is really soil-to-soil water-to-plant transfer and since the soil-to-soil water distribution factor, K_d, can vary widely, the large uncertainty in uptake from soil is not surprising. Another significant contributor to the uncertainty in uptake from soil is the environmental removal rate of radionuclides from soil. Since leaching, a principal consideration, is highly dependent on K_d, there is a strong correlation between the transfer factor and the leaching rate. Overall, a gsd of 5 might be considered reasonable for deposition to soil-to-plant transfers.

12.5.6 Transfers to Milk and Meat

Intakes of feed and water by animals can best be determined on a site-specific basis. While typical values can be assigned, the uncertainties in these values may represent differences in specific management practices rather than random variation. The transfer factor for iodine from feed to milk, f_m, has a gsd of about 1.7. Other radionuclides would have comparable uncertainties. Taking into account other uncertainties such as that in the milk production rate, the overall transfer from feed to milk could reasonably be assigned a geometric standard deviation (gsd) of about 2. Similarly, on the basis of data for cesium, the uncertainty in the meat transfer coefficient, f_f, can be considered to be represented by a gsd of about 2.3. Considering other associated uncertainties, an overall gsd of about 3 would be reasonable for feed-to-meat transfers.

12.5.7 Carbon-14

Because atmospheric carbon dioxide is the primary source of carbon in plants, there is little uncertainty in the transfer of carbon-14 to plants and animal products. At equilibrium, plants and animals will have the same specific activity as the atmosphere to which they are exposed. What uncertainty there is has to do with considerations affecting the atmospheric concentration where the different crops providing food and feed are grown. Such considerations have much more to do with scenario considerations than with model uncertainty.

12.5.8 Summary

The overall uncertainties in food chain models can be considerable. The uncertainty for leafy vegetables and pasture feed can be represented by a geometric standard deviation (gsd) of about 2.3. For other produce where direct deposition is the source of contamination, a gsd of 3.2

would be appropriate. For food products where soil-to-crop transfer is the dominant contamination mechanism, a gsd of about 5 would be reasonable unless site-specific parameters can be used. The gsd values for transfers of activity directly deposited onto forage to milk and meat would be about 3 and 3.8, respectively. For transfers from other feeds, these values would increase to about 3.8 and 4.4, respectively. With specific activity models such as that used for C-14, the principal uncertainties are usually associated with the postulated scenario rather than model assumptions. In any case, the uncertainty estimates in this section should not be considered as authoritative, verified values but as aids to evaluating the potential significance of the food pathways.

12.6 Uncertainty Due to Estimation of Organ Doses

As mentioned in Chapter 6, the primary sources of uncertainty in estimating doses to organs of individuals exposed through ingesting or inhaling radionuclides are associated with: (1) ICRP model formulation and (2) parameter variability caused by measurement and sampling errors or natural variations. It was also mentioned that the Agency's ability to quantify these uncertainties is extremely limited because of the lack of experimental data.

This difficulty can be attributed to several factors. First, most of the ICRP models for estimating doses to organs of individuals in the general population were developed from animal experiments; the metabolic behavior of radionuclides in animals and man often differs significantly. Differences are also observed in the anatomical structure of organs and tissues in animals and man. To quantitatively determine the uncertainties associated with using animal-based models requires extensive animal and human data and a means for properly extrapolating animal results to humans. Data and methods are both lacking in this area. Second, most of the assumptions used in the ICRP modeling approach (e.g., for handling ingrowth of radioactive daughters, for relating similar nuclides with different metabolic patterns, or for estimating doses to organs consisting of heterogeneous cell populations) have not been properly tested and verified. Generally, the experimental data supporting these assumptions are very sparse as well.

In addition, for those models that are assumed from the outset to be correct, considerable uncertainties are expected in estimating organ doses because of the variability in anatomical and physiological parameters. Parameter variability primarily relates to age differences in the general population. The parameters employed for EPA modeling purposes were obtained from persons with anatomical or metabolic characteristics similar to "Reference Man" and represent "best estimates" or "average" values from parameter distributions. The parameter values are normally scaled for other age groups in the general population. This method ignores the recognized variability among individuals, and it automatically introduces bias when extending these models to other members of the population. Many of the parameters used for estimating

doses to organs, such as radionuclide intake rates (I), organ masses (m), blood transfer factors (f_1), organ uptake rates (f_2'), and biological half-lives of ingested radionuclides, vary with age. In addition, considerable variability in these parameters can exist among individuals of the same age group. To properly ascertain the magnitude of this uncertainty requires knowing how these parameters vary with age and obtaining a parameter distribution for each age group. Again, there are limited data upon which to base such an analysis.

If we restrict our attention to an "average" individual, some parameter uncertainties will be greatly reduced, and the overall uncertainty may be better obtained. In particular, for radioiodine, the variability of the target organ (thyroid) mass is quite large, especially when all age groups are considered; nevertheless, the average thyroid mass is known to be within perhaps $\pm$ 20 percent. The major sources of error with respect to I-129 dosimetry appear to be related to assumptions regarding intake volume and f_2'. Actual average daily water intake is probably between 1 and 1.5 L, rather than 2 L, as assumed here. Based on more recent studies, the assumed value for f_2' (0.3) is probably high, perhaps by a factor of 2. Both of these errors would tend to bias the dose estimates high. Thus, the dose conversion factor for I-129 should be regarded as an upper bound "conservative" estimate.

The major source of dosimetric uncertainty with respect to C-14 is uncertainty over retention time in the body. The models used here assume that the C-14 released from the waste site into ground water and subsequently ingested in drinking water is handled by the body like carbon ingested in food. This assumption is highly conservative. Carbon-14 ingested in an inorganic (carbonate/bicarbonate) form will be rapidly eliminated from the body through exhalation. Furthermore, organic C-14 compounds originating from the waste site may not, for the most part, be in a form which the body can utilize as a carbon source; hence, the average retention time may also be low for ingested organic compounds. In conclusion, the dose conversion factors for C-14 should be regarded as upper bounds and may have overestimated the actual average dose.

12.7 Uncertainty Due to Health Effects Conversion Factors

The uncertainties in the risk estimates for radiogenic cancer are discussed in Section 7.5. The chief sources of uncertainty associated with a uniform whole-body dose of low-LET radiation to the general population, and their estimated magnitudes, are summarized in Table 7-10. The estimated combined uncertainty, due to all sources, encompasses the range from 23 to 160 percent of the central estimate. Based on the central estimate of 395 fatal cancers per million person-rad (see Table 7-3), the overall uncertainty range is 91 to 630 fatal cancers per million person-rad. For risks to individual organs, the percent uncertainties may be much larger.

The uncertainties in the risk estimates for radiation-induced genetic effects are discussed in Sections 7.6.4 through 7.6.7. A list of the sources of uncertainty and their magnitudes is given in Table 7-17. As noted in Section 7.6.7, the EPA genetic risk estimate is believed to be uncertain by about a factor of 4 either way. Based on limited human data, however, it is more likely to be on the conservative (high) side.

12.8 Uncertainty of the Overall Health Effects and Maximum CPG Dose Analyses

Based on the uncertainty analyses discussed for each of the components in Sections 12.2 through 12.7, one may calculate the overall uncertainties using the method to be presented in this section. Since uncertainties for all of the components are not quantifiable, the analyses of the overall uncertainty of health effects and maximum CPG dose cannot be performed accordingly. Nevertheless, the analysis calculates an example of the overall uncertainty by using the quantified uncertainties and the assigned uncertainties made for those components that are not quantifiable. The selected example represents the disposal technology specified by the 10 CFR 61 regulations (NRC82a) applied to a site in a humid permeable hydrogeological setting.

12.8.1 Method of Analysis

For the purpose of the uncertainty analysis, the cumulative health effects and the maximum CPG dose analyses may be expressed, respectively, as:

$$HE = [IR]\ [AT]\ [FCC]\ [DCF]\ [HCF] \qquad (12\text{-}3)$$

and

$$BD = [IR]\ [MC]\ [FCC]\ [DCF]\ [BDC] \qquad (12\text{-}4)$$

where:

HE = the cumulative health effects;

IR = the radionuclide inventory in the waste;

AT = the cumulative radioactivities being discharged into the regional river basin based on the unit curie of disposal;

MC = the maximum concentration of radionuclides due to the unit curie disposal;

FCC = the food chain factor;

DCF = the organ dose conversion factors;

HCF = the health risk conversion factor;

BD = the maximum CPG dose;

BDC = the CPG dose conversion factor; and

[] designates a random variable.

Equations 12-3 and 12-4 are a linear multiplicative chain of independent parameters. Therefore, the overall uncertainty (geometric standard deviation) of the assessment can be calculated from the standard deviation for each individual parameter without undergoing a time consuming calculation of joint probability functions. However, in order to apply this method, one has to assume that the probability density distribution for each component as discussed in Section 12.1.3 is in the form of a log-normal distribution. For the purpose of this analysis, this assumption is thought to be acceptable, judging from the results of observations on the distribution of general environmental parameters. The mean value and its standard deviation for the overall assessment may be calculated by employing the Theorem of Variance for the joint distribution of random variables as:

$$\mu_{HE} = \mu_{IR} + \mu_{MC} + \mu_{FCC} + \mu_{DCF} + \mu_{HCF}$$

and (12-5)

$$\sigma^2_{HE} = \sigma^2_{IR} + \sigma^2_{MC} + \sigma^2_{FCC} + \sigma^2_{DCF} + \sigma^2_{HCF}$$

for the cumulative health effect analysis; and

$$\mu_{BD} = \mu_{IR} + \mu_{MC} + \mu_{FCC} + \mu_{DCF} + \mu_{BDC}$$

and (12-6)

$$\sigma^2_{BD} = \sigma^2_{IR} + \sigma^2_{MC} + \sigma^2_{FCC} + \sigma^2_{DCF} + \sigma^2_{BDC}$$

for the maximum CPG dose assessment. In the above equations:

μ = the log transformed mean value; and

σ = the log transformed standard deviation; and

the subscripts HE, IR, FCC, DCF, HCF, BD, MC, and BDC are the same as those defined for Equations 12-3 and 12-4.

Therefore, the uncertainty of the assessment can be calculated from Equations 12-5 or 12-6, if the mean values and the standard deviations for all the components of the various calculations are defined.

12.8.2 Uncertainties of Assessment Components

As was stated in Sections 12-2 through 12-7, the quantification of the uncertainty for each component is extremely difficult to obtain and the uncertainty was not quantified except for the components of the transport through the geosphere and health effects conversion factors. This impeded the quantification of the overall uncertainties for the cumulative health effects analysis and the maximum CPG dose analysis. Nevertheless, the best estimate on the uncertainties for each component, through expert judgment, was made to obtain an order of magnitude quantification for the overall uncertainties of the assessments. The analyzed and estimated standard deviations for C-14 and I-129 for each component, together with the mean values extracted from the results of the assessment using PRESTO-EPA models, are listed in Tables 12-1 and 12-2 for the cumulative health effects analysis and the maximum CPG dose analysis, using the example case of the disposal technology specified by the 10 CFR 61 regulations at a humid permeable site.

12.8.3 Results of the Overall Uncertainty Analysis

As indicated in Section 12.8.2, the quantification of the uncertainties for all components of the uncertainty analysis cannot be conducted at this time. Quantification of the uncertainties for the radionuclide transport through the geosphere and for the health effects conversion factors has been obtained from detailed analyses, while best expert judgment is used to estimate the uncertainties for the remaining components. These results were presented in Tables 12-1 and 12-2 and served as input to the overall assessment of uncertainties carried out with Equations 12-5 and 12-6.

Parallel analyses are also conducted for the minor contributors, I-129 for the cumulative health effects analysis and C-14 for the maximum CPG dose analysis. The results of the analyses indicated that the upper bound, or the uncertainties for the cumulative health effects analysis for both C-14 and I-129, is identical and equal to 161 percent of the central value, while the uncertainties for the maximum CPG dose analysis are 147 percent of the central value for I-129 and 132 percent of the central value for C-14.

The combined results of the assessment for the 10 CFR 61 disposal technology applied to a site in a humid permeable region are presented in Table 12-3. Note that these results reflect the combined effects from both C-14 and I-129. It is also interesting to note that the major share of fatal health effects (about 98 percent) is estimated to originate from C-14, while I-129 is the major contributor (92 percent) for maximum CPG dose.

Table 12-1. Summary of estimated mean and standard deviation for cumulative health effects analysis for C-14

Items	Estimated value	Unit	Estimated standard deviation	
Radionuclide Inventory	6.681E+02	Ci	6.681E+02	(100%)
Cumulative Activities Discharged	5.984E-01	(Ci/yr) x10,000 yr/Ci	0.00	(0%)
Food Chain Factor	1.604E+04	man-pCi/yr / Ci/yr	0.00	(0%)
Dose Equivalent Conversion Factor	1.540E-06	mrem/yr/pCi/yr	9.240E-07	(60%)
Health Risk Conversion Factor	2.806E-01	death/yr / man-mrem/yr	2.370E-01	(60%)

Table 12-2. Summary of estimated mean and standard deviation for maximum CPG dose analysis for I-129

Items	Estimated value	Unit	Estimated standard deviation
Radionuclide Inventory	7.899E+00	Ci	7.899E+00 (100%)
Maximum Concentration Factor	2.609E-09	$Ci/m^3/Ci$	1.044E-09 (40%)
Food Chain Conversion Factor	4.819E+11	$pCi/yr/Ci/m^3$	0.00 (0%)
Dose Equivalent Conversion Factor	8.568E-04	mrem/yr/pCi/yr	5.141E-04 (60%)

Table 12-3. Results of uncertainty analyses: 10 CFR 61 technology at a humid permeable site

Analyses	Calculated value	Standard deviation
Fatal Health Effects	3.9 deaths	6.3 deaths (161%)
Maximum CPG Dose	9.2 mrem/yr	13.4 mrem/yr (146%)

Based on the above results (Table 12-3), the upper bounds of the results of analysis for the analyzed scenario are estimated to be 10.2 cumulative fatal health effects and 22.6 mrem/yr for the maximum CPG dose.

The cumulative health effects presented in Chapter 9 for the analyzed scenario are the combined result of fatal cancers and serious genetic effects. Serious genetic effects comprise, in general, only a small fraction of total health effects (fatal plus serious genetic). Therefore, for the purpose of the uncertainty analysis, one may reasonably assume that the uncertainty for the serious genetic effects is the same as that for the fatal cancers. Thus, the upper bound of the combined total population health effects (fatal plus serious genetic) is calculated to be 11.5 health effects, with a central value of 4.4 health effects.

12.9 Conclusion

Despite the difficulty of quantifying the uncertainty of the cumulative health effects and the maximum CPG dose, an effort was made to quantify the uncertainties. Since multidisciplinary processes were involved, the analysis was divided into five components to permit the uncertainty analysis for each component to be conducted by experts in the field. The results of the analysis for each component are summarized as follows:

12.9.1 Source Term Concentration

The analysis centered on C-14 and I-129 because these predominant radionuclides are projected to be the major contributors to the cumulative health effects and the maximum CPG dose. Since the concentrations of these radionuclides, particularly I-129, are in a trace amount, the accuracy of their measurement was found to be poor.

It is believed that the uncertainty of the C-14 and I-129 concentrations in daily samples collected from each waste stream may vary considerably from one to another. However, the uncertainty of the concentrations of all radionuclides from all waste streams for 20 years is expected to be much smaller than that for any daily sample of any waste stream. Based upon a detailed evaluation of the EPA LLW source term (Gr86) and an extensive study of the occurrence of C-14 and I-129 in waste samples (Cl85), a qualitative uncertainty of a factor of 2 was assigned to C-14 and I-129.

12.9.2 Radionuclide Transport in Geosphere

It was found that the major uncertainty for the geosphere transport will be dominated by the selection of site scenarios. This finding occurs because the amount of available dilution water in the underlying aquifer will greatly affect the results of the assessment. Any uncertainties resulting from these site scenarios were not considered

because the risk assessments were concentrated on typical sites for three distinct hydrogeological settings. When this simplification is imposed, the major uncertainties will be governed by the rate of infiltration, the distribution coefficients, and the distance of transport.

Fortunately, the model output parameters are controlled by integrated effects either over an extremely long period of time (for the cumulative health effects assessment) or over the entire disposal area (for the maximum CPG dose assessment), which have greatly converged the uncertainties of the output parameters, at least for mobile radionuclides. Our results concluded that the uncertainty of the cumulative C-14 and I-129 radionuclide releases for the health effect assessment is near zero, and the uncertainty of the maximum CPG dose assessment for I-129 and C-14 is approximately 40 percent and 10 percent of the mean values, respectively.

12.9.3 Radionuclide Transport through the Food Chain

Radionuclide transport through the food chain pathway is divided into two categories, the drinking water pathway and the nondrinking water pathway. The radionuclide transport in the food chain for nondrinking water includes the deposition from the atmosphere or from irrigation water, plant uptake from soil, transfer to milk and meat, and finally, human consumption.

When the overall uncertainties of the transport through the food chain are considered, the uncertainties resulting from the nondrinking water pathway are negligible. Whereas the drinking water pathway accounts for the major portion of exposures, C-14 and I-129 account for approximately 99 percent of the total exposure from the analyzed scenario, which uses the 10 CFR 61 technology in a humid permeable site. Therefore, the uncertainties of radionuclide transport through the food chain pathway are dominated by the uncertainty of the daily consumption of drinking water.

Furthermore, since the PRESTO-EPA model calculates the radiation exposure to average persons in the United States, the variation in per capita consumption of drinking water resulting from individual differences should not be considered part of the uncertainty. Therefore, the overall uncertainty of the radionuclide transport in food chain pathways for the analyzed scenario is small and can be neglected.

12.9.4 Organ Dose Conversion Factor

The primary sources of uncertainty in estimating doses to organs of individuals exposed to radionuclides are associated with: (1) ICRP model formulation and (2) parameter variability caused by measurement and sampling errors or natural variations. The quantification of these uncertainties is extremely difficult because of the lack of experimental data. Therefore, for many radionuclides of interest no quantitative

statement can be made about the uncertainties associated with the use of EPA dose conversion factors. Data are lacking and more research is needed to test models and determine the variability in model parameters.

12.9.5 Health Effects Conversion Factor

The estimated overall uncertainty resulting from all sources encompasses the range from 23 to 160 percent of the central value. Based on the central estimate of 395 fatal cancers per million person-rad, the overall uncertainty range is from 91 to 630 fatal cancers per million person-rad. For risk to individual organs, the percent uncertainty may be much larger.

12.9.6 Results of Overall Uncertainty Analysis

Realizing that the uncertainties for all of the components are not quantifiable, the analysis calculated an overall upper bound of the results for a selected scenario by using the best estimated uncertainties for each component. The analyzed scenario, which applied the disposal technology specified in 10 CFR 61 for the humid permeable site, was selected. The results of the analyses for combined effects from C-14 and I-129 are 161 percent of the central value of 3.9 deaths for the cumulative fatal health effects over 10,000 years, and 146 percent of the central value of 9.2 mrem/yr for the maximum CPG dose.

The upper bounds of the combined results for the analyzed scenario are estimated to be 11.5 total health effects (10.2 for fatal health effects and 1.3 for the genetic effects) for the cumulative health effects analysis and 22.6 mrem/yr (21.0 mrem/yr from I-129 and 1.6 mrem/yr from C-14) for the maximum CPG analysis.

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Chapter 13: PREDISPOSAL WASTE MANAGEMENT OPERATIONS

13.1 Introduction

EPA is proposing an annual dose limit for the CPG for LLW management operations prior to disposal. Predisposal management operations include preparation of the waste for disposal, i.e., compaction, incineration, solidification, packaging, handling, storage, and placement. These activities could be carried out, for example, by LLW generators (power plants, industries, hospitals, medical centers, or DOE sites), at or adjacent to operating LLW disposal facilities, or at regional LLW processing facilities designed to serve a State or an entire Compact. The following analysis assesses the potential exposures to the public from radiological releases during predisposal management operations.

Waste generators increasingly are opting for volume reduction and waste processing to meet NRC's waste stabilization requirements (NRC82), to reduce disposal costs, and to stay within volume limits imposed by host States for existing disposal facilities (LLR86). This trend in the processing of LLW is being met in a number of ways. Large LLW generators are building their own processing facilities and small generators are being serviced by mobile processing units (i.e., compactors, solidifiers). States and commercial companies are planning to establish regional facilities solely for processing LLW, with the processed LLW then shipped to a disposal site.

These trends toward (1) widespread processing of wastes at a large number of diverse facilities and generators, and (2) possible long-term, aboveground storage present an area in which environmental protection standards for certain releases from these activities are lacking. In some cases, the processing and storage of LLW done at various uranium fuel cycle facilities, such as power reactors and fuel fabrication plants, will be covered under the 40 CFR 190 standards (EPA77a). These standards encompass all activities carried out at these facilities, and many LLW processing and storage operations at these facilities use the same techniques as are found at any LLW processing facility, such as evaporation, incineration, compaction, handling, and storage. Several of these operations were evaluated for the UFC standards (EPA73a,b). In addition, atmospheric releases of radionuclides from LLW processing and storage operations at those DOE- and NRC-regulated facilities are covered by the Clean Air Act standards pursuant to 40 CFR 61 (EPA85). These standards also encompass all airborne activities at these facilities, which include many of the same LLW processing and storage operations as are carried out at specific LLW processing facilities (EPA84). However, releases through such other pathways as water and gamma exposure from processing operations, as well as long-term storage at NRC-regulated large, away-from-generator or central processing facilities and at DOE facilities, would not be covered under 40 CFR 61.

EPA has not performed the comprehensive quantitative risk assessment necessary to determine the health impacts of the various predisposal management operations. However, some limited analysis has been done on operational spillage (see Section 13.4.1). Such comprehensive assessments would first require identification of potential exposure pathways that are not already limited by existing regulations or standards. For example, since the proposed standard represents a limit on the cumulative dose through all pathways, the contribution of the air pathway, even though limited by the Clean Air Act emissions standards, would need to be further quantified, including exposure resulting from surface spillage, followed by resuspension and offsite transport.

It is theoretically possible that offsite contamination could occur as a result of spillage and surface runoff during a rainstorm (or flood). Finally, if waste treatment or storage vessels are located near the boundary of the site, external direct gamma radiation could also cause exposure to individuals at the site boundary.

Therefore, the proposed predisposal management standard would probably result in actions to control spillage (e.g., by the use of good housekeeping practices and proper design of handling equipment) or to limit direct gamma radiation (e.g., by placing storage facilities away from the facility boundary).

13.2 Basic Assumptions

In this analysis, we have examined the most likely major steps in the management of LLW. They include: evaporation, incineration, liquid storage, packaging, solid waste storage, compaction, and solidification processes.

As indicated earlier, many of the operations already take place at various generators' facilities, and in some cases these operations were analyzed in connection with the UFC 40 CFR 190 and CAA 40 CFR 61 standards (EPA73a,b, EPA84). The data presented here come from reports of the DOE, NRC, and EPA. Some of the data deals with hypothetical generic facilities; other data are concerned with actual operations at DOE or commercial facilities.

The basic assumption underlying this analysis is that the major radiation dose to the critical population group from the facilities is through airborne discharges to the atmosphere based on present practices. Some gamma exposure could be present and some minimal liquid releases could occur. In almost all cases, however, operations today recycle many liquids for use, and waste liquids are usually solidified and disposed of as solids.

The exposure pathways, demography, and other parameters, as well as the mathematical models relating dose to man for the estimated radionuclide releases from the generic facilities, are described in

DOE79. (These documents were reviewed by EPA in 1979 and were found to be adequate.) During many waste management operations, some of the radionuclides in the wastes are released as volatile gases and particulates. Before these gases and particulates are released to the atmosphere, they are routed to treatment systems designed to remove the majority of the radionuclides. Those releases cited from specific facilities are discussed in EPA84 and include LLW operations such as evaporation and incineration and LLW disposal sites. The maximum annual CPG doses are based on hypothetical area residents whose habits would tend to maximize the dose.

Several major factors that can affect the potential radiation dose to the CPG and populations as a result of release of radionuclides to the atmosphere are as follows: proximity to the plant, the pathways by which the radionuclides can reach people, the length of time during which the radionuclides continue to pose a health hazard, decay time, meteorological factors, facility capacity, and off-gas treatment.

13.3 General Air Emissions Pathway

A review was made of the previous evaluations by EPA, in connection with its regulations for radionuclide air emissions (40 CFR 61). The results of this review are given in this section (EPA84, EPA85).

13.3.1 Department of Energy Facilities

The DOE administers many government-owned, contractor-operated facilities that emit radionuclides to the air. Operations at these facilities include research and development; production of nuclear weapons; enrichment of uranium and production of plutonium for nuclear weapons and reactors; and processing, storing, and disposing of radioactive wastes. Not all of these operations take place at all sites, of course. Certain of these facilities are on large sites, some of which cover hundreds of square miles in remote areas; several States are host to such sites. Some smaller facilities resemble typical industrial sites and are located in suburban areas. As indicated earlier, many of these facilities use, or are expected to use, the same processing, management, and storage techniques as one would expect to find at a large commercial centralized LLW processing center.

Each facility differs in emission rates, site size, nearby population densities, and other parameters that directly affect the offsite dose from radionuclide emissions. Many different radionuclides are emitted to the atmosphere. Six sites have multipurpose operations spread over very large areas. Another 12 or so sites are primarily research and development facilities located in more populated areas. Reactor and accelerator operations at these sites may release radioactive noble gases and tritium; other operations may release small amounts of other radionuclides. Several facilities are primarily engaged in weapons development and production, and may release small amounts of tritium and

certain long-lived radionuclides. Finally, two sites are dedicated entirely to gaseous diffusion plants that enrich uranium for use in commercial electric power reactors and for defense purposes. They primarily emit uranium.

At 15 of the smaller DOE facilities, which are considered as a group in the Radionuclides Emissions BID (EPA84) because of their relatively small health impact, the doses to the nearby individuals are estimated to be considerably less than 1 mrem/yr. These small doses were also reported at less than 1 mrem/yr in a previous report (EPA77b).

A second group, which consists of the 13 facilities having the largest emissions of radionuclides, was studied in more detail. The collective dose to the populations living around these sites is also low, no higher than about 10 person-rem after 1 year of site operation.

The doses from these facilities to the CPG are generally estimated to range from 2 to 10 mrem/yr, although two facilities indicated doses of greater than 25 mrem/yr. These exposure results reflect all operations (resulting in airborne releases) carried on at these sites, and the major releases are those from the principal activities carried on at these facilities, e.g., reactor operation, fuel reprocessing, enrichment, etc. Therefore, the various LLW processing, management, and storage operations carried on at the sites contribute only a small percentage of the radioactivity to the offsite population. A rough estimate would be 10 to 15 percent. Therefore, it is expected that the doses to the CPG from LLW operations at these DOE facilities will also be a small percentage (probably several orders of magnitude less) of that reported for the overall health impact from air emissions from all operations on that site. Chapter 3 lists the various DOE facilities throughout the U.S., and also presents the volume and radionuclide characteristics of the various LLW generated and disposed of at these facilities.

13.3.2 Nuclear Regulatory Commission-Licensed and Non-DOE Federal Facilities

NRC-licensed and non-DOE Federal facilities include research and test reactors, shipyards, the radiopharmaceutical industry, and other research and industrial facilities. This category includes both facilities licensed by NRC and those licensed by a State under an agreement with NRC. These facilities number in the thousands and are located in all 50 States. Uranium fuel-cycle facilities are not included because radionuclide emissions from these facilities are limited by EPA standards (40 CFR 190). See the discussion in Sections 13.1 and 13.2. The principal differences among these various types of activities are their emission characteristics and rates, their sizes, and the population densities of the surrounding areas.

The vast majority of NRC-licensed and non-DOE Federal facilities emit relatively small quantities of radionuclides, which cause correspondingly low doses to people living nearby. From EPA studies and contractor-supported analysis, the maximum radiation doses from these facilities were less than 1 mrem/yr, with the total dose to the population living around a site rarely exceeding 1 or 2 person-rem/yr of operation (EPA77b, EPA84). Various LLW processing, management, and storage operations are also carried out at these facilities. In many cases, the quantities of waste and hence the waste management operations are small. Chapter 3 presents a description of a number of the waste streams generated by these different operations.

Waste management and storage operations take place at all facilities where radionuclides are used, and the size of the waste operations can be either small or large depending on the annual throughput of materials and waste generated. We want to emphasize that these doses are calculated from all operations taking place at a specific facility. It is expected that the doses from the various LLW management operations will be only a small percentage of the overall amount, in many cases probably less than 10 to 20 percent.

13.3.3 Air Emissions from Compaction

DOE estimates of doses to the CPG from gaseous effluents released from a generic model fuel bundle residue compaction facility are in the range of 1E-11 to 1E-09 mrem/yr (DOE79).

13.3.4 Air Emissions from Incineration

A DOE estimate of doses to the CPG from gaseous effluents released from a generic model solvent incineration facility was in the range of 1E-09 to 1E-05 mrem/yr, whereas for a generic model LLW incineration facility the range was 1E-18 to 1E-07 mrem/yr (DOE79). Further generic analysis for intermediate level waste (or what might be considered greater-than-Class C waste) found the dose to the CPG ranged between 1E-05 and 20 mrem/yr.

Another environmental impact analysis (Ph84) of incineration of institutional LLW indicated that the radionuclide air emissions due to incineration are very small and that either the CPG organ doses or whole-body doses will also be small (less than 0.001 mrem/yr).

In a generic licensing report to the NRC, the Newport News Industrial Corporation and Energy Incorporated presented an environmental impact analysis of their incinerator system for LLW volume reduction at nuclear power plants. The environmental analysis is based on a system capable of processing up to 1,200 m^3/yr and incinerating up to 91 kg/h of LLW. Their analysis indicated that the maximum dose to the CPG would be less than 0.1 mrem/yr (En77).

13.3.5 Packaging

DOE estimates of doses to the CPG from a generic model packaging facility are in the range of 1E-12 to 1E-06 mrem/yr (DOE79).

DOE also analyzed a generic package facility for intermediate-level waste (or what might be considered greater-than-Class C waste) and found the doses to the CPG to be in the range of 1E-04 to 4 mrem/yr (DOE79).

13.3.6 Solidification

DOE estimates for the immobilization of LLW by bitumen and cement solidification systems indicate doses to the CPG should be in the range of 1E-05 to 0.1 mrem/yr (DOE79).

13.3.7 Storage

DOE also considered storage as an option for its many generic processing operations, such as those discussed in Sections 13.3.3 through 13.3.6. The analysis of doses to the CPG was in the same range as that indicated for the previous packaging, solidification, and incineration operations, 1E-18 to 0.1 mrem/yr. For intermediate-level radioactive wastes, the dose analysis resulted in a range of 1E-05 to 1 mrem/yr (DOE79).

13.4 LLW Disposal Facilities Operations

Normal operational releases from an LLW disposal facility can potentially occur through small spills and releases resulting from normal waste handling and disposal operations. Releases have also occurred at some existing sites as a result of water management programs involving evaporation and treatment of trench leachate. Since the need for active maintenance programs is expected to be eliminated in the future, releases resulting from such programs were not analyzed. Also considered was the processing of waste at a regional processing center, which for purposes of analysis is assumed to be located at the disposal facility.

13.4.1 Operational Spillage

Small leaks and spills from waste containers during normal operations can potentially be released to the air or contaminate the ground surface, which can then be carried from the site by the actions of wind or precipitation runoff. It is believed that the contamination of the ground surfaces at the Maxey Flats facility was caused by earlier cases of inadequate waste handling and site maintenance procedures. It is known that waste packages delivered to the facility frequently failed to properly contain the waste within the packages and/or ruptured during emplacement operations.

At currently operating facilities, however, considerably more attention is being paid to minimizing potential surface contamination. For example, disposal facilities in operation have procedures to survey facility areas on a routine basis, as well as when possible contamination is suspected. Allowable contamination limits have been established at operating facilities. In addition, monitoring programs at all operating facilities have been improved and routinely sampled for onsite surface contamination.

Of interest are environmental monitoring results for the Barnwell, South Carolina, disposal facility. This facility accepts approximately 50 to 70 percent of the LLW in the country. Given the large volume of waste received at the facility, most of the operational impacts associated with LLW disposal would be expected to be associated with this facility. For example, the concentrations of Co-60 and Cs-137 measured onsite are within the range of measurements of samples collected offsite (NRC81).

Thus, there appear to be no significant releases of radionuclides from the operating sites from surface contamination. This is principally due to the increased attention paid by facility operators to minimizing facility contamination. The practice of delivering bulk liquids to disposal facilities for solidification has been discontinued. All disposal facilities have license conditions that restrict wastes delivered to the disposal facilities to dry solids, and include restrictions on the amount of free-standing liquids allowed in the waste. Compliance with DOT regulations is also required. Improvements in waste form and packaging required by 10 CFR 61 will also reduce the potential for surface contamination and subsequent release to offsite areas.

Since releases during normal operations caused by spills have not been significant and are not expected to be significant in the future, NRC conducted no detailed analysis of these potential pathways of release and potential public impacts in its EIS for 10 CFR 61 (NRC81). However, in EPA's risk assessment methodology, spillage was taken into account (see Chapters 8 and 11). EPA's results indicated that any operational spillage would account for less than 0.1 mrem/yr to any offsite individual.

13.4.2 Operational Airborne Emissions

Analysis done at four LLW disposal sites for exposures to offsite individuals from airborne migration indicated less than 0.2 mrem/yr (Ad78, FBD78a, FBD78b, FBD78c).

13.4.3 Evaporator Operation

A site investigation and analysis by EPA of the LLW evaporation system operation in 1974-1975 at the Maxey Flats burial site indicated

that the maximum individual dose rate received by the nearest resident was less than 3 mrem/yr (Mo77). An NRC computer model analysis of the Maxey Flats evaporator for both airborne and aquatic pathways found the individual dose rate to be less than 0.01 mrem/yr (Ad78). The difference in these two analyses is probably based on parameter data used, especially annual throughput of materials evaporated.

13.4.4 Offsite Gamma Radiation

Offsite external radiation monitoring is done by South Carolina for the Barnwell waste disposal site. During the period 1983-1985, the annual background gamma exposure rate at stations not influenced by the Barnwell LLW burial site was 6E+01 mR/yr. The annual gamma exposure (background included) rates at the Barnwell exclusion fence and at stations less than 8 kilometers were 6.6E+01 mR/yr and 5.9E+01 mR/yr. Thus, the exposure rates at the exclusion fence and the other stations appear to be within background levels. The operations of the Barnwell LLW burial site do not appear to influence the annual gamma exposure levels in the immediate vicinity (SC86).

13.5 Regional Processing Facility

One of the viable options addressed was that of processing waste on a regional basis at a central processing facility. Such a facility could be located at or be separate from the disposal facility.

Such waste processing activities can lead to potential airborne releases of radionuclides and subsequent exposures to the public in the neighborhood of the regional processing facilities. NRC analyzed the potential population exposures due to the assumed operation of a central waste processing facility (an incinerator) that was co-located with the disposal facility. These exposures were estimated to be approximately 2 person-rem/yr, arising from the assumed incineration of 100,000 m^3 of combustible trash per year. The total population assumed to be exposed was 480,000 within an 80-kilometer radius of the processing facility. This would be in the neighborhood of less than 0.01 mrem/yr to the nearest individual (NRC81).

13.6 Summary

Analyses for the various pre-disposal LLW management and storage operations have not been extensive and, as shown by our review, have been somewhat fragmented. In some cases though, the analyses has been very detailed and based on actual operating data (e.g., the data collected by EPA), while the generic data presented by DOE are based on existing or available technology applied to proposed facilities. It must be kept in mind, however, that the doses presented here are related to specific sizes of facilities, although most studies have shown that these sizes are probably the optimum. It is not likely that the waste volume processed in super-sized facilities would be two to three times what is being planned or handled today.

In dealing with waste management processing and storage operations, only a few major alternatives generally will be used. Storage is a common operation and shielding to prevent gamma exposure is an acceptable and widely used technique. Evaporation is a well known process and has been used in the nuclear industry since its inception. Liquids are usually evaporated and the residues, along with other semiliquids, are solidified. The solidification process is also well known and has been in use at nuclear installations since the 1960's (HO76).

General trash is commonly packaged, either with or without compaction depending on the materials. Compaction is being used more frequently for radioactive waste generation (DOE79, Jo86). In essence, packaging consists of containing general trash in steel drums or boxes for interim storage or disposal.

Incineration is another important technique for treating LLW. Incineration consists of burning the waste and treating the off-gas for removal of radionuclides and other noxious materials, thereby reducing the waste volume and rendering it noncombustible (DOE79, Jo86). In determining the health impacts from incineration versus direct disposal, the use of incineration considerably reduces the health effects and CPG doses over direct disposal (in our analyses, by a factor of 2 - see Chapter 10, Section 10.7.1(B)).

The underlying reason for this reduction in health impacts is that incineration transforms those radionuclides that are volatile (in many cases the majority of the radionuclides present in the waste) from being present in the water ingestion pathway, as the result of direct disposal of waste without incineration, to being present in the air inhalation pathway. In the airborne pathway, the radionuclides are diluted considerably and the body response health impact from inhalation is always much less than when the material is ingested, as through the water pathways.

In summary, potential releases from the airborne pathway and the waterborne carry-off pathway from contaminated surfaces are expected to be on the order of a few mrem/yr. Even when combining these various operations, the overall CPG should be less than 10 mrem/yr for processing Class A, B, and C wastes. Where greater-than-Class C waste is processed, improved technology and techniques may be required to keep the CPG doses to less than 20 to 25 mrem/yr.

Overall, it is felt that these types of CPG exposures can be further reduced by:

- the continued practice of strict housekeeping procedures to maintain potential contamination of equipment and surfaces to ALARA levels; and

- improvements in waste form and packaging.

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APPENDIX A

ACRONYMS, ABBREVIATIONS, CONVERSION FACTORS, NOTATION, AND GLOSSARY

APPENDIX A

APPENDIX A: ACRONYMS, ABBREVIATIONS, CONVERSION FACTORS, NOTATION, AND GLOSSARY

A.1 Acronyms

AEA	Atomic Energy Act of 1954, as amended
AEC	U.S. Atomic Energy Commission
AECB	Atomic Energy Control Board of Canada
AIF	Atomic Industrial Forum
ALAP	As low as practicable
ALARA	As low as reasonably achievable
ANL	Argonne National Laboratory
BEAR	Biological Effects of Atomic Radiation
BEIR	Biological Effects of Ionizing Radiation
BID	Background Information Document
BRC	Below Regulatory Concern
BWR	Boiling water reactor
CC	Concrete canister
CFR	Code of Federal Regulations
CPG	Critical population group
CRCPD	Conference of Radiation Control Program Directors
CW	Consumer waste
DGD	Deep geologic disposal
DOD	U.S. Department of Defense
DOE	U.S. Department of Energy
DOT	U.S. Department of Transportation
DREF	Dose rate effectiveness factor
DWI	Deep-well injection
EIS	Environmental Impact Statement
EMCB	Earth-mounded concrete bunker
EPA	U.S. Environmental Protection Agency
ERDA	U.S. Energy Research and Development Administration
FRC	Federal Radiation Council
GI	Gastrointestinal
GW(e)	Gigawatts of electric power
HANF	Hanford, Washington
HECF	Health effects conversion factors
HEW	U.S. Department of Health, Education, and Welfare
HF	Hydrofracture
HIC	High-integrity container
HLW	High-level radioactive waste
IAEA	International Atomic Energy Agency
ICRP	International Commission on Radiological Protection
IDD	Intermediate-depth disposal
INEL	Idaho National Engineering Laboratory
ISD	Improved shallow-land disposal
L	Pulmonary lymph
LANL	Los Alamos National Laboratory
LET	Linear energy transfer
LLI	Lower large intestine

LLRWPA	Low-Level Radioactive Waste Policy Act
LLW	Low-level radioactive waste
LQ	Linear quadratic
LWR	Light-water reactor
MD	Municipal dump
MIRD	Medical Internal Radiation Dose
MTHM	Metric tons of heavy metal
NARM	Naturally occurring and accelerator-produced radioactive materials
NAS	National Academy of Sciences
NCHS	National Center for Health Statistics
NCRP	National Council on Radiation Protection and Measurements
N-P	Naso-pharyngeal
NRC	U.S. Nuclear Regulatory Commission
NRPB	National Radiological Protection Board
NTS	Nevada Test Site
OMB	Office of Management and Budget
ORNL	Oak Ridge National Laboratory
ORP	EPA's Office of Radiation Programs
P	Pulmonary
PWR	Pressurized water reactor
RBE	Relative biological effectiveness
RFP	Rocky Flats Plant
S	Stomach
SAB	Science Advisory Board
SF	Suburban sanitary landfill
SI	Suburban sanitary landfill with incineration
SI	Small intestine (Chapter 6 only)
SLD	Shallow-land disposal
SLF	Regulated sanitary landfill
SRP	Savannah River Plant
SS	Source and special nuclear material
T-B	Tracheo-bronchial
TRU	Transuranic
TSCA	Toxic Substances Control Act
UF	Urban sanitary landfill
UI	Urban sanitary landfill with incineration
UIC	Underground injection control
ULI	Upper large intestine
UNSCEAR	United Nations Scientific Committee on the Effects of Atomic Radiation
USGS	U.S. Geological Survey
WL	Working level
WLM	Working level month

A.2 Metric-to-English Conversion Factors

Efforts have been made to use metric units of measure throughout this volume of the EIS. Meteorological data and calculations are examples of subject areas commonly reported in the metric system. To assist the reader in converting from metric values to the more familiar English values, the following conversion table is provided.

To Convert from	To	Multiply by
Centimeters (cm)	Inches (in)	0.394
Centimeters (cm)	Feet (ft)	0.0328
Cubic centimeters (cm^3)	Cubic feet (ft^3)	0.0000353
Cubic meters (m^3)	Cubic feet (ft^3)	35.314
Degrees Centigrade (°C)	Degrees Fahrenheit (°F)	*
Grams (g)	Ounces (oz)	0.0353
Grams (g)	Pounds (lb)	0.00220
Hectare (ha)	Acres	2.471
Kilograms (kg)	Pounds (lb)	2.204
Kilometers (km)	Miles (mi)	0.621
Liter (L)	Cubic feet (ft^3)	0.0353
Liter (L)	Gallons (gal)	0.264
Meter (m)	Feet (ft)	3.281
Meters per second (m/s)	Miles per hour (mi/h)	2.237
Milligrams (mg)	Ounces (oz)	0.000035
Milliliters (mL)	Ounces (oz)	0.0338
Millimeter (mm)	Inches (in)	0.0394
Square meter (m^2)	Square feet (ft^2)	10.764
Tonne (t)	Kilograms (kg)	1,000

*°F = (°C x 9/5) + 32

A.3 Scientific Notation

The conventional notation, when dealing with very large or very small numbers, is awkward and cumbersome. Writing 0.000000000000001, for example, is undesirable, as is calling this number "a millionth of a billionth." Another system would indicate the above number as 1×10^{-15}. This notation then can be converted back to the original number by moving the decimal point according to the power of ten. If the power of ten is positive, for example, the decimal is moved right the number of places indicated by the power. If the power of ten is negative, the decimal is moved left the number of places indicated by the power. An example of a positive and negative power of ten follows:

$$1.25 \times 10^{5} = 125000$$

$$1.25 \times 10^{-4} = 0.000125$$

The notation system used in this volume of the EIS utilizes a value followed by the letter E. After the E is another number, which represents a power of ten. The number 1.055E+03, for example, is 1.055×10^{3}. The number 1.08E-08 is identical to 1.08×10^{-8}.

Prefixes are often added to units (such as curies or grams) to indicate the magnitude of the value. Prefixes used in this statement, their values, and their abbreviations are as follows:

Prefix	Power	Value	Symbol
giga	10^{9}	1,000,000,000	G
mega	10^{6}	1,000,000	M
kilo	10^{3}	1,000	k
centi	10^{-2}	0.01	c
milli	10^{-3}	0.001	m
micro	10^{-6}	0.000001	μ
nano	10^{-9}	0.000000001	n
pico	10^{-12}	0.000000000001	p
femto	10^{-15}	0.000000000000001	f

Thus, 1 kilogram (kg) = 10^{3} grams = 1,000 grams

and 1 microcurie (μCi) = 10^{-6} curie = 0.000001 curie.

A.4 Glossary

activation product: An element made radioactive by bombardment with neutrons, protons, or other nuclear particles.

alpha particle: Positively charged particle emitted by certain radioactive materials. It is made up of two neutrons and two protons, identical to the nucleus of a helium atom. It is the least penetrating type of ionizing radiation.

aquifer: A water-bearing formation below the surface of the earth that can furnish an appreciable supply of water for a well or spring.

arid site: A term often applied to a shallow-land waste disposal site located in an area that receives very little annual precipitation, typically less than 25 cm/yr. In these sites there is little potential for radionuclide transport by rainwater moving downward through the soil.

barrier: (natural or engineered) A material object or substance that delays or prevents migration of water and/or radionuclides into the general environment.

beta particle: A subatomic particle emitted from a nucleus during radioactive decay with a single electrical charge. A negatively charged beta particle is identical to an electron. A positively charged beta particle is called a positron.

biointrusion barriers: An engineered barrier designed to prevent plant roots or burrowing animals from coming into contact with buried waste, and thereby prevent transport of radionuclides by these vectors.

biosphere: That portion of the Earth's environment inhabited by any living organisms. It comprises parts of the atmosphere, the hydrosphere (ocean, seas, inland waters, and subterranean waters), and the lithosphere.

buffer zone: An area surrounding a nuclear facility (e.g., a waste disposal site) established to provide an isolation area between the facility and places used by or accessible to the public.

compaction: The reduction in bulk volume of a material; hence, an increase in its density (weight per unit volume), by application of external pressure. Often it is an economical way to aid in the safe handling of low-level solid wastes.

conditioning of waste:	Those operations that transform waste into a form suitable for transport and/or storage and/or disposal. The operations may include converting the waste to another form, enclosing the waste in containers, and providing additional packaging.
containment:	The confinement of radioactive material in such a way that it is prevented from being dispersed into the environment or is released only at a specified rate.
contamination, radioactive:	The presence of a radioactive substance or substances in or on a material or in a place where they are undesirable or could be harmful.
controlled area:	An area into which access is limited and personnel are subject to appropriate controls (such as individual assessment of dose and special health supervision).
criteria:	Principles or standards on which a decision or judgment can be based. They may be qualitative or quantitative.
critical organ:	The most exposed human organ or tissue or the organ of interest in an analysis, whichever is appropriate.
critical pathway:	The dominant environmental pathway through which a given radionuclide reaches humans.
critical population group (CPG):	For a given radiation source, the members of the public whose exposure is reasonably homogeneous and is typical of individuals receiving the highest effective dose equivalent or organ dose equivalent (whichever is relevant) from the source.
cumulative population health effects:	Fatal cancers or serious genetic effects (i.e., disorders and traits that cause serious handicap at some time during lifetime).
curie (Ci):	A unit rate of radioactive decay; the quantity of any radionuclide that undergoes 3.7×10^{10} (3.7E+10) disintegrations/second. Several fractions of the curie are in common usage, i.e., millicurie, picocurie, etc.
daughter:	Synonym for radioactive decay product.
decay product:	A nuclide (daughter) resulting from the radioactive disintegration of a radionuclide (parent), being formed either directly or as the result of successive transformations in a radioactive series. Also called a daughter. Decay products may be stable or radioactive.

deep-well injection:	The discharge of liquid wastes via deep wells into permeable but confined geological formations deep underground as a means of isolating the wastes from the human environment.
disposal:	The permanent isolation of radioactive waste from the accessible environment whether or not recovery is possible.
distribution coefficient:	The ratio of the concentration of a radionuclide (Ci/g) adsorbed by a solid to the concentration of the same radionuclide (Ci/mL) in solution (water) when the liquid and solid are in contact and the respective radionuclide concentrations have reached equilibrium.
documentation:	Written, recorded, or pictorial information describing, defining, specifying, reporting, or certifying activities, requirements, procedures, or results.
dose, radiation:	The amount of energy imparted to matter by ionizing radiation per unit mass of the matter, usually expressed as the unit rad, or in SI units, 100 rad = 1 gray (Gy).
dose assessment:	An estimate of the radiation dose to an individual or a population group usually by means of predictive modeling techniques, sometimes supplemented by the results of measurements.
dose equivalent:	A term used to express the effective radiation dose when modifying factors have been considered; the product of absorbed dose multiplied by a quality factor multiplied by a distribution factor. It is expressed numerically in rems, or in SI units, 100 rems = 1 sievert (Sv).
dosimetry:	Quantification of radiation doses to individuals or populations resulting from specified exposures.
effective dose equivalent:	The sum of risk-weighted dose equivalents to a specified set of organs, normalized to the risk to the whole body.
effective half-life ($t_{1/2}$):	The time required for one-half of a radioactive material originally present in the body to be removed by biological clearance and radioactive decay.
electron volt (eV):	A unit of energy equivalent to the energy gained by an electron in passing through a potential difference of one volt.
engineered storage:	A method of radioactive waste storage utilizing sealed containers placed in any of a variety of structures especially designed to protect the integrity of containers from accidents and environmental processes.

environmental transfer models:	Mathematical descriptions of the movement of radionuclides through the environment to an end point (usually to man).
evapotranspiration:	The sum total of water lost from the land by evaporation and plant transpiration.
fissile:	Any nucleus capable of undergoing fission due to interaction with neutrons.
fission:	The splitting of a heavy nucleus into approximately equal parts (which are nuclei of lighter elements), accompanied by the release of a relatively large amount of energy. Fission can occur spontaneously, but usually is caused by nuclear absorption of gamma rays, neutrons, or other particles.
fission products:	The nuclides resulting from the fission of heavy elements.
fuel cycle:	The series of steps involved in supplying fuel for nuclear power reactors. It includes mining, refining, the original fabrication of fuel elements, their use in a reactor, chemical processing to recover the fissionable material remaining in the spent fuel, re-enrichment of the fuel material, and refabrication into new fuel elements.
gamma ray:	High-energy, short-wavelength electromagnetic radiation emitted from the nucleus of a decaying radionuclide. Gamma radiation frequently accompanies alpha and beta emissions and always accompanies fission. Gamma rays are very penetrating and are most effectively stopped by dense materials.
general environment:	The total terrestrial, atmospheric, and aquatic environments outside sites within which any activity, operation, or process under the authority of the Atomic Energy Act of 1954, as amended, is conducted.
geometric mean:	The Nth root of the product of a set of N positive numbers; equivalently, the exponential of the arithmetic mean of their logarithms.
geometric standard deviation:	The exponential of the standard deviation of the logarithms of a set of positive numbers.
geosphere:	The solid portion of the earth, synonymous with the lithosphere.
ground water:	Subsurface water within a zone of saturation.

ground-water transport:	The principal means by which radionuclides can be mobilized from an underground repository and moved into the biosphere. Avoiding such transport is the basis for selecting and designing disposal systems.
health impacts:	For the purpose of this analysis, health impacts consist of cumulative population health effects and maximum CPG risk.
heavy metal:	All uranium, plutonium, or thorium placed into a nuclear reactor.
high-level radioactive waste:	Waste whose radioactivity is predominantly characterized by high-energy radiation; consists of the by-products of nuclear reactors and wastes generated by spent fuel processing operations of the nuclear fuel cycle. These are highly radioactive materials resulting from the reprocessing of spent nuclear fuel, including liquid waste produced directly in reprocessing and any solid material derived from such liquid waste.
humid site:	An area from which annual precipitation exceeds water loss by evaporation; hence, there is a significant downward flux of moisture through the soil which could transport radionuclides.
hydraulic conductivity:	Ratio of flow velocity to the gradient of driving force for viscous flow under saturated conditions of a specified liquid in a porous medium.
hydrofracture process:	A process for permanent disposal of radioactive liquid waste in which wastes in the form of a slurry containing hydraulic binders (grouts) are injected by means of fracturing into a deep underground formation (such as a nearly impermeable shale formation) considered to be isolated from the surface. The slurry solidifies in situ, ensuring fixation of the waste.
hydrogeology:	The study of the geological factors relating to the Earth's water.
hydrology:	The study of all waters in and upon the Earth. It includes underground water, surface water, and rainfall, and embraces the concept of the hydrological cycle.
immobilization of waste:	Conversion of a waste to a solid form that reduces the potential for migration or dispersion of radionuclides by natural processes during storage, transport, and disposal.

incineration:	The process of burning a combustible material to reduce its volume and yield an ash residue.
incinerator ash:	The residue remaining after burning waste in a specially designed unit. The volume of radioactive ash will be much less than that of the original waste, and the ash will usually be incorporated into a solid matrix for disposal.
ingest:	Take into the body by way of the digestive tract.
ionizing radiation:	Any electromagnetic or particulate radiation capable of producing ions, directly or indirectly, in its passage through matter.
irregularly inherited disorders:	Genetic conditions with complex causes, constitutional and degenerative diseases, etc.
isotope:	One of two or more atoms with the same atomic number (the same chemical element) but with different atomic weights. Isotopes usually have very nearly the same chemical properties, but some have somewhat different physical properties.
light-water reactor (LWR):	A nuclear reactor whose heat removal system is based on the use of ordinary water as the moderator and reactor coolant.
linear energy transfer (LET):	The rate at which charged particles transfer their energy to the atoms in a medium; expressed as energy lost per distance traveled in the medium.
lognormal distribution:	A normal distribution (i.e., bell-shaped, symmetrical, and of infinite extent) of the logarithms of a set of numbers.
management and storage:	All activities, operations, or processes, administrative and operational, except for transportation, conducted to prepare radioactive wastes for storage or disposal, the storage of any of these materials, or activities associated with disposal of these wastes.
maximum CPG risk:	The probability of contracting a fatal cancer or serious genetic effect, and which is based on the maximum annual effective whole-body dose equivalents to the CPG.
member of the public:	Any individual who is not engaged in operations involving the management, storage, and disposal of materials regulated by these standards. A worker so engaged is a member of the public except when on duty at a site regulated by these standards.

monitoring:	The methodology and practice of measuring levels of radioactivity either in environmental samples or en route to the environment. Examples include ground-water monitoring, gaseous effluent (stack) monitoring, and personnel monitoring.
neutron:	An uncharged elementary particle with a mass slightly greater than that of a proton, and found in the nucleus of every atom heavier than hydrogen. Neutrons sustain the fission chain reaction in a nuclear reactor. Bombardment of materials by neutrons can cause them to become radioactive.
nonstochastic effect:	Those health effects that increase in severity with increasing dose and usually have a threshold.
operational period:	The period during which a nuclear facility is being used for its intended purpose until it is shut down and decommissioned.
pathways model:	A mathematical description, usually in the form of a computer algorithm, that allows estimation of the magnitude and direction of possible radionuclide transport vectors.
rad (radiation absorbed dose):	A measure of the energy imparted to matter by ionizing radiation; defined as 100 ergs/g. A millirad (mrad) is 1E-03 of a rad. In SI units, 100 rad = 1 gray (Gy).
radioactive decay:	A process whereby an atom emits particles or excess energy. This emission is referred to as radioactivity. The energy is usually in the form of alpha or beta particles, gamma or X rays, or neutrons.
radioactive waste:	Any material that contains or is contaminated with radionuclides at concentrations or radioactivity levels requiring regulation by the competent authorities and for which no use is foreseen.
radioactivity:	The property of certain nuclides of spontaneously emitting alpha or beta particles or gamma or X-radiation, or of undergoing spontaneous fission.
radionuclide:	A radioactive nuclide.
relative biological effectiveness (RBE):	The ratio of the dose (rad) of high-LET radiation to the dose of low-LET radiation, which expresses the effectiveness of high-LET compared to low-LET radiation in causing the same biological endpoint.

rem (roentgen equivalent man): A measure of dose equivalence for the biological effect of radiations of different types and energies on man compared to the effect of X rays. In SI units, 100 rem = 1 sievert (Sv).

retrievability: The capability to remove waste from where it has been stored.

risk: For the purposes of radiation protection, the probability that a given individual will incur any given deleterious effect as a result of radiation exposure.

risk analysis: An analysis of the risks associated with a technology wherein the possible events and their probabilities of occurrence are considered, together with their potential consequences, the distribution of these consequences within the affected population(s), the time factor, and the uncertainties of these estimates.

risk projection: Absolute - risk projection based on the assumption that the excess risk from radiation exposure adds to the underlying (base-line) risk by a constant increment dependent on dose but independent of the base-line risk.

Relative - risk projection based on the assumption that the excess risk from radiation exposure is proportional to the base-line risk.

roentgen (R): A unit of measurement of exposure to gamma or X rays in air, equivalent to an absorbed dose in tissue of approximately 0.9 rad. The milliroentgen (mR) is 1E-03 of a roentgen.

saturated zone: A subsurface zone in which all the interstices are filled with water under pressure greater than that of the atmosphere. This zone is separated from the unsaturated zone, i.e., zone of aeration, by the water table.

sensitivity analysis: An analysis of the variation of the solution of a problem with changes in the values of the variables involved. For example, in simple parameter variation, the sensitivity of the solution is investigated for changes in one or more input parameters within a reasonable range about selected reference or mean values.

shallow-ground disposal: Disposal of radioactive waste, with or without engineered barriers, above or below the ground surface, where the final protective covering is of the order of a few meters thick.

solidified waste, radioactive:	Liquid waste or otherwise mobile waste materials (ion exchange resins, etc.) that have been immobilized by incorporation (either physical or chemical) into a solid matrix by some specific treatment.
spent nuclear fuel:	Any nuclear fuel removed from a nuclear reactor after it has been irradiated and whose constituent elements have not been separated by reprocessing.
stochastic effect:	A health effect for which the probability of occurrence is a function of the dose received, but for which the severity of the effect is independent of the dose received.
storage:	Placement of radioactive wastes with planned capability to readily retrieve such materials.
surface water:	Water that fails to penetrate into the sub-soil and flows or gathers on the surface of the ground.
target:	Material subjected to particle bombardment or irradiation in order to induce a nuclear reaction.
teratogenesis:	Production of congenital abnormalities or defects by irradiation of the fetus.
transmissivity, hydraulic:	Rate at which water is transmitted through a unit width of aquifer under a unit hydraulic gradient. It is expressed as the product of the hydraulic conductivity and the thickness of the saturated portion of the aquifer.
transuranic waste:	Waste containing more than 100 nanocuries of alpha-emitting transuranic isotopes, with half-lives greater than 20 years, per gram of waste.
unsaturated flow:	The flow of water in undersaturated soil by capillary action and gravity.
unsaturated zone:	A subsurface zone in which at least some interstices contain air or water vapor, rather than liquid water. Also referred to as "zone of aeration." (See: saturated zone.)
volume reduction:	A treatment that decreases the physical volume of a waste. Volume reduction is used to facilitate subsequent handling, storage, transportation, or disposal of the waste. Typical treatments are mechanical compaction, incineration, or evaporation. Volume reduction results in a corresponding increase in radionuclide concentration.

working level (WL): Any combination of short-lived radon daughters (through Po-214) per liter of air that will result in the emission of 1.3E+05 MeV of alpha energy. An activity concentration of 100 picocuries per liter of Rn-222 in equilibrium with its daughters, corresponds approximately to one WL. A working level month (WLM) is an exposure to a concentration of one WL for 170 hours (about 21 work days).

X ray: Penetrating electromagnetic radiation whose wavelengths are shorter than those of visible light. In nuclear reactions, it is customary to refer to photons originating in the nucleus as gamma rays, and those originating in the extranuclear part of the atom as X rays.

APPENDIX B

NRC LOW-LEVEL RADIOACTIVE WASTE CLASSIFICATION SYSTEM

APPENDIX B: NRC LOW-LEVEL RADIOACTIVE WASTE CLASSIFICATION SYSTEM

The following is the U.S. Nuclear Regulatory Commission's waste classification as set forth in Title 10, Code of Federal Regulations, Part 61.55.

§61.55 Waste Classification

(a) Classification of waste for near surface disposal.

(1) Considerations. Determination of the classification of radioactive waste involves two considerations. First, consideration must be given to the concentration of long-lived radionuclides (and their short-lived precursors) whose potential hazard will persist long after such precautions as institutional controls, improved waste form, and deeper disposal have ceased to be effective. These precautions delay the time when long-lived radionuclides could cause exposures. In addition, the magnitude of the potential dose is limited by the concentration and availability of the radionuclide at the time of exposure. Second, consideration must be given to the concentration of shorter-lived radionuclides for which requirements on institutional controls, waste form, and disposal methods are effective.

(2) Classes of waste.

(i) Class A waste is waste that is usually segregated from other waste classes at the disposal site. The physical form and characteristics of Class A waste must meet the minimum requirements set forth in §61.56(a). If Class A waste also meets the stability requirements set forth in §61.56(b), it is not necessary to segregate the waste for disposal.

(ii) Class B waste is waste that must meet more rigorous requirements on waste form to ensure stability after disposal. The physical form and characteristics of Class B waste must meet both the minimum and stability requirements set forth in §61.56.

(iii) Class C waste is waste that not only must meet more rigorous requirements on waste form to ensure stability but also requires additional measures at the disposal facility to protect against inadvertent intrusion. The physical form and characteristics of Class C waste must meet both the minimum and stability requirements set forth in §61.56.

(iv) Waste that is not generally acceptable for near-surface disposal is waste for which waste form and disposal methods must be different, and in general more stringent, than those specified for Class C waste. In the absence of specific requirements in this part, proposals for disposal of this waste may be submitted to the Commission for approval, pursuant to §61.58 of this part.

(3) Classification determined by long-lived radionuclides. If radioactive waste contains only radionuclides listed in Table 1, classification shall be determined as follows:

(i) If the concentration does not exceed 0.1 times the value in Table 1, the waste is Class A.

(ii) If the concentration exceeds 0.1 times the value in Table 1, but does not exceed the value in Table 1, the waste is Class C.

(iii) If the concentration exceeds the value in Table 1, the waste is not generally acceptable for near-surface disposal.

(iv) For wastes containing mixtures of radionuclides listed in Table 1, the total concentration shall be determined by the sum of fractions rule described in paragraph (a)(7) of this section.

(4) Classification determined by short-lived radionuclides. If radioactive waste does not contain any of the radionuclides listed in Table 1, classification shall be determined based on the concentrations shown in Table 2. However, as specified in paragraph (a)(6) of this section, if radioactive waste does not contain any nuclides listed in either Table 1 or Table 2, it is Class A.

(i) If the concentration does not exceed the value in Column 1, the waste is Class A.

(ii) If the concentration exceeds the value in Column 1, but does not exceed the value in Column 2, the waste is Class B.

(iii) If the concentration exceeds the value in Column 2, but does not exceed the value in Column 3, the waste is Class C.

(iv) If the concentration exceeds the value in Column 3, the waste is not generally acceptable for near-surface disposal.

(v) For wastes containing mixtures of the nuclides listed in Table 2, the total concentration shall be determined by the sum of fractions rule described in paragraph (a)(7) of this section.

(5) Classification determined by both long- and short-lived radionuclides. If radioactive waste contains a mixture of radionuclides, some of which are listed in Table 1 and some of which are listed in Table 2, classification shall be determined as follows:

(i) If the concentration of a nuclide listed in Table 1 does not exceed 0.1 times the value listed in Table 1, the class shall be that determined by the concentration of nuclides listed in Table 2.

Table 1.

Radionuclide	Concentration curies per cubic meter
C-14	8
C-14 in activated metal	80
Ni-59 in activated metal	220
Nb-94 in activated metal	0.2
Tc-99	3
I-129	0.08
Alpha-emitting transuranic nuclides with half-life greater than five years	[1]100
Pu-241	[1]3,500
Cm-242	[1]20,000

[1]Units are nanocuries per gram.

Table 2.

Radionuclide	Concentration, curies per cubic meter		
	Col. 1 A	Col. 2 B	Col. 3 C
Total of all nuclides with less than 5-year half-life	700	([1])	([1])
H-3	40	([1])	([1])
Co-60	700	([1])	([1])
Ni-63	3.5	70	700
Ni-63 in activated metal	35	700	7000
Sr-90	0.04	150	7000
Cs-137	1	44	4600

[1]There are no limits established for these radionuclides in Class B or C wastes. Practical considerations such as the effects of external radiation and internal heat generation on transportation, handling, and disposal will limit the concentrations for these wastes. These wastes shall be Class B unless the concentrations of other nuclides in Table 2 determine the waste to be the Class C independent of these nuclides.

(ii) If the concentration of a nuclide listed in Table 1 exceeds 0.1 times the value listed in Table 1 but does not exceed the value in Table 1, the waste shall be Class C, provided the concentration of nuclides listed in Table 2 does not exceed the value shown in Column 3 of Table 2.

(6) Classification of wastes with radionuclides other than those listed in Tables 1 and 2. If radioactive waste does not contain any nuclides listed in either Table 1 or 2, it is Class A.

(7) The sum of the fractions rule for mixtures of radionuclides. For determining classification for waste that contains a mixture of radionuclides, it is necessary to determine the sum of fractions by dividing each nuclide's concentration by the appropriate limit and adding the resulting values. The appropriate limits must all be taken from the same column of the same table. The sum of the fractions for the column must be less than 1.0 if the waste class is to be determined by that column. Example: A waste contains Sr-90 in a concentration of 50 Ci/m^3 and Cs-137 in a concentration of 22 Ci/m^3. Since the concentrations both exceed the values in Column 1, Table 2, they must be compared to Column 2 values. For Sr-90 fraction, 50/150 = 0.33; for Cs-137 fraction, 22/44 = 0.5; the sum of the fractions = 0.83. Since the sum is less than 1.0, the waste is Class B.

(8) Determination of concentrations in wastes. The concentration of a radionuclide may be determined by indirect methods such as use of scaling factors which relate the inferred concentration of one radionuclide to another that is measured, or radionuclide material accountability, if there is reasonable assurance that the indirect methods can be correlated with actual measurements. The concentration of a radionuclide may be averaged over the volume of the waste, or weight of the waste if the units are expressed as nanocuries per gram.

REFERENCE

NRC82 U.S. Nuclear Regulatory Commission, Licensing Requirements for Land Disposal of Radioactive Waste, 10 CFR 61, Federal Register, 47(248):57446-57482, December 27, 1982.

APPENDIX C

INPUT PARAMETERS AND PARAMETER VALUES USED IN THE PRESTO-EPA ANALYSES

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APPENDIX C: INPUT PARAMETERS AND PARAMETER VALUES USED IN THE PRESTO-EPA ANALYSES

C.1 Input Parameters and Parameter Values Used in the PRESTO-EPA Analyses

A listing of the input parameters in the PRESTO-EPA codes and the values used in the LLW analyses are summarized in this appendix. A complete listing and short description of each of the input parameters is contained in Table C-1. This table also contains parameter values for those parameters whose values remain constant over the various analyses. These are followed by listings of parameter values that vary by setting, waste form, disposal method, and radionuclide (Tables C-2 through C-7). Input parameters for the PATHRAE-EPA code are not included in this Appendix. For information pertaining to the PATHRAE-EPA input parameters, see the User's Manual (EPA87f).

Some of the input parameters are complicated factors whose description is beyond the scope of this summary. Examples are the resuspension factors (RE1, RE2, and RE3) in Table C-2 and the cap performance factors (NYR1, NYR2, PCT1, and PCT2) in Table C-4. The various PRESTO methodology and user's manuals (EPA87a through EPA87e) should be consulted for a complete description. Values for some parameters are based on various equations. These equations are listed in Table C-8. Additional information for some important PRESTO-EPA-BRC input parameters is contained in Section C.2.

Abbreviations are used throughout the data tables. The following key identifies these abbreviations:

	Key
Setting	
	HP - Humid Permeable
	AP - Arid Permeable
	HI - Humid Impermeable
Disposal Method	
Shallow Options:	CS - Conventional Shallow
	IS - Improved Shallow (10 CFR 61)
	ID - Intermediate Depth
	SL - Sanitary Landfill
	EM - Earth-Mounded Tumulus
	CB - Concrete Bunker
	CC - Concrete Canister
Deep Options:	HF - Hydrofracture (Solidified Waste Form)
	DG - Deep Geologic (Solidified Waste Form)
	DI - Deep-Well Injection (Absorbing Waste Form)

BRC Options:

- MD - Municipal Dump
- SF - Suburban Sanitary Landfill
- SI - Suburban Sanitary Landfill w/Incineration
- UF - Urban Sanitary Landfill
- UI - Urban Sanitary Landfill w/Incineration

Waste Form

- AW - Absorbing Waste
- AM - Activated Metal
- SW - Solidified Waste
- TR - Trash
- I/S - Incinerated/Solidified
- ASH - Incinerated
- HIC - Emplaced in High Integrity Containers

Table C-1. Listing and description of all input parameters and the values for parameters which remain constant over the analyses

Parameter	Explanation[1]	Values used in code[2] POP	CPG	DEEP	BRC
Hydrogeologic Parameters					
SINFL	Infiltration rate for non-cap areas (m/yr)	A	A	A	A
DTRAQ	Distance from trench bottom to aquifer (m)	C	C	C	C
SSAT	Fraction of saturation (default equal to 0)	0	0	0	0
RESAT	Fraction of residual saturation	A	A	C	A
DWELL	Distance from trench to well (m)	A	C	A	1.61E+3
GWV	Ground-water velocity (m/yr)	A	A	A	A
AQTHK	Aquifer thickness at well (m)	A	A	A	A
AQDISP	Dispersion angle of pollutant plume (rad)	A	A	A	A
PORA	Aquifer porosity	A	A	A	A
PORV	Sub-trench porosity	A	A	C	A
PERMV	Sub-trench permeability (m/yr)	A	A	A	A
DWS	Distance from well to basin stream (m)	A	N/A	A	3220
VWV	Vertical water velocity (m/yr)	N/A	0	C	N/A
HGRAD	Hydraulic gradient (dimensionless)	N/A	0	1.0	N/A
ALV	Dispersivity in confining stratum (m)	N/A	0	C	N/A
ALH	Dispersivity in the aquifer (m)	N/A	0	0.3	N/A
BDENV	Density of confining stratum (g/cm^3)	N/A	0	C	N/A
RAINF	Rainfall factor (R)	A	A	A	A
ERODF	Soil-erodibility factor (tons/acre-R)	A	A	A	A
SEDELR	Sediment delivery ratio	1.0	1.0	1.0	1.0
PORS	Porosity of surface soil	A	A	A	A
BDENS	Bulk density of soil (g/cm^3)	A	A	A	A
STFLOW	Stream flow rate (m^3/yr)	A	A	A	A
ADEPTH	Active depth of soil (m)	0.1	0.1	0.1	0.1
PD	Distance from trench to local stream (m)	A	C	A	50
PPN	Total annual precipitation (deep option)	N/A	N/A	0	N/A
RUNOFF	Fraction of precipitation that runs off	A	A	A	A
SEEP	Fraction of precip. that becomes deep infiltration (deep option)	N/A	N/A	0	N/A
STPLNG	Slope steepness-length factor	A	A	A	A
COVER	Crop management factor	A	A	A	A
CONTROL	Erosion control practices factor	A	A	A	A
EXTENT	Cross-slope extent of surface spillage (m)	C	C	C	0.45
Engineered Parameters					
NYR1, NYR2	Beginning and ending year of cap failure, respectively	C	C	N/A	1,40
PCT1	Beginning percentage of cap failure at NYR1	C	C	N/A	0
PCT2	Ending percentage of cap failure at NYR2	C	C	N/A	0.3

Table C-1. Listing and description of all input parameters and the values for parameters which remain constant over the analyses (continued)

Parameter	Explanation[1]	Values used in code[2] POP	CPG	DEEP	BRC
Engineered Parameters (continued)					
TAREA	Waste facility surface area (m^2)	C	C	C	0.2
TDEPTH	Depth of operating trench (m)	C	C	C	6.6
OVER	Thickness of trench overburden (m)	C	C	C	0.6
CFT1	Number of years before waste containers fail	B	C	B	0
DCFT	Number of years after CFT1 that containers fail fully	B	C	B	0
Waste-Related Parameters					
PORT	Porosity of material in trench	B	B	B	B
DENCON	Density (mean) of waste materials (g/cm^3)	B	B	B	B
RELFAC	Annual fraction of trench inventory released	B	N/A	B	1.0*
IOPVWV	Option for forming into waste (-1 = no farming; 0 = farming)	-1	-1	-1	-1
TRAM	Amount of each nuclide in trench at t = 0 (Ci)	1.0	N/A	1.0	1.0
SOAM	Amount of surface spillage of each nuclide (Ci)	E	N/A	E	E
STAM (I)	Amount of each nuclide placed in stream (Ci)	0	N/A	0	0
ATAM (I)	Amount of each nuclide placed in air (Ci)	0	N/A	0	0
DECAY (I)	Radiological decay constant (yr^{-1})	E	E	E	E
SOL (I)	Radionuclide solubility (g/ml) (if LEOPT = 5)	N/A	N/A	N/A	N/A
CON (I)	Global health effects conversion factor (HE/Ci)	E	N/A	E	E
XKD1 (I)	Surface Kd for nuclide I (ml/g)	E	E	E	E
XKD2 (I)	Waste Kd for nuclide I (ml/g)	E	E	E	E
XKD3 (I)	Vertical zone Kd for nuclide I (ml/g)	E	E	E	E
XKD4 (I)	Aquifer Kd for nuclide I (ml/g)	E	E	E	E
Atmospheric Pathway Parameters					
H	Atmospheric source height (m)	1.0	1.0	1.0	1.0
VG	Contaminated soil particles' fall velocity (m/s)	A	A	A	A
U	Annual average wind speed (m/s)	A	A	A	A
VD	Deposition velocity (m/s)	A	A	A	A
XG	Distance from trench to local population (m)	A	C	A	C
HLID	Height of inversion layer (m)	300	300	300	300
ROUGH	Hosker's roughness parameter (m)	0.01	0.01	0.01	0.01
FTWIND	Fraction of time wind blows toward population	A	A	A	C
CHIQ	User-specified dispersion coefficient	A	0	A	C
RE1, RE2, RE3	Resuspension rate equation factors	A	A	A	A
RR	Resuspension rate from farming (sec^{-1})	0	0	0	C
FTMECH	Resuspension rate modifier	0	0	0	0.24

*0.25 in AP setting

Table C-1. Listing and description of all input parameters and the values for parameters which remain constant over the analyses (continued)

Parameter	Explanation[1]	Values used in code[2] POP	CPG	DEEP	BRC
Atmospheric Pathway Parameters (Continued)					
IT	Stability class formulation (1 = Pasquill-Gifford)	1	1	1	1
IS	Stability category indicator (4 = neutral stability)	4	4	4	4
FTWND2	Fraction of time wind blows toward population of interest, during incineration	N/A	N/A	N/A	C
CHIQ2	Dispersion coefficient during incineration	N/A	N/A	N/A	C
RMECH	Dust resuspension for onsite operations (kg/sec)	N/A	N/A	N/A	C
RINC	Waste incineration rate (kg/sec)	N/A	N/A	N/A	C
POPG	Number of equivalent full-time, full-exposure onsite visitors exposed to gamma	N/A	N/A	N/A	C
POPDST	Number of equivalent full-time, full-exposure workers and visitors exposed to dust	N/A	N/A	N/A	C
Source Term Parameters					
RELFRAC (1)	Release fraction for absorbing waste (CPG)	N/A	0	N/A	N/A
RELFRAC (2)	Release fraction for activated metals (CPG)	N/A	E	N/A	N/A
RELFRAC (3)	Release fraction for the trash (CPG)	N/A	E	N/A	N/A
RELFRAC (4)	Release fraction for solidified waste (CPG)	N/A	E	N/A	N/A
RELFRAC (5)	Release fraction for incin./solid. waste (CPG)	N/A	E	N/A	N/A
FRTRSH	Fraction of waste that is not in water-tight containers (CPG)	N/A	0.555	N/A	N/A
FTRAB	Fraction of trash that is absorbing waste (CPG)	N/A	1.0	N/A	N/A
SPLAW	Spillage fraction for absorbing waste (CPG)	N/A	C	N/A	N/A
SPLAM	Spillage fraction for activated metal (CPG)	N/A	0	N/A	N/A
SPLTR	Spillage fraction for trash (CPG)	N/A	C	N/A	N/A
SPLSW	Spillage fraction for solidified waste (CPG)	N/A	0	N/A	N/A
SPLIS	Spillage fraction for incin./solid. waste (CPG)	N/A	0	N/A	N/A
CIAW (I)	Absorbing waste activity (Ci) (CPG)	N/A	E	N/A	N/A
CIAM (I)	Activated metal activity (Ci) (CPG)	N/A	E	N/A	N/A
CITR (I)	Trash activity (Ci) (CPG)	N/A	E	N/A	N/A
CISW (I)	Solidified waste activity (Ci) (CPG)	N/A	E	N/A	N/A
CIIS (I)	Incinerated/solidified activity (Ci) (CPG)	N/A	E	N/A	N/A
FVOLAT (I)	Fraction of each radionuclide released to the atmosphere through incineration (BRC)	N/A	N/A	N/A	F
Biological Pathway Parameters					
WWATL	Fraction of irrigation supplied by contaminated well (1 = 100%)	A	A	A	A
WWATA	Fraction of water for livestock supplied by contaminated well	A	A	A	A

Table C-1. Listing and description of all input parameters and the values for parameters which remain constant over the analyses (continued)

Parameter	Explanation[1]	Values used in code[2] POP	CPG	DEEP	BRC
Biological Pathway Parameters (Continued)					
WWATH	Fraction of water for humans supplied by contaminated well	A	A	A	A
SWATL	Fraction of irrigation water supplied by contaminated stream	A	A	A	A
SWATA	Fraction of water for livestock supplied by contaminated stream	A	A	A	A
SWATH	Fraction of water for humans supplied by contaminated stream	A	A	A	A
Y1, Y2	Agricultural productivity for pasture grass and other consumed vegetation, respectively (kg/m^2)	A	A	A	A
PP	Surface density of farmed soil (kg/m^3)	240	240	240	240
XAMBWE	Weathering removal decay constant (h^{-1})	0.0021	0.0021	0.0021	0.0021
TE1, TE2	Period of time pasture grass or vegetables are exposed to contaminated air while in fields (h)	720 1440	720 1440	720 1440	720 1440
TH1	Delay time between harvest and consumption of: pasture grass	A	A	A	A
TH2	Stored feed	2160	2160	2160	2160
TH3	Leafy vegetables (maximum individual)	24	24	24	24
TH4	Produce (maximum individual)	1440	1440	1440	1440
TH5	Leafy vegetables (population)	336	336	336	336
TH6	Produce (population)	336	336	336	336
FP	Fraction of year animals graze on pasture grass	A	A	A	A
FS	Fraction of daily feed which is fresh grass	A	A	A	A
QFC	Amount of feed consumed daily by cattle (kg)	50	50	50	50
QFG	Amount of feed consumed daily by goats (kg)	6	6	6	6
TF1, TF2	Transport time from animal feed via milk to maximum individual and general population (h)	48 96	48 96	48 96	48 96
TS	Time between slaughter and consumption (h)	A	A	A	A
ABSH	Absolute humidity of atmosphere (g/m^3)	A	A	A	A
P14	Fractional equilibrium ratio for C-14	1.0	1.0	1.0	1.0
XRTH	Maximum root depth for onsite farming (m)	0	0	0	0
RTGR	Root growth rate constant (yr^{-1})	0	0	0	0
FI	Fraction of year that crops are irrigated	A	A	A	A
WIRATE	Irrigation rate (L/m^2-h)	A	A	A	A
QCW, QGW, QBW	Amount of water consumed by milk cows, milk goats, and beef cattle, respectively (L/d)	60, 8 50	60, 8 50	60, 8 50	60, 8 50
ULEAFY	Human uptake of leafy vegetables (kg/yr)	A	A	A	A
UPROD	Human uptake of produce (kg/yr)	A	A	A	A

Table C-1. Listing and description of all input parameters and the values for parameters which remain constant over the analyses (continued)

Parameter	Explanation[1]	Values used in code[2] POP	CPG	DEEP	BRC
Biological Pathway Parameters (Continued)					
UCMILK	Human uptake of cow milk (L/yr)	A	A	A	A
UGMILK	Human uptake of goat milk (L/yr)	0	0	0	0
UMEAT	Human uptake of meat (kg/yr)	A	A	A	A
UWAT	Human uptake of drinking water (L/yr)	A	A	A	A
UAIR	Inhalation rate (m^3/yr)	8035	8035	8035	8035
POP	Local population size	A	1.0	A	C
RA (I)	Radionuclide retention fraction for air	2.0E-1	2.0E-1	2.0E-1	2.0E-1
RW (I)	Radionuclide retention fraction for irrigation	2.5E-1	2.5E-1	2.5E-1	2.5E-1
BV (I)	Radionuclide soil-to-plant uptake factor for vegetative parts	E	E	E	E
BR (I)	Radionuclide soil-to-plant uptake factor to grain	E	E	E	E
FMC (I)	Nuclide forage-to-milk transfer factor for cows	E	E	E	E
FMG (I)	Nuclide forage-to-milk transfer factor for goats	E	E	E	E
FF (I)	Nuclide forage-to-beef transfer factor	E	E	E	E
Administrative Parameters					
MAXYR	The number of years simulation will run	10,000	1000	10,000	10,000
NONCLD	The number of nuclides (I) used in run	40	40	40	40
IDISP	Indication variable for mode of disposal (1 = CPG)	N/A	1	C	N/A
IPRT1, IPRT2	Beginning and ending year of printed summaries	0 1000	0 1000	0 10,000	0 1000
IDELT	Time step between printed summaries	100	100	1000	100
LIND	Option parameter to specify max. indv. or pop. H.E. for DARTAB (1 = POP, 0 = max. indv.)	1	0	1	1
IAVG1, IAVG2	Beginning and ending years for averaging nuclide concentration values	0 1000	1 1000	1 10,000	1 1000
IAQSTF	Control parameter for flow from aquifer to basin (0 = yes, - = no)	0	-	-	0
ITWO	Secondary year for which organ dose summary table will be printed	N/A	-	-	N/A
CPRJ	Fraction of unused local water flowing to basin (0 = 100%)	0	0	0	0
INTYR	Maximum number of years for which detailed output summaries will be printed, in addition to 1,000-year summary	100, 500	N/A	1000, 5000	100, 500

Table C-1. Listing and description of all input parameters and the values for parameters which remain constant over the analyses (continued)

Parameter	Explanation[1]	Values used in code[2] POP	CPG	DEEP	BRC
Miscellaneous Parameters					
LEOPT	Leaching option (2 = immersed fraction, 5 = release fraction)	B	2	B	B
IRRES1,	Beginning and ending year of mech. suspension for	0	0	0	0
IRRES2	farming	0	0	0	0
IBSMT	Basement gamma exposure correction factor (0 = yes, -1 = no)	-1	-1	-1	-1
IRST	Number of years of restricted site use	100	100	100	C
FACTIM	Number of years of active site operation	20	20	40*	20
TMN	Number of years of active site maintenance	0	0	0	0
FGAM	Factor of intensity and duration of gamma exposure from basement	0	0	0	1.0
FRACB	Fraction of waste impacted (1 = 100%)	N/A	1.0	1.0	N/A
		*20 for DEEP Geologic Disposal			
Infiltration Parameters[3]					
TWT	Trench width (m)	A	A	N/A	A
SLOP	Trench cap slope	A	A	N/A	A
XKI	Permeability of trench cover (m/h)	A	A	N/A	A
EGSG	Porosity in gravity zone	A	A	N/A	A
EPSP	Porosity in pellicular zone	A	A	N/A	A
YGMAX	Trench cap thickness (m)	A	A	N/A	A
XDE	Equivalent upward diffusivity (m/h)	A	A	N/A	A
XKE	Equivalent upward hydraulic conductivity (m/h)	A	A	N/A	A
YPI	Pellicular infiltration capacity (m/h)	A	A	N/A	A
YGI	Gravity infiltration capacity (m/h)	A	A	N/A	A
DTH (I)	Maximum day length for month I	*	*	N/A	*
TMP (I)	Daily mean temperature for month I	*	*	N/A	*
P (MO, IDA)	Hourly precipitation for month MO and day of month IDA	*	*	N/A	*

*see PRESTO-EPA User's Manuals (EPA87a,b,c,d,e)

Table C-1. Listing and description of all input parameters and the values for parameters which remain constant over the analyses (continued)

Key for Table C-1

A - Input parameter which varies by setting (see Table C-2).
B - Input parameter which varies by waste form (see Table C-3).
C - Input parameter which varies by disposal method (see Table C-4).
E - Input parameter which varies by radionuclide (see Tables C-5 through C-7).
F - See additional information relating to PRESTO-EPA-BRC, in Section C.2 of this Appendix.

1 - Explanations are in summary form; for a more detailed description of each input parameter, see the various PRESTO-EPA Methodology and User's Manuals (EPA87a through EPA87e).
2 - POP - PRESTO-EPA-POP (Population Health Effects Code)
CPG - PRESTO-EPA-CPG (Critical Population Group Dose Analysis Code)
DEEP - PRESTO-EPA-DEEP (Health Effects from Deep Disposal Code)
BRC - PRESTO-EPA-BRC (Health Effects from Unregulated Disposal Code)
3 - Parameters from the INFIL data set (sub-program in PRESTO-EPA). See PRESTO-EPA Methodology Manual for details (EPA87a,b).

N/A - Not Applicable.

Table C-2. Input parameters and parameter values which vary by setting (parameters listed with an "A" in Table C-1)

Parameter	Explanation	Value by setting[1]		
		HP	AP	HI
WWATL	Fraction of irrigation water supplied by contaminated well (1 = 100%)	0	1	0
WWATA	Fraction of water for livestock supplied by contaminated well	0.5	1	0
WWATH	Fraction of water for humans supplied by contaminated well	1.0	1	0
SWATL	Fraction of irrigation water supplied by contaminated stream	0	0	0.1
SWATA	Fraction of water for livestock supplied by contaminated stream	0	0	0.1
SWATH	Fraction of water for humans supplied by contaminated stream	0	0	1.0
SINFL	Annual infiltration rate for non-cap portions of site (m/yr)	0.43	0	3.0E-3
RSAT	Fraction of residual saturation	0.17	0.03	0.31
DWS[a]	Distance from well to basin stream (m)	457	30,000	250
DWELL[a]	Distance from trench to nearest well (m)	457	29,000	250
GWV	Velocity of ground water in aquifer (m/yr)	27.8	90	0.03
AQTHK	Thickness of aquifer at well location (m)	30.5	37	11
AQDISP	Dispersion angle of pollutant plume in aquifer (radians)	0.3	0.3	0.1
PORA	Aquifer porosity	0.39	0.40	0.25
PORV[a]	Sub-trench porosity	0.35	0.40	0.32
PERMV	Sub-trench permeability (m/yr)	2.2	63.4	0.019
VG	Fall velocity of contaminated soil particles due to gravity (m/sec)	0.01	0.027	0.01
U	Annual average wind speed toward critical population (m/sec)	2.01	4.8	5.0
VD	Deposition velocity (m/sec)	0.01	0.027	0.01
XG[a]	Distance from trench to population of interest (m)	480	29,000	7240
FTWIND[a]	Fraction of time wind blows toward population of interest (values from RADE program which reflect weighted population)	0.4458	19.477	0.288
CHIQ[a]	User-specified dispersion coefficient	3.856E-5	5.186E-8	3.25E-7
RE1	Factors in resuspension rate equation	1E-6	1E-4	1E-6
RE2	Factors in resuspension rate equation	-0.15	-0.15	-0.15
RE3	Factors in resuspension rate equation	1E-11	1E-9	1.0E-10
RAINF	Rainfall factor	250	20	100
ERODF	Soil-erodibility factor (tons/acre x R) R = RAINF	0.23	0.5	0.19
STPLNG	Slope steepness - length factor	0.27	0.26	0.54
COVER	Crop management factor	0.30	0.30	0.10
CONTRL	Erosion control practices factor	0.30	0.40	1.0
PORS	Porosity of the surface soil	0.39	0.30	0.30
BDENS	Bulk density of the soil (g/cm^3)	1.6	1.55	1.49

Table C-2. Input parameters and parameter values which vary by setting (parameters listed with an "A" in Table C-1) (continued)

Parameter	Explanation	Value by setting[1]		
		HP	AP	HI
STFLOW	Annual flow rate of nearest stream (m^3/yr)	3.57E+5	1000	3.65E+8
PD[a]	Distance from trench to nearest stream (m)	460	4000	50
RUNOFF	Fraction of annual precipitation that runs off annually	0.29	0.005	0.56
TH1	Delay time between harvest and consumption of pasture grass (h)	0	0	330
Y1	Agricultural productivity for pasture grass (kg/m^2)	0.67	0.04	0.336
Y2	Agricultural productivity for consumed vegetation (kg/m^2)	0.65	0.76	0.56
FP	Fraction of year animals graze on pasture grass	1.0	0.47	0.27
FS	Fraction of animal's feed that is fresh grass during period animals are in pasture	0.83	1.0	0.10
TS	Length of time between slaughter of animal and consumption of meat (h)	480	480	336
ABSH	Absolute humidity of atmosphere (g/m^3)	9.9	4.4	6.4
FI	Fraction of year that crops are irrigated	0.40	0.65	0.08
WIRATE	Irrigation rate (L/m^2 - h)	0.015	0.114	0.042
ULEAFY	Human uptake of leafy vegetables (kg/yr)	14.0	16.5	13.9
UPROD	Human uptake of produce (kg/yr)	88.5	94.2	84.9
UCMILK	Human uptake of cow milk (L/yr)	89.4	122.7	112.3
UMEAT	Human uptake of meat (kg/yr)	62.8	61.6	62.1
UWAT	Human uptake of drinking water (L/yr)	481.9	467.9	391.6
POP[a]	Population in local area for first 1,000 years	25	15	4,285
TWT	Trench width (m)	30.5	12.2	87.8
SLOP	Trench cap slope	0.01	0.0	0.25
XKI	Permeability of trench cover (m/h)	0.02	4.0	3.6E-5
EGSG	Porosity in gravity zone	0.25	0.35	0.01
EPSP	Porosity in pellicular zone	0.24	0.03	0.47
YGMAX	Trench cap thickness (m)	1.2	1.5	3.1
XDE	Equivalent upward diffusivity (m/h)	3.5E-4	2.0E-3	8.0E-5
XKE	Equivalent upward hydraulic conductivity (m/h)	1.4E-6	1.0E-4	9.0E-5
YPI	Pellicular infiltration capacity (m/h)	0.01	0.1	0.01
YGI	Gravity infiltration capacity (m/h)	1.2	1.5	3.1

[a]Values only for POP - see Table C-1.
[1]See key on page C-3.

Table C-3. Input parameters and parameter values which vary by waste form (parameters listed with a "B" in Table C-1)

Parameter	Explanation	Value by waste form[1]									
		AW	TR	AM	SW	I/S	ASH	HIC	HF	DI	DG
LEOPT[a]	Leaching option (1 = total contact, 2 = immersed fraction, 5 = release fraction). See PRESTO methodology manual for more information (EPA85a).	2	2	5	5	5	2	N/A	5	1	5
PORT	Porosity of material within trench	0.4	0.6	0.5	0.2	0.2	0.35	N/A	0.25	0.13	0.295
DENCON	Mean density of waste in trench (g/cm^3)	0.8	0.8	3.5	1.8	2.0	0.89	N/A	1.645	2.102	1.70
CFTI[b]	Number of years before waste container starts to fail	20	0	0	20	20	0	300	0	0	20
DCFT[b]	Number of years after CFTI that all containers have failed	50	0	0	50	50	0	0	0	0	50
RELFAC	Annual fraction of waste inventory released	N/A	N/A	*	*	*	N/A	N/A	1.0E-4	0.005	1.0E-6

[a] Values for POP and BRC only - see Table C-1.
[b] Values for POP and DEEP only - see Table C-1.

[1]See key on page C-3.

N/A - Not Applicable.

*See Table C-8.

Table C-4. Input parameters and parameter values which vary by disposal method (parameters listed with a "C" in Table C-1)

Parameter	Explanation	Input values by disposal method[1]													
		CS	IS	ID	SL	EM	CB	CC	HF	DG	DI	MD	SF	SI	UF	UI
NYR1	Beginning year of cap failure	100	100	100	1	100	100	100	0	0	0	1	1	1	1	1
NYR2	Ending year of cap failure	300	300	300	40	300	300	300	0	0	0	40	40	40	40	40
PCT1	Fraction of cap failed in year NYR1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
PCT2	Fraction of cap failed in year NYR2 - between NYR1 and NYR2 a linear interpolation between PCT1 and PCT2 determines fraction of cap that has failed	0.2	0.1	0.1	0.3	0.15	0.075	0.075	0	0	0	0.3	0.3	0.3	0.3	0.3
TAREA	Total combined facility surface area	0.2	0.143	0.167	0.5	0.133	0.167	0.133	2	0.33	0.0167	0.2	0.2	0.2	0.2	0.2
TDEPTH	Nominal depth of trench in shallow options, and waste thickness in deep options (m)	7.0	12	16	2.6	9.5	8.0	11.5	0.5	3.0	60	6.6	6.6	6.6	6.6	6.6
OVER	Thickness of trench overburden (m)	2.0	5	10	0.6	2.0	2.0	4.0	300	300	900	0.6	0.6	0.6	0.6	0.6
EXTENT	Surface length of trench as disposal area parallel to stream (on a unit volume basis) (m)	0.45	0.38	0.41	0.71	0.36	0.41	0.36	1.414	0.574	0.129	0.45	0.45	0.45	0.45	0.45
DTRAQ	Distance from trench bottom to nominal aquifer depth (m)	*	*	*	*	*	*	*	261	176	848	*	*	*	*	*
PORV	Sub-trench porosity	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.32	0.25	0.20	N/A	N/A	N/A	N/A	N/A
DWELL	Distance from trench to well (m)	*	*	*	*	*	*	*	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A

*See Table C-8.

Table C-4. Input parameters and parameter values which vary by disposal method (parameters listed with a "C" in Table C-1) (continued)

Parameter	Explanation	Input values by disposal method[1]													
		CS	IS	ID	SL	EM	CB	CC	HF	DG	DI	MD	SF	SI	UF	UI
PD	Distance from trench to local stream (m)	*	*	*	*	*	*	*	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
CFT1	Number of years before waste containers fail	20	20	20	20	20	100	100	0	20	0	N/A	N/A	N/A	N/A	N/A
DCFT	Number of years after CFT1 that containers fail fully	50	50	50	50	50	200	200	0	50	0	N/A	N/A	N/A	N/A	N/A
XG	Distance from trench to local population - atmospheric (m)	*	*	*	*	*	*	*	*	*	*	1.6E+4	1.8E+4	1.8E+4	2.2E+4	2.2E+4
SPLAW	Spillage fraction for absorbing waste (CPG only)	1E-7	1E-7	1E-7	1E-3	1E-7	1E-7	1E-7	1E-7	N/A	N/A	N/A	N/A	N/A	N/A	N/A
SPLTR	Spillage fraction for trash waste (CPG only)	1E-7	1E-7	1E-7	1E-3	1E-7	1E-7	1E-7	1E-7	N/A	N/A	N/A	N/A	N/A	N/A	N/A
POPG	Number of equivalent full-time, full-exposure onsite visitors exposed to gamma (BRC only)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	5E-7	4E-7	2E-6	3E-7	1E-5
POPDST	Number of equivalent full-time, full-exposure workers and visitors exposed to dust (BRC only)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	1E-6	1E-6	3E-5	1E-6	1E-5
RINC	Waste incineration rate (kg/h) (BRC only)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0	0	1E-6	0	1E-6
IDISP	Indicator variable for the mode of disposal	N/A	N/A	N/A	N/A	N/A	N/A	N/A	3	4	2	N/A	N/A	N/A	N/A	N/A
VWV	Vertical water velocity (m/yr)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	1.0	5	0.5	N/A	N/A	N/A	N/A	N/A
ALV	Dispersivity in confining stratum (m)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	5	25	40	N/A	N/A	N/A	N/A	N/A

Table C-4. Input parameters and parameter values which vary by disposal method (parameters listed with a "C" in Table C-1) (continued)

Parameter	Explanation	Input values by disposal method[1]													
		CS	IS	ID	SL	EM	CB	CC	HF	DG	DI	MD	SF	SI	UF	UI
BDENV	Density of confining stratum (g/cm^3)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	2.3	1.85	1.5	N/A	N/A	N/A	N/A	N/A
RSAT	Fraction of residual saturation	N/A	N/A	N/A	N/A	N/A	N/A	N/A	0.31	0.02	0.1	N/A	N/A	N/A	N/A	N/A
POP	Local population size	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	6E4	1.75E5	1.75E5	1E6	1E6
CHIQ2	Dispersion coefficient during incineration (BRC only)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	F	F	F	F	F
FTWIND2	Fraction of time wind blows toward population during incineration (BRC only)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	F	F	F	F	F
RMECH	Dust resuspension for onsite operations (BRC only)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	F	F	F	F	F
RR	Resuspension rate for farming (BRC only)	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	5E-4	4E-4	4E-4	4E-4	4E-4

*See Table C-8.

[1]See key on page C-3.

N/A - Not Applicable.

F - See additional information relating to PRESTO-EPA-BRC, in Section C.2 of this Appendix.

Table C-5. Input parameters and parameter values which vary by radionuclide (parameters listed with an "E" in Table C-1)

Parameter	Explanation	Input value
TRAM(I)	Amount (Ci) of each radionuclide found in the trench at the beginning of the simulation. One curie is assumed, with constant rate of deposit and decay over 20-year operating life of site	Table C-6
SOAM(I)	Amount of spillage onto the surface that exists at the beginning of simulation, as a fraction of TRAM	1.0E-7*
STAM(I)	Amount (Ci) of radioactivity placed into stream at beginning of simulation	0
ATAM(I)	Amount (Ci) of radioactivity placed in air above trench at beginning of simulation	0
DECAY(I)	Radiological decay constant (yr^{-1})	Table C-6
CON(I)	Conversion factor for basin health effects (health effects/Ci released)	Table C-7
XKD1(I)	Surface K_d of radionuclide I (ml/g)	Table C-7
XKD2(I)	Waste K_d of radionuclide I (ml/g)	Table C-7
XKD3(I)	Vertical zone K_d of radionuclide I (ml/g)	Table C-7
XKD4(I)	Aquifer K_d of radionuclide I (ml/g)	Table C-7
RA(I)	Radionuclide retention fraction for air	2.0E-1
RW(I)	Radionuclide retention fraction for irrigation	2.5E-1
BV(I)	Radionuclide soil-to-plant uptake factors for vegetative parts (d/kg)	Table C-6
BR(I)	Radionuclide soil-to-plant uptake factors for grain (d/kg)	Table C-6
FMC(I)	Radionuclide forage-to-milk transfer factors for cows (d/L)	Table C-6
FMG(I)	Radionuclide forage-to-milk transfer factors for goats (d/L)	Table C-6
FF(I)	Radionuclide forage-to-beef transfer factors (d/kg)	Table C-6

*1.0E-3 for sanitary landfill scenarios.

Table C-6. Input parameters and parameter values which vary by radionuclide (parameters listed with an "E" in Table C-1)

Radionuclide	Parameter values							
	TRAM (Ci)[1]	SOAM (Ci)[2]	DECAY (yr-1)	BV	BR	FMC	FMG	FF
Hydrogen-3	6.17E-1	6.17E-8	5.64E-02	4.80	4.80	1.00E-2	1.70E-1	1.20E-2
Carbon-14	1.00	1.00E-7	1.21E-04	5.50	5.50	1.20E-2	1.00E-1	3.10E-2
Manganese-54	9.52E-2	9.52E-9	8.09E-01	2.50E-1	5.00E-2	3.50E-4	2.50E-4	4.00E-4
Iron-55	2.21E-1	2.21E-8	2.57E-01	4.00E-3	1.00E-3	2.50E-4	1.30E-4	2.00E-4
Nickel-59	1.00	1.00E-7	8.66E-06	6.00E-2	6.00E-2	1.00E-3	6.70E-3	6.00E-3
Cobalt-60	3.76E-1	3.76E-8	1.32E-01	2.00E-2	7.00E-3	2.00E-3	1.00E-3	2.00E-2
Nickel-63	1.00	1.00E-7	7.53E-03	6.00E-2	6.00E-2	1.00E-3	6.70E-3	6.00E-3
Strontium-90	8.02E-1	8.02E-8	2.42E-02	2.50	2.50E-1	1.50E-3	1.40E-2	3.00E-4
Niobium-94	1.00	1.00E-7	3.47E-05	2.00E-2	5.00E-3	2.00E-2	2.50E-3	2.50E-1
Technetium-99	1.00	1.00E-7	3.25E-06	9.50	1.50	1.00E-2	2.50E-2	8.50E-3
Ruthenium-106	1.04E-1	1.04E-8	6.89E-01	7.50E-2	2.00E-2	6.00E-7	1.30E-4	2.00E-3
Antimony-125	2.26E-1	2.26E-8	2.50E-01	2.00E-1	3.00E-2	1.00E-4	1.50E-3	1.00E-3
Iodine-129	1.00	1.00E-7	4.08E-08	1.00	1.00	1.00E-2	3.00E-1	7.00E-3
Cesium-134	1.77E-1	1.76E-8	3.36E-01	8.00E-2	3.00E-2	7.00E-3	3.00E-1	2.00E-2
Cesium-135	1.00	1.00E-7	2.30E-07	8.00E-2	3.00E-2	7.00E-3	3.00E-1	2.00E-2
Cesium-137	8.10E-1	8.10E-8	2.31E-02	8.00E-2	3.00E-2	7.00E-3	3.00E-1	2.00E-2
Cerium-144	9.06E-2	9.06E-9	8.90E-01	1.00E-2	4.00E-3	2.00E-5	5.00E-6	7.50E-4
Europium-154	6.84E-1	6.84E-8	4.33E-02	2.50E-3	2.50E-3	2.00E-5	2.00E-5	4.80E-3
Radium-226	1.00	1.00E-7	4.34E-04	1.50E-2	1.50E-3	4.50E-4	5.00E-6	2.50E-4
Uranium-234	1.00	1.00E-7	2.83E-06	8.50E-3	4.00E-3	6.00E-4	5.00E-4	2.00E-4
Uranium-235	1.00	1.00E-7	9.85E-10	8.50E-3	4.00E-3	6.00E-4	5.00E-4	2.00E-4
Neptunium-237	1.00	1.00E-7	3.30E-07	4.30E-3	4.30E-3	5.00E-6	5.00E-6	5.50E-5
Uranium-238	1.00	1.00E-7	1.55E-10	8.50E-3	4.00E-3	6.00E-4	5.00E-4	2.00E-4
Plutonium-238	1.00	1.00E-7	7.90E-03	4.50E-4	4.50E-5	1.00E-7	1.50E-6	5.00E-7
Plutonium-239	1.00	1.00E-7	2.87E-05	4.50E-4	4.50E-5	1.00E-7	1.50E-6	5.00E-7
Plutonium-241	6.36E-1	6.36E-8	5.25E-02	4.50E-4	4.50E-5	1.00E-7	2.50E-6	5.00E-7
Americium-241	1.00	1.00E-7	1.51E-03	5.50E-3	2.50E-4	4.00E-7	0.0	3.50E-6
Plutonium-242	1.00	1.00E-7	1.83E-06	4.50E-4	4.50E-5	1.00E-7	1.50E-6	5.00E-7
Americium-243	1.00	1.00E-7	9.40E-05	5.50E-3	2.50E-4	4.00E-7	0.0	3.50E-6
Curium-243	8.20E-1	8.20E-8	2.17E-02	8.50E-4	1.50E-5	2.00E-5	0.0	3.50E-6
Curium-244	7.06E-1	7.06E-8	3.94E-02	8.50E-4	1.50E-5	2.00E-5	0.0	3.50E-6

[1]One curie of each nuclide is disposed of. These values assume the disposal rate is constant over the 20-year life of the site, with decay during that period.

[2]Spillage fraction is 1.0E-7 of the value of TRAM (1.0E-3 for sanitary landfill scenarios, 0.2 for BRC municipal dump scenarios, and 0.1 for BRC sanitary landfill scenarios).

Table C-7. Input parameters and parameter values which vary by radionuclide and setting (parameters listed with an "E" in Table C-1)

Radionuclide	Humid Permeable Setting (HP)					Arid Permeable Setting (AP)					Humid Impermeable Setting (HI)				
	XKD1	XKD2	XKD3	XKD4	CON(I)	XKD1	XKD2	XKD3	XKD4	CON(I)	XKD1	XKD2	XKD3	XKD4	CON(I)
H-3	0.01	0.01	0.01	0.01	5.36E-6	0.01	0.01	0.01	0.01	6.43E-6	0.01	0.01	0.01	0.01	4.49E-6
C-14	0.01	0.01	0.01	0.01	5.39E-3	0.01	0.01	0.01	0.01	5.39E-3	0.01	0.01	0.01	0.01	5.38E-3
Fe-55	6000	50	6000	6000	3.98E-4	2000	50	2000	2000	3.72E-4	1500	50	1500	1500	3.94E-4
Ni-59	150	50	150	150	1.95E-5	3000	50	3000	3000	4.49E-4	150	50	150	150	1.87E-5
Co-60	55	50	55	55	4.85E-4	5000	50	5000	5000	7.07E-2	40	40	40	40	6.80E-4
Ni-63	150	50	150	150	4.81E-5	3000	50	3000	3000	3.07E-4	150	50	150	150	4.55E-5
Sr-90	150	30	150	20	2.70E-4	150	30	150	150	5.97E-3	30	30	30	30	2.48E-4
Nb-94	350	70	350	350	8.41E-2	350	70	350	350	1.67E+0	350	70	350	350	8.55E-2
Tc-99	0.5	0.5	0.5	0.5	1.44E-4	0.1	0.1	0.1	0.1	1.46E-1	0.033	0.033	0.033	0.033	2.52E-4
Ru-106	220	70	220	220	6.81E-4	220	70	220	220	8.54E-5	220	70	220	220	5.94E-4
Sb-125	45	45	45	45	4.10E-7	45	45	45	45	4.10E-7	45	45	45	45	2.49E-5
I-129	3	3	3	3	5.64E-3	0.1	0.1	0.1	0.1	1.17E+0	0.01	0.01	0.01	0.01	6.05E-3
Cs-134	100	100	1000	500	7.81E-2	5000	2000	10,000	5000	7.41E-2	200	200	250	250	7.77E-2
Cs-135	100	100	1000	500	8.01E-3	5000	2000	10,000	5000	2.79E-2	200	200	250	250	7.97E-3
Cs-137	100	100	1000	500	5.35E-2	5000	2000	10,000	5000	1.46E-1	200	200	250	250	5.32E-2
Ba-137m	100	100	1000	500	5.35E-2	5000	2000	10,000	5000	1.46E-1	200	200	250	250	5.32E-2
Eu-154	4000	4000	2000	4000	1.34E-4	4000	2000	4000	4000	1.04E-1	4300	2000	4300	4300	4.55E-4
Ti-208	60,000	60,000	60,000	60,000	1.09E-3	60,000	60,000	60,000	60,000	2.94E+0	60,000	60,000	60,000	60,000	3.70E-3
Po-210	220	220	220	220	1.38E-1	220	220	220	220	2.19E-1	220	220	220	220	1.35E-1
Pb-210	220	220	220	220	1.44E-1	220	220	220	220	4.33E-1	220	220	220	220	1.36E-1
Pb-212	60,000	60,000	60,000	60,000	1.55E-3	60,000	60,000	60,000	60,000	1.56E-1	60,000	60,000	60,000	60,000	1.60E-3
Bi-214	220	220	220	220	1.16E-5	220	220	220	220	1.29E+0	220	220	220	220	1.20E-3
Pb-214	220	220	220	220	2.94E-5	220	220	220	220	2.39E-1	220	220	220	220	2.47E-4
Ra-226	220	220	220	220	3.03E-2	220	220	220	220	7.38E-2	220	220	220	220	2.77E-2
Th-228	60,000	60,000	60,000	60,000	2.59E-3	60,000	60,000	60,000	60,000	7.08E-3	60,000	60,000	60,000	60,000	2.33E-3
Ac-228	220	220	220	220	3.27E-4	220	220	220	220	8.79E-1	220	220	220	220	1.11E-3
Ra-228	220	220	220	220	2.40E-2	220	220	220	220	5.34E-2	220	220	220	220	2.20E-2
Th-232	60,000	60,000	60,000	60,000	4.56E-3	60,000	60,000	60,000	60,000	8.82E-3	60,000	60,000	60,000	60,000	4.10E-3
U-234	750	750	750	750	1.78E-4	6	6	6	6	1.19E-3	50	50	50	50	1.56E-4
U-235	750	750	750	750	2.12E-4	6	6	6	6	1.57E-1	50	50	50	50	3.25E-4
Np-237	5	5	5	5	2.80E-1	5	5	5	5	3.63E-1	5	5	5	5	2.76E-1
U-238	750	750	750	750	2.22E-5	6	6	6	6	1.07E-3	50	50	50	50	1.55E-4

Table C-7. Input parameters and parameter values which vary by radionuclide and setting (parameters listed with an "E" in Table C-1) (continued)

Radionuclide	Humid Permeable Setting (HP)					Arid Permeable Setting (AP)					Humid Impermeable Setting (HI)				
	XKD1	XKD2	XKD3	XKD4	CON(I)	XKD1	XKD2	XKD3	XKD4	CON(I)	XKD1	XKD2	XKD3	XKD4	CON(I)
Pu-238	3500	700	3500	3500	3.96E-3	2000	700	2000	2000	6.97E-2	1800	700	1800	*1800	3.35E-2
Pu-239	3500	700	3500	3500	3.73E-2	2000	700	2000	2000	7.86E-3	1800	700	1800	1800	3.65E-3
Pu-241	3500	700	3500	3500	6.65E-4	2000	700	2000	2000	1.19E-3	1800	700	1800	1800	5.79E-4
Am-241	80,000	80	80,000	80,000	4.45E-2	2E+5	80	2E+5	2E+5	1.42E-1	4700	80	4700	4700	5.72E-2
Pu-242	3500	700	3500	3500	3.54E-2	2000	700	2000	2000	7.67E-3	1800	700	1800	1800	3.46E-3
Am-243	80,000	80	80,000	80,000	8.21E-2	2E+5	80	2E+5	2E+5	1.77E-1	4700	80	4700	4700	7.66E-2
Cm-243	3300	700	3300	3300	3.95E-2	3300	700	3300	3300	9.03E-2	3300	700	3300	3300	3.54E-2
Cm-244	3300	700	3300	3300	8.81E-3	3300	700	3300	3300	5.62E-2	3300	700	3300	3300	2.98E-2

Table C-8. Equations relating to various input parameters

- RELFAC - Annual fraction of waste inventory released (POP only)

  RELFAC = (site factor) x (waste factor) x (disposal factor)

Site factors		Waste factor		Disposal factor	
1.0	HI	0.0	TR	1.0E-03	SL
1.0	HP	0.0	AW	4.0E-04	CS
0.25	AP	1.0	AM	2.0E-04	IS
		1.0	SW	2.0E-04	ID
		1.0	I/S	4.0E-04	EM
				5.0E-05	CB
				1.0E-04	CC

- DTRAQ - Distance from trench bottom to nominal aquifer depth (m)

  DTRAQ = (site factor) - TDEPTH

Site factors	
21.6	HP
87	AP
28	HI

- DWELL - distance from trench to well (m) (CPG only)

$$DWELL = \frac{A}{2} + 100$$

$$\text{where, } A = \frac{\text{waste volume } (m^3)}{(TDEPTH - OVER)^* \text{ P.E.}}$$

where, P.E. = placement efficiency =		
	0.5	SL, CS, HF, DG, DI
	0.68	EM, CC
	0.27	CB
	0.80	IS, ID

- PD - distance from trench to local stream (m) (CPG only)

$$PD = PD^* + \frac{A}{2}$$

where PD* =		
	460	HP
	4000	AP
	100	HI

- XG - distance from trench to local population - atmospheric (m) (CPG only)

  XG = PD + 1

C.2 Additional Information for PRESTO-EPA-BRC Parameters

The following parameters are important to the BRC risk analysis.

In the PRESTO-EPA-BRC analysis of collective population exposures, the effects of waste form (En84) on radionuclide transport are reflected primarily in the variables' waste porosity (PORT) and waste density (DENCON).

	Trash	Ash
PORT (unitless)	0.60	0.35
DENCON (gm/cc)	0.80	0.89

The release rate of the radionuclides out of the trench, in individual BRC waste streams, has been simulated in a dynamic release submodel through the use of distribution coefficients (XKD2). The submodel assumes that the total mass of radionuclides contaminating the waste will be in two forms, solids and dissolved solids, after water infiltrates through the trench. The radionuclides in the solid phase will remain stationary and those in the dissolved phase will become mobile.

Other radionuclide-specific input data requirements related to waste form and processing include DECAY (I) and FVOLAT (I). DECAY (I) is the radiological decay constant ($year^{-1}$) for each radionuclide I.

For scenarios involving incineration of wastes at the sanitary landfill or pathological incinerator prior to disposal, the variable FVOLAT (I) is employed. This variable is defined as the fraction of each radionuclide released to the atmosphere through the incineration process. Values for FVOLAT (Oz84) are listed below:

Radionuclide	FVOLAT
Hydrogen-3	0.90
Carbon-14	0.75
Technetium-99	0.01
Ruthenium-106	0.01
Iodine-129	0.01

For other isotopes, the value of FVOLAT is 0.005 (sanitary landfill incinerator) or 0.0025 (pathological incinerator)(Oz84).

Incinerator control efficiencies for radionuclides present in surrogate BRC waste streams are defined by the expression:

$$\text{Control Efficiency (I)} = 1 - \text{FVOLAT (I)}$$

Those radionuclides that escape through the stack are subjected to atmospheric dispersion and may reach the local population. The rate of incineration used in the modeling is continuous over the 20 years of operation of the SF.

After incineration, the ash and rubble are landfilled, using methods similar to those described in Section 4.3.2. The major differences from a computer simulation viewpoint are the porosity (0.35) and density (0.89 g/cm^3) for ash after being placed in the trench.

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APPENDIX D

HYDROGEOLOGIC/CLIMATIC DESCRIPTIONS FOR SPECIFIC COMMERCIAL DISPOSAL FACILITIES.

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APPENDIX D: HYDROGEOLOGIC/CLIMATIC DESCRIPTION FOR SPECIFIC COMMERCIAL DISPOSAL FACILITIES

The descriptions of general geology, hydrogeology, surface water hydrology, climatic settings, and potential hydrogeologic pathways for commercial disposal facilities located at Barnwell, Beatty, and West Valley are presented in a general, qualitative manner. These sites are generally representative of conditions in those regions, and much is already known concerning site-specific conditions.

D.1. Barnwell

The Barnwell Low-Level Radioactive Waste Disposal facility is located in Barnwell County, South Carolina, approximately 60 km southeast of Augusta, Georgia, and along the eastern boundary of the Barnwell Nuclear Fuel Plant (FB78).

D.1.1 General Geology

The Barnwell facility is located in the Southern Atlantic Coastal Plain Province, approximately 65 km southeast of the Fall Line that separates the Piedmont Plateau of the southern Appalachians from the coastal plain sediments. The Barnwell site lies on the Brandywine Pleistocene Coastal Terrace which has gently rolling topography cut in Tertiary sedimentary rocks. Figure D-1 is a generalized northwest-southeast cross-section that shows how the Coastal Plain deposits lap onto the crystalline rocks of the Appalachian Piedmont. Cenozoic and Mesozoic sedimentary rocks thin to the northwest and thicken to the east and southeast toward the Atlantic Ocean (Fe77).

Of the geologic formations that occur beneath the Barnwell plant area, the Triassic red bed sequence in the area was deposited in a fault basin like those created in the Northern Coastal Plain in the mid-Atlantic New England area (Fe77, FB78).

The rocks younger than the Precambrian-Paleozoic sequence are a varied sequence of clastic sedimentary rocks displaying a variety of size and sorting ranges. Most units contain some amount of clay and silt and the Eocene rocks have some thin limestone beds present. The "cleanest" unit is the Tuscaloosa Formation, a quartzose, arkosic sand unit with intervening beds of kaolinitic clay. Recent to Pliocene deposits consist of alluvium and gravelly terrace deposits in stream valleys.

D.1.2 Hydrogeology

Ground water is found in varying amounts and qualities in almost all sedimentary formations in the area. The water table in the Barnwell area occurs in the Hawthorne Formation, although this unit is a poor aquifer and is not generally suitable for even domestic use except where sand or

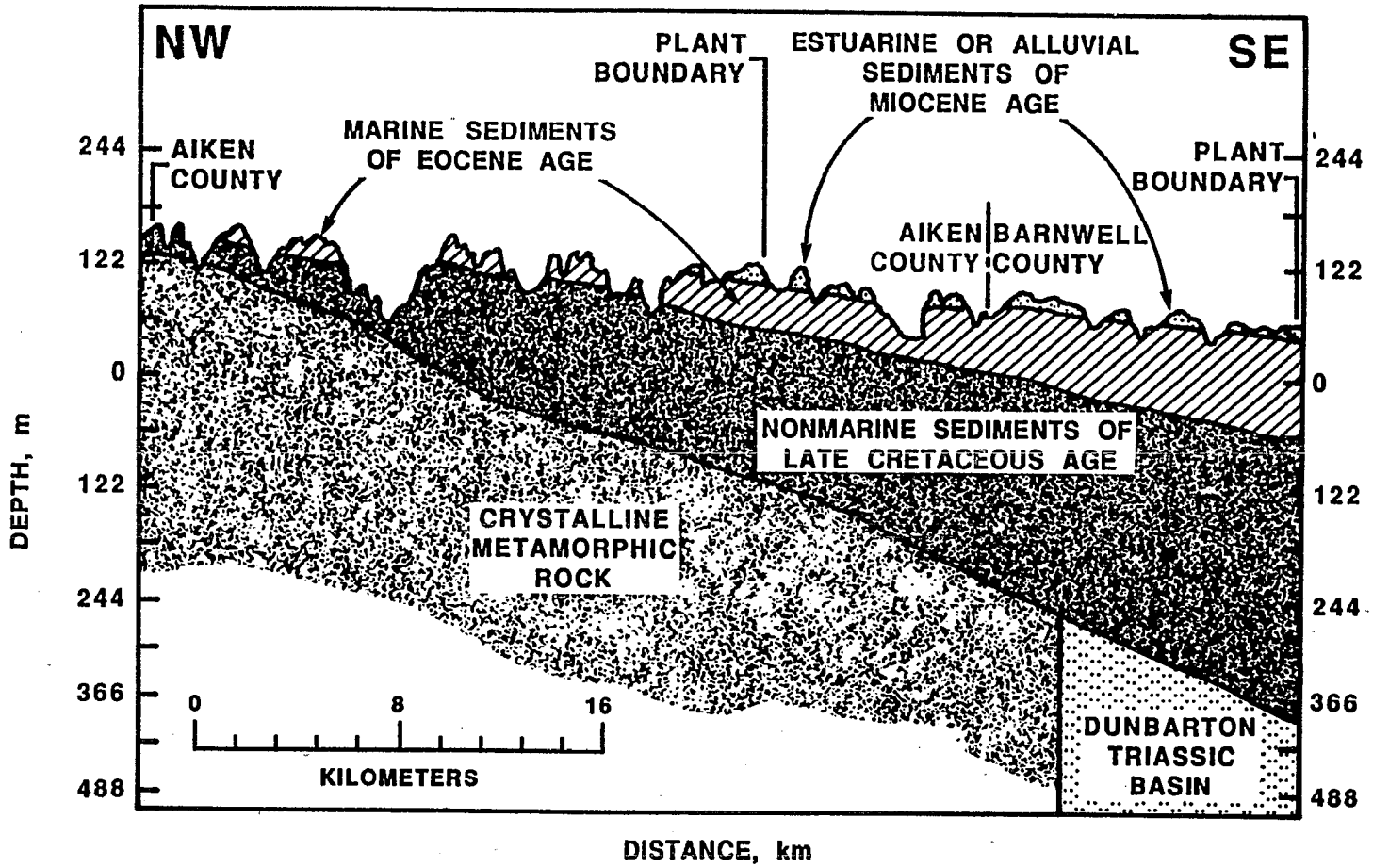

Figure D-1. Profile of Geologic Formations Beneath the Savannah River Plant (Fe77)

gravel channels occur. The water table fluctuates seasonally, but is approximately 23 m below the surface at Barnwell. The ground-water flow pattern is generally north to south across the facility, but rises closer to the surface in the central area, where a large number of burial trenches have been constructed and filled. It is common to have enhanced infiltration in disturbed areas like landfills, and this pattern appears to be true at Barnwell because this same area does not have a corresponding topographic high. The other sedimentary aquifers beneath the facility are the Barnwell Formation, the McBean-Congaree aquifer, and the Tuscaloosa Formation. The Tuscaloosa Formation is the principal regional aquifer in the area and lies approximately 100 m below the Barnwell site. The artesian system may yield as much as 7,570 L/min to municipal and industrial wells (FB78). The McBean and Congaree Formation consists of an alternating sequence of sands, marls, clays, and limestone. The sand and limestone beds are water bearing and supply water for industrial and municipal supplies in the area around Barnwell. These two formations discharge via springs and seep directly to surface water drainages such as Lower Three Runs Creek and the Savannah River to the south. The Eocene Barnwell Formation is a clayey sand to sandy clay and is not a significant source of water supply but is used for limited, rural, domestic supplies.

The character and thickness of the unsaturated zone is of particular interest for a LLW facility because this is the medium through which leachate from the trenches will flow. At the Barnwell site, the unsaturated zone has a thickness of approximately 9 to 15 m. Given the 6- to 7-m depth of the trenches, this thickness is great enough to prevent trench flooding except in an extremely wet year. The material in the unsaturated zone is composed of sand and clay from the Hawthorn Formation. The average permeability of these sediments is 8E-05 m/min (NRC82), which corresponds to a silty sand. The grain size distribution supports the permeability figures because the sediments are composed of 75 percent sand and 25 percent silt and clay (principally kaolinite). The distribution coefficients (Kd) for most of the radionuclides at the Barnwell site are generally lower than those of the montmorillonite and zeolite-rich western soil (Ne83). An exception, however, is uranium, which may have greater retention because of less bicarbonate concentration in the ground water (Wo83).

D.1.3 Surface Water Hydrology

Two major river systems are in the Barnwell area - the Savannah River to the south and the Salkehatchie River to the northeast. The Barnwell facility is located on the edge of the Lower Three Runs Creek watershed, a southerly flowing tributary of the Savannah River, and only 1 to 2 percent of the area is within the Salkehatchie watershed. There are numerous swamps on the Brandywine Terrace, many of them in the Carolina bays geomorphic features. The bays are local, circular depressions with closed drainage systems that often support swamps or ponds (NRC82). The closest perennial stream is Mary's Creek, a tributary to Lower Three Runs Creek,

about 1 km south of the facility (McD84). West of the Barnwell facility on the Savannah River Plant site, Lower Three Runs Creek is dammed to form a large lake, Par Pond, the largest impoundment in the area, covering over 110 km^2. Flow into Lower Three Runs Creek is controlled by the discharge system at Par Pond. Surface drainage from the Barnwell site would not impact Par Pond, but would most likely flow south to Mary's Creek, and from there to Lower Three Runs Creek and the Savannah River. A summary of USGS data that characterize flow rates and drainage areas for Lower Three Runs Creek and the Savannah River is presented in Table D-1.

Generally, the sandy soil and surficial material in the Barnwell Formation aids infiltration and controls surface runoff except during extreme precipitation events such as hurricanes and thunderstorms. Most precipitation infiltrates through the unsaturated zone to the water table and then moves laterally to the surface discharge system. Some water-holding soils occur in the Carolina bays area, but these soils account for less than 10 percent of the area at Barnwell.

D.1.4 Climatic Setting

The climate at Barnwell can be characterized as a warm, humid type with all seasons represented. The main chain of the Appalachians protects the area from the more severe winters of the Tennessee Valley, but the humid, semitropical summers of the southeast are not moderated by any local topography or geographic feature. The most extensive data on climate of the area have been collected at the nearby SRP and at a Class A weather station at Augusta, Georgia. The following climatic summary is a synthesis of information derived from NRC82, Fe77, FB78, CN80, LE71, and NOAA80.

Climatic features that directly affect the transport of radionuclides through air, ground water, and surface water pathways include wind speed and direction, atmospheric stability, mixing depth, temperature, humidity, rainfall, and solar radiation (sunshine). The general climate at Barnwell is, except for the hot and humid summers, moderate. Winters are mild with little snow and spring and fall have temperate weather. Severe weather is not unknown, with tornados, hurricanes, and hailstorms occurring with regularity. Precipitation is distributed fairly evenly throughout the year and the average annual humidity is 66 percent.

The average daily temperature ranges from 1 to 33°C, with extremes of -15°C to +41°C. The average relative humidity ranges from 45 to 92 percent. The average annual rainfall at Barnwell is about 1.2 m/yr. The propensity for flooding is low at Barnwell, but surface erosion could result from extremely heavy precipitation events.

The average annual atmospheric mixing depth is 938 m (NRC82). The prevailing wind at the SRP is from the southwest with a secondary direction from the northeast (McD84). Data at SRP indicate that wind speeds less than 2 m/sec occur 15 percent of the time.

Table D-1. Summary of discharge data for Lower Three Runs Creek and the Savannah River (USGS81)

	Lower Three Runs Creek below Par Pond at Savannah River	Lower Three Runs Creek near Snelling, South Carolina	Savannah River at Augusta, Georgia	Savannah River at Clyo, Georgia
Location				
Latitude	33°,14',07"	33°, 0',35"		32°,31',33"
Longitude	81°,31',60"	81°,28',50"		81°,15',45"
Drainage Basin Area, km^2	90.4	153.6	19,446	25,511
Average Stream Flow, m^3/sec	0.93	2.71	292	346
Maximum Stream Flow, m^3/sec	4.31	21	9,930	7,660
Minimum Stream Flow, m^3/sec	0.05	0.45	18	55

D.1.5 Hydrogeologic Pathways

The host soils at the Barnwell LLW disposal site are moderately permeable and well drained, with a low natural attenuation as described earlier. Although measurable water is found in the trench only after prolonged storm events, samples of this trench water indicate that oxidizing conditions are present and that microbiological action is taking place to reduce levels of organic components in the waste. The primary potential pathway from the waste at Barnwell is leaching and drainage from the trenches, migration to the water table in the Hawthorn Formation, and flow down-gradient to Mary's Spring and Mary's Creek. Leached radioactive material would then be available for human or animal ingestion or plant uptake through contaminated surface or ground water.

The ground-water velocity (approximately 27.8 m/yr) would give a travel time of approximately 36 yr for those highly mobile radionuclides to migrate from the trenches to Mary's Creek after the material had infiltrated to the aquifer. Assuming placement of a well midway between the trenches and Mary's Creek, contaminants could reach the well within approximately 18 yr. Vertical flow downward to the Barnwell Formation and the McBean-Congaree aquifer would also be possible because the hydrologic head in this area drives water from the Hawthorn to the McBean-Congaree. The pathways to humans through this system would be longer in space and in time. The most likely avenue would be through surface water discharge, although the possibility exists of flow to an irrigation well drilled to the McBean-Congaree.

The possibility that either of these scenarios will occur under current demographic distribution is remote because Mary's Creek and Lower Three Runs Creek are, in this area, on SRP land. As long as this land remains part of this facility, dilutions and decay will ensure that LLW leachate will not significantly impact the local population. However, health effects to the population of communities that use the Savannah River downstream of the Barnwell disposal facility for human and animal consumption and/or agricultural irrigation are possible if contaminants escape the disposal facility and migrate into the Savannah River.

D.2 Beatty

The Beatty Low-Level Radioactive Waste Disposal facility is a 32.4-ha tract located about 18.4 km southeast of Beatty, Nevada, and midway between Beatty and Lathrop Wells, Nevada.

D.2.1 General Geology

The Beatty site is on the upper northeast border of the northwest part of the Amargosa Desert. The Amargosa Desert is a large, northwest trending valley that is both a topographic and hydrographic province. The LLW disposal area of the valley is bounded on the northeast by Bare Mountain and on the southwest by the Grapevine and Funeral Mountains. At

the head of the valley to the northwest, the Bullfrog Hills separate the Amargosa Desert from Sarcobatus Flats.

The Amargosa River is the principal drainage and enters the valley from Oasis Valley and Amargosa Narrows. The town of Beatty is in Oasis Valley. The Amargosa River is an intermittent stream that flows southeast along the desert valley floor immediately west of the site. Water rarely flows past the site except in floods, and surface-water flow is seldom seen less than 16 km from the facility (Cle62). The main tributaries of the Amargosa River are Beatty Wash, Forty Mile Canyon Wash, and Carson Slough.

The topography of the disposal site is nearly flat or gently sloping (10 m vertical/ 1.6 km horizontal) toward the Amargosa River. The southwest side of Bare Mountain, northeast of the site, is a pediplain with armored gravels protecting the erosional surface. Caliche zones are not present in the soil horizon near Beatty, thus allowing direct infiltration to take place unimpeded by a dense soil horizon. Numerous intermittent valleys are present along the slope, but only receive moisture during snowmelt and convective storms. The Beatty site lies between the Amargosa River channel and a secondary drainage channel, along the north side of U.S. Highway 95, which carries the drainage from these intermittent streams to the Amargosa River.

The Amargosa Desert, like most bolson valleys in the Great Basin, consists of a fault-controlled Pleistocene and Tertiary valley fill overlying volcanic and sedimentary basement rocks. Bare Mountain, north and east of the disposal area, is composed mainly of Paleozoic carbonates and metasediments and various Tertiary volcanics, and rises over 915 m up from the valley floor to elevations greater than 1,800 m.

The valley floor contains a variety of alluvial materials ranging from silt through gravels. The soil zone is usually capped by a thin, low-density soil zone comprised of a large number of air vesicles between the soil particles, with lag sands and gravels armoring the surface. Soil moisture is estimated at 6 to 10 percent. Few deep wells are located near the facility, but analysis of the one deep well at the site, coupled with regional and geophysical analyses, indicates that between 150 and 180 m of valley fill may be present on a largely irregular bedrock surface. Many Great Basin valleys of this type are fault controlled, and a large fault probably lies along the northeast side of the valley along the Bare Mountain front.

The valley fill material consists of pebbles, cobbles, and boulders representing the full range of bedrock units, namely, sandstone, siltstone, dolomite, limestone, shale, phyllite, schist, quartzite, and marble. The bedrock beneath the valley appears to be a quartzite similar to that found on Bare Mountain, as evidenced by samples collected near the base of the one deep well drilled on the property (Cle62).

D.2.2 Hydrogeology

The hydrogeologic area of interest with respect to the low-level disposal site at Beatty is the 500 feet of valley fill material on which the site is located. Material filling Great Basin bolson valleys is largely heterogeneous, having been derived from mudflows, floods, and other desert sediment transport mechanisms. Permeabilities in such material are usually estimated by statistical analyses which predict zones of percent permeable material. Based on such work to the east in the Ash Meadows flow system, this portion of the Amargosa Desert valley fill can be expected to have an estimated average transmissivity of approximately 1.9E+05 L/da/m. Typically, the water table is 91 to 98 m below land surface, with unsaturated zone moisture contents of 15 to 20 percent of saturation. A depth to water of 87 m was reported in a well at the Beatty facility (Wa63). The casing in the well was perforated from 138 to 150 m and 156 to 175 m below the surface (Cle62). If the casing was properly installed and if the water levels are correct, then an artesian head exists in the deep sediments of the valley fill. The major clay section of the well, from 111 to 99 m, could serve in part as a confining bed for this artesian water. Clebsch refers to an upper and lower aquifer and notes that the water levels in these zones are 6 m apart, with the higher potential measured in the upper section (Cle62). This would indicate a potential for downward flow in this portion of the alluvial fill. Both "aquifers" are beneath the clay layer noted in the well, which would insulate the water-bearing sections from direct access from recharge. However, because no other detailed well logs are available nearby, the extent of the clay layer is unknown. Thus, the aquifer zones may only be semiconfined locally, a condition commonly found in Great Basin valley fill material.

The general direction of ground-water flow in the Amargosa Desert valley fill is from northwest to southeast. The Oasis Valley area and the Spring Mountains are the major recharge sources to both the valley fill and bedrock systems in the area. A large recharge source exists in the Amargosa River north of Beatty, and the flow is channeled through Amargosa narrows into the valley fill of the Amargosa Desert (Wh79).

Some desert valleys in the Great Basin contain permeable sedimentary rocks beneath the valleys. The Amargosa Desert has been shown to be the regional discharge area for a large regional flow system. At present, no deep bedrock wells are available to assess the bedrock beneath the Beatty site to determine whether formations there are part of a regional hydrologic system (Cle62, Ni82). However, a bedrock flow system appears to be beneath the Amargosa Desert and it is part of the Oasis Valley-Forty Mile Canyon ground-water basin that receives water from the Pahute Mesa-Timber Mountain area (Ba72). Based on deep drilling data at the NTS, bedrock units carrying the water could be either Paleozoic carbonates, such as are found in other regional aquifers in Nevada, or fractured Tertiary volcanics. The Amargosa Desert is a regional discharge area much the same as the Ash Meadows area to the southeast (Ba72). However, no

major springs in the area have been specifically tied to the Pahute Mesa flow system. The total discharge from the Amargosa Desert region is estimated at 3E+07 m^3 per year, the majority of which is discharge through evapotranspiration (Wa63).

Based on well tests (Cle62) and a regional gradient of 5.7E-03, it is estimated that a ground-water velocity in the valley fill is 1.22 m/da, a rather high value. The tests also indicate transmissivity ranges from 1,500 to 19,000 L/da/m. The closest producing wells are about 25 km east-southeast of the disposal facility, and they produce 1,100 to 3,800 L/min from the valley fill.

D.2.3 Surface Water Hydrology

The Amargosa River does not flow perennially in the area of the disposal site, with the closest gauging station at Beatty, 0.16 km below Amargosa Narrows. Table D-2 shows the USGS statistics for the Amargosa River near Beatty, Nevada. Stream flow, when recorded, was usually the result of local, high-intensity storms. These storms can provide a large discharge as indicated by the maximum recorded discharge of 120 m^3/sec.

Other surface streams in the area only flow during storms and snowmelt. Most surface flow readily infiltrates the ground, and continuous flow is not usually seen on the Amargosa River except immediately adjacent to some perennial springs that are a long distance downstream of Beatty.

D.2.4 Climatic Setting

The climate at Beatty is a warm-to-hot, arid desert climate characterized by low humidity and large seasonal temperature fluctuations. The area generally has low precipitation and high evaporation. There is a seasonal precipitation variation, with the winter months being the wettest. The higher elevations near the Amargosa Desert, such as Spring Mountains, have much more precipitation than the lowlands. The average annual precipitation is estimated at 5 to 13 cm/yr (Cle62). During the year, the winter precipitation brings the most moisture to the area, with low pressure storm impulses from the Pacific Ocean the most common event. Every few years, major storms from the Gulf of Mexico account for especially wet weather in winter and the early spring months. These storms serve to raise the total annual precipitation significantly in those years. Summer precipitation is not uniformly distributed with respect to area. The localized convective storms that develop during these months can cause high amounts of precipitation in isolated areas and leave other locations dry and unaffected. Most of this moisture originates from the south and southeast (Wa63).

The average monthly temperatures in the Beatty/Lathrop Wells area (Amargosa Desert) range between 3°C and 29°C. The recorded extremes of temperatures are -17°C to 46°C in the Beatty/Lathrop Wells area.

Table D-2. Summary of discharge data for Amargosa River near Beatty, Nevada (USGS68)

Location:	Latitude, 36° 52' 55" Longitude, 116° 45' 05"
Drainage Base Area:	1217 km^2
Average Stream Flow:	None most of the year
Maximum Stream Flow:	120 m^3/sec
Minimum Stream Flow:	0

According to USGS records, rainfall at Beatty averages 0.117 m/yr. The combination of high temperatures and low humidity in the Amargosa Desert means that the Beatty area has a high evaporation rate. Clebsch reports a conservative estimate of 25 m/yr of evaporation. The highest evaporation is in the summer months and the lowest in the winter months (Cle62).

The high evaporation and low rainfall, coupled with the moderate permeability of the valley fill, indicate that flooding of burial trenches is not a problem at the Beatty facility. The main climate-related problem would be erosion from high intensity storms.

D.2.5 Hydrogeologic Pathways

The remote location suggests very few pathways to humans exist at the Beatty facility in the short-term period. If water were to leak from the trenches, the high evaporation rate could retard the water and radionuclides from migrating downward. During the course of migration, the radionuclides would also be adsorbed in the clay in the valley fill sediments. Once in the flow system, the velocity is fairly rapid in the sediments at Beatty, although the closest water use is miles away and local populations are small. However, because of the scarcity of water available for human and animal consumption and agricultural irrigation in the southwest region, a large percentage of the potentially contaminated aquifer is thought to be utilized by downstream populations.

The other possibilities of release from Beatty would be through wind erosion on disturbed trenchland and ground disturbance due to earthquake activity.

D.3 West Valley

The West Valley Low-Level Radioactive Waste Disposal facility is located at the West Valley Nuclear Service Center in western New York, 48 km south of Buffalo in Cattaraugus County.

D.3.1 General Geology

The West Valley site location is on the gently sloping flank of a bedrock ridge. Local elevation is about 419 m above sea level, and local drainage is north toward Buttermilk Creek, a tributary of Cattaraugus Creek, which flows into Lake Erie.

The West Valley LLW waste disposal site is located in the Allegheny Plateau physiographic province. A pre-existing erosional surface was moderately to deeply dissected during the Pleistocene era, and a highly variable thickness of till, outwash, and glacial lake deposits up to 180 m thick was deposited on the area. The waste disposal site is located in a thick sequence of till gravel and glacial lake deposits estimated at from 90 to 150 m thick (Pr77, FB78). The area is underlain

with a thick, flat-lying sequence of shales, siltstones, limestones, and sandstones. Except for the glacial deposits, rocks younger than Pennsylvanian are usually not reported in the area. The Paleozoic sedimentary sequence may be as much as 2,745 m thick and rests on crystalline Precambrian sequence at depths estimated in the 2,450-m range. The only bedrocks exposed in the West Valley area are the shales and siltstones exposed on Buttermilk Creek Valley. These rocks are in the Machias Formation of the Upper Devonian Canadaway Group (EPA77).

Little is known of the potential bedrock aquifers in the disposal site area. Associated bedrock units are poorly productive, and the low quantities of water observed are from brackish to brine in quality. Carbonates in the lower Devonian have produced large amounts of water in wells several hundred meters deep, but the quality of this water is variable, being fresher near recharge areas well to the west of the West Valley area (FB78).

The major stratigraphic materials of interest at West Valley are the complex series of Pleistocene glacial deposits that overlay the bedrock in the area. The dominant glacial topographic feature is Buttermilk Creek Valley, which contains from 2 to 170 m of glacial deposits. Originally, Buttermilk Creek was a deep bedrock valley, but with successive Pleistocene glaciations, the valley has been filled with a heterogeneous series of glaciofluvial material. The principal deposits found in the area are (EPA77):

- Till--a very fine-grained, compact, dark blue-gray, heterogeneous mixture of clay and silt, containing minor amounts of sand and stones;

- Coarse granular deposits--a mixture of sand and pebbles up to several inches in diameter, which also contains minor amounts of silt and clay;

- Outwash--coarse, granular deposits of stratified, well-sorted sands and gravels. Some deposits appear to be thin-bedded units of both sand and gravel; and

- Lake deposits--fine-grained, thin-bedded sands, silts, and clays with minor amounts of fine pebble.

The West Valley LLW disposal facility was constructed in the glacial till (Figure D-2). The till has been leached of calcium carbonate and is oxidized and weathered up to 5 m below the surface, which is littered with pebbles, cobbles, and boulders. Unweathered till extends down to about 27 m below the surface and lies on an unknown thickness of glacial lake deposits. On average, the till contains 50 percent clay, 27 percent silt, 10 percent sand, and 13 percent gravel. Coarse, granular surficial material found above the first "tight" till is thin (0-7.6 m thick) and discontinuous, often isolated and perched by local stream valleys.

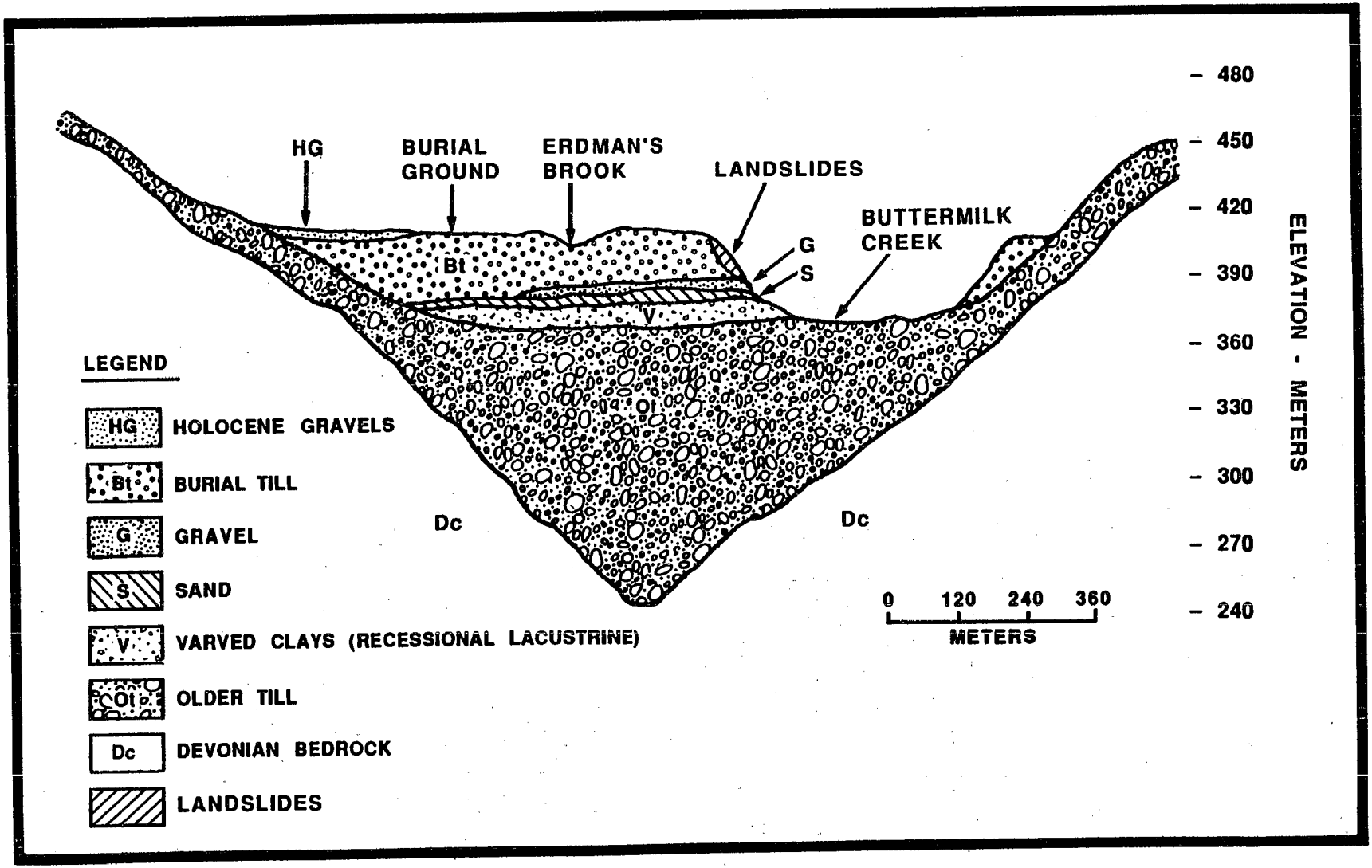

Figure D-2. Cross-Section of Glacial Deposits at the West Valley Disposal Site (FB78)

D.3.2 Hydrogeology

The principal ground-water occurrences and movement in the West Valley area are in the glacial deposits. While some bedrock units can and are to be utilized as aquifers, use of these units as water sources depends on their proximity to the surface and the lack of sufficient surface and shallow ground-water supplies. The presence of relatively impermeable and low yielding Upper Devonian siltstones and shales immediately below the LLW disposal facility makes bedrock interaction with leachate from the facility unlikely. There is presently no evidence of a hydrologic connection between the till and bedrock, except possibly in the weathered zone at the top of the bedrock as described below.

Several water-bearing zones have been noted in the glacial deposits at West Valley: principally an artesian unit 12 to 15 m below land surface; an outwash zone from 30 to 38 m; and a poorly defined, apparently permeable zone at depths greater than 60 m, which may possibly be used for individual supply (FB78).

In addition to the confined water-bearing zones, an unconfined water table unit occurs in the coarse, granular, near-surface sediments. As with most unsaturated units, the water-table surface mimics the topography. This unit is the one most directly connected to evapotranspiration and direct discharge to streams. The transmissibility of this material is very low and is estimated to be approximately 1,300L/da/m with a 25 percent porosity (EPA77).

The shallow artesian unit is found between Buttermilk Creek and Frank's Creek. This condition, however, does not occur in the disposal site area. The unit is confined by the upper till, and the depth to water varies between 2 and 6 m below land surface (EPA77). This unit is of minor importance and is not found directly beneath the LLW disposal area.

The third water-bearing zone is a confined unit that is present at the base of the till and comprises the base of the gravelly till and the top of the fractured and weathered bedrock. Not much is known about this unit, but it is believed to be sufficiently permeable to support low-yield (4 to 40 L/min) wells in the area (FB78).

The silty, clayey till in which the LLW trenches are emplaced is not a water-yielding zone. It is, however, saturated and allows only extremely slow water movement. Slug tests conducted by the USGS in observation wells were analyzed by a variety of methods to arrive at horizontal till permeabilities of 2E-08 to 6E-08 cm/sec. Distortion and disturbance in the stratification due to loading and clay expansion do not appear to seriously affect the permeability of the till.

The oxidized till has a higher overall permeability (up to 10 times greater) than the unweathered till. The greater permeability may be in part due to the presence of numerous fractures in the oxidized till to a depth of about 5 m. All the till material is anisotropic on a gross scale, with the unweathered till having a horizontal permeability on the order of 100 times greater than the vertical permeability. Silty lenses may occur in the till, which, if they intersect trenches, may conduct water away from the trenches toward surface drainages. Coarse sand lenses are reported to exist on some trench walls but not in others (EPA77). Even so, the permeability is very low, with an estimated 160 to 200 yr being the ground-water travel time to the nearest surface drainage, Buttermilk Creek, where springs do exist along with the valley walls. Unweathered till without sand or gravel lenses is less strongly anisotropic than the weathered till, with very low vertical and horizontal permeabilities. All hydraulic gradients observed at the burial site indicate that the potential for migration from the trenches under undisturbed conditions is down and away from the trenches toward surface water drainages or the deeper glacial deposits (Pr77).

The main source of past leakage and contamination at West Valley was cap failure and the consequent filling and overflowing of some of the trenches. Reworking of the covers and pumping operations resulted in spillage and radionuclide redistribution, causing contamination of the soil and weathered till zone over most of the site (Cla81).

D.3.3 Surface Water Hydrology

The West Valley area falls within the Cattaraugus Creek drainage basin, and the closest tributary to the site is Frank's Creek, which is a tributary to Buttermilk Creek. Several small, marshy areas and minor drainages can be found on the northwest side of the facility, but these will probably not be a factor in any future excursions. Buttermilk Creek is a tributary to Cattaraugus Creek, which flows westerly about 65 km to Lake Erie. The main drainages flow over glacial deposits that fill deeply incised bedrock valleys. The data are summarized in Table D-3.

Seasonal variations of flow at the Gowanda and Buttermilk Creek gauging stations are quite similar, with high flows occurring in fall and spring and low flows in summer. Flow is somewhat controlled by surface impoundments, and direct ground-water discharge from deep water-bearing zones forms a small percentage of surface-water flow. Precipitation, snowmelt, and soil saturation along the stream beds are the main sources of surface water in the area.

Streams in the northeast are subjected to periodic flooding, an important consideration at West Valley because of its past history of contamination leakage. A review of U.S. Army Corps of Engineers records indicates that even a 100-yr flood on Buttermilk Creek would

Table D-3. Summary of USGS discharge data for Buttermilk Creek and Cattaraugus Creek (USGS81)

	Cattaraugus Creek at Gowanda, New York	Buttermilk Creek near Springville, New York
Location: Latitude	42° 27' 50"	42° 28' 21"
Longitude	78° 56' 10"	78° 39' 54"
Drainage Basin Area, km^2	1118	76
Average Stream Flow m^3/sec	20.9	1.32
Maximum Stream Flow m^3/sec	98	111
Minimum Stream Flow m^3/sec	0.17	0.06

not affect the West Valley disposal site (FB78). The disposal site area is well above local flood plains. Flash flood surface runoff is not as common in the east as in the west where different soil types and infiltration rates prevail. Nonetheless, because high intensity thunderstorms can occur in the Northeast, care should be taken in grading the disposal facility to prevent localization of surface drainage in the area around the trenches. Dikes around the trenches should also be used to control trench overflow in the event of future cap failure due to a combination of seal cracking and heavy rain. Two water supply earthen dams are located at the southeastern part of the Nuclear Service Center. Their overflow is below the highest elevation at the facility. Therefore, in the event of dam failure during an extreme precipitation event, flooding would be contained in the Buttermilk Creek Valley and pose no danger to the disposal facility.

D.3.4 Climatic Setting

The prevailing climate at West Valley is a cool humid type with approximately 1.2 m of annual precipitation, much of which occurs as snow. The nearby Great Lakes Region and local topography have an influence on weather in the area, which is subject to the "lake effect," which can produce up to 380 cm of snowfall per year in western New York. The average annual temperature ranges from -18°C to 32°C, with a mean of 7.2°C. Winds occur with an average speed of about 19 km/h from the southwest (NOAA79, Ri78).

The area has high rainfall and a high percentage of overcast and partly cloudy weather which minimizes direct evaporation (FB78). Pan evaporation in this area of the United States is 89 to 102 cm/yr. Vegetation flourishes in this climate, and plant transpiration is expected to be high most of the year, but may exceed soil moisture availability during hot, dry periods. The high annual rainfall usually ensures a low incidence of soil moisture deficiency.

D.3.5 Hydrogeologic Pathways

The potential major hydrogeological pathway that could occur at the West Valley site is the overflow of trench water, with subsequent contamination of the ground surface. The radionuclides that contaminate the ground surface could potentially be transported into the local and/or downstream water supplies if communities or individuals draw their water from surface streams. The abovementioned trench-water overflow is potentially caused by the extremely low hydraulic conductivity of the host formation and rainwater infiltration allowed by excessive trench cap failure.

Sand lenses have been observed underneath some disposal trenches; however, there is no evidence that these sand lenses are connected to any ground water. Therefore, the potential radionuclide transport pathway from the trench to the biosphere through this pathway was not considered for this analysis.

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APPENDIX E

A DESCRIPTION OF THE RADRISK AND CAIRD COMPUTER CODES USED BY EPA TO ASSESS DOSES AND RISKS FROM RADIATION EXPOSURE

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E.1 Introduction

This appendix provides a brief overview of the RADRISK (Du80) and CAIRD (Co78) computer codes used by the EPA to assess the health risk from radiation exposures. It describes the mechanics of the life table implementation of the risk estimates derived in Chapter 7.

E.2 Life Table Analysis to Estimate the Risk of Excess Cancer

Radiation effects can be classified as stochastic or nonstochastic (NAS80, ICRP77). For stochastic effects, the probability of occurrence of the effect, as opposed to the severity, is a function of dose; induction of cancer, for example, is considered a stochastic effect. Nonstochastic effects are those health effects for which the severity of the effect is a function of dose; examples of nonstochastic effects include cell killing, suppression of cell division, cataracts, and nonmalignant skin damage.

At the low levels of radiation exposure attributed to radionuclides in the environment, the principal health detriments are the induction of cancers (solid tumors and leukemia) and the expression, in later generations, of genetic effects. In order to estimate these effects, instantaneous dose rates for each organ at specified times are calculated for use in a subroutine adaptation of CAIRD contained in the RADRISK code. This subroutine uses annual doses derived from these dose rates to estimate the number of incremental fatalities in the cohort due to radiation-induced cancer in the reference organ. The calculation of incremental fatalities is based on estimated annual incremental risks, computed from annual doses to the organ, together with radiation risk factors such as those given in the 1980 NAS report, BEIR-3 (NAS80). Derivation of the risk factors in current use is discussed in Chapter 7.

An important feature of this methodology is the use of actuarial life tables to account for the time dependence of the radiation insult and to allow for competing risks of death in the estimation of risk due to radiation exposure. A life table consists of data describing age-specific mortality rates from all causes of death for a given population. This information is derived from data obtained on actual mortality rates in a real population; mortality data for the U.S. population during the years 1969-1971 are used throughout this study (HEW75).

The use of life tables in studies of risk due to low-level radiation exposure is important because of the time delay inherent in radiation risk. After a radiation dose is received, there is a minimum induction

period of several years (latency period) before a cancer is clinically observed. Following the latency period, the probability of occurrence of a cancer during a given year is assumed to be constant for a specified period, called a plateau period. The length of both the latency and plateau periods depends upon the type of cancer.

During or after radiation exposure, a potential cancer victim may experience years of life in which he/she is continually exposed to risk of death from causes other than incremental risk from radiation exposure. Hence, some individuals in the population will die from competing causes of death, and are not victims of radiation-induced cancer.

Each member of the hypothetical cohort is assumed to be exposed to a specified activity of a given radionuclide. In this analysis, each member of the cohort annually inhales or ingests 1 pCi of the radionuclide, or is exposed to a constant external concentration of 1 pCi/cm^3 in air or 1 pCi/cm^2 on ground surfaces. Since the models used in RADRISK are linear, these results may be scaled to evaluate other exposure conditions. The cohort consists of an initial population of 100,000 persons, all of whom are simultaneously liveborn. In the scenario employed here, the radiation exposure is assumed to begin at birth and continue throughout the entire lifetime of each individual.

No member of the cohort lives more than 110 yr. The span from 0 to 110 yr is divided into 9 age intervals, and dose rates to specified organs at the midpoints of the age intervals are used as estimates of the annual dose during the age interval. For a given organ, the incremental probability of death due to radiation-induced cancer is estimated for each year using radiation risk factors and the calculated doses during that year and relevant preceding years. The incremental probabilities of death are used in conjunction with the actuarial life tables to estimate the incremental number of radiation-induced deaths each year.

The estimation of the number of premature deaths proceeds in the following manner. At the beginning of each year, m, there is a probability p^N of dying during that year from nonradiological causes, as calculated from the life table data, and an estimated incremental probability p^R of dying during that year due to radiation-induced cancer of the given organ. In general, for the m-th year, the calculations are:

$M(m)$ = total number of deaths in cohort during year m,

$= [p^N(m) + p^R(m)] \times N(m)$

$Q(m)$ = incremental number of deaths during year m due to radiation-induced cancer of a given organ,

$= p^R(m) \times N(m)$

$N(m+1)$ = number of survivors at the beginning of year m + 1

$= N(m) - M(m)$

p^R is assumed to be small relative to p^N, an assumption that is reasonable only for low-level exposures such as those considered here (Bu81). The total number of incremental deaths for the cohort is then obtained by summing Q(m) over all organs for 110 yr.

In addition to providing an estimate of the incremental number of deaths, the life table methodology can be used to estimate the total number of years of life lost to those dying of radiation-induced cancer, the average number of years of life lost per incremental mortality, and the decrease in the population's life expectancy. The total number of years of life lost to those dying of radiation-induced cancer is computed as the difference between the total number of years of life lived by the cohort assuming no incremental radiation risk, and the total number of years of life lived by the same cohort assuming the incremental risk from radiation. The decrease in the population's life expectancy can be calculated as the total years of life lost divided by the original cohort size (N(1)=100,000).

Either absolute or relative risk factors can be used. Absolute risk factors, given in terms of deaths per unit dose, are based on the assumption that there is some absolute number of deaths in a population exposed at a given age per unit of dose. Relative risk factors, the percentage increase in the ambient cancer death rate per unit dose, are based on the assumption that the annual rate of radiation-induced excess cancer deaths, due to a specific type of cancer, is proportional to the ambient rate of occurrence of fatal cancers of that type. Either the absolute or the relative risk factor is assumed to apply uniformly during a plateau period, beginning at the end of the latent period.

The estimates of incremental deaths in the cohort from chronic exposure are identically those that are obtained if a corresponding stationary population (i.e., a population in which equal numbers of persons are born and die in each year) is subjected to an acute radiation dose of the same magnitude. For example, the total person-years lived by the 1970 life table cohort is approximately 7.07 million, the estimates of incremental mortality in the cohort from chronic irradiation also apply to a 1-year dose of the same magnitude to a population of this size, age distribution, and age-specific mortality rates. More precise life table estimates for a specific population can be obtained by altering the structure of the cohort to reflect the age distribution of a particular population at risk.

E.3 Risk Analysis Methodology

Risk estimates in current use at EPA are based on the 1980 BEIR-3 report of the National Academy of Sciences Advisory Committee on the

Biological Effects of Ionizing Radiation (NAS80). The form of these risk estimates is, to some extent, dictated by practical considerations, e.g., a desire to limit the number of cases that must be processed for each environmental analysis and a need to conform to limitations of the computer codes in use. For example, rather than analyze male and female populations separately, the risk estimates have been merged for use with the general population; rather than perform both an absolute and a relative risk calculation, average values have been used.

The derivation of the risk estimates from the BEIR-3 report is presented in Chapter 7. A brief outline of the general procedure is presented below. Tables referenced from Chapter V of NAS80 are designated by a V prefix.

(1) The total number of premature cancer fatalities from lifetime exposure to 1 rad per year of low-LET radiation is constrained to be equal to the relative risk value (403 per million person-rad) given in Table V-25 of the BEIR-3 report for the L-L and $\overline{L\text{-}L}$ models for leukemia and solid cancers, respectively (NAS80).

(2) For cancers other than leukemia and bone cancer, the age and sex-specific incidence estimates given in Table V-14 were multiplied by the mortality/incidence ratios of Table V-15 and processed through the life table code at constant, lifetime dose rates of 1 rad/yr. The resulting number of deaths are averaged, using the male/female birth ratio, and proportioned for deaths due to cancer in a specific organ as described in Chapter 7. These proportional risks are then used to allocate the organ risks among the 358.5 deaths per million person-rad remaining after the 44.5 leukemia and bone cancer fatalities (Table V-17) are subtracted from the 403 given in Table V-25.

(3) The RADRISK code calculates dose rates for high- and low-LET radiations independently. A quality factor of 20 has been applied to all alpha doses to obtain the organ dose equivalent rates in rem per year (ICRP77). For high-LET radiation risk estimates, the risk from alpha particles is considered to be eight times that for low-LET radiation to the same tissue except for bone cancer, for which the risk coefficient is 20 times the low-LET value. Additional discussion was included in Chapter 7.

A typical environmental analysis requires that a large number of radionuclides and multiple exposure models be considered. The RADRISK code has been used to obtain estimates of cancer risk for unit intakes of about 200 radionuclides and unit external exposures by approximately 500 radionuclides. The calculated dose rates and mortality coefficients described in the preceding sections are processed through the life table subroutine of the RADRISK code to obtain lifetime risk estimates. At the low levels of contamination normally encountered in the environment, the life table population is not appreciably perturbed by the excess radiation deaths calculated and, since both the dose and risk models are

linear, the unit exposure results may be scaled to reflect excess cancers due to the radionuclide concentrations predicted in the analysis of a specific source.

As noted in the discussion of the life table analysis, risk estimates for chronic irradiation of the cohort may also be applied to a stationary population having the same age-specific mortality rates as the 1970 U.S. population. That is, since the stationary population is formed by superposition of all age groups in the cohort, each age group corresponds to a segment of the stationary population with the total population equal to the sum of all the age groups. Therefore, the number of excess fatal cancers calculated for lifetime exposure of the cohort at a constant dose rate would be numerically equal to that calculated for the stationary population exposed to an annual dose of the same magnitude. Thus, the risk estimates may be reported as a lifetime risk (the cohort interpretation) or as the risk ensuing from an annual exposure to the stationary population. This equivalence is particularly useful in analyzing acute population exposures. For example, estimates for a stationary population exposed to annual doses which vary from year to year may be obtained by summing the results of a series of cohort calculations at various annual dose rates.

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APPENDIX F

MAXIMUM CPG DOSES FOR BRC WASTE DISPOSAL SCENARIOS

APPENDIX F: MAXIMUM CPG DOSES FOR BRC WASTE DISPOSAL SCENARIOS

This Appendix details, in the form of tables, the maximum annual dose to the CPG, the radionuclide providing the major dose, and the year in which the maximum CPG dose occurs for each of the ten major pathways (see Table 10-2) at each of the three hydrogeologic/climatic settings for the 15 BRC waste disposal scenarios. Section 4.4 describes the scenarios in detail.

The CPG doses are calculated over a 10,000-year time span, during which the maximum individual in any given year may be either an onsite worker, an onsite resident, or an offsite resident. As explained in Section 10.7.2, the onsite worker while employed at the BRC waste disposal facility is also considered a member of the general public. The onsite resident is a member of the general public living onsite and/or growing crops for human consumption. The offsite resident is a member of the general public who lives away from the BRC waste disposal site, but is subjected to the various pathways capable of transporting radionuclides to the human population. The pathways can be separated into onsite and offsite workers and residents as follows (see also Section 8.5.4):

Onsite Worker Pathways (Pre-closure)

(1) Direct gamma exposure
(2) Dust inhalation

Onsite Resident Pathways (Post-closure)

(1) Food grown onsite
(2) Biointrusion

Offsite Resident Pathways (Post-closure)

(1) Ground water to river
(2) Ground water to well
(3) Surface erosion
(4) Facility overflow or bathtub effect

Offsite Resident Pathways (Pre-closure)

(1) Spillage
(2) Atmospheric inhalation.

Three time periods are involved in the CPG analysis. In all cases it is assumed that the disposal site has a maximum 20-year inventory of BRC wastes, with radioactive decay taken into consideration. The 0 year represents the last year of pre-closure. In the 0 year, the CPG is to the onsite worker and some offsite residents.

The year 1 represents the first year of the post-closure phase and the CPG is to onsite residents. Finally, there are the variable years, beyond year 1 of the post-closure phase, in which the CPG is to offsite residents.

The ground-water to river, the bathtub effect, and spillage pathways are applicable only to the humid impermeable hydrogeologic/climatic setting. This is because this setting deals solely with surface water flow, whereas the other two settings use the ground-water migration pathways.

The erosion pathway for the arid permeable setting does not appear within the 10,000-year analysis performed.

For the scenarios involving urban demographic disposal settings, it was assumed that there would be no food grown onsite after site closure.

The four scenarios, Tables F-12 through F-15, were analyzed for reference purposes only. These four scenarios were not relevant to the analysis for regulatory considerations and were used for comparison purposes only.

Table F-1. Maximum annual CPG dose, dominant radionuclide, and year of occurrence for Scenario 1. Three-Unit Pressurized-Water Power Reactor Complex - Municipal Dump (PWR-MD)

Site pathways	Humid Impermeable Site			Humid Permeable Site			Arid Permeable Site		
	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year
GW to River	7.5E-07	I-129	3060	*			*		
GW to Well	3.1E-02	I-129	3060	1.5E-01	I-129	184	7.3E-04	I-129	556
Spillage	2.1E-03	Co-60	0	*			*		
Erosion	2.8E-06	Pu-239	3900	3.3E-03	Pu-239	3900	*		>13000
Bathtub	2.1E-04	I-129	100	*			*		
Food Onsite	3.2E-01	Cs-137	1	1.9E-01	Cs-137	1	2.7E-01	Cs-137	1
Biointrusion	1.1E+00	Cs-137	1	6.4E-01	Cs-137	1	9.0E-01	Cs-137	1
Direct Gamma	1.2E+01	Co-60	0	1.2E+01	Co-60	0	1.2E+01	Co-60	0
Dust Inhalation	3.2E-02	Am-241	0	3.2E-02	Am-241	0	3.2E-02	Am-241	0
Atmosphere	2.1E-06	Am-241	0	4.4E-06	Am-241	0	5.0E-06	Am-241	0

*Pathway not applicable.

Note: In all tables, GW means ground water.

Table F-2. Maximum annual CPG dose, dominant radionuclide, and year of occurrence for Scenario 2. Two-Unit Boiling-Water Power Reactor Complex - Municipal Dump (BWR-MD)

	Humid Impermeable Site			Humid Permeable Site			Arid Permeable Site		
Site pathways	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year
GW to River	1.7E-06	I-129	3060	*			*		
GW to Well	7.1E-02	I-129	3060	3.3E-01	I-129	184	1.7E-03	I-129	556
Spillage	3.5E-03	Cs-137	0	*			*		
Erosion	1.8E-06	I-129	3900	1.9E-03	I-129	3900	*		>13000
Bathtub	4.7E-04	I-129	100	*			*		
Food Onsite	8.1E-01	Cs-137	1	4.8E-01	Cs-137	1	6.7E-01	Cs-137	1
Biointrusion	2.7E+00	Cs-137	1	1.6E+00	Cs-137	1	2.2E+00	Cs-137	1
Direct Gamma	1.1E+01	Co-60	0	1.1E+01	Co-60	0	1.1E+01	Co-60	0
Dust Inhalation	1.1E-02	Co-60	0	1.1E-02	Co-60	0	1.1E-02	Co-60	0
Atmosphere	1.8E-06	Co-60	0	3.8E-06	Co-60	0	4.3E-06	Co-60	0

*Pathway not applicable.

Table F-3. Maximum annual CPG dose, dominant radionuclide, and year of occurrence for Scenario 3. University and Medical Center Complex - Urban Sanitary Landfill (LUMC-UF)

Site pathways	Humid Impermeable Site			Humid Permeable Site			Arid Permeable Site		
	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year
GW to River	7.3E-07	C-14	2360	*			*		
GW to Well	1.5E-03	C-14	2360	1.2E-01	C-14	100	6.4E-05	C-14	247
Spillage	5.4E-04	Cs-137	0	*			*		
Erosion	1.1E-06	C-14	3900	1.7E-03	C-14	3900	*		>13000
Bathtub	6.3E-04	C-14	100	*			*		
Food Onsite	*			*			*		
Biointrusion	*			*			*		
Direct Gamma	1.8E-01	Co-60	0	1.8E-01	Co-60	0	1.8E-01	Co-60	0
Dust Inhalation	1.6E-04	Am-241	0	1.6E-04	Am-241	0	1.6E-04	Am-241	0
Atmosphere	1.1E-09	H-3	0	2.3E-09	H-3	0	2.6E-09	H-3	0

*Pathway not applicable.

Table F-4. Maximum annual CPG dose, dominant radionuclide, and year of occurrence for Scenario 4. Metropolitan Area and Fuel-Cycle Facility - Suburban Sanitary Landfill (MAFC-SF)

	Humid Impermeable Site			Humid Permeable Site			Arid Permeable Site		
Site pathways	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year
GW to River	1.4E-06	C-14	2360	*			*		
GW to Well	1.8E-02	C-14	2360	1.1E+00	C-14	47	1.4E-04	C-14	231
Spillage	5.2E-04	Cs-137	0	*			*		
Erosion	4.1E-06	U-234	3900	5.6E-03	U-234	3900	*		>13000
Bathtub	5.7E-04	C-14	100	*			*		
Food Onsite	2.3E-02	Cs-137	1	1.4E-02	Cs-137	1	1.9E-02	Cs-137	1
Biointrusion	7.6E-02	Cs-137	1	4.6E-02	Cs-137	1	6.4E-02	Cs-137	1
Direct Gamma	8.9E-01	Co-60	0	8.9E-01	Co-60	0	8.9E-01	Co-60	0
Dust Inhalation	5.3E-02	U-234	0	5.3E-02	U-234	0	5.3E-02	U-234	0
Atmosphere	8.0E-06	U-234	0	1.7E-05	U-234	0	2.0E-05	U-234	0

*Pathway not applicable.

Table F-5. Maximum annual CPG dose, dominant radionuclide, and year of occurrence for Scenario 5. Metropolitan Area and Fuel-Cycle Facility - Suburban Sanitary Landfill with Incineration (MAFC-SI)

	Humid Impermeable Site			Humid Permeable Site			Arid Permeable Site		
Site pathways	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year
GW to River	8.0E-07	C-14	2320	*			*		
GW to Well	6.6E-02	C-14	2320	1.5E+00	C-14	25	9.4E-05	C-14	224
Spillage	2.8E-04	Cs-137	0	*			*		
Erosion	3.4E-06	U-234	3900	4.5E-03	U-234	3900	*		>13000
Bathtub	1.5E-04	C-14	100	*			*		
Food Onsite	9.0E-02	Cs-137	1	5.5E-02	Cs-137	1	7.7E-02	Cs-137	1
Biointrusion	3.0E-01	Cs-137	1	1.8E-01	Cs-137	1	2.6E-01	Cs-137	1
Direct Gamma	5.4E+00	Co-60	0	5.4E+00	Co-60	0	5.4E+00	Co-60	0
Dust Inhalation	2.1E-01	U-234	0	2.1E-01	U-234	0	2.1E-01	U-234	0
Atmosphere	3.6E-02	U-234	0	6.0E-02	U-234	0	3.7E-02	U-234	0

*Pathway not applicable.

Table F-6. Maximum annual CPG dose, dominant radionuclide, and year of occurrence for Scenario 6. Two-Unit Power Reactor, Institutional, and Industrial Facilities - Municipal Dump (PWRHU-MD)

	Humid Impermeable Site			Humid Permeable Site			Arid Permeable Site		
Site pathways	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year
GW to River	5.0E-07	I-129	3060	*			*		
GW to Well	2.0E-02	I-129	3060	6.7E-01	C-14	32	4.9E-04	I-129	556
Spillage	1.6E-03	Co-60	0	*			*		
Erosion	2.0E-06	Pu-239	3900	2.5E-03	Pu-239	3900	*		>13000
Bathtub	2.5E-04	I-129	100	*			*		
Food Onsite	2.4E-01	Cs-137	1	1.5E-01	Cs-137	1	2.0E-01	Cs-137	1
Biointrusion	8.1E-01	Cs-137	1	4.8E-01	Cs-137	1	6.8E-01	Cs-137	1
Direct Gamma	8.8E+00	Co-60	0	8.8E+00	Co-60	0	8.8E+00	Co-60	0
Dust Inhalation	2.1E-02	Am-241	0	2.1E-02	Am-241	0	2.1E-02	Am-241	0
Atmosphere	1.2E-06	Am-241	0	2.6E-06	Am-241	0	2.9E-06	Am-241	0

*Pathway not applicable.

Table F-7. Maximum annual CPG dose, dominant radionuclide, and year of occurrence for Scenario 7. Uranium Hexafluoride Facility - Municipal Dump (UHX-MD)

Site pathways	Humid Impermeable Site			Humid Permeable Site			Arid Permeable Site		
	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year
GW to River	9.0E-09	U-234	22200	*			*		
GW to Well	3.7E-04	U-234	22200	1.2E-03	U-234	22200	4.7E-05	U-234	22200
Spillage	5.9E-05	U-234	0	*			*		
Erosion	2.7E-06	U-234	3900	3.6E-03	U-234	3900	*		>13000
Bathtub	1.5E-05	U-234	100	*			*		
Food Onsite	1.2E-04	U-234	1	1.1E-04	U-234	1	1.3E-04	U-234	1
Biointrusion	4.2E-04	U-234	1	3.8E-04	U-234	1	4.4E-04	U-234	1
Direct Gamma	2.4E-02	U-235	3900	2.4E-02	U-235	3900	2.6E-03	U-235	>10000
Dust Inhalation	1.3E-01	U-234	0	1.3E-01	U-234	0	1.3E-01	U-234	0
Atmosphere	6.7E-06	U-234	0	1.5E-05	U-234	0	1.6E-05	U-234	0

*Pathway not applicable.

Table F-8. Maximum annual CPG dose, dominant radionuclide, and year of occurrence for Scenario 8. Uranium Foundry - Municipal Dump (UF-MD)

	Humid Impermeable Site			Humid Permeable Site			Arid Permeable Site		
Site pathways	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year
GW to River	4.2E-09	U-238	>7E+06	*			*		
GW to Well	1.7E-04	U-238	>7E+06	5.7E-04	U-238	>7E+06	2.2E-05	U-238	>7E+06
Spillage	1.8E-05	U-238	0	*			*		
Erosion	8.5E-07	U-238	3900	1.1E-03	U-238	3900	*		>13000
Bathtub	4.7E-06	U-238	100	*			*		
Food Onsite	3.8E-05	U-238	1	3.5E-05	U-238	1	4.1E-05	U-238	1
Biointrusion	1.3E-04	U-238	1	1.2E-04	U-238	1	1.4E-04	U-238	1
Direct Gamma	4.9E-03	U-235	3900	4.9E-03	U-235	3062	4.7E-04	U-235	>13000
Dust Inhalation	3.9E-02	U-238	0	3.9E-02	U-238	0	3.9E-02	U-238	0
Atmosphere	1.1E-05	U-238	0	2.4E-05	U-238	0	2.7E-05	U-238	0

*Pathway not applicable.

Table F-9. Maximum annual CPG dose, dominant radionuclide, and year of occurrence for Scenario 9. Large University and Medical Center with Onsite Incineration and Disposal (LURO-3)

	Humid Impermeable Site			Humid Permeable Site			Arid Permeable Site		
Site pathways	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year
GW to River	1.1E-06	C-14	2320	*			*		
GW to Well	5.4E-01	C-14	2320	2.4E+00	C-14	16	1.3E-04	C-14	221
Spillage	1.4E-05	Cs-137	0	*			*		
Erosion	6.4E-08	C-14	>8400	1.0E-04	C-14	>6600	*		>28000
Bathtub	8.9E-05	C-14	100	*			*		
Food Onsite	*			*			*		
Biointrusion	*			*			*		
Direct Gamma	1.6E-01	Co-60	0	1.6E-01	Co-60	0	1.6E-01	Co-60	0
Dust Inhalation	7.6E-03	Am-241	0	7.6E-03	Am-241	0	7.6E-03	Am-241	0
Atmosphere	1.3E-03	H-3	0	3.2E-03	H-3	0	1.3E-03	H-3	0

*Pathway not applicable.

Table F-10. Maximum annual CPG dose, dominant radionuclide, and year of occurrence for Scenario 10. Large Metropolitan Area with Consumer Wastes - Suburban Sanitary Landfill with Incineration (LMACW-SI)

	Humid Impermeable Site			Humid Permeable Site			Arid Permeable Site		
Site pathways	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year
GW to River	7.1E-07	I-129	2940	*			*		
GW to Well	6.1E-02	I-129	2940	1.1E+00	C-14	25	7.3E-04	I-129	553
Spillage	6.9E-04	Co-60	0	*			*		
Erosion	2.1E-06	Pu-239	3900	2.5E-03	Pu-239	3900	*		>13300
Bathtub	2.4E-04	I-129	100	*			*		
Food Onsite	2.1E-01	Cs-137	1	1.2E-01	Cs-137	1	1.7E-01	Cs-137	1
Biointrusion	6.8E-01	Cs-137	1	4.1E-01	Cs-137	1	5.7E-01	Cs-137	1
Direct Gamma	2.1E+01	Co-60	0	2.1E+01	Co-60	0	2.1E+01	Co-60	0
Dust Inhalation	3.9E-02	Am-241	0	3.9E-02	Am-241	0	3.9E-02	Am-241	0
Atmosphere	1.8E-03	Am-241	0	3.0E-03	Am-241	0	1.8E-03	Am-241	0

*Pathway not applicable.

Table F-11. Maximum annual CPG dose, dominant radionuclide, and year of occurrence for Scenario 11. Large Metropolitan Area with Consumer Wastes - Urban Sanitary Landfill with Incineration (LMACW-UI)

	Humid Impermeable Site			Humid Permeable Site			Arid Permeable Site		
Site pathways	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year
GW to River	5.5E-07	C-14	2320	*			*		
GW to Well	7.5E-03	C-14	2320	4.9E-01	C-14	46	3.0E-04	I-129	562
Spillage	9.0E-04	Cs-137	0	*			*		
Erosion	2.5E-06	Pu-239	3900	3.1E-03	Pu-239	3062	*		>13300
Bathtub	3.5E-04	C-14	100	*			*		
Food Onsite	*			*			*		
Biointrusion	*			*			*		
Direct Gamma	4.4E+00	Co-60	0	4.4E+00	Co-60	0	4.4E+00	Co-60	0
Dust Inhalation	1.3E-02	Am-241	0	1.3E-02	Am-241	0	1.3E-02	Am-241	0
Atmosphere	5.8E-04	Am-241	0	9.4E-04	Am-241	0	6.0E-04	Am-241	0

*Pathway not applicable.

Table F-12. Maximum annual CPG dose, dominant radionuclide, and year of occurrence for Scenario 12. Consumer Product Wastes - Suburban Sanitary Landfill (CW-SF)

	Humid Impermeable Site			Humid Permeable Site			Arid Permeable Site		
Site pathways	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year
GW to River	6.5E-11	Np-237	>7E+06	*			*		
GW to Well	9.3E-07	Np-237	>7E+06	4.3E-02	H-3	23	2.6E-08	Np-237	>7E+06
Spillage	2.6E-05	H-3	0	*			*		
Erosion	5.1E-08	Am-241	3900	6.2E-05	Am-241	3900	*		>13000
Bathtub	1.5E-06	Am-241	100	*			*		
Food Onsite	1.3E-05	Am-241	1	1.4E-05	Am-241	1	1.5E-05	Am-241	1
Biointrusion	4.5E-05	Am-241	1	4.6E-05	Am-241	1	5.1E-05	Am-241	1
Direct Gamma	3.9E-06	Am-241	3900	3.9E-06	Am-241	3062	1.4E-09	Am-241	>13000
Dust Inhalation	1.8E-03	Am-241	0	1.8E-03	Am-241	0	1.8E-03	Am-241	0
Atmosphere	2.7E-10	Am-241	0	6.0E-10	Am-241	0	6.7E-10	Am-241	0

*Pathway not applicable.

Note: Scenarios 12, 13, 14, and 15 are reference scenarios where the waste streams are already deregulated.

Table F-13. Maximum annual CPG dose, dominant radionuclide, and year of occurrence for Scenario 13. Consumer Product Wastes - Urban Sanitary Landfill (CW-UI)

Site pathways	Humid Impermeable Site			Humid Permeable Site			Arid Permeable Site		
	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year
GW to River	1.5E-10	Np-237	>7E+06	*			*		
GW to Well	3.8E-07	Np-237	>7E+06	1.7E-02	H-3	23	6.1E-08	Np-237	>7E+06
Spillage	1.4E-04	H-3	0	*			*		
Erosion	2.9E-07	Am-241	3900	3.5E-04	Am-241	3900	*		
Bathtub	8.5E-06	Am-241	100	*			*		
Food Onsite	*			*			*		
Biointrusion	*			*			*		
Direct Gamma	3.7E-06	Am-241	3900	3.7E-06	Am-241	3062	1.3E-09	Am-241	>13000
Dust Inhalation	1.7E-03	Am-241	0	1.7E-03	Am-241	0	1.7E-03	Am-241	0
Atmosphere	3.6E-10	Am-241	0	7.8E-10	Am-241	0	8.7E-10	Am-241	0

*Pathway not applicable.

Table F-14. Maximum annual CPG dose, dominant radionuclide, and year of occurrence for Scenario 14. Large University and Medical Center with Onsite Incineration and Disposal (LURO-1)

	Humid Impermeable Site			Humid Permeable Site			Arid Permeable Site		
Site pathways	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year
GW to River	*			*			*		
GW to Well	*			*			*		
Spillage	*			*			*		
Erosion	*			*			*		
Bathtub	*			*			*		
Food Onsite	*			*			*		
Biointrusion	*			*			*		
Direct Gamma	*			*			*		
Dust Inhalation	*			*			*		
Atmosphere	3.1E-04	H-3	0	7.6E-04	H-3	0	3.2E-04	H-3	0

*Pathway not applicable.

Table F-15. Maximum annual CPG dose, dominant radionuclide, and year of occurrence for Scenario 15. Large University and Medical Center with Onsite Incineration and Disposal (LURO-2)

Site pathways	Humid Impermeable Site			Humid Permeable Site			Arid Permeable Site		
	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year	Dose (mrem/yr)	Nuclide	Year
GW to River	1.8E-05	C-14	2320	*			*		
GW to Well	9.1E+00	C-14	2320	4.0E+01	C-14	16	2.2E-03	C-14	221
Spillage	2.3E-05	C-14	0	*			*		
Erosion	1.1E-06	C-14	>8400	1.7E-03	C-14	>6600	*		>28000
Bathtub	1.4E-03	C-14	100	*			*		
Food Onsite	*			*			*		
Biointrusion	*			*			*		
Direct Gamma	*			*			*		
Dust Inhalation	3.8E-06	H-3	0	3.8E-06	H-3	0	3.8E-06	H-3	0
Atmosphere	1.5E-04	H-3	0	3.8E-04	H-3	0	1.6E-04	H-3	0

*Pathway not applicable.