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BACKGROUND INFORMATION DOCUMENT FOR AMENDMENTS TO 40 CFR PART 191

ENVIRONMENTAL STANDARDS FOR THE MANAGEMENT AND DISPOSAL OF SPENT NUCLEAR FUEL, HIGH-LEVEL AND TRANSURANIC RADIOACTIVE WASTES

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Chapter 1: INTRODUCTION

The U.S. Environmental Protection Agency (EPA) is responsible for developing and issuing environmental standards and criteria to ensure that the public and environment are adequately protected from potential radiation impacts. With these objectives in mind, the EPA is proposing generally applicable environmental standards for the management and disposal of spent nuclear fuel and high-level and transuranic radioactive wastes. These standards provide the basic framework to control, in the long-term, the management and disposal of three types of radioactive wastes:

1. Spent nuclear reactor fuel, if ultimately disposed without reprocessing;

2. High-level radioactive liquid or solid wastes from the reprocessing of spent nuclear fuel; and

3. Transuranic wastes containing long-lived radionuclides of elements heavier than uranium; defined as containing more than 100 nanocuries per gram of wastes of alpha-emitting transuranic nuclides, with half-lives greater than 20 years.

1.1 EPA AUTHORITIES FOR THE RULEMAKING

These final amended standards have been developed pursuant to the Agency's authorities under the Atomic Energy Act (AEA) of 1954, as amended, and Reorganization Plan No. 3 of 1970 (NI70). The basic authority under the AEA, as transferred to the EPA by the Reorganization Plan of 1970, includes the mandate of:

> "establishing generally applicable environmental standards for the protection of the general environment from radioactive materials. 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 materials."

The Nuclear Waste Policy Act (NWPA) of 1982 established formal procedures regarding the evaluation and selection of sites for geologic repositories, including procedures for the interaction of State and Federal Governments; reiterated the existing responsibilities of the Federal Agencies involved in the national program; and provided a time table for several key milestones to be met by the Federal agencies in carrying out the program. As part of this national program, the EPA, pursuant to its authorities under other provisions of law, was required to:

> "by rule, promulgate generally applicable standards for the protection of the general environment from off-site releases from radioactive material in repositories."

In December 1987, Congress enacted the Nuclear Waste Policy Amendments Act (NWPA87). The 1987 Amendments Act redirects the nuclear waste program to consider Yucca Mountain, located in the State of Nevada, as the prime site for the nation's first high-level waste and spent nuclear fuel repository. Activities at all other potential sites were to be phased out. If Yucca Mountain is found to be suitable, the President is to submit a recommendation to Congress to develop a repository at this site. The Secretary of Energy is also required to inform Congress and the State if the site characterization activities indicate that Yucca Mountain is unsuitable. The Amendments Act prohibits the Department of Energy from conducting site-specific activities for a second repository unless authorized by Congress. Finally, the Act established a Commission to study the need and feasibility of a monitored retrievable storage facility to complement the nation's nuclear waste management program. The Commission submitted to Congress (as required under the original Act, as amended by Public Law 100-507) a report outlining their recommendations on November 1, 1989 (NWPA88, RMRS89).

In October 1992, the Waste Isolation Pilot Plant Land Withdrawal Act (WIPP LWA) was enacted and reinstated the 1985 disposal standards except sections 191.15 and 191.16. The WIPP LWA directs EPA to issue final disposal standards and specifies that such regulations shall not be applicable to the characterization, licensing, construction, operation or closure of any site required to be characterized under section 113(a) of the NWPA (Public Law 97-425).

1.2 HISTORY OF THE EPA RULEMAKING

Since the inception of the nuclear age in the 1940s, the Federal government has assumed ultimate responsibility for the care and disposal of high-level radioactive wastes regardless of whether they are produced by commercial or national defense activities. In 1949, the Atomic Energy Commission (AEC) initiated research and development work aimed at developing systems for the conversion of high-level liquid wastes into a stable form. Then, in 1955, at the request of the AEC, a National Academy of Sciences - National Research Council (NAS-NRC) Advisory Committee was established to consider the disposal of high-level radioactive wastes within the United States. Its report (NAS57), issued in 1957, recommended, that:

1. The AEC continue to develop processes for the solidification of high-level radioactive liquid wastes, and

2. Naturally occurring salt formations are the most promising medium for the long-term isolation of these solidified wastes.

Project Salt Vault, conducted from 1965 to 1967 by the AEC in an abandoned salt mine near Lyons, Kansas, demonstrated the safety and feasibility of handling and storing solid wastes in salt formations (MC70).

In 1968, the AEC again requested the NAS-NRC to establish a Committee on Radioactive Waste Management (CRWM) to advise the AEC concerning its long-range radioactive waste management plans and to evaluate the feasibility of disposing of solidified

radioactive wastes in bedded salt. The CRWM convened a panel to discuss the disposal of radioactive wastes in salt mines. Based on the recommendations of the panel, the CRWM concluded that the use of bedded salt is satisfactory for the disposal of radioactive wastes (NAS70).

In 1970, the AEC announced the tentative selection of a site at Lyons, Kansas, for the establishment of a national radioactive waste repository (AEC70). During the next two years, however, in-depth site studies raised several questions concerning the safe plugging of old exploratory wells and proposed expanded salt mining activities. These questions and growing public opposition to the Lyons site prompted the AEC in late 1971 to pursue alternatives to the salt site at Lyons (DO72).

In 1976, the Federal government intensified its program to develop and demonstrate a permanent disposal method for high-level radioactive wastes. The Office of Management and Budget (OMB) established an interagency task force on commercial wastes in March 1976. The OMB interagency task force defined the scope of the responsibility of each Federal agency's activities on high-level waste management, including the preparation of environmental standards for high-level wastes by the EPA (LY76, EN77a, EN77b).

A status report on the management of commercial radioactive nuclear wastes, published in May 1976 by the President's Federal Energy Resources Council (FERC), emphasized the need for coordination of administration policies and programs relating to energy. The FERC 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 and recommended the formation of 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 disposal activities, including high-level radioactive wastes that must be delivered to Federal repositories for long-term management (FERC76).

In 1976, President Ford issued a major policy statement on nuclear waste. As part of his comprehensive statement, he announced new steps to assure that the United States has the facilities for the long-term management of nuclear waste from commercial power plants. The President's actions were based on the findings of the OMB interagency task force formed in March 1976. He announced that the experts had concluded that the most practical method for disposing of high-level radioactive wastes is in geologic repositories located in stable formations located deep underground. Among the EPA's responsibilities, the Agency was to issue general environmental standards governing nuclear waste facility releases to the biosphere above natural background radiation levels (FO76). These standards were to place a numerical limit on long-term radiation releases outside the boundary of the repository.

In December 1976, the EPA announced its intent to develop environmental radiation protection criteria for radioactive wastes to assure the protection of public health and the general environment (EPA76). These efforts resulted in a series of radioactive waste disposal workshops, held in 1977 and 1978 (EPA77a, EPA77b, EPA78a, EPA78b).

In 1978, President Carter established the Interagency Review Group (IRG) to develop recommendations for the establishment of an administrative policy to address the long-term management of nuclear wastes and supporting programs to implement the policy. The IRG report re-emphasized EPA's role in developing generally applicable standards for the disposal of high-level wastes, spent nuclear fuel, and transuranic wastes (DOE79). In a message to Congress on February 12th, 1980, the President outlined the content of a comprehensive national radioactive waste management program based on the IRG recommendations. The message called for an interim strategy for disposal of high-level and transuranic wastes that would rely on mined geologic repositories. 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, the EPA published proposed "Criteria for Radioactive Wastes," which were intended as Federal Guidance for storage and disposal of all forms of radioactive wastes (EPA78c). In March 1981, however, the EPA withdrew the proposed criteria because the many different types of radioactive wastes made the issuance of generic disposal guidance too problematic (EPA81).

In 1982, under the authority of the Atomic Energy Act of 1954, the EPA proposed a set of standards under 40 CFR Part 191, "Environmental Standards for the Management and Disposal of Spent Nuclear Fuel, High-Level and Transuranic Radioactive Wastes" (EPA82). Shortly after the publication of the EPA's proposed rule, Congress passed the Nuclear Waste Policy Act of 1982 (Public Law 97-425), wherein the EPA was to "...promulgate generally applicable standards for the protection of the general environment from off-site releases from radioactive material in repositories..." not later than January 1984 (NWPA83).

After the first comment period on the proposed rule ended on May 2, 1983, the EPA held two public hearings on the proposed standards - one in Washington, D.C., on May 12-14, 1983, one in Denver, CO, on May 19-21, 1983 - and during a second public comment period requested post-hearing comments (EPA83a, EPA83b). More than 200 comment letters were received during these two comment periods and 13 oral statements were made at the public hearings. Responses to comments received from the public were subsequently published and released in August 1985 (EPA85a).

In parallel with its public review and comment effort, the Agency conducted an independent scientific review of the technical basis for the proposed 40 CFR Part 191 standards through a special Subcommittee of the Agency's Science Advisory Board (SAB). The Subcommittee held nine public meetings from January 18, 1983 through September 21, 1983 and later prepared and released a final report on February 17, 1984 (EPA83c, SAB84). Although the SAB review found that the Agency's analyses in support of the proposed standards were comprehensive and scientifically competent, the report contained several findings and recommendations for improvement. The report was publicly released on May 8, 1984 and the public was encouraged to comment on the findings and recommendations (EPA84). Responses to the SAB report were subsequently presented and released in August 1985 (EPA85b).

On February 8, 1985, the Natural Resources Defense Council, the Environmental Defense Fund, the Environmental Policy Institute, the Sierra Club, and the Snake River Alliance brought suit against the Agency and the Administrator because they had failed to comply with the January 7, 1984 deadline mandated by the NWPA for promulgation of the standards. A consent order was negotiated with the plaintiffs that required the standards to be promulgated on or before August 15, 1985. The EPA issued the final rule under 40 CFR Part 191 on that date (EPA85c, EPA85d).

The EPA standards were divided into two main sections, Subparts A and B. Subpart A addressed the management and storage of wastes. For any disposal facility operated by the Department of Energy and that is not regulated by the Nuclear Regulatory Commission or by Agreement States, under Subpart A of the Standard, the exposure limits to any member of the general public are 25 millirem (mrem) to the whole body and 75 mrem to any critical organ. For facilities which are regulated by the Nuclear Regulatory Commission or Agreement States, the Standards endorsed the annual dose limits given in 40 CFR Part 190, the environmental standards for the uranium fuel cycle, 25 mrem to the whole body, 75 mrem to the thyroid, and 25 mrem to the critical organ.

Subpart B imposed limits associated with the release of radioactive materials into the environment following closure of the repository. The key provisions of Subpart B were:

- Limits on cumulative releases of radioactive materials into the environment over 10,000 years;
- Assurance requirements to compensate for uncertainties in achieving the desired level of protection;
- Individual exposure limits based on the consumption of ground water and any other potential exposure pathways for 1,000 years after disposal; and
- Ground-water protection requirements in terms of allowable radionuclide concentrations and associated doses for 1,000 years after disposal.

Under Sections 191.15 and 191.16 of Subpart B, the annual dose to any member of the general public was limited to 25 mrem to the whole body and 75 mrem to any critical organ. The ground-water concentration for beta or gamma emitters is limited to the equivalent yearly whole body or organ dose of 4 mrem. The allowable water concentration for alpha emitters (including radium-226 and radium-228, but excluding radon) was 15 picocuries/liter. For radium-226 and radium-228 alone, the concentration limit was 5 picocuries/liter. Appendix A of the standards provided acceptable radionuclide cumulative release limits.

In March 1986, five environmental groups led by the Natural Resources Defense Council and four States filed petitions for a review of 40 CFR Part 191 (EPA85c, USC87). These suits were consolidated and argued in the U.S. Court of Appeals for the First Circuit in Boston. The main challenges concerned:

1. Violation of the Safe Drinking Water Act (SDWA) underground injection section;

2. Inadequate notice and comment opportunity on the ground-water protection requirements; and

3. certain aspects of the standards were thought to be arbitrary, not supported in the record, or not adequately explained.

In July 1987, the Court rendered its opinion and noted three findings against the Agency and two favorable judgements. The Court's action resulted in the remand of Subpart B. The Court began by looking at the definition of "underground injection," which is the "subsurface emplacement of fluids by well injection." A "well" is defined by the SDWA and the EPA as a shaft "bored, drilled, or driven where the depth is greater than the largest surface dimension." A "fluid" is a material or substance which flows or moves whether in a semi-solid, sludge, gas, or any other form or state." In the view of the Court, the method envisioned by DOE for disposal of radioactive wastes in underground repositories might fit both of the latter definitions and would "likely constitute an underground injection under the SDWA." Under the SDWA, the Agency is required to assure that underground sources of drinking water will not be endangered by any underground injection. With regard to such potential endangerment, the Court supported part, but not all, of the Agency's approach. A dichotomy appeared in the rationale when endangerment was considered inside the "controlled area" versus beyond the controlled area (i.e., in the accessible environment). Inside the controlled area, the Court ruled that Congress - through the EPA - had allowed endangerment of ground water. Therefore, the EPA's approach of using the geological formation as part of the containment was validated. However, outside the controlled area, the Court found that Section 191.15 would allow endangerment of drinking water supplies. In the context of the SDWA, "endangerment" is considered when doses higher than that allowed by the Primary Drinking Water Regulations may occur. Section 191.15 permits an annual dose of 25 mrem to the whole body and 75 mrem to any critical organ from all pathways. On the other hand, the regulations under the SDWA allow four mrem from drinking water. The Court recognized that less than four mrem may result from the ground-water pathway, however, it rejected this possibility because the Agency stated that radioactivity may eventually be released into the ground-water system near the repository which could result in substantially higher doses. Therefore, the Court decided that it seemed clear that a large fraction of the 25 mrem could be received through the ground-water exposure pathway. Accordingly, the Court found that the high-level wastes standards should have been consistent with the SDWA or the Agency should have explained that a different standard was adopted and its position should have been justified.

The Court also noted that the Agency is not necessarily incorrect in promulgating the proposed standards, however, the Agency never acknowledged the interrelationship of the SDWA and HLW rules nor did it present a reasonable explanation for the divergence between them. The Court also supported the petitioner's argument that the Agency arbitrarily selected the 1,000-year limit for individual protection requirements (Section 191.15) under undisturbed performance. The Court indicated that the 1,000-year criterion is not inherently flawed, but rather that the administrative record and the Agency's explanations do not adequately support

this choice. The criterion was remanded for reconsideration and the Agency must provide a more thorough explanation for its basis. Finally, the Court found that the Agency did not provide sufficient opportunity for notice and comments on Section 191.16 (Ground-water Protection Requirements) since that section was added to Subpart B after the standards were proposed. This section was remanded for a second round of notice and comments. There were, however, no rulings issued on technical grounds about Section 191.16.

In August 1987, the Justice Department asked the First Circuit Court to reinstate all of 40 CFR Part 191 except for Sections 191.15 and 191.16, which were originally found defective. The Natural Resources Defense Council filed an opposing opinion. The Court then issued an Amended Decree that reinstated Subpart A, but continued the remand of Subpart B.

In October 1992, the WIPP LWA was signed by the President. This Act reinstates Subpart B of 40 CFR Part 191, except Sections 191.15 and 191.16, and requires the Administrator to issue final disposal standards no later than 6 months after enactment. The reinstatement of these regulations is not applicable to the characterization, licensing, construction, operation, or closure of any site required to be characterized under the NWPA Section 113(a) of Public Law 97-425. The amended standards represent the Agency's response to the above legislation and to the issues raised by the court pertaining to individual and ground-water requirements. In so doing, EPA is not revisiting any of the regulations reinstated by the WIPP LWA.

1.3 PURPOSE AND SCOPE OF THE BACKGROUND INFORMATION DOCUMENT

This document provides the necessary background information, technical analyses, and justifications in support of the proposed amendments to 40 CFR Part 191.

The scope of this Background Information Document (BID) encompasses the conceptual framework for assessing radiation exposures and associated health risks. In general terms, this assessment examines the radioactive source term characterization, analysis of the movement of radionuclides from the repository through the appropriate environmental exposure pathways, and doses received by members of the general public. Consistent with the reinstatement provision of the WIPP LWA the only release mechanism considered in this document is normal ground-water flow because only individual doses and ground-water protection are addressed in this rulemaking. Transuranic waste is used as our source term instead of spent fuel because the WIPP LWA provision stated that the reinstatement is not applicable to the characterization, licensing, construction, operation, or closure of any site required to be characterized under the NWPA Section 113(a) of Public Law 97-425. Most of the waste under the NWPA is spent fuel and HLW. The majority of the waste not covered by the NWPA is transuranic waste. This document used transuranic waste for individual dose and ground-water protection analysis. A separate technical support document contains the individual and population dose analyses for spent fuel and HLW.

1.4 ANALYTICAL COMPUTER CODES

The principal computer code used in the risk assessments is NEFTRAN-S. This code was preceded by the NEFTRAN, NWFT/DVM, and NWFT codes, all developed by Sandia National Laboratories. (The NWFT/DVM code was used to support the 1985 promulgation of 40 CFR Part 191.) All of the codes model network flow and transport using the distributed velocity method. The codes have evolved such that each code contains both the capabilities of its predecessor and new features to enhance the modeling capability. The NEFTRAN code expanded the capability to simulate transport through saturated, dual-porosity fields or fractured media. The NEFTRAN-S version further enhanced the code capability by including statistical analysis of radionuclide transport. Chapters 7 and 8 further describe the capabilities of NEFTRAN-S.

1.5 PROGRAM TECHNICAL SUPPORT DOCUMENTS

A number of technical support documents have been used and published during the history of the rulemaking activities to establish the technical basis of the standards. The following list presents the documents which have been used to support the current rulemaking activities.

1. Technical Support of Standards for High-Level Radioactive Waste Management - Volume A, Source Term Characterization, EPA 520/4-79-007A, March-July 1977.

This report provides a characterization of commercial spent nuclear fuel and high-level wastes, including comparisons of source terms from various fuel cycles and fuel mixes; a characterization of government high-level and transuranic wastes; a comparison with commercial wastes; and an estimate of existing and projected quantities of spent nuclear fuel and high-level and transuranic wastes. The data are presented in several formats and by specific basis (per unit of fuel used or energy generated), as well as on a total basis for a given number of nuclear power plants.

2. Technical Support of Standards for High-Level Radioactive Waste Management - Volume B, Engineering Controls, EPA 520/4-79-007B, March-August 1977.

This report reviews the technology for engineering control of spent fuel and high-level and TRU wastes and projected costs of the various disposal technologies. Analyses include processing and packaging technologies, alternative geologic disposal techniques, effectiveness of engineering controls, and associated cost considerations.

3. Technical Support of Standards for High-Level Radioactive Waste Management - Volume C, Migration Pathways, EPA 520/4-79-007C, March - July 1977.

This report assesses geologic site selection factors; quantification of the potential migration and dispersion of radionuclides through the biosphere; and dose implications of a repository containing radioactive wastes at high concentrations.

4. Technical Support of Standards for High-Level Radioactive Waste Management - Volume D, Release Mechanisms, EPA 520/4-79-007D, March 1980.

This report analyzes the potential radionuclide releases from a generic deep-mined repository for radioactive wastes. Five different geologic media are considered: bedded salt, dome salt, granite, basalt, and shale. A range of potential containment failure mechanisms was evaluated and compared. The results combine radionuclide transport and dose calculations in assessing potential health effects of a repository.

5. Technical Support of Standards for High-Level Radioactive Waste Management - Volume E, Addendum to Volumes C and D, EPA 520/4-79-007E, March 1982.

This report updates the information and issues relevant to the conclusions reached in Volumes C and D.

6. Assessment of Waste Management of Volatile Radionuclides, EPA ORP/CSD-79-2, May 1979.

This report reviews waste management technologies in terms of immobilization, containment, and disposal of I-129, Kr-85, H-3, and C-14. Included are alternative disposal options that may be applied to isolate these wastes from human exposures and the environment.

7. Radiation Exposures From Solidification Processes for High-Level Radioactive Liquid Wastes, EPA 520/3-80-007, May 1980.

This report provides an assessment of a generic high-level liquid waste solidification plant and the potential environmental impact of atmospheric discharges during normal operations involving four different solidification processes.

8. A Review of Radiation Exposure Estimates From Operations in the Management and Disposal of High-Level Radioactive Wastes and Spent Nuclear Fuel, EPA 520/3-80-008, August 1980.

This report provides an analysis of the estimated radioactive releases during normal waste management operations (i.e., preparation for storage, disposal, and emplacement) and resulting radiation exposures and doses.

9. Economic Impacts of 40 CFR 191: Environmental Standards and Federal Guidance for Management and Disposal of Spent Nuclear Fuel, High-Level and Transuranic Radioactive Wastes, EPA 520/4-80-014, December 1980.

This report develops a methodology for examining the potential economic impacts of the proposed environmental standards.

10. Population Risks from Uranium Ore Bodies, EPA 520/3-80-009, October 1980.

This report presents a methodology for estimating the radiological releases and potential health impact of deep-lying uranium ores.

11. High-Level and Transuranic Radioactive Wastes - Background Information Document for Final Rule, EPA 520/1-85-023, August 1985.

This report presents estimates of population doses and risks associated with disposal of radioactive wastes in geologic repositories and describes the methodologies used to derive these estimates.

12. Final Regulatory Impact Analysis - 40 CFR Part 191: Environmental Standards for the Management and Disposal of Spent Nuclear Fuel, High-Level and Transuranic Radioactive Wastes, EPA 520/1-85-027, August 1985.

This report reviews the project costs associated with the management and disposal of high-level radioactive wastes. The reports also addresses the containment, and ground-water and individual protection requirements from such wastes.

13. High-Level and Transuranic Radioactive Wastes - Response to Comments for Final Rule, Volume I, EPA 520/1-85-024-1, August 1985.

This report presents a compilation of public comments and the EPA's responses in support of the promulgation of the proposed environmental standards.

14. High-Level and Transuranic Radioactive Wastes - Response to Comments for Final Rule, Volume II, EPA 520/1-85-024-2, August 1985.

This report presents a compilation of comments generated by the Science Advisory Board and the EPA's responses in support of the promulgation of the proposed environmental standards.

15. Environmental Pathway Models for Estimating Health Effects From Disposal of High-Level Radioactive Waste in Geologic Repositories, EPA 520/5-85-026, May 1986.

This report presents detailed methodology and models to characterize the mobilization, environmental transport, exposure pathways, and doses associated with potential releases of radioactive materials from high-level waste repositories.

16. Risk Assessment for TRU Waste Disposal in Bedded Salt; Prepared by Rogers & Associates Engineering Corporation, under contract with Sanford Cohen & Associates, Inc., Contract No. 68D90170, Work Assignment 2-29, March 1992.

This report augments the analysis of TRU waste disposal provided in the 1985 BID for 40 CFR Part 191. It expands the discussion of uncertainty and sensitivity.

17. Risk Assessments of Spent Fuel, Transuranic, and High-Level Radioactive Wastes in Mined Repositories, Technical Support Document. Prepared by Rogers & Associates Engineering Corporation, under contract with Sanford Cohen & Associates, Inc., Contract No. 68D20155, Work Assignment 1-6, Report No. RAE-9231/1-3, January 20, 1993.

This document updates the 1985 BID technical analyses for evaluating population and individual risks. In addition, it performs a similar multi-media analysis on population and individual risks from a TRU waste disposal facility.

18. Economic Impact Analysis for Amendments to EPA's Radioactive Waste Standards (40 CFR Part 191), EPA 402-R-92-007, December 1992.

This report assesses the economic impact from the proposed amendments and additions to 40 CFR Part 191.

19. NEFTRAN-S: A Network Flow and Contaminant Transport Model for Statistical and Deterministic Simulations Using Personal Computers, Sandia Report, SAND-90-1987, UC-502, May 1991.

This report describes the NEFTRAN-S computer code and was written to provide a comprehensive discussion of the code including its history, the theory, its use and examples of possible applications.

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This report describes how NEFTRAN-S Code was used to provide estimates of releases to the environment that could result from disposal of radioactive waste in an unsaturated tuff zone.

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Chapter 2: CURRENT REGULATORY PROGRAMS AND STRATEGIES

2.1 INTRODUCTION

Human beings and all other living organisms have always been exposed to ionizing radiation from cosmic rays and the naturally occurring radioactivity contained in the Earth. These two sources of radiation or radioactivity make up the natural radiation background environment in which all life forms have evolved. Our experience with radiation dates back only to the end of the last century, when X-rays were discovered in 1895 and naturally occurring radioactivity was observed in 1896. These discoveries marked the beginning of the deliberate use of radioactivity and radioactive materials in science, medicine, and industry.

The findings of radiation science rapidly led to the development of medical radiology, industrial radiography, nuclear physics, and nuclear medicine. By the 1920s, the use of X-rays in diagnostic medicine and industrial applications was widespread. Radium was being routinely used in luminescent dials and by doctors in therapeutic procedures. By the 1930s, biomedical and genetic research scientists were studying the effects of radiation on living organisms and physicists were beginning to understand the mechanisms of spontaneous fission and radioactive decay. In the 1940s, research in nuclear physics had advanced to the point where a self-sustaining fission reaction was demonstrated under laboratory conditions. These events led directly to the construction of the first nuclear reactors and the development of atomic weapons.

Since the end of World War II, research and development activities in all aspects of nuclear physics have been accelerating. Today the use of radiation or radioactivity, be it naturally-occurring or man-made, is widespread and reaches every segment of our society. The uses or applications include:

- Nuclear reactors, which generate electricity and power ships and submarines; produce radioisotopes for research, medical and industrial applications, space, and national defense; and are used as research tools for nuclear engineering and physics.

- Particle accelerators, which produce radioisotopes and radiation, are used to study the structure of matter, atoms, and common materials.

- The radio-pharmaceutical industry, which provides the radioisotopes used in nuclear medicine, biomedical research, and medical treatment.

- Nuclear medicine, which uses radioisotopes for the diagnosis and treatment of numerous diseases.

- X-rays and gamma rays, which are widely used as diagnostic tools in medicine and in diverse industrial applications, such as industrial radiography, luggage x-ray inspections, and non-destructive materials testing.

- Radionuclides, which are used in common consumer products, such as smoke detectors, luminous-dial wrist watches, luminous markers and signs, cardiac pacemakers, lightning rods, static eliminators, welding rods, lantern mantles, and optical glass.

As the use of radioactive materials and radiation became widespread, it was recognized that their use would have to be controlled to protect the users, public, and the environment. The following sections present a brief history of the evolution of radiation protection activities, principles and concepts used in radiation protection, and regulatory programs and strategies. These activities are summarized for two basic types of organizations - those responsible for direct regulation and oversight and those that only provide technical guidance and regulatory recommendations without the force of law.

2.2 INTERNATIONAL ATOMIC ENERGY AGENCY, 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 1920s, sufficient direct evidence (from 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 an official body should be established to make recommendations concerning human protection against exposure to X-rays and radium.

At the Second International Congress of Radiology meeting in Stockholm, Sweden, 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. It was charged with developing recommendations concerning radiation protection. In 1950, to better reflect its role in a changing world, the Commission was reorganized and renamed the International Commission on Radiological Protection (ICRP).

During the Second International Congress of Radiology, the newly created Commission suggested to the nations represented at the Congress that they appoint national advisory committees to represent their viewpoints before the Commission, 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 advisory group for the United States. This Committee operated until 1964 when it was Congressionally chartered as the National Council on Radiation Protection and Measurements (NCRP).

Throughout their existence, the ICRP and the NCRP have worked together closely to develop radiation protection recommendations that reflect the current understanding of the risks associated with exposure to ionizing radiation (ICRP34, ICRP38, ICRP51, ICRP59, ICRP65).

In 1977, the ICRP released recommendations which are currently in use. In ICRP Publication No. 26 (ICRP77), it adopted the weighted whole-body dose equivalent (defined as the effective dose equivalent) concept for limiting occupational exposures. This change reflected the increased understanding of the differing radio-sensitivities of various organs and tissues, and was intended to sum exposures from external sources and from internally deposited nuclides. (Note: the concept of summing internal and external exposures to arrive at total dose had been mentioned as early as ICRP Publication No. 1 [ICRP59]). The occupational overall annual exposure limit is now 5 rem (as an effective dose equivalent).

The ICRP report also introduced the concept of stochastic and non-stochastic radiation effects, and defined the aim of radiation protection as to "...prevent detrimental non-stochastic effects and to limit the probability of stochastic effects to levels deemed to be acceptable..." The concept of collective dose equivalent for populations was also discussed. Also significant is the fact that the ICRP 26 recommendations represent the first explicit attempt to relate and justify permissible radiation exposures with quantitative levels of acceptable risk. The ICRP concluded that "...the mortality risk factor for radiation-induced cancers is about 10^{-4} per rem, as an average for both sexes and all ages..." Thus, the risks of average occupational exposures (about 0.5 rem/year) are roughly comparable to risks experienced in safe industries, 10^{-4} annually. At the permissible limit of 5 rem/year, the risk is comparable with that experienced by some workers in occupations having higher-than-average risk.

For members of the public, the ICRP considered that an annual risk in the range of 10^{-6} to 10^{-5} would likely be acceptable. This would imply the restriction of the annual dose to an individual of 100 mrem. The existing recommended annual dose limit of 500 mrem, applied to critical groups, was found to provide an adequate degree of safety, even though a few individuals exposed to the limit could have an annual risk in the range of 10^{-5} to 10^{-4}.

The ICRP recommended the continued use of the 500 mrem annual limit for individuals, under specified conditions. No dose limits for populations were proposed; the Commission felt that the system of dose limitation specified in ICRP 26 was "...likely to ensure that the average dose equivalent to the population will not exceed 50 mrem per year..."

In 1979, the ICRP issued Publication No. 30 (ICRP79), which established the Annual Limit on Intake (ALI) system for limiting the intake of radionuclides by workers. The ALI is the activity of a given nuclide which would irradiate a person to the limit set in ICRP No. 26 for each year of occupational exposure. It is a secondary limit, based on the primary limit of equivalent whole-body irradiation, and applies to intake by either ingestion or inhalation. The recommendations of ICRP No. 30 applied only to occupational exposures. In 1983, the ICRP issued a statement (ICRP84) to clarify the use of ALIs and DACs for members of the public. It was recommended that the appropriate authorities should assess each specific situation.

In 1985, the ICRP issued a statement (ICRP85) commenting on dose limits for members of the public. ICRP No. 26 had endorsed an annual limit of 500 mrem, subject to certain conditions. In making this endorsement, it was assumed that the conditions would, in practice, restrict the average annual dose to about 100 mrem. In the 1985 statement, the Commission stated that the principal limit was 100 mrem, while occasional and short-term exposures up to 500 mrem were thought to be acceptable. More recently, the Commission

has published additional guidance for waste disposal (ICRP85b) and for general radiological protection (ICRP91). The first of these "Radiation Protection for the Disposal of Solid Radioactive Waste" emphasizes an individual risk approach that considers both the probability of an event and its consequence.

In 1987, the NCRP issued Report No. 91 (NCRP87), which acknowledged the assumptions and the basic thrust of the recommendations in ICRP Reports 26 and 30. In discussing risk estimates, the NCRP noted that (in 1987) new data were becoming available which might require changes in the current estimates. However, the value of 10^{-4} per rem, recommended in ICRP No. 26, was retained for a nominal lifetime somatic risk for adults.

The NCRP also noted that continuous annual exposure to 100 mrem, which approximates the average whole-body background exposure, gives a person a mortality risk of about 10^{-5} annually, or approximately 10^{-3} in a lifetime. Annual limits of 500 mrem were recommended for infrequent exposures and 100 mrem for continuous (or frequent) exposures. These limits do not include natural background or medical exposures.

In 1989, the International Atomic Energy Agency (IAEA) issued reports 96 and 99 in its Safety Series (IAEA89a, IAEA89b). These documents presented criteria and guidance for the underground disposal of nuclear wastes. Safety Series No. 99, "Safety Principles and Technical Criteria for the Underground Disposal of High-Level Radioactive Wastes," sets out basic design objectives to ensure that "humans and the human environment will be protected after closure of the repository and for the long periods of time for which the wastes remain hazardous." It states that for releases from a repository due to gradual processes, the dose upper bound should be less than an annual average dose value of 1 mSv for prolonged exposures for individuals in the critical group (defined as the members of the public whose exposure is relatively homogeneous and is typical of individuals receiving the highest effective dose equivalent or dose equivalent from a given radiation source). It suggests a risk upper bound of 10^{-5} per year for an individual for disruptive events.

2.3 INTERNATIONAL STANDARDS-SETTINGS

As with the United States, countries which are committed to use nuclear power (or in which nuclear power already makes up a significant fraction of the total electrical generating capacity) are establishing long-term programs for the safe management and disposal of spent reactor fuel and high-level radioactive and transuranic wastes (collectively referred to here as HLW). Such programs include adopting a national strategy, assigning the technical responsibility for research and development activities usually to a state-owned agency, and setting regulatory standards to protect public health and the environment. HLW management strategies may include spent fuel storage at and away from reactor sites, spent fuel reprocessing, HLW vitrification and storage, and ultimate HLW disposal in deep geological media. For illustrative purposes, the institutional/regulatory programs of eight countries are summarized below (IAEA91, NEA86, NEA88, NEA91, SCH88, SCH91, IEAL87). These countries are Canada, the United Kingdom, France, Germany, Belgium, Switzerland, Sweden, and Japan. A summary of these countries' planned HLW disposal programs is also provided in Chapter 4.

2.3.1 Canada

In 1990, Canada produced about 15% of its electrical needs through nuclear power (19 pressurized heavy water cooled and moderated reactors). Canada relies on the CANDU reactor design which operates using natural uranium in a once-through fuel cycle, i.e., the fissile material is not recycled or reprocessed. It is estimated that by the year 2000, Canada will have produced about 34,000 metric tons (heavy metal) of spent fuel.

Atomic Energy of Canada Limited (AECL) has the lead role in developing a HLW disposal facility. The AECL has reached a cooperative agreement with Ontario Hydro (a provincially owned utility) for developing interim technologies for the storage and transportation of spent fuel. The Atomic Energy Control Board (AECB) is the lead regulatory agency for assessing and determining the long-term performance of the disposal facility. The AECB also develops and issues policy statements and regulatory guidance for the eventual licensing of the HLW repository.

Between 1985 and 1987, the AECB issued three regulatory documents containing statements of policy on nuclear waste disposal and guidance on HLW repository siting and waste disposal. The overall regulatory objective expressed in these documents is to ensure that there is a small probability that radiation doses to the public associated with the repository will exceed a small fraction of natural background radiation doses. The burden on future generations is to be minimized without relying on long-term institutional controls, and there should be no future impacts on the environment that would not currently be accepted. Predicted radiological risk to individuals from a waste repository must not exceed 1×10^{-6} fatal cancers and serious genetic effects per year. As a guideline, calculations of individual risks should be made using the risk conversion factor of 2×10^{-2} per sievert. For the purpose of demonstrating compliance with the individual risk requirement, the time period need not exceed the first 10,000 years.

2.3.2 United Kingdom

In 1990, the United Kingdom (Britain) produced about 20% of its electrical needs through nuclear power. Britain depends primarily on gas cooled reactors (36 units), but it is also considering other reactor designs, including breeder reactors (one unit in operation) and pressurized light water reactors (one unit under construction). The government-owned utility Nuclear Electric proposes to begin construction of three additional PWRs in the 1990s. British Nuclear Fuels Ltd. (BNFL), another government-owned corporation, reprocesses spent fuel at its Sellafield facility on behalf of both domestic and foreign utilities, and since 1952 over 30,000 metric tons (heavy metal) of metal Magnox fuel have been reprocessed. Of this total, about 15,000 metric tons of uranium have been recycled into new reactor fuel. BNFL plans to begin operating a new reprocessing plant at Sellafield for oxide fuel, the Thermal Oxide Reprocessing Plant (THORP), in 1992. Britain's current plans are to solidify reprocessing wastes in glass and then dispose of them in deep geologic media. It is estimated that by the year 2000, Britain will have about 4,000 cubic meters (about 140,000 cubic feet) of HLW destined for storage or disposal due to the reprocessing of some 60,000 metric tons of spent fuel.

The responsibility for the disposal and safeguard of radioactive wastes is shared by several governmental agencies. The regulatory functions are performed by the Nuclear Installations Inspectorate, which is part of the Health and Safety Executive; the Radiochemical Inspectorate of the Department of the Environment; the Ministry of Agriculture, Fisheries, and Food; the UK Atomic Energy Authority; and the Secretaries of State of Scotland and Wales. The government also takes advice from several independent expert and advisory committees, including the Radioactive Waste Management Advisory Committee. In 1982, the government established the Nuclear Industry Radioactive Waste Executive (NIREX) to develop and operate intermediate and low-level radioactive waste disposal facilities. NIREX was originally established as a partnership consisting of private firms and governmental agencies. In 1985, NIREX was restructured as an independent legal entity as UK NIREX. The BNFL has the lead responsibility for management of HLW from reprocessing, and began operating a vitrification plant at Sellafield in 1990. Reprocessed and solidified wastes will be stored for at least 50 years prior to disposal. The need for a high-level waste repository is not contemplated until the year 2040.

The Atomic Energy Act of 1946 establishes the authority and responsibility to control and regulate the development of nuclear power in Britain. The Act has since been amended several times to establish new requirements, including those addressing the management and disposal of radioactive wastes. The government has not, however, issued detailed regulations for HLW disposal, since British policy is to store HLW for at least 50 years. Current guidance suggests that radiation exposure limits for members of the general public would most likely be based on ICRP guidance, or about 10 mrem per year.

2.3.3 France

In 1990, France produced about 75% of its electrical needs through nuclear power. The French nuclear power program relies primarily on pressurized light water reactors (52 units). Older gas cooled reactors are being phased out, while research and development activities and demonstration projects focus on an alternate reactor design (liquid metal fast breeder reactor) for power production. France reprocesses spent fuel, and from 1976 through 1990 had reprocessed over 20,000 metric tons (heavy metal) of metal and oxide fuel. The new UP3 reprocessing line began operation in 1990, and an expansion of the UP2 facility is scheduled to be completed in 1994. Current plans are to solidify reprocessing wastes in glass before placement and disposal in deep geological formations. A vitrification plant for UP2 entered service in 1990 and a plant for UP3 entered service in July 1992. It is estimated that by the year 2000, France will accumulate about 3,000 cubic meters of HLW and 47,000 cubic meters of alpha waste. Like Britain, France provides reprocessing services to foreign customers in addition to its domestic market.

The French nuclear power industry is controlled by several agencies, some of which are quasi-governmental agencies. The key agencies include the French Atomic Energy Commission (CEA) and its subsidiaries, the Institute for Nuclear Protection and Safety (IPSN), the National Radioactive Waste Management Agency (ANDRA), COGEMA (operator of spent fuel reprocessing and HLW immobilization facilities), and SGN (architect and engineering services); the Directorate for the Safety of Nuclear Installations (DSIN) within the

Ministry of Industry; the Bureau of Geological and Mineral Research; and Electricité de France (the national electric utility).

ANDRA was formed in 1979 to be responsible for all radioactive waste disposal activities and long-term management, and is chartered to design, build, and operate waste disposal facilities. ANDRA must comply with CEA requirements as well as those promulgated by DSIN, which is an independent agency under the Ministry of Industry. DSIN issued "Fundamental Safety Rule III.2.f.," pertinent to high-level and alpha waste disposal, on June 10, 1991. The rule requires, among other things, that the impact of a deep geologic disposal facility be as low as reasonably achievable; that individual dose equivalent due to the facility be limited to 0.25 millisieverts (25 millirem) per year for likely events; that the stability of geologic barriers be demonstrated for at least 10,000 years; and that HLW packages prevent the release of radioactive contents during the period when short- and medium-lived radionuclides dominate total radioactivity.

2.3.4 Germany

In 1990, Germany produced about 33% of its electrical needs through nuclear power. The German nuclear power program relies primarily on pressurized light water reactors (14 units) and boiling water reactors (7 units). Research and development activities and demonstration projects are also evaluating alternate reactor designs (high temperature gas-cooled reactors and liquid metal fast breeder reactors) for power production. Germany's plan for a domestic reprocessing facility was abandoned in 1989, but German utilities ship their spent fuel to France and Britain for reprocessing. It is estimated that by the year 2000, Germany will have generated about 9,000 metric tons (heavy metal) of spent fuel. Vitrified waste will be returned to Germany and stored in metal casks prior to disposal.

In Germany, the institutional and legal framework for the regulation of nuclear facilities is based on the joint participation of Federal and State governments. The Atomic Energy Act and the Radiation Protection Ordinance establish the principles and requirements regarding the safe utilization and application of atomic energy and radioactive materials, including the disposal of radioactive wastes. The key agencies include the Federal Ministry for Environment, Protection of Nature and Reactor Safety (BMU), the Federal Ministry for Research and Technology (BMFT), the Federal Institute for Radiation Protection (BfS), which is responsible for repository construction and operation, the Federal Institute for Geosciences and National Resources, and the host state's ministry for environmental protection. In addition, a consortium of Germany's nuclear utilities and engineering firms has been formed to meet the industry's responsibilities for spent fuel storage, reprocessing, waste management, and waste disposal.

Vitrified HLW will be disposed of in a salt dome at Gorleben in the State of Lower Saxony if the site proves to be acceptable. The disposal of radioactive wastes in deep geological media is governed by safety criteria issued by the Federal government in 1982. The regulations provide specific objectives to be met for each phase of the development of the repository. Additional licensing procedures and guidance will be issued in support of the licensing activities. The long-term performance objectives for the repository require that doses to members of the general population be limited to 30 mrem per year following closure.

2.3.5 Belgium

In 1990, Belgium produced about 60% of its electrical needs through nuclear power. The Belgian nuclear power program relies on seven pressurized light water reactors. From 1966 to 1974, Belgium reprocessed spent fuel at its Eurochemic facility. The company Belgoprocess was created to reactivate the Eurochemic plant in a consortium with foreign firms, but these efforts failed in the mid-1980s and Belgoprocess is now responsible for decommissioning the plant. Belgium is currently shipping some of its spent fuel to France for reprocessing and storing some of it in reactor pools. It is estimated that by the year 2000, Belgium will have produced about 2,500 metric tons (heavy metal) of spent fuel. A vitrification plant, PAMELA, began processing wastes from the Eurochemic plant in 1985.

The independent National Agency for Radioactive Waste and Fissile Materials (ONDRAF) was established in 1982 for the long-term management and disposal of radioactive wastes, including spent fuel, high-level wastes, and reprocessing wastes returned from the French facility. In addition to ONDRAF, the other key organizations or agencies with responsibilities related to waste management include the Ministry of Public Health, the Ministry of Economic Affairs and the Ministry of Employment. An inter-ministerial commission was also established to coordinate all related activities within each ministry. The Nuclear Research Center (CEN), under the Ministry of Economic Affairs, provides technical assistance in basic and applied R&D in nuclear energy and technology.

ONDRAF intends to begin operation of a shallow land burial facility for LLW in the mid-1990s and has established an underground laboratory in a clay formation at Mol to evaluate the site's suitability as a HLW repository. There are currently no specific regulatory requirements or criteria governing the disposal of spent fuel and high-level wastes.

2.3.6 Switzerland

In 1990, Switzerland's five nuclear power plants supplied about 43% of the country's electrical power needs. The Swiss nuclear power program relies on a mix of pressurized and boiling light water reactors (3 PWRs and 2 BWRs). It is estimated that by the year 2000, the Swiss will have produced about 1,800 metric tons (heavy metal) of spent fuel. Switzerland is currently shipping its spent fuel to France and Britain for reprocessing and holds contracts to reprocess all spent fuel produced through 1993. For spent fuel generated after 1993, Switzerland maintains the options of spent fuel management both with and without reprocessing.

A joint government and utility cooperative agency (NAGRA) was established in 1972 to manage the disposal of radioactive wastes, including spent fuel, HLW and other reprocessing wastes returned from the French and British reprocessing facilities. In addition to NAGRA, other key organizations or agencies with direct responsibilities in waste management include the Nuclear Safety Division (HSK) of the Federal Energy Office (BEW) within the Federal Department of Transport, Communications, and Energy (EVED), the Federal Commission for the Safety of Nuclear Installations (KSA), the Federal Department of Interior (EDI) and the Institute for Reactor Research (EIR). An interagency working group

(AGNEB) was also established to coordinate activities in support of Government decisions on the licensing of nuclear waste facilities.

A central interim storage facility for spent fuel and low-, intermediate- and high-level wastes is planned at Würenlingen, which has agreed to host the facility. NAGRA plans to begin construction of an intermediate-depth repository for low- and intermediate-level wastes by 2000; one of four candidate sites is to be selected for detailed characterization in 1993. With regard to the high-level waste repository, NAGRA is considering crystalline and sedimentary rock formations; repository operation will not begin before 2020 to allow a 40-year waste cooling period. HSK and KSA published safety goals for the disposal of all categories of radioactive waste in 1980. The goals are to limit individual doses due to radionuclide releases from a repository, from realistically assumed processes and events, to 10 mrem/year; and that a repository must be designed so that it can be sealed at any time within a few years, after which it must be possible to go without institutional controls.

2.3.7 Sweden

In 1990, nine boiling water reactors and three pressurized water reactors supplied about 46% of Sweden's electrical power needs. Under a 1980 referendum, the Swedish nuclear power program is to be phased out by the year 2010. By that time, Sweden will have produced nearly 8,000 metric tons (heavy metal) of spent fuel. Swedish utilities had contracted in the 1970s for foreign reprocessing of spent fuel, but this approach was abandoned after the 1980 referendum and the utilities have since sold their contracts or traded HLW from reprocessing for other spent fuel. A centralized spent fuel storage facility went into operation in 1985 and will eventually hold all Swedish spent fuel for about 40 years. A repository for short-lived low- and intermediate-level wastes, SFR, began operating in 1988. Three candidate sites for a high-level waste repository are to be identified in 1993, followed by detailed characterization of two sites beginning in 1997 and the filing of a license application for one site in 2003. Construction is anticipated to begin around 2010 and operation around 2020.

A joint utility consortium, the Swedish Nuclear Fuel and Waste Management Company (SKB), manages the disposal of radioactive wastes. The key government entities with direct responsibilities in waste management, operating under the Ministry of the Environment and Energy, include the Swedish Nuclear Power Inspectorate (SKI), the National Board for Spent Nuclear Fuel (SKN), the National Institute for Radiation Protection (SSI) and the Swedish Consultative Committee for Nuclear Waste Management (KASAM).

The SKI is now developing regulatory principles and criteria for geologic disposal of HLW, in cooperation with SSI. The SKI intends to develop increasingly detailed guidelines for the repository system during the 1990s. However, both SKI and SSI favor a total systems approach, without specifying detailed sub-system quantitative criteria in early phases of repository development. Criteria for the waste package and other components will be developed eventually, in time for use in the licensing procedure beginning around 2003.

2.3.8 Japan

In 1990, Japan produced about 27% of its electrical needs through nuclear power. The Japanese nuclear power program relies primarily on pressurized light water reactors (19 units) and boiling water reactors (21 units). Research and development activities and demonstration projects are also evaluating alternate reactor designs (gas cooled reactor, heavy water moderated reactor, and liquid metal fast breeder reactor) for power production. It is estimated that by the year 2000, Japan will have discharged about 20,000 metric tons (heavy metal) of spent fuel from its reactors. Japanese utilities have secured reprocessing services from France and Britain. In addition, a small reprocessing plant has been operating in Japan since 1977 and a large plant is scheduled to begin operating by about 1998. Japan plans to recycle recovered plutonium in thermal reactors and eventually in breeder reactors. Vitrified HLW will be stored 30-50 years for cooling before ultimate disposal in a geologic repository.

The Atomic Energy Basic Law of 1955 established the Japan Atomic Energy Commission (AEC) and the principles and requirements regarding the safe utilization and application of atomic energy and radioactive materials, including the disposal of radioactive wastes. In addition to the AEC, other key agencies or organizations include the Nuclear Safety Commission (NSC), the Ministry of International Trade and Industry (MITI), the Science and Technology Agency (STA), the Power Reactor and Nuclear Fuel Development Corporation (PNC), the Japan Atomic Energy Research Institute (JAERI) and the Japan Nuclear Fuel Services Company (JNFS). In addition, the Japanese nuclear utilities and engineering firms have formed two consortia (JAIF and FEPCO) to meet the industry's responsibilities, including spent fuel storage, reprocessing and waste management.

Radioactive wastes are managed in accordance with Japan's "Long Term Program for the Development and Utilization of Nuclear Energy," most recently updated in 1987. The AEC published reports in 1985 describing waste management plans, and STA issued a research and development program for HLW disposal in 1986. The PNC and JAERI, which are both under STA jurisdiction, share responsibilities for HLW management: PNC is the lead organization implementing the research and development program that will lead to site selection, while JAERI performs research in support of the government's safety evaluation of geological disposal, as well as research on advanced waste management technologies. The government has not yet determined which organization will make site selection decisions. Furthermore, it has not yet been decided whether MITI or STA will have the responsibility to license a HLW repository. Regulatory requirements for the HLW repository have not yet been established. No formal individual dose limits have been issued, but a dose limit of 5 mrem per year has been proposed following closure. The time period for complying with regulatory criteria has not yet been specified.

2.4 FEDERAL RADIATION COUNCIL GUIDANCE

The ICRP and the NCRP function as non-governmental 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 1950s 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 activities conducted by the Federal government (FRC60). 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..."

The Council's first recommendations concerning radiation protection guidance for Federal agencies were approved by the President in 1960. Based largely on the work and recommendations of the ICRP and NCRP, the guidance established occupational exposure limits, which differed only slightly from those recommended by NCRP and ICRP at the time (NCRP54, NCRP59).

- Whole body, head and trunk, active blood forming organs, gonads or lens of the eyes are not to exceed 3 rem in 13 weeks and the total accumulated dose is limited to 5 times the number of years beyond age 18, expressed as 5(N-18), where N is the current age.

- Skin of the whole body and thyroid are not to exceed 10 rem in 13 weeks or 30 rem per year.

- Hands, forearms, feet, and ankles are not to exceed 25 rem in 13 weeks or 75 rem per year.

- Bone is not to exceed 0.1 microgram of radium-226 or its biological equivalent.

- Any other organs are not to exceed 5 rem per 13 weeks or 15 rem per year.

The guidance also established exposure limits for members of the public. These were set at 0.5 rem per year for the whole body for an individual and an average gonadal dose of 5 rem in 30 years.

In addition to the formal exposure limits, the guidance also established as 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 exposures to a minimum was based on the possibility that there is no threshold for radiation. The linear non-threshold dose response relationship was assumed to place an upper limit on the estimate of radiation risk. However, the FRC explicitly recognized that it might also represent the actual level of risk. If so, then any radiation exposure carried some risk, and it was necessary to avoid all unproductive exposure and to keep all productive exposures as "far below this guide as practicable."

2.5 THE ENVIRONMENTAL PROTECTION AGENCY

In 1970, the functions of the Federal Radiation Council were transferred to the U.S. Environmental Protection Agency (EPA). Since then, the EPA has issued Federal guidance for the control of radiation hazards in underground mining (EPA71), for setting occupational exposure limits (EPA81), for occupational exposures of workers subject to federal regulations (EPA87), standards and technical information regarding radionuclide intake and air concentration limits, occupational radiation doses, biological parameters, and dose conversion factors (EPA88).

In addition to the statutory responsibility to provide Federal guidance on radiation protection, the EPA has various statutory responsibilities regarding regulation of exposure to radiation. The standards and the regulations that EPA has promulgated and proposed with respect to controlling radiation exposures and which are related to 40 CFR Part 191 are summarized here.

2.5.1 Atomic Energy Act

The Atomic Energy Act of 1954, as amended, and Reorganization Plan No. 3 granted the EPA the authority to establish generally applicable environmental standards for exposure to radionuclides AEA54, NI70). Pursuant to this authority, in 1977 the EPA issued standards limiting exposures from operations associated with the light-water reactor fuel cycle (EPA77). These standards, under 40 CFR Part 190, cover normal operations of the uranium fuel cycle, excluding mining and radioactive waste disposal. The standards limit the annual dose equivalent to any member of the public from all phases of the uranium fuel cycle (excluding radon and its daughters) to 25 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 standard also sets 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 standard cover all exposures resulting from radiation and radionuclide releases to air and water from operations of 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.

2.5.2 Safe Drinking Water Act

Under the authority of the Safe Drinking Water Act, the EPA issued interim regulations (40 CFR Part 141, Subpart B) covering the permissible levels of radium, gross alpha, man-made beta, and photon-emitting contaminants in community water supply systems (EPA76). The limits are expressed both in terms of average and maximum concentration limits (picocurie/liter) and annual doses to the whole body or organs. The allowable limit for radium-226 and radium-228, combined, is 5 picocuries per liter. For total gross alpha activity, including radium-226 but excluding radon and uranium, the maximum concentration limit is 15 picocuries per liter. The standard also specifies maximum concentration limits for strontium-90 and tritium. The dose limits chosen for man-made beta and photon emitters is 4 mrem/year to the whole body or organ dose for the most exposed individual. The supporting

information for the standard justifies the 4 mrem/year dose limit on the basis of existing man-made sources of contamination of drinking water (nuclear testing and nuclear power reactors) and compares it to the recommended population exposure of 170 mrem/year per capita. The conclusion reached is that when considering all exposure pathways, a 40-fold decrease is appropriate for this single pathway.

In 1991, the EPA issued a Notice of Proposed Rulemaking (NPRM) to update the 1976 interim regulations for radionuclide water pollution control (EPA91). The NPRM, under the Safe Drinking Water Act, proposed the establishment of Maximum Contaminant Level Goals (MCLGs) and Maximum Contaminant Levels (MCLs). The MCLGs and MCLs target radium-226, radium-228, natural uranium, radon, gross alpha, and gross beta, and photon emitters. As proposed, MCLGs are not enforceable health goals while MCLs are enforceable standards. The EPA concluded that radionuclide MCLGs should be set at zero to avert known or anticipated adverse health effects while providing an adequate margin of safety. In setting the MCLGs, the EPA also committed itself to evaluate the feasibility, costs, and availability of water treatment technologies, as well as other practical considerations. The proposed regulations provide the following MCLs: radium-226, 20pCi/l; radium-228, 20 pCi/l; radon-222, 300 pCi/l; uranium, 20 micro g/l; adjusted gross alpha, 15 pCi/l; and beta and photon emitters, 4 mrem ede/yr. In general, these limits yield doses of between 4 mrem/yr and 20 mrem/yr to individuals drinking the contaminated water.

2.5.3 Clean Air Act

Section 112 of the Clean Air Act (CAA) Amendments of 1977 (Public Law 95-95) directed the EPA Administrator to review all relevant information and to determine if airborne emissions of hazardous pollutants will cause or contribute to air pollution that may reasonably be expected to endanger public health. In December 1979, the EPA designated radionuclides as hazardous air pollutants under Section 112 of the Act (EPA79). In April 1983, the EPA proposed standards regulating radionuclide emissions from four source categories, one of which included DOE facilities. The rule established annual airborne emission limits for radioactive materials and specified that annual doses resulting from such emissions do not exceed 25 mrem to the whole body and 75 mrem to any critical organ to members of the general public. The EPA also proposed not to regulate several other categories of facilities, including high-level radioactive waste disposal facilities.

In October 1984, following a court order to promulgate final radionuclide emission standards or make a finding that radionuclides are not hazardous air pollutants, the EPA withdrew the proposed emission standards based on the findings that the control practices already in effect protected the public from radionuclide releases with an ample margin of safety. The Agency also affirmed its position not to regulate other categories of emission sources, including uranium fuel facilities and high-level radioactive wastes.

In December of 1984, a U.S. District Court found the EPA in contempt of its order and directed the EPA to either issue final radionuclide emission standards or make a finding that radionuclides are not hazardous air pollutants. The EPA complied with the Court order in 1985 by issuing standards for selected sources, National Emission Standards for Hazardous Air Pollutants (NESHAPs) (EPA85a, EPA85b). As a result of the decision in National

Resources Defense Council Inc. vs. EPA, which concluded EPA had improperly promulgated vinyl chloride regulations under Section 112 of the CAA by considering cost and technological feasibility, the Agency in November 1987 moved the Court for a voluntary remand of the NESHAPs for the four original categories of emission sources. The EPA agreed to re-examine all issues raised by the parties to the litigation. In December 1987, the Court granted the EPA's motion for voluntary remand and established a schedule to propose new regulatory standards within one year. The Court decision also defined the analytical process under which the EPA was to re-evaluate its standards. Two steps were identified: 1) first determine what is safe, based exclusively on health risk; and 2) adjust the level of safety downward to provide a greater or ample margin of safety.

In March 1989, the EPA issued a proposed rule for regulating radionuclide emissions under NESHAPs following the re-examination of the regulatory issues associated with the use of Section 112 (EPA89). The draft rule proposes four policy alternatives to control emissions and risks from 12 categories of sources, including DOE facilities. Each of the four approaches treats the acceptable risk criterion differently. The four approaches were:

- Case-by-Case Approach - Acceptable risk considers all health information, risk measures, potential biases, assumptions, and quality of the information. The preferred level of maximum individual lifetime risk must be 10^{-4} or less.

- Incidence-Based Approach - Based on the best estimate of the total incidence of fatal cancer. The proposed acceptable level of incidence must not exceed more than 1 fatal cancer per year per source category.

- Maximum Individual Risk Approach (10^{-4} or less) - Only parameter being considered is the best estimate of the maximum individual lifetime risk of fatal cancer. The acceptable maximum individual lifetime risk must not exceed 1×10^{-4}.

- Maximum Individual Risk Approach (10^{-6} or less) - This approach is similar to the previous one. The acceptable risk, however, must not exceed 1×10^{-6}.

The definition of the ample margin of safety is established separately after the safe level has been determined based solely on health risks. In reaching its final decision, the EPA must consider all health risk measures as well as technological feasibility, costs, uncertainties, economic impacts of control technologies, and any other relevant information. This decision process may also require the EPA to determine whether or not to require all technologically feasible controls which are affordable, no matter how small the risk reduction.

Based on the comments and the record developed in the rulemaking, EPA selected an approach announced in the notice on benzene standards published on September 14,1989 (54 FR 38044). Thus, in the first step of *Vinyl Chloride* inquiry, EPA has considered the extent of the estimated risk were an individual exposed to the maximum level of a pollutant for a lifetime. The EPA has generally presumed that if the risk to that individual is no higher

than approximately 1 in 10 thousand, that risk level is considered acceptable and EPA then considers the other health and risk factors to complete an overall judgement on acceptability. The presumptive level provides a benchmark for judging the acceptability of maximum individual risk, but does not constitute a rigid line for making that determination.

The rule concludes that there is no need to establish NESHAP standards for high-level waste disposal repositories since the releases and consequently the risks are very low and therefore constitute a margin of safety. The reason why the emissions and risks are so low is that radioactive materials received at such facilities are sealed in containers. Normal operations do not require additional processing or handling because spent fuels or high-level wastes are received and emplaced into the ground in their original containers. Operations at the disposal site which may require additional waste processing or repackaging, before the site is declared a disposal facility must comply with NESHAPs Subpart I.

2.5.4 Resource Conservation and Recovery Act

Some of the radioactive wastes covered by this rulemaking also contain hazardous wastes subject to the Resource Conservation and Recovery Act (RCRA); these materials are known as "mixed wastes." RCRA wastes are primarily governed by EPA regulations under 40 CFR Parts 260, 262, 263, 264, 265, 268, and 270. Section 6001 of RCRA explicitly subjects all Federal facilities and their activities to State and Federal regulations under RCRA. However, RCRA Section 1006(a) relieves facilities operating under the authority and control of the Atomic Energy Act of 1954 (AEA) from compliance with RCRA for conditions which could be inconsistent with the requirements of the AEA.

In 1987, the EPA formed the Mixed Energy Waste Study (MEWS) task force to evaluate DOE's proposed option to exempt mixed high-level radioactive wastes (HLW) and transuranic wastes (TRU) from RCRA, Subtitle C (EPA87). The MEWS task force concluded that, with some exceptions, current DOE management of mixed HLW/TRU wastes is equivalent to RCRA requirements. In other words, the management of these wastes would not change significantly if they were required to comply with RCRA Subtitle C requirements for hazardous wastes. The task force, however, noted that there were a few aspects which would not meet RCRA standards. For example, the task force noted that some waste forms do not fit "normal" management practices, particularly when dealing with submarine reactor components, classified TRU wastes, and TRU wastes unacceptable for disposal. For those aspects which do not meet RCRA standards, the task force gave the following examples: waste chemical analyses, ground-water monitoring, TRU waste retrievability, disposal of classified TRU wastes, and self-inspection. Some States were also concerned about the DOE self-regulating its HLW/TRU waste disposal activities under the proposed option, but were willing to consider case-by-case variances with specific requirements.

Since July 1986, the Agency has required states to obtain mixed waste authorization as part of their RCRA programs. Procedures for considering disposal of mixed wastes are now being developed and the Office of Solid Waste is issuing authorizations for States to regulate such types of mixed wastes. The EPA's Office of Radiation Programs and Office of Solid Waste are maintaining cognizance of these developments with the State programs.

2.5.5 Waste Isolation Pilot Plant Land Withdrawal Act

Under the Waste Isolation Pilot Plant Land Withdrawal Act, besides setting the terms and conditions for the Department of Energy's (DOE) activities at the WIPP, the new law contains numerous provisions pertinent to the Agency's role in overseeing DOE's activities at the WIPP and to the Agency's handling of the 40 CFR Part 191 disposal standards. For instance, the new law reinstates all of the disposal standards issued by the Agency in 1985 except the three aspects of the individual and ground-water protection requirements which were the subject of the court remand. It, then, puts the Agency on a schedule for issuing final disposal standards. The new law provides an extensive role for EPA in reviewing and approving various phases of DOE activities at the WIPP and requires EPA to certify whether the WIPP repository will meet the final 40 CFR Part 191 standards.

2.6 NUCLEAR REGULATORY COMMISSION

Under the authority of the Atomic Energy Act of 1954, as amended, the U.S. Nuclear Regulatory Commission (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 (AEA54). The Federal guidance on radiation protection applies directly to the NRC. Therefore, the NRC must assure that none of the operations of its licensees expose an individual of the public to more than 0.5 rem/year from all pathways.

The dose limits imposed by the EPA's standards for uranium fuel-cycle facilities (40 CFR Part 190) apply to the fuel-cycle facilities licensed by the NRC (See Section 2.5 for a summary of EPA regulations). These facilities are prohibited from releasing radioactive effluents in amounts that would result in doses greater than the 25 mrem/year limit imposed by that standard. Also, NRC facilities are required to operate in accordance with the requirements of the Clean Air Act (40 CFR Part 61), which limits radionuclide emissions to air.

The NRC exercises its statutory authority over licensees 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 activities.

2.6.1 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 licensees 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 maintained as low as is reasonably achievable (ALARA) by imposing design criteria for effluent control systems and equipment. After a license has been issued, fuel-cycle licensees must monitor

their emissions and set up an environmental monitoring program to assure that the design criteria and license conditions have been met.

2.6.2 Radioactive Waste Disposal Licenses

The authority for the NRC to regulate high-level waste disposal originates from Public Law 97-425, also known as the "Nuclear Waste Policy Act of 1982." The Act requires the NRC to promulgate regulations governing 1) construction authorization for a repository, 2) license to receive and dispose of wastes in the repository, and 3) authorization for repository closure (NWPA83).

This Act also requires the EPA to promulgate, "...generally applicable standards for the protection of the general environment from off-site releases of radioactive material in repositories..." The Act also requires that the NRC regulations be consistent with the EPA standards. See Sections 1.2 and 2.5 for a detailed discussion of the EPA's role and responsibilities.

The NRC regulations governing deep geologic disposal are contained in the Code of Federal Regulations, Title 10, Part 60, titled, "Disposal of High-level Radioactive Wastes in Geologic Repositories." These regulations are summarized below. In addition, the NRC certifies (under 10 CFR Part 71) packaging for the transportation of spent nuclear fuel, high-level and transuranic radioactive wastes.

Similar to the licensing of power reactors, 10 CFR Part 60 requires the waste repository operator (DOE) to submit a safety analysis report (SAR) and an Environmental Impact Statement (EIS) in order to obtain a license to construct a repository (NRC81, NRC85). The EIS must meet the requirements of 10 CFR Part 51, (under NEPA) "Environmental Protection Requirements for Domestic Licensing and Related Regulatory Functions" (NEPA70).

The SAR is required to contain a description of the characteristics of the proposed repository site, including fractures, geomechanics, geochemistry and thermal loading effects. It must also include a description of the natural resources of the site, and an assessment of the waste isolation properties of the proposed site. A program of site characterization field work is required to support the preparation of the SAR. The general plan for this program of characterization is presented in a Site Characterization Plan (SCP). This plan contains the description of the studies to be conducted, their sequencing and possible interferences, and the impacts of the studies on the ability of the site to isolate and contain the waste. Before beginning site characterization, the SCP receives extensive reviews by the NRC, the host state, and other interested parties. Progress during site characterization and any changes to the plans for site characterization are reported in semiannual progress reports which are also reviewed by the NRC and other interested parties. The SAR is then prepared using the information developed during site characterization.

Upon receipt of the SAR, the NRC will conduct a safety review. The planned repository will be evaluated against the technical criteria specified in the NRC regulations in 10 CFR Part 60. If the NRC determines from this evaluation that there is reasonable

assurance that the waste can be received, possessed, and disposed of safely, that the common defense and security can be protected, and that environmental values are protected, an authorization will be given to the DOE to begin construction of the repository.

After construction has been completed, the DOE will update the SAR and the environmental report and this information will be reviewed by the NRC to determine if a license to receive, possess, and dispose of waste can be granted. At this stage, the NRC will confirm that construction has been completed in conformity with the license application, and that the repository poses no unreasonable risk to public health and safety. Likewise, at the end of the operating period, the license application and environmental report are updated and an application to amend the license application is submitted by the DOE. This application and the associated updated information are reviewed by the NRC to determine if the repository may be permanently closed.

- Technical Criteria

At each stage of the licensing process, the SAR is reviewed to determine if the technical criteria specified in Subpart E of the NRC regulations are satisfied. These technical criteria include performance objectives and other criteria (e.g., requirements on land ownership and control, siting criteria, and design criteria) intended to ensure that the performance objectives are met. The performance objectives are set to ensure radiological safety and waste retrievability during the operating period, waste isolation and containment by the overall system after permanent closure, and adequate performance of particular barriers after permanent closure. These performance objectives require that radiation exposures, radiation levels, and releases of radioactive materials conform to the applicable environmental standards established by the EPA. Therefore, demonstration of compliance with these standards will be an integral part of DOE's license application. The NRC regulations also specify requirements for monitoring during the institutional control period (NRC83) and provisions for the retrievability of any emplaced wastes. Other requirements deal with land ownership and waste package design criteria.

The performance objective for protection against radiation exposures and releases during the operating period requires that the repository be designed so that radiation exposures, radiation levels, and releases of radioactive materials to unrestricted areas meet the applicable environmental standards; these standards are specified in Subpart A of 40 CFR Part 191. The performance objective for waste isolation containment by the overall geologic repository system requires that releases to the accessible environment following permanent closure conform to environmental standards that apply to this period; these standards in this case are specified in Subpart B of 40 CFR Part 191.

- Waste Isolation Pilot Plant (WIPP)

The WIPP project is a DOE facility located near Carlsbad, NM for the disposal of defense-produced transuranic wastes. The NRC has no regulatory authority over the WIPP project. However, the certification of the containers used to ship the TRU wastes from DOE facilities to the WIPP site is under the authority of the NRC as specified in 10 CFR Part 71. Two types of shipping containers have been designed, one for contact-handled wastes and one

for remote-handled wastes. Both designs are currently being reviewed and evaluated by the NRC.

2.6.3 Center for Nuclear Waste Regulatory Analysis

In the fall of 1987, the NRC created the Center for Nuclear Waste Regulatory Analysis to support repository licensing activities. Traditionally, the NRC has relied on the national laboratories for this support. Since the laboratories are largely under DOE control, their involvement in repository licensing could present a potential conflict of interest. The Center is operated by the Southwest Research Institute and is located in San Antonio, Texas.

In supporting the NRC, the Center is charged with providing long-term continuity in technical assistance and research. Also, it is to provide central capabilities for integrating all aspects of the high-level waste licensing program. Current projects at the Center include identifying priority areas of the site characterization plan (SCP) for NRC staff review, analyzing technical uncertainties pertaining to repository siting, recommending candidate areas for additional rulemaking, and assessing the importance of various regulatory requirements.

The Center is currently working on a number of special reports. These include a long-range plan, an open-item tracking system, and an issue resolution monitoring report. Besides these special reports, the Center is also preparing a number of format and content guides, and standard review plans related to the license application.

2.6.4 Other Activities

The current NRC repository licensing program is divided into two areas - proactive activities and reactive activities. These are described briefly below.

Proactive activities are those that do not depend on DOE action. These include developing and reviewing regulatory requirements and guidance to identify and resolve uncertainties. Regulatory uncertainties exist where regulatory requirements are ambiguous and could be subject to various interpretations. Technical uncertainties are related to demonstrating compliance with a particular regulation. These are currently being addressed so that the NRC can meet the three-year license review schedule mandated by Public Law 97-245 (NWPA83).

In another area, the NRC staff is developing and implementing performance assessment models using Yucca Mountain site data. This will help develop technical assessment capability, as well as identify areas of regulatory and technical uncertainty.

These activities have produced licensing review plans in anticipation of the DOE submittals. They include the SCP Review Plan, Study Plan Review Plan, and Quality Assurance Review Plan. The License Application Review Plan is still in preparation.

Other proactive activities include the evaluation of progress on actions required by NWPA. This ongoing evaluation is documented in the Quarterly Progress Reports to the Commission on the High-Level Radioactive Waste Management Program. This evaluation

complements other actions and more specific reviews and consultations by taking a broad view of progress and identifying fundamental concerns.

The reactive part of the NRC program consists of pre-licensing reviews that follow DOE's sequence and schedule of activities. To date, this includes reviews of quality assurance programs for DOE and DOE contractors. Quality assurance issues need to be resolved before significant data collection activities are performed at the Yucca Mountain site.

The next major activity will involve the NRC's review of the SCP and will focus on DOE's strategies, assumptions, and programs. For the more detailed Study Plans, prepared by the DOE, the NRC will conduct a completeness review on each. However, a detailed review will be made on only a sample (about 20%) of the hundred or so Study Plans. During site characterization, the NRC will conduct on-site reviews of selected testing activities and selected data.

As site characterization activities proceed, DOE's semiannual progress reports on the site characterization program will be reviewed by the NRC. These reviews will focus on the resolution of previously identified concerns and will evaluate new information about the site and repository design. In addition, the NRC will review selected DOE study reports and position papers that document the detailed results of work performed to date. The NRC will review DOE's topical reports and issue resolution reports, which summarize the site characterization work for specific licensing topics. These will be used to evaluate compliance with NRC regulations.

All concerns identified by the NRC will be tracked by the staff as open items. The tracking system, presently being implemented, will focus on root causes and DOE's progress toward resolution. The system will also provide and maintain a licensing record of all NRC and DOE actions related to resolving specific issues.

2.7 DEPARTMENT OF ENERGY

The U.S. Department of Energy (DOE) operates facilities for the enrichment of nuclear fuels for commercial and defense reactors, the production and testing of nuclear weapons, the management and disposal of radioactive wastes generated in national defense activities, and research and development, including several national laboratories. In addition, the DOE is conducting several remedial action programs, such as the program for the management of uranium mill tailings and the cleanup of sites formerly used for nuclear activities. These facilities and activities are not licensed by the NRC. However, to protect public health and the environment, the DOE has implemented orders and procedures that are consistent with NRC regulations under 10 CFR Part 20 (NRC60), standards promulgated by the EPA, and other applicable Federal regulations and guidelines.

The DOE is also responsible for the disposal of spent nuclear fuel and high-level radioactive wastes from defense activities and the generation of electricity by commercial nuclear reactors. The facilities, developed by the DOE for the management and disposal of these wastes, will eventually be licensed by the NRC.

2.7.1 DOE Programs for the Environment, Health, and Safety

The DOE is responsible for operating its facilities in a manner that is safe and environmentally sound, as stated in DOE Order 5400.1 (DOE88a). To this end, it has issued a number of orders specifying procedures and standards. (See Table 2.7-1) It should be noted that many of these DOE procedures and standards are currently being reviewed and revised to conform with NRC and EPA regulations and standards (DOE89). Mandatory standards for the protection of public health and the environment are established by DOE Order 5480.4 (DOE84a). These standards apply to all DOE and DOE contractor operations during facility design, construction, operation, modification, and decommissioning. The order mandates compliance with the standards promulgated by the Occupational Health and Services Administration in 29 CFR Parts 1910, 1915, 1918, 1926, and 1928 (DOL74). DOE Order 5480.1B (DOE86a) establishes procedures for the preparation and review of safety analyses for DOE operations, including the identification and control of hazards and risk assessments. DOE Order 5400.2A (DOE87) establishes specific requirements for the coordination of DOE and contractor activities to ensure the timely resolution of significant environmental compliance issues.

DOE Order 5820.2A (DOE88b) establishes policies and guidelines by which the DOE assures that all DOE facilities, including surplus facilities, involving the use of radioactive or mixed waste or waste by-products are operated in a manner that protects the health and safety of the public and the environment. The DOE is developing specific orders for the management of hazardous and radioactive mixed wastes and for environmental surveillance of radioactive effluents.

Under 5482.1B (DOE86b), the DOE established a program for environmental quality assurance; its objective is to ascertain that the DOE's environmental, safety, and health policies are properly interpreted and implemented. The DOE also complies with the national standards established jointly by the American National Standards Institute and the American Society of Mechanical Engineers (ANSI86) for quality assurance in nuclear facilities.

Under the Atomic Energy Act of 1954, as amended (AEA54), the DOE is responsible for keeping radionuclide emissions at its facilities as low as is reasonably achievable (ALARA). Under the authority of the Clean Air Act, the EPA has issued, in 40 CFR Part 61, standards (EPA89a) that limit airborne radionuclide emissions from DOE facilities to any member of the public in any year an effective dose equivalent of 10 mrem per year. The current emission levels achieved by emission control technologies and practices at DOE facilities are within these limits. In order to comply with these standards and the maximum permissible concentrations established by the National Council on Radiation Protection and Measurements for radioactive material in air and water (NCRP54, NCRP59, NCRP71), the DOE has issued Orders 5400.3 (DOE89) and 5480.11 (DOE88c) to protect the general population and workers at DOE facilities, respectively, from radioactivity in air and water. These orders set a limit of 10 millirem per year for the effective dose equivalent.

2.7.2 Compliance with Federal Regulations

The DOE has developed orders to ensure the compliance of its facilities and programs with the applicable Federal environmental regulations. (see Table 2.7-2) DOE Order 5440.1C (DOE85) establishes procedures for implementing the requirements of the National Environmental Policy Act of 1970 (NEPA70). New facilities and modifications to existing facilities are subject to extensive design criteria reviews and require the preparation of environmental impact statements. In existing facilities, the DOE has implemented a systematic program for reducing the releases of gaseous and liquid radionuclides to the environment.

In addition, the DOE is subject to the Resource Conservation and Recovery Act (RCRA), which requires that all radioactive wastes containing RCRA-hazardous materials are subject to regulations under both the RCRA and the Atomic Energy Act of 1974 (AEA54). The DOE is also preparing an order for demonstrating compliance with the requirements of the Comprehensive Environmental Response, Compensation, and Liability Act.

2.8 DEPARTMENT OF TRANSPORTATION

The U.S. Department of Transportation (DOT) has statutory responsibility for regulating shipments of radioactive materials, including radioactive wastes. DOT its regulatory activities are coordinated with those of the NRC. Its authority includes the responsibility to protect the public from exposure to radioactive materials while they are in transit. The DOT has implemented its authority by specifying performance standards for shipping containers, setting maximum exposure rates for any package containing radioactive materials, and managing the routing of radioactive materials shipments to avoid densely populated areas.

The regulatory authority of the DOT derives from several laws. For the transportation of radioactive waste, the primary laws are the Hazardous Materials Transportation Act of 1974 (HMTA75) and the Federal Railroad Safety Act (FRSA70). These laws authorize the Secretary of Transportation to issue regulations for the safe transportation of hazardous materials, including radioactive materials, and to define the specific relationship between the DOT and state and local authorities.

The regulations promulgated by the DOT are contained in Title 49 of the Code of Federal Regulations. Those directly applicable to the transportation of radioactive wastes are mainly included in 49 CFR Parts 171-177 (DOT83). They define the types of materials that are regulated; specify the DOT's enforcement authority, including potential sanctions; and state specific requirements for materials handling, the marking and labeling of packages, the placarding of shipments, the routing of shipments, and the training of drivers.

Specific provisions cover carriage by rail (49 CFR Part 174), carriage by vessel (49 CFR Part 176), and carriage by public highway (49 CFR Part 177). Transportation by barge is regulated by the standards promulgated under 49 CFR Chapter 2. In addition to the regulations established under the HMTA, the DOT's Federal Highway Administration has established, in 49 CFR Part 300, general standards for highway transportation.

2.9 OFFICE OF THE NUCLEAR WASTE NEGOTIATOR

The Office of the Nuclear Waste Negotiator, created by the 1987 Amendments of the 1982 Nuclear Waste Policy Act, is an independent Federal entity. The Nuclear Waste Negotiator is appointed by the President. The mission of the Negotiator is to seek a dialogue with the Governor of every State and the leaders of all federally recognized Indian tribes to explore upon what terms and conditions, if any, they might willingly host a facility for the permanent or temporary storage of nuclear waste.

The Negotiator is authorized to negotiate with the Governor or tribal leader of the interested potential host jurisdictions to determine the terms and conditions under which they would agree to host either a Monitored Retrievable Storage facility or a repository. Preparation of an environmental assessment and consultation with Federal agencies concerning a site's technical suitability are required when a negotiation begins. The negotiation is to result in a written agreement that will be submitted to Congress and enacted into law before it becomes effective.

2.10 STATE AGENCIES

States have played an important role in protecting the public from hazards associated with ionizing radiation. Twenty-nine States have assumed the NRC's inspection, enforcement, and licensing responsibilities for users of nuclear source and by-product materials and users of small quantities of special nuclear material. These "NRC Agreement States," are bound by formal agreements to adopt requirements consistent with those imposed by the NRC.

2.10.1 Federal Provisions for State Participation

State and public participation in the planning and development of geologic disposal is essential to promote public confidence in the safety of geologic repositories for spent nuclear fuel and high-level radioactive wastes. The Congress has provided for public participation in the NWPA and in the Nuclear Waste Policy Act Amendments Act of 1987 (Amendments Act) (NWPA87). Specific provisions of the NWPA, as amended, govern the notification of potentially affected States and Indian Tribes (Section 116(a)). Other provisions require the Secretary of Energy to hold hearings in the vicinity of the repository before selection takes place (Section 114(a)(10)).

2.10.2 Programs in the State of New Mexico

The New Mexico Environmental Evaluation Group (EEG) was created in 1978 to conduct independent scientific reviews and to evaluate the potential impact on public health and environment from the Waste Isolation Pilot Plant (WIPP) project (EEG88, EEG89, NEI89). The WIPP facility is a repository designed to demonstrate the disposal of national defense-related TRU wastes. The EEG was formed in response to the authorizing legislation for the Waste Isolation Pilot Plant, since Congress specifically excluded DOE from the licensing requirements of the Nuclear Regulatory Commission for the WIPP facility.

The EEG is a full-time, multi-disciplinary group funded entirely by the U.S. Department of Energy for the State of New Mexico. The EEG is the only independent oversight group monitoring the WIPP site and its activities; however, it does not have any regulatory authority on the WIPP facility and it can only recommend actions to DOE for its consideration. In spite of these constraints, the EEG has been influential in making recommendations which led to the relocation of the repository, redesign of the waste shipping containers, consideration and evaluation of transportation issues, and monitoring of WIPP site activities. The EEG has also organized several technical forums to evaluate technical issues and consider alternate approaches.

In October 1988, the EEG was assigned to the New Mexico Institute of Mining and Technology to provide a better climate for technical independence. Up to that point, the EEG had been attached to the Environmental Improvement Division, a component of the New Mexico Health and Environment Department. The EEG Director is appointed by and reports directly to the President of the Institute and the Director appoints all other EEG staff. Scientific disciplines represented in the EEG group include engineering, geology, hydrology, health physics, environmental monitoring, radiation protection, radiological health, and quality assurance. The EEG has offices both in Carlsbad and Albuquerque, NM.

Since 1978, the EEG staff have conducted several evaluations to assess the suitability of the WIPP site, including identifying potential environmental problems, suitability of facility design, suitability of the proposed waste shipping containers, waste form characterizations and other related technical topics. EEG responsibilities also include the conduct of an environmental radiation surveillance program to establish a background base line for naturally occurring radioactivity present in air, water, and soils for both on and off-site locations and within the surrounding communities. Both EEG and DOE have independent monitoring stations located in the exhaust stacks of the WIPP facility to characterize and document airborne emissions.

EEG disseminates its findings and analyses by publishing reports, articles in professional journals, presentations to scientific society meetings, public hearings, and by issuing pamphlets and brochures to the public. The EEG has published over 40 major reports since 1978. It also distributes the results of its analyses to DOE, the Governor's Office, the New Mexico Legislature, Congress, the scientific community, and general public. Typically, EEG reports have addressed the following technical issues: site characterization; performance assessment; facility operations; monitoring; and transportation. Several of these reports present independent evaluations and analyses of DOE studies, models, assumptions, and plans.

2.11 INDIAN TRIBES

Indian Tribes have a unique sovereign status in U.S. law, and this status was recognized by the NWPA and the Amendments Act. This government-to-government relationship between the Federal government and Indian Tribes obligates the DOE to interact directly and specifically with Indian Tribes in areas where repository or MRS siting activities will occur. The NWPA, as amended, under Section 2(2), defines:

"...affected tribe as (1) any Indian Tribe within whose reservation boundaries an MRS, test and evaluation facility, or a repository for high-level wastes or spent fuel is proposed to be located, or (2) whose federally defined possessory or usage rights to other lands outside of the reservation's boundaries arising out of congressionally ratified treaties may be substantially and adversely affected by the locating of such a facility. Provided, that the Secretary of the Interior finds, upon the petition of the appropriate governmental officials of the tribe, that such effects are both substantial and adverse to the tribe..."

As noted above, many of the sections of the NWPA, as amended, that delineate the participation activities and rights of affected States in repository and MRS siting decisions also apply to affected Indian Tribes. The means to disapprove of the site selection and designation process is given in Section 118(a). An affected Indian Tribe is also eligible to receive the same grants, financial and technical assistance, and payments equal to taxes for which a State is eligible under Section 116(c).

Since the passage of the Amendments Act, no Indian Tribes have been designated as affected tribes. However, the DOE is cooperating with Indian Tribes that may be located near the transportation routes or the WIPP facility. The DOE informs Indian Tribes of the status of the program through a cooperative agreement with the National Congress of American Indians. Finally, to ensure compliance with the American Indian Religious Freedom Act, the National Historic Preservation Act and related statutes, and the National Environmental Policy Act, the DOE will consult with Indian Tribes that have current or traditional religious or cultural ties to the Yucca Mountain site (DOE88g).

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Chapter 3: QUANTITIES, SOURCES, AND CHARACTERISTICS OF SPENT NUCLEAR FUEL AND HIGH-LEVEL AND TRANSURANIC WASTES

3.1 INTRODUCTION

This chapter presents current inventories of commercial spent fuels, commercial and U.S. Department of Energy (DOE) high-level radioactive wastes, and DOE transuranic wastes. Although spent fuel and high-level radioactive wastes remain covered under 40 CFR Part 191, these standards are not applicable to the characterization, licensing, construction, operation, or closure of any site required to be characterized under section 113(a) of Public Law 97-425. The inventories were compiled from the most reliable Federal government information sources publicly available (BUR82, DOE89a, DOE88b, EIA88, JAN83, LIT79, STO79). Estimates of generated wastes and spent fuel to the year 2020, based on DOE information and projected U.S. commercial nuclear power growth, are also presented. The spent fuel and wastes are characterized according to their volumes (or quantities) and their nuclear, physical, and chemical properties.

The wastes are broadly characterized as high-level waste (HLW) and transuranic (TRU) waste. In addition, an inventory of commercial reactor spent fuel may also require an expansion of current storage or the construction of additional facilities for interim storage, pending the availability of commercial reprocessing facilities, permanent disposal facilities, or monitored retrievable storage.

Both spent fuel and high-level radioactive wastes from reprocessing are intensely radioactive and generate substantial quantities of heat. The radioactivity and heat production continue for long periods of time because the wastes contain a number of long-lived radionuclides. The transuranic elements in particular have long radiological half-lives, generate very little heat relative to spent fuel, and present a potential health hazard for tens of thousands of years. Transuranic elements are nuclides with an atomic number greater than 92 and include plutonium, curium, americium, and neptunium.

3.2 SPENT NUCLEAR FUEL

In this standard, spent nuclear fuel is defined as fuel that has been withdrawn from a nuclear reactor following irradiation and whose constituent elements have not been separated by reprocessing (EPA85). The generators of spent nuclear fuel are: 1) commercial light-water reactors (LWRs), 2) government sponsored research and demonstration programs, universities, and industry, 3) experimental reactors, i.e., liquid metal fast breeder reactor (LMFBR) and high temperature gas cooled reactors (HTGR), 4) U.S. Government nuclear weapons production reactors, and 5) Department of Defense (DOD) reactors.

Approximately 96 percent of the spent fuel from commercial power reactors is stored in pools at reactor sites. The rest is stored at the West Valley Demonstration Project (WVDP) in New York, and at the Midwest Fuel Recovery Plant (MFRP) at Morris, Illinois. The WVDP facility is currently being decommissioned. All utility-owned spent fuel assemblies previously stored there have been returned to the utilities, and the fuel remaining

is DOE-owned material. Spent fuels from one-of-a-kind reactors are currently stored at Hanford (HANF) and the Idaho National Engineering Laboratory (INEL). Spent fuel from the Fort St. Vrain HTGR is stored at the Idaho Chemical Processing Plant (ICPP) at INEL. Other types of special spent fuel are stored at the Savannah River Plant (SRP) and INEL. These fuels are government-owned and are not scheduled for reprocessing in support of DOE/defense activities.

The fuel currently used in commercial light-water reactors consists of a mixture of uranium-238 and uranium-235 dioxides encased in zirconium alloy (zircaloy) or stainless steel tubes. During reactor operation, fission of the uranium-235 produces energy, neutrons, and radioactive materials. The neutrons produce further fission reactions and thus sustain the chain reaction. The neutrons also convert some of the uranium-238 into plutonium-239, which can fission as uranium-235 does. In time, the fissile uranium-235, which originally constituted some 3 to 4 percent of the enriched fuel, is depleted to such a low level that power production becomes inefficient. Once this occurs, the fuel bundles are deemed "spent" and are removed from the reactor. Typical removal rate is one-third of the fuel, or 30 metric tons per year and per reactor. Reprocessing of commercial spent fuel has been proposed to recover the unfissioned uranium-235 and the plutonium for reuse as a fuel resource, but such reprocessing is not currently taking place.

The radioactive materials associated with spent fuel fall into three categories - a) fission products, b) actinide elements, and c) activation products. Typically, fresh spent fuel contains more than 100 radionuclides as fission products. Fission products are of particular importance, because of the quantities produced, their radiological half-lives, their heat production, and their potential biological hazard. Such fission products include: strontium-90; technetium-99; iodine-129 and -131; the cesium isotopes, such as cesium-134, -135, and -137; tin-126; and krypton-85 and other noble gases.

The activation products include tritium (hydrogen-3), carbon-14, and other radioactive isotopes created by neutron activation of fuel assembly materials and impurities in cooling water or in the spent fuel. The actinides consist of uranium isotopes and transuranic elements (i.e., isotopes with an atomic number greater than 92, including plutonium, curium, americium, and neptunium formed by neutron capture, and their decay products). The exact composition of radionuclides in any given spent fuel sample depends on the reactor type, the initial fuel composition, the length of time the fuel was irradiated, and the elapsed time since its removal from the reactor core.

3.2.1 Spent Fuel Inventory and Projection

By the end of 1988, there were 17,607 metric tons (MT) of spent fuel in inventory from commercial reactor operation (DOE89a). Of this amount, 27 MT are stored at the WVDP facility and 668 MT are stored at the MFRP. The remainder is stored at each reactor site. The historical and projected quantities of the spent fuel inventory and accumulated radioactivity are given in Table 3.2-1.

The radioactivity in spent fuel depends primarily on its age. As the spent fuel ages, many of the short-lived fission products decay away. Calculations of waste activities 10

years after removal from the reactor, with consideration being given only to radionuclides (fission products and heavy elements) with half-lives greater than 20 years, show that the 1988 activity of the 17,607 MT of spent fuel corresponds to about 18.6 billion curies.

The projected inventory of spent fuel (Table 3.2-1) was based on DOE's lower reference case projections for installed nuclear capacities given in Table 3.2-2 and for the burnup rate and duration assumed for those reactors. DOE's projection assumes 15 reactors now in the construction pipeline will become operational at the year 2005 (EIA88). This is in addition to the 107 reactors already operating by the end of 1988. The DOE also assumes that two reactors currently on order will eventually be built and become commercially operable by 2005. The DOE's inventory projections assume the startup of a MRS Facility and a Commercial Repository in the year 2003.

It is estimated that by the year 2020 the total nuclear electrical capacity will reach 122.7 gigawatts (DOE89a, EIA89). The position that new reactor orders will resume assumes that future changes, driven by political, environmental, and economic issues as well as the decreasing availability of oil, will present nuclear power as a better alternative. For example, it is assumed that clean air standards will become stricter principally in response to the acid rain issue and the uncertainty about the greenhouse effect associated with the build up of atmospheric CO_2 resulting from the combustion of fossil fuels. These factors could enhance the choice of nuclear over fossil-fueled (coal and oil) power plants.

These projections do not include potential contributions from spent naval propulsion reactor fuel. Although the current plans do not include such a possibility, modifications to the nation's strategy for nuclear weapons may result in the availability of fuel-grade material without reprocessing. In this case, disposal of spent propulsion reactor fuel may be considered for the repository.

3.3 HIGH-LEVEL RADIOACTIVE WASTES

The EPA standards (40 CFR Part 191) define high-level radioactive wastes as the highly radioactive materials resulting from the reprocessing of spent nuclear fuel, including liquid wastes produced directly in reprocessing, and any solid material derived from such liquid wastes (EPA85). This definition is the same as that given in the NWPA (NWPA83). NRC regulations require that commercial high-level radioactive wastes generated in the future be converted to a solid form within 5 years (NRC88).

The fission products, actinides, and neutron-activated products of particular importance are the same for HLW as those listed for the spent fuel assemblies (DOE89a, DOE88b, LIT79, STO79).

Weapons program reactors are operated mainly to produce plutonium. Reprocessing to recover the plutonium is an integral part of the weapons program operations. Naval propulsion reactor fuel elements may also be reprocessed to recover the highly enriched uranium that still remains after use.

High-level radioactive waste that is generated by the reprocessing of spent reactor fuel and targets would contain more than 99 percent of the non-volatile fission products produced in the fuel or targets during reactor operation. It generally would contain about 0.5 percent of the uranium and plutonium originally present in the fuel. Most of the current HLW inventory, which is the result of DOE national defense activities, is stored at the Savannah River Plant (SRP), the ICPP at the Idaho National Engineering Laboratory, and the Hanford sites. A small amount of commercial HLW was generated at the Nuclear Fuel Services Plant at West Valley, New York, from 1966 to 1972. That facility is now referred to as the West Valley Demonstration Project (WVDP) and is under the responsibility of the DOE Idaho Operations, West Valley Project Office. These wastes have been through one or more treatment steps (e.g., neutralization, precipitation, decantation, evaporation). Their total volumes depend greatly on the steps to which they have been subjected during the various processing stages. Such wastes must be incorporated into a stable solid medium (e.g., glass) for final disposal, and the volumes of these interim wastes will be greatly reduced once this has been accomplished.

The DOE defense HLW at INEL results from reprocessing nuclear fuels from naval propulsion reactors and special research and test reactors. The bulk of this waste, which is acidic, has been converted to a stable, granular solid (calcine). At SRP and HANF, the acidic liquid wastes from reprocessing defense reactor fuel is or has been made alkaline by the addition of caustic soda and stored in tanks. During storage, these alkaline wastes separate into three phases: liquid, sludge, and salt cake. The relative proportions of liquid and salt cake depend on how much water is removed by waste treatment evaporators during waste management operations. The condensed water is currently sent to seepage basins and holding ponds.

The commercial HLW at West Valley consists of both alkaline and acidic wastes. The alkaline wastes were generated by reprocessing commercial power reactor fuels and some Hanford N-Reactor fuels, whereas acidic wastes were generated by reprocessing a small amount of commercial fuel containing thorium.

The inventories of HLW in storage at the end of 1987 are listed in Table 3.3-1 (by volume) and Table 3.3-2 (by radioactivity). Projected volume and radioactivity data for DOE defense, West Valley, and future commercial HLW are given in Table 3.3-3.

3.3.1 HLW Inventories at SRP

Approximately 128,000 m^3 of alkaline HLW that has accumulated at the SRP over the past three decades is currently stored underground in high-integrity, double-walled, carbon-steel tanks. The current inventories (Tables 3.3-1 and 3.3-2) consist of alkaline liquid, sludge, and salt cake that were generated primarily by the reprocessing of nuclear fuels and targets from plutonium production reactors. As generated, most of the waste is acid. The sludge is formed after treatment with caustic agents. Salt cake results when the supernatant liquor is concentrated in waste treatment evaporators.

3.3.2 HLW Inventories at INEL

About 11,000 m^3 of HLW is currently stored at the Idaho Chemical Processing Plant (ICPP) at INEL; this volume consists of 7,600 m^3 of liquid wastes and 3,400 m^3 of calcine materials (Tables 3.3-1 and 3.3-2). Liquid HLW is generated at ICPP primarily by the reprocessing of spent fuel from the national defense (naval propulsion nuclear reactors) and reactor testing programs; a small amount is also generated by reprocessing fuel from non-defense research reactors. This acidic waste is stored in large, doubly contained, underground, stainless steel tanks. The waste is then converted to a calcine, after which it is stored in retrievable stainless steel bins housed in reinforced concrete vaults.

3.3.3 HLW Inventories at HANF

The alkaline HLW (243,500 m^3) located at HANF is stored in four phases: liquid, sludge, slurry, and salt cake. This waste, which has been accumulating since 1944, was generated by reprocessing production reactor fuel for the recovery of plutonium, uranium, and neptunium for defense and other Federal programs. Fuel reprocessing was suspended from 1972 until November 1983. Most of the high-heat-emitting isotopes (Sr-90 and Cs-137, and their decay products) have been removed from the old wastes, converted to solids as strontium fluoride and cesium chloride, placed in double-walled capsules, and stored in water basins. The liquid, sludge, slurry, and salt cake wastes (Tables 3.3-1 and 3.3-2) are stored in underground concrete tanks with carbon steel liners.

3.3.4 HLW Inventories at WVDP

About 2,116 m^3 of HLW is stored at the WVDP Facility and consists of 2,066 m^3 of alkaline wastes and only 50 m^3 of acid wastes. The alkaline wastes were generated by reprocessing commercial and a few Hanford N-Reactor spent fuel elements. Initially, all of the wastes were highly acid; treatment with excess sodium hydroxide led to the formation of an alkaline sludge. The acid wastes now in storage were generated by reprocessing a small batch of thorium-uranium fuel from the Indian Point-1 Reactor. The alkaline wastes are stored in an underground carbon-steel tank, and the acid wastes are stored in an underground stainless steel tank. Reprocessing at the WVDP plant was discontinued in 1972, and no additional HLW has been generated since. The current inventories of HLW at WVDP are presented in Tables 3.3-1 and 3.3-2.

3.3.5 Waste Characterization

It is difficult to characterize HLW generically at any site because such wastes have been generated by several different processes and several methods have been used to condition the wastes for storage (e.g., evaporation and precipitation). In some instances, several different wastes have been blended. Nonetheless, representative chemical and radionuclide compositions of the HLW at SRP, ICPP, HANF, and WVDP can be found in other sources (DOE88a, DOE88b).

As with spent fuel, HLW radioactivity levels depend on age. To bring the level of radioactivity into perspective, the activity of fission products and heavy element radionuclides

with half-lives exceeding 20 years in existing HLW is estimated to be about 700 million curies.

3.3.6 HLW Projections

Projections for HLW (volume and radioactivity) by source are presented in Table 3.3-3. The projections for SRP are based on the assumption that three reactors will be operating through the year 2000. After the year 2000, these three reactors are expected to be replaced by a single new production reactor, and the Defense Waste Processing Facility (DWPF) is expected to begin to produce wastes in a glass form by late 1990. The HLW glass will be stored on-site until a national HLW repository becomes available. Current plans call for the DWPF to produce approximately 5,700 canisters of glass between 1990 and the end of 2020.

The ICPP projections are premised on predicted fuel deliveries and estimates of fuel reprocessing and waste management operations. The HANF projections assume that the fuel reprocessing plant will operate through 1993. A Hanford Waste Vitrification Plant (HWVP) may begin operation in 1999.

The projections of HLW for Hanford do not include vitrification, since material balances for such processes are not yet available. At the WVDP, vitrification of the HLW is scheduled to begin in 1994 and to be completed in 1995.

3.4 TRANSURANIC WASTES

The EPA standards (40 CFR Part 191) define transuranic wastes as those wastes containing more than 100 nanocuries per gram of alpha-emitting transuranic isotopes, with half-lives greater than 20 years (EPA85).

Alpha-emitting transuranic nuclides present a hazard because of their long radiological half-lives and high chemical toxicity. Most of the radionuclides that are contained in TRU wastes have very long half-lives and are typically present at low concentrations (DOE88a, LIT79, DOE88b, JAN83, BUR82, BRY81). Although a few decay products have energetic gamma emissions, their most significant hazard is due to alpha radiation emissions. Most TRU wastes can be handled with just the shielding that is provided by the waste package itself. These wastes are classified as "contact-handled" TRU wastes. A smaller volume may be contaminated with sufficient beta, gamma, or neutron activity to require remote handling. Heat generation in stored TRU waste is not a factor affecting how closely packages can be stored; however, avoiding the assembly of a critical mass as a result of densely-stored material must always be considered.

Relative to other radioactive wastes, TRU wastes represent a group of liquid and solid materials with widely varying chemical and physical properties. These wastes are categorized as contact-handled (CH), i.e., having a surface dose rate of less than 200 milliRoentgen per hour (mR/h); or remote-handled (RH), i.e., having a surface dose rate of greater than 200 mR/h.

Most TRU wastes are generated in DOE defense-related activities at the Rocky Flats Plant (RFP), Hanford facilities, and the Los Alamos National Laboratory (LANL). Nearly one-half of all TRU waste comes from weapons components manufactured at RFP and subsequent plutonium recovery at all three sites. Smaller amounts are generated at the Oak Ridge National Laboratory (ORNL), SRP, INEL, Argonne National Laboratory (ANL), Mound Facility, Bettis Atomic Power Laboratory, Lawrence Livermore National Laboratory, and Battelle-Columbus Laboratory. It should be noted that TRU wastes originating from the Mound Facility, Bettis and Argonne Laboratory, and from the Rocky Flats Plant are shipped to INEL for interim storage. The second largest source of TRU wastes is decontamination and decommissioning projects which account for one-fourth of the total. About one-fifth of TRU wastes come from laboratory activities, which can produce exotic TRU isotopes.

The amounts of TRU wastes from fuel cycle activities are in fact quite small because of the current moratorium on reprocessing and plutonium recycle. The Nuclear Fuel Services' reprocessing of nuclear fuel at West Valley, New York, produced some TRU waste that was disposed at that site. A small amount of TRU waste is also being generated in industrial and government-sponsored fuel fabrication and research.

3.4.1 Inventories and Characterization

Before March 1970, TRU wastes were disposed by shallow-land burial at AEC (now DOE) and commercial sites in pits and trenches and covered with soil. Beginning in 1970, the AEC initiated a policy of retrievable storage for TRU wastes since it concluded that such wastes must disposed using methods which provide greater confinement. Consequently, since 1970 TRU wastes have been stored in facilities for easy retrieval. Storage facilities have been built to suit the needs of each DOE site selecting methods which considered local climate, waste forms, existing volumes, and future generation rates. In addition, a program was established to characterize all previously disposed wastes and to identify long-term waste management options since early burial practices were not governed by current requirements. Such wastes, as well as newly generated wastes, would eventually be disposed at a dedicated transuranic waste disposal site such as is being considered at the Waste Isolation Pilot Plant, located in New Mexico. (DOE89).

The estimated buried volume mass of contained TRU elements and their associated alpha activities for each DOE site are given in Table 3.4-1. Storage facilities and waste disposal containers are designed for a 20-year lifetime, during which time, the necessary measures will be taken regarding the identification of permanent disposal options. According to the DOE, all of the stored retrievable wastes are located at the DOE sites listed in Table 3.4-2. Also given in this table are waste volumes, the mass of TRU elements, and the radioactivity as of December 31, 1988. Estimates of the radioactivity of this waste are based upon emplacement records and a knowledge of the types of operations at each disposal site or for each waste generator.

Over the years, some of the buried waste containers have been breached, and the surrounding soil has been contaminated. Accurately determining the volume of contaminated soil is a difficult task, and the estimated amounts cover a rather broad range (Table 3.4-3). Also, in the early days at HANF, ORNL, and LANL, some liquid wastes containing TRU

elements were spilled or drained into the ground. Further characterization studies are needed to provide a better estimate of the total volume of soil that is contaminated with TRU elements.

From ongoing characterization studies, several DOE sites have estimated that their buried and retrievable TRU solid wastes are composed primarily of the physical species given in Table 3.4-4. Most of the storage sites have relatively large fractions of combustible material and contaminated metal.

Estimated isotopic compositions for buried and retrievable wastes at the several DOE sites where TRU wastes are emplaced are given in Table 3.4-5. These estimates reflect information of DOE site operations and commercial TRU waste sources to characterize waste compositions when documented data are not available. Separate data for contact-handled and remote-handled waste were available for all sites that store both types of such wastes; however, composition data were not available for buried TRU waste at ORNL and portions of the waste buried at SRP. The radioactivity of the wastes buried at ORNL was assumed to be the same as that of the contact-handled waste. These data represent the best site estimates of the isotopic compositions of existing TRU wastes at government sites. The mix categories represent variations on major waste stream composition based on the total volume in storage plus the estimated waste volume generation through the year 2013 for each of the listed DOE sites.

3.4.2 TRU Waste Projections

TRU waste inventories and projected accumulations at government sites, of contact and remote-handled wastes from DOE defense activities, are listed in 5-year increments in Table 3.4-6. Projections are given, starting in 1988, for buried and stored wastes up to the year 2013. By 1990, when the Waste Isolation Pilot Plant, located in New Mexico, was expected to start receiving TRU wastes (DOE89), about 190,000 m^3 of such wastes were expected to have accumulated at the several DOE facilities.

It should be noted that TRU waste inventories could increase significantly as a result of uranium and/or plutonium recovered from post-Cold War dismantlement of nuclear warheads. The U.S. military inventories are believed to hold approximately 550 MT of highly-enriched uranium and 100 MT of plutonium (AL92). These inventories have not been factored into the quantities in Table 3.4-6.

Table 3.2-1. Historical and projected mass and radioactivity of commercial spent fuel (DOE89a)

End of Calendar year	Mass accumulated (Mt)	Radioactivity accumulated (10^6 Ci)
1970	55	215
1975	1,556	3,273
1980	6,534	10,159
1985	12,607	14,052
1988	15,607	18,654
1990	21,400	21,400
1995	30,800	25,600
2000	40,200	31,200
2005	48,700	31,900
2010	58,500	37,300
2015	70,200	42,400
2020	84,400	48,400

*Lower Reference Case projected capacity includes all existing reactors completed or under construction plus additional new reactors beyond the year 2005.

Table 3.2-2. Historical and projected* installed nuclear electric power capacity (DOE89a)

End of calendar year	Total GW(e)	End of calendar year	Total GW(e)
1960	0.2	1985	79.3
1965	0.3	1988	93.5
1970	6.4	1990*	99.6
1975	36.9	2000*	103.9
1980	51.4	2010*	100.6
		2020*	122.7

*Lower Reference Case projected capacity includes all existing reactors, completed or under construction, plus additional new reactors beyond the year 2005.

Table 3.3-1. Current volume of HLW in storage by site through 1988 (DOE89a)

Site	Volume (10^3 m^3) Liquid	Sludge	Salt cake	Slurry	Calcine	Capsules[a]	Total
Defense							
Savannah River Plant	64.2	14.1	50.0	[b]	[b]	[b]	128.3
Idaho Chemical Processing Plant	7.6	[b]	[b]	[b]	3.4	[b]	11.0
Hanford	26.8	46.0	93.0	77.7	[b]	0.004	243.5
Subtotal	98.6	60.1	143.0	77.7	3.4	0.004	382.8
Commercial							
West Valley Demonstration Project							
Acid waste	0.05	[b]	[b]	[b]	[b]	[b]	0.05
Alkaline waste	2.02	0.046	[b]	[b]	[b]	[b]	2.066
Subtotal	2.07	0.046	[b]	[b]	[b]	[b]	2.116
Grand total	100.67	60.15	143.0	77.7	3.4	0.004	384.92

[a]Capsules contain either strontium (^{90}Sr-^{90}Y) fluoride or cesium (^{137}Cs-^{137}Ba) chloride.
[b]Not Applicable.

Table 3.3-2. Current radioactivity of HLW in storage by site through 1988 (DOE89a)

Site	Radioactivity (10^6 Ci)						
	Liquid	Sludge	Salt cake	Slurry	Calcine	Capsules[a]	Total
Defense							
Savannah River Plant	99.0	400.0	162.1	[b]	[b]	[b]	661.1
Idaho Chemical Processing Plant	10.1	[b]	[b]	[b]	56.9	[b]	67.0
Hanford	23.3	121.4	12.6	111.1	[b]	177.1	445.5
Subtotal	132.4	521.4	174.7	111.1	56.9	177.1	1,173.6
Commercial							
West Valley Demonstration Project							
Acid waste	1.8	[b]	[b]	[b]	[b]	[b]	1.8
Alkaline waste	13.1	13.7	[b]	[b]	[b]	[b]	26.8
Subtotal	14.9	13.7	[b]	[b]	[b]	[b]	28.6
Grand total	147.3	535.1	174.7	111.1	56.9	177.1	1,202.2

[a]Capsules contain either strontium (^{90}Sr-^{90}Y) fluoride or cesium (^{137}Cs-^{137}Ba) chloride.
[b]Not Applicable.
[c]Includes strontium and cesium in capsules and separated concentrates that are awaiting encapsulation. The quantity of ^{90}Sr-^{90}Y is 61.3 x 10^6 Ci and that of ^{137}Cs-^{137m}Ba is 141.8 x 10^6 Ci.

Table 3.3-3. Historical and projected volume and associated radioactivity of HLW in storage by site through 2020 (DOE89a)

End of calendar year	Volume (10^3 m^3) Liquid	Sludge	Salt cake	Slurry	Calcine	Capsules[a]	Glass[b]	Total	Radio-activity (10^6 Ci) Total
				Savannah River Plant					
1980	59.8	10.5	26.4	–	–	–	–	97	699
1985	71.3	13.8	37.6	–	–	–	–	123	841
1988	64.2	14.1	50.0	–	–	–	–	128	661
1990	57.5	14.4	49.9	–	–	–	–	124	664
1995	39.9	10.0	40.2	–	–	–	1.0	94	849
2000	34.7	6.0	29.7	–	–	–	2.0	73	837
2005	41.0	3.4	15.1	–	–	–	2.6	62	863
2010	36.8	1.6	16.4	–	–	–	3.3	58	858
2015	36.8	1.5	10.3	–	–	–	3.4	52	888
2020	36.8	1.5	7.5	–	–	–	3.5	49	884
				Idaho Chemical Processing Plant					
1980	9.3	–	–	–	2.1	–	–	11	53
1985	7.1	–	–	–	3.0	–	–	10	69
1988	7.6	–	–	–	3.4	–	–	11	67
1990	7.6	–	–	–	3.9	–	–	12	77
1995	6.3	–	–	–	5.2	–	–	12	85
2000	6.7	–	–	–	6.3	–	–	14	128
2005	5.5	–	–	–	9.5	–	–	15	172
2010	2.0	–	–	–	11.3	–	–	13	182
2015	0.4	–	–	–	14.5	–	–	15	251
2020	0.4	–	–	–	16.8	–	–	17	284

[a]Includes strontium and cesium in capsules and separated concentrates that are to be encapsulated.

[b]Glass may be in storage at the site, in transit to a repository, or in a repository.

Table 3.3-3. Historical and projected volume and associated radioactivity of HLW in storage by site through 2020 (DOE89a) (continued)

End of calendar year	Volume (10^3 m^3)								Radio-activity (10^6 Ci)
	Liquid	Sludge	Salt cake	Slurry	Calcine	Capsules[a]	Glass[b]	Total	Total
				Hanford					
1980	39.0	49.0	95.0	4.0	--	0.002	--	187	558
1985	28.1	46.0	93.0	55.1	--	0.004	--	222	554
1988	26.8	46.0	93.0	77.7	--	0.004	--	244	446
1990	25.3	46.0	93.0	81.3	--	0.004	--	246	415
1995	7.9	46.0	93.0	95.6	--	0.004	--	242	392
2000	6.9	46.0	93.0	92.1	--	0.004	--	238	342
2005	6.9	46.0	93.0	94.1	--	0.004	--	240	303
2010	6.9	46.0	93.0	95.7	--	0.004	--	58	858
2015	6.9	46.0	93.0	97.4	--	0.004	--	52	888
2020	36.8	46.0	93.0	97.4	--	0.004	--	49	884
				West Valley Demonstration Project					
1980	2.15	0.046	--	--	--	--	--	2.196	35
1985	2.15	0.046	--	--	--	--	--	2.196	31
1988	2.07	0.046	--	--	--	--	--	2.129	29.3
1990	1.44	0.046	--	--	--	--	--	1.596	28.0
1995	--	--	--	--	--	--	0.21	0.21	24.7
2000	--	--	--	--	--	--	0.21	0.21	22.0
2005	--	--	--	--	--	--	0.21	0.21	19.7
2010	--	--	--	--	--	--	0.21	0.21	17.6
2015	--	--	--	--	--	--	0.21	0.21	15.6
2020	--	--	--	--	--	--	0.21	0.21	13.9

[a]Includes strontium and cesium in capsules and separated concentrates that are to be encapsulated.

[b]Glass may be in storage at the site, in transit to a repository, or in a repository.

Table 3.4-1. Inventories and characteristics of DOE/defense TRU waste buried through 1988 (DOE89a)

Burial site	Values reported by burial site as of Dec. 31, 1988		
	Volume (m^3)	Mass of TRU elements (kg)	Alpha radioactivity (Ci)
HANF	109,000	346	29,200
INEL	57,100	357	73,267
LANL	14,000	53.5	9,230
ORNL	6,200	5.6	270
SAND	3	<<1	1
SRP	4,534	9.1	9,831
Total	190,837	771.2	121,799

Table 3.4-2. Inventories and characteristics of DOE/defense TRU waste in retrievable storage through 1988[a] (DOE89a)

Burial site	Values reported by burial site as of Dec. 31, 1988		
	Volume (m^3)	Mass of TRU elements (kg)	Alpha radioactivity (Ci)
Contact-handled			
HANF	15,161	436	35,830
INEL	63,975	747.8	261,417
LANL	7,451.6	541.7	187,717
NTS	596	4.1	705
ORNL	625.2	26.6	17,505
SRP	6,489	195.0	653,191
Subtotal	94,297.8	1,951.2	1,156,365
Remotely-handled			
HANF	137	6	855
INEL	53.8	0.42	115
LANL	11.1	1.8	150
ORNL	1,304	106.2	2,920
Subtotal	1,505.9	114.42	4,040
Total	95,803.7	2,065.62	1,160,405

[a]Values cited are total quantities which represent the combined value of certified TRU waste and TRU waste managed as LLW (i.e., waste that is stored as TRU but falls below the 100 nCi/g alpha activity level).

Table 3.4-3. Inventories and characteristics of soil contaminated with DOE/defense TRU waste buried through 1988 (DOE89a)

Burial site	Values reported by burial site as of Dec. 31, 1988(a)		
	Volume (m^3)	Mass of TRU elements (kg)	Alpha radioactivity (Ci)
HANF	31,960	190.2	16,706
INEL	56,000-156,000	unknown	unknown
LANL	1,140	unknown	unknown
MOUND	300-1,000	0.009-0.029	150-526
ORNL	13,000-61,000[b]	unknown	unknown
SRP	38,000	unknown	unknown
Total	140,400-289,100	unknown	unknown

[a]See text for details.

[b]If soil containing TRU waste can be isolated from 1,600,000 m^3 of soil containing TRU and LLW waste.

Table 3.4-4. Estimated physical composition of retrievably stored, newly generated, and buried TRU waste at DOE/defense sites (DOE89a)

		Waste composition, vol %				
		Contact-handled		Remote-handled		
Waste type	Site	RSW[a]	NGW[b]	RSW[a]	NGW[b]	Buried
	ANL-E					
Absorbed liquids or sludges			36			
Combustibles			32		50	
Glass, metal, or similar noncombustibles			32		50	
	HANF					
Absorbed liquids or sludges						8
Combustibles		43	43	69.5	17	20
Concreted or cemented sludge		6	6	0.1		5
Filters or filter media						1
Glass, metal, or similar. noncombustibles		48	48	30.4	75	48
Other						18
Dirt, gravel, or asphalt		3				
	INEL					
Absorbed liquids or sludges		12	18.5			23.4
Combustibles		25	41.9	8	8	31.8
Concreted or cemented sludges		13	1.0			3.9
Dirt, gravel, or asphalt			2.3			6.7
Filters or filter media		5	6.9	11.2	11.2	1.3
Glass, metal, or similar noncombustibles		35	24.1	80[d]	80[d]	10.5
Other		10	5.3	0.8	0.8	22.4
	LANL					
Absorbed liquids or sludges		22	10			4
Combustibles		8	25	50	50	7
Concreted or cemented sludges		36	15			44
Dirt, gravel, or asphalt			1			30
Filters or filter media		4	1			2
Glass, metal, or similar noncombustibles		30	48	50	50	13
	LLNL					
Combustibles			73			
Concreted or cemented sludges			1			
Filters or filter media			7			
Glass, metal, or similar noncombustibles			15			
Other			4			
	MOUND					
Combustibles			1			
Concreted or cemented sludges						
Dirt, gravel, or asphalt			89			
Glass, metal, or similar noncombustibles			10			
	NTS					
Combustibles		51.5				
Concreted or cemented sludges		1				
Glass, metal, or similar noncombustibles		47.5				

Table 3.4-4. Estimated physical composition of retrievably stored, newly generated, and buried TRU waste at DOE/defense sites (DOE89a) (continued)

Waste type	Waste composition, vol %				
	Contact-handled		Remote-handled		
	RSW[a]	NGW[b]	RSW[a]	NGW[b]	Buried
ORNL					
Absorbed liquids or sludges			65		
Combustibles	59	55	20	55	40
Dirt, gravel or asphalt	1	1			
Filters or filter media	5	5	14	1	
Glass, metal, or similar noncombustibles	35	39	1	42	30
Other				2	30
RFP					
Absorbed liquids or sludges					
Combustibles		15.5			
Concreted or cemented sludges		36.3			
Dirt, gravel, or asphalt		0.7			
Filters or filter media		0.7			
Glass, metal, or similar noncombustibles		41.3			
Other		5.5			
SRP					
Absorbed liquids or sludges	[e]	3.5		[e]	
Combustibles	[e]	70		[e]	
Filters or filter media	[e]	5		[e]	
Glass, metal, or similar noncombustibles	[e]	27.5		[e]	
Other	[e]	1.5		[e]	

[a]Retrievably stored waste (RSW).
[b]Newly generated waste (NGW).
[c]Not reported, assumed to be same as stored waste.
[d]This is alpha hot-cell waste.
[e]Not reported; assumed to be same as newly generated waste.

Table 3.4-5. Calculated isotopic composition (wt %) of buried and retrievably stored TRU waste for each site[a] (DOE89a)

	Retrievably stored and newly generated wastes, wt %									
	Contact-handled							Remote-handled		Buried
Isotopes[b]	Mix-1[c]	Mix-2	Mix-3	Mix-4	Mix-5	Mix-6	Mix-7	Mix-8	Mix-9	Mix-10
ANL-E[d]										
^{235}U							56.0			
^{239}Pu	87	85					40.0			
^{240}Pu	11	8					4.0			
^{241}Am	1	1								
^{237}Np	5									
^{241}Pu	1	1								
MFP[e]							<1.0			
HANF[f][g]										
^{239}Pu	2.2						2.2	4.8		2.2
^{240}Pu	0.1						0.1	0.7		0.1
^{241}Pu								0.1		
^{232}Th	3.1						3.1	16.0		3.1
U depleted	72.8						72.8	21.6		72.8
U enriched	1.8						1.8	54.3		1.8
U normal	19.9						19.9	2.4		19.9
Other	0.1						0.1	0.1		0.1
INEL[h]										
^{241}Am	0.08	5.0		Trace						0.0001
^{238}Pu	Trace		80							
^{239}Pu	92.99	80	12.49	16			5.0	1.35	1.0	0.0017
^{240}Pu	5.8	10	2.5	4			1.0	0.15		0.0001
^{241}Pu	0.40	0.08								Trace
^{242}Pu	0.03									
^{232}Th									25.0	
^{233}U									5.0	
^{235}U			10.19				38.20	39.40	69.0	0.060
^{238}U			74.74				55.20	59.10		99.800
MFP							0.6			
Other	0.70	5.00								0.1

Table 3.4-5. Calculated isotopic composition (wt %) of buried and retrievably stored TRU waste for each site[a] (DOE89a) (continued)

	Retrievably stored and newly generated wastes, wt %										
	Contact-handled							Remote-handled			Buried
Isotopes[b]	Mix-1[c]	Mix-2	Mix-3	Mix-4	Mix-5	Mix-6	Mix-7	Mix-8	Mix-9	Mix-10	
LANL[i]											
^{235}U							47	47			
^{238}U							28	28		5.0	
^{238}Pu	5	0.5	1.2	0.5						0.01	
^{239}Pu	92	21.5	98.8	93	100		22.7	22.7		91	
^{240}Pu							2.1	2.1			
^{241}Pu							0.2	0.2			
^{241}Am	3	78		6.5						3.3	
MFP							[j]	[k]			
Other										0.69	
LLNL[l]											
^{238}Pu	0.014	0.013	0.011	0.009	0.004						
^{239}Pu	92.477	86.531	73.657	59.661	24.962						
^{240}Pu	5.965	11.941	24.896	14.915	49.922						
^{241}Pu	0.532	0.498	0.424	0.343	0.144						
^{242}Pu	0.023	0.022	0.018	0.015	0.006						
^{241}Am	0.990	0.995	0.994	25.057	24.962						
MOUND[m]											
^{238}Pu	79.894										
^{239}Pu	17.1										
^{240}Pu	3.0										
^{241}Pu	0.006										
NTS[n]											
^{238}Pu	0.5										
^{239}Pu	93.0										
^{240}Pu	6.0										
^{241}Pu	0.5										

Table 3.4-5. Calculated isotopic composition (wt %) of buried and retrievably stored TRU waste for each site[a] (DOE89a) (continued)

	Retrievably stored and newly generated wastes, wt %										
	Contact-handled							Remote-handled			Buried
Isotopes[b]	Mix-1[c]	Mix-2	Mix-3	Mix-4	Mix-5	Mix-6	Mix-7	Mix-8	Mix-9	Mix-10	
ORNL[o]											
^{233}U	67.34	6.13						94.57	52.55		
^{235}U	11.95	2.99									
^{238}U	7.69										
^{238}Pu	4.58										
^{239}Pu	20.71	42.44	99.98				98.77	2.57	29.99		
^{241}Am	1.68							2.11			
^{244}Cm	0.96	0.02						0.75			
^{252}Cf		0.15									
^{137}Cs									0.95		
^{90}Sr									15.18		
^{232}Th											
^{152}Eu											
^{154}Eu											
^{237}Np		13.03									
^{240}Pu		15.42									
^{241}Pu		3.78									
Other		1.15					1.23		1.33		
RFP[p]											
^{238}Pu	Trace										
^{239}Pu	91.00										
^{240}Pu	5.7										
^{241}Pu	0.3										
^{242}Pu	Trace										
^{241}Am	1.7										
^{235}U	0.6										
SIS[q]											
^{238}Pu	0.02	83.7	80.4								
^{239}Pu	93.18	14.0	16.2								
^{240}Pu	6.0	2.0	2.5								
^{241}Pu	0.5	0.3	0.7								
^{242}Pu			0.2								
^{241}Am	0.3				100						
^{237}Np				100							
^{244}Cm						100					

Table 3.4-5. Calculated isotopic composition (wt %) of buried and retrievably stored TRU waste for each site[a] (DOE89a) (continued)

Footnotes

[a] Data from Joseph Lippis, U.S. Department of Energy, Albuquerque Operations Office, WIPP, Carlsbad, New Mexico, memorandum to J.A. Klein, Oak Ridge, Tennessee, "TRU Waste Programs IDB Submittal through December 31, 1988," dated Sept. 15, 1989.

[b] Isotopes listed are those that are either >1%, by weight, or >1%, by activity, of the total.

[c] The mixes represent major waste stream composition variations.

[d] At ANL-E, 46 vol % of the contact-handled TRU waste is Mix-1, 54 vol % is Mix-2, and 100 vol % of the remote-handled TRU waste is Mix-7.

[e] Assumed ^{137}Cs to determine weight percent. ^{137}Cs chosen because it is the longest-lived major isotope in Mixed Fission Product (MFP).

[f] At HANF, 100 vol % of the contact-handled TRU waste is Mix-1, 12.4 vol % of the remote-handled TRU waste is Mix-7, 5.1 vol % is Mix 8, 82.5 vol % is Mix-9 (the composition of Mix-9 is unknown), and 100 vol % of the TRU-contaminated buried waste is Mix-10.

[g] HANF reported isotropic composition of uranium as U depleted, U enriched, and U normal. For radionuclide decay calculations, the data were converted to ^{235}U and ^{238}U by assuming 99.5%, 97.0%, and 99.3% ^{238}U, respectively.

[h] At INEL, 90.7 vol % of the contact handled TRU waste is Mix-1, 3.6 vol % is Mix-2, 0.3 vo; % is Mix-3, and 5.4 vol % is Mix-4; 7.9 vol % of the remote-handeled TRU waste has the same isotopic composition as Mix-1, 45.9 vol % is Mix-7, and 46.2 vol % is Mix-8. A portion of Mix-8 may also contain some Mix-9; and 100 vol % of the TRU contaminated buried waste is Mix-10.

[i] At LANL, 27.5 vol % of the contact-handled TRU waste is Mix-1, 6.6 vol % is Mix-2, 5.2 vol % is Mix-3, 54.0 vol % is Mix-4, and 6.7 vol % is Mix-5; 78.6 vol % of the remote-handled TRU waste is Mix-7, and 21.4 vol % is Mix-8. Also, 100 vol % of the TRU-contaminated buried waste is Mix-10. Mix-2 contains trace weight % MFP but 10.7 activity % MFP.

[j] Trace by weight percent, 85% by activity.

[k] Trace by weight percent, 95% by activity.

[l] At LLNL, 92.4 vol % of the contact-handled TRU waste is Mix-1, 1 vol % is Mix-2, 3.8 vol % is Mix-3, 2.3 vol % is Mix-4, and 0.5 vol % is Mix-5.

[m] At MOUND, 100 vol % of the contact-handled TRU waste is Mix-1.

[n] At NTS, 100 vol % of the contact-handled TRU waste is Mix-1.

[o] At ORNL, 25.6 vol % of the contact-handled TRU waste is Mix-1, 38.7 vol % is Mix-2, and 37.5 vol % is Mix-3; 27.6 vol % of the remote-handled TRU waste is Mix-7, 6.4 vol % is Mix-8, and 66 vol % is Mix-9. No information available on buried waste at ORNL.

[p] At RFP, 100 vol % of the contact-handled TRU waste is Mix-1. Weight percent totals less than 100% due to traces and round off.

[q] At SRS, 57.0 vol % of the contact-handled TRU waste is Mix-1, 31.2 vol % is Mix-2, 2.2 vol % is Mix-3, and 8.9 vol % is Mix-4, 0.6 vol % is Mix-5, and 0.1 vol % is Mix-6. No information available on buried waste at SRS.

Table 3.4-6. Current inventories and projections of DOE buried and stored TRU waste from defense activities (DOE89a)

End of calendar year	Volume (10^3 m^3) Accumulation	Radioactivity (10^3 Ci) Accumulation	Mass (kg) Accumulation
		Buried[a]	
1988	190.8	62.3	771.2
1990	190.8	62.3	771.2
1995	190.8	62.3	771.2
2000	190.8	62.3	771.2
2005	190.8	62.3	771.2
2010	190.8	62.3	771.2
2015	190.8	62.3	771.2
2020	190.8	62.3	771.2
		Stored[a]	
1987	57.7	3,871.1	2,064.3
1990	67.8	6,921.7	2,078.9
1995	83.1	12,010.6	3,785.8
2000	99.2	17,118.3	4,873.2
2005	114.6	22,211.9	5,952.7
2010	129.7	27,296.2	7,027.0
2013[b]	138.7	30,346.8	7,671.6

[a]Certified TRU waste (excludes waste managed as LLW).
[b]The destination of TRU waste after 2013 will not be defined until 2002.

Chapter 3 References

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Chapter 4: PLANNED PROGRAMS FOR THE MANAGEMENT AND DISPOSAL OF SPENT NUCLEAR FUEL, HIGH-LEVEL AND TRANSURANIC RADIOACTIVE WASTES

4.1 INTRODUCTION

As discussed in Chapter 1, the U.S. Department of Energy (DOE) is responsible for the care and disposal of government-produced TRU wastes and spent nuclear fuel and high-level wastes, regardless of their source. The DOE is conducting two programs to this end: (1) the Civilian Radioactive Waste Management Program, which pertains to the management and disposal of spent fuel from commercial nuclear reactors, commercial high-level waste, and any other wastes deemed by the U.S. Nuclear Regulatory Commission (NRC) to require geologic disposal, and (2) a program for the management and disposal of high-level and transuranic wastes generated in DOE atomic energy and defense activities.

The Congressionally selected disposal method for commercial HLW is burial in repositories excavated in geologically stable rock formations. Geologic disposal was selected after an evaluation (DOE80a) of several alternative concepts, including transmutation; disposal in space; the rock-melt concept; disposal in continental ice sheets, very deep holes, or isolated islands; and disposal under the ocean floor (the subseabed concept). In a recent rulemaking, the NRC has concluded that there is reasonable assurance that a repository located in deep geological media can provide safe disposal for spent fuel and high-level wastes (NRC84).

Disposal in deep geologic media has also been selected as the disposal method for much of the defense high-level and transuranic wastes. The DOE plans to dispose of the transuranic wastes in the Waste Isolation Pilot Plant (WIPP), located in New Mexico, if that facility is found suitable (DOE89a), and to a limited extent into the Greater Confinement Disposal Facility, located in Nevada. High-level wastes requiring disposal will be emplaced in the repository developed for commercial wastes.

A geologic repository will consist of surface facilities, underground facilities, and shafts or ramps connecting the surface and the underground facilities. When the repository is prepared for permanent closure, seals will be constructed for the shafts, ramps, and exploratory boreholes. The underground facilities is expected to consist of entry drifts and disposal rooms excavated deep (hundreds to thousands of feet) beneath the surface, with boreholes drilled vertically into the floors or horizontally into the walls of the disposal rooms for the emplacement of waste canisters.

The repository will be prepared for permanent closure by backfilling the underground areas and permanently sealing the shafts and ramps. The surface facilities will be decontaminated and decommissioned, and the site will be eventually returned to its natural state. Permanent site markers, records, and other passive institutional controls will be erected to warn future generations of the presence of the repository and its contents.

4.2 CIVILIAN RADIOACTIVE WASTE MANAGEMENT PROGRAM

In response to the Amendments Act, the DOE reports it is developing a waste management system consisting of: 1) a geologic repository, 2) an MRS facility, and 3) a transportation system. The DOE plans, as referenced, for each of these elements are briefly summarized below.

Geologic repository -- The geologic repository program is currently focused on site characterization activities at Yucca Mountain. The DOE is preparing to construct two exploratory shafts and to conduct surface-based and in-situ tests designed to provide information about the suitability of the site. A detailed site characterization plan (DOE88a) based on conceptual repository and waste package designs has been published. This plan is based on a ranked hierarchy system reflecting regulatory requirements and DOE strategies for resolving technical and licensing issues. It provides a framework for conducting the testing needed to resolve issues about the design and performance of the repository and the waste package. It also describes the DOE's plans for assessing the pre-closure and post-closure performance of the repository, including the development and validation of the necessary models. In a parallel effort, the DOE will develop advanced design concepts for the repository and waste packages, including designs or design changes which will be included in the license application to the NRC.

The Yucca Mountain site lies in the southern part of the Great Basin, an arid region with linear mountain ranges and valleys, very little rainfall, sparse vegetation, and a sparse population. At Yucca Mountain, the water table is very deep, lying as much as 2,500 feet below the land surface. The repository is currently planned to be located in the unsaturated zone. The unsaturated zone is the rock mass between the surface of the land and the water table. At Yucca Mountain, the unsaturated zone is thick enough to allow the construction of a repository at a depth of about 1,050 feet while remaining about 660 to 1,300 feet above the top of the water table. The rock formation selected as the potential host media is volcanic tuff, which is a moderately to densely welded and devitrified rock (see Section 4.4 for more details). The host rock formation is known as the Topopah Spring Member of the Paintbrush Tuff.

The unsaturated rock of the Topopah Spring tuff is expected by DOE to provide a suitable environment for the long-term performance of the waste package. For example, the pressure exerted on the disposal containers is estimated to be equal to that of atmospheric pressure. There will be no hydrostatic pressure because the repository is to be located above the water table, and the waste packages will not be subjected to loads induced by the creeping (plastic movement) of the rock because the host rock is not plastic enough. Any water available for the corrosion of containers and waste dissolution is expected to be limited to minimal amounts. These and other pertinent features of the site will be subject to thorough investigations during site characterization by the DOE.

MRS facility -- The DOE has conducted a series of systems studies to determine the preferred configuration for the MRS facility (DOE89b). The preferred configuration is that of a facility which receives and temporarily stores spent fuel and initiates spent fuel shipments to an operational repository. The capability of packaging wastes is an option which could be added at some later time. It would consist of a facility and equipment needed for additional

functions, such as waste consolidation and repackaging into disposal containers. This option would provide added flexibility and facilitate the operations of the waste management system. The MRS facility should be able to start receiving spent fuel more than 3 years earlier than the repository. The Amendments Act established two different paths for siting the MRS facility: 1) siting through a DOE-directed screening process or 2) siting through the Office of the Nuclear Waste Negotiator.

Transportation -- The transportation system will rely on the use of shipping casks, to be developed by the private sector, and transportation support services, which may include a cask maintenance facility, operational facilities, and support equipment. Specific facility needs will be identified as the designs for other facilities proceed. In order to increase cask capacity and reduce the number of shipments and overall disposal costs, the DOE has embarked on a cask development program designed to support the shipment of the following types of wastes:

1. Spent fuel from reactor sites to the facilities of the Civilian Radioactive Waste Management Program.

2. Spent fuel and other types of radioactive materials from the MRS facility to the repository.

3. Non-standard spent fuel and non-fuel bearing components from reactor sites to the MRS facility or the repository.

4. Defense and commercial high-level radioactive wastes from other storage locations to the repository.

The DOE will use the private industry to the maximum extent possible in both the development and the acquisition of transportation equipment and services. It plans to contract with private industry for cask development, certification, and the fabrication of prototype casks. After the development has been completed, the DOE will contract with the private sector to supply a fleet of casks for its transportation operations.

4.3 PROGRAMS FOR THE MANAGEMENT AND DISPOSAL OF DEFENSE WASTES

High-level wastes are produced during the reprocessing of spent reactor fuel and irradiated targets to recover uranium and plutonium. Most of the high-level wastes produced in the United States through 1987 have originated from atomic energy research and defense activities, such as the manufacture of nuclear weapons, and are stored at three DOE facilities: the Hanford Site in the State of Washington; the Savannah River Plant near Aiken, South Carolina; and the Idaho National Engineering Laboratory (DOE88b). These wastes are in the form of alkaline liquids, salt cake, slurry, sludge, acid liquid, and calcine; they are stored in underground tanks or bins. Most of them have already been subjected to some treatment (e.g., neutralization with caustic soda, which produces the sludges), and most will require incorporation into a stable solid medium, such as glass or ceramic. The solid waste will be packaged in stainless-steel canisters.

4.3.1 Defense High-Level Wastes

In accordance with the provisions of the Nuclear Waste Policy Act of 1982, the DOE performed a comparative evaluation of two disposal options for defense high-level wastes: 1) disposal in a commercial geologic repository and 2) disposal in a geologic repository constructed for defense wastes only. The two options were compared in terms of criteria specified in the Act: 1) cost efficiency, health and safety, regulation, transportation, public acceptability, and 2) national security. The results (DOE85b) indicated that there are no compelling reasons to develop a repository for defense wastes only, but the only factor that showed a clear advantage for disposal in a commercial repository was cost efficiency. The Secretary of Energy recommended to the President, and the President agreed, that a combined repository option could be implemented (DOE85a).

Typically, defense high-level radioactive wastes will be shipped in containers to a commercial repository for disposal after solidification, e.g., in glass or some other solid forms. At the repository, the containers will then be transferred underground for emplacement and final disposal. Transportation to the repository will be conducted by the DOE's Office of Civilian Radioactive Waste Management, and the costs of geologic disposal will be paid by DOE contributions to the Nuclear Waste Fund.

Hanford Site -- Since the early 1940s, DOE Hanford operations have resulted in the generation of large volumes of solid and liquid wastes. Solid wastes are routinely disposed of on site by ground burial. Liquid wastes are stored either temporarily or permanently in special tanks, processed, and discharged into surface ponds, cribs, ditches, or released into the Columbia River. There are four major operating areas at the Hanford Site which support a diverse range of production and research activities. They are: the 100-Areas, which include the N-Reactor and eight deactivated production reactors; the 200-Areas (West and East), which include two reactor fuel reprocessing plants and waste treatment facilities; the 300-Areas, which include the reactor fuel manufacturing and research and development facilities; and the 400-Areas, which includes the Fast Flux Test Facility.

The high-level radioactive wastes stored at Hanford may be divided into five groups: liquids (11% by volume), sludges (19%), and salt cake (38%) - stored in single-shell tanks; slurry (32%) - stored in double-shell tanks; and encapsulated wastes which contain heat producing radionuclides (e.g., strontium-90 and cesium-137, and their decay products) (DOE89d). Encapsulated wastes are stored in water basins for cooling. Encapsulated wastes, however, represent a very small fraction of the total waste volume (less than 0.002%), but about 40% of the total activity for the waste forms identified above (DOE89d). Table 4.3-1 presents a summary of the Hanford waste storage methods, volumes, TRU activity, and number of sites (see Chapter 3 for more details).

Much of this waste has been accumulating since the 1940s and was initially stored in single-shell tanks. More recently, double-shell tanks were built to reduce the possibility of leakage into the environment since it was known that a number of the single-shell tanks (26) were leaking (DOE87b). The double-shell tanks provide redundant containment by incorporating primary and secondary carbon steel walls which are supported and encased in a reinforced concrete outer shell. The radioactive decay-heat is removed by a closed-loop

cooling coil system within which circulates cooling water. The contents of each tank are continuously mixed by air circulators and air ballast tanks which provide an intermittent flushing action to prevent sediments accumulating at the bottom of the tank.

In total, there are five types of tanks (Types I through IV are single-shell and Type V is double-shell) used in Hanford with capacities ranging from 210 to 3,800 m^3. Typically, seven single-shell tanks (3,800 m^3) are needed to provide a 5-year storage capacity associated with the processing of about 2,000 metric tons of spent fuel (DOE80a). Currently, there are 28 double-shell tanks in use in addition to the existing 149 single-shell tanks (DOE87b). These tanks are located in 17 tank-farm sites located in both the East and West 200-Areas. The tanks are placed underground and are covered with about 2 meters of soil. On the average, the tanks are located about 60 meters above the local water table.

The wastes, destined for storage in the double-shell tanks, are typically pre-treated prior to being pumped into the tanks. The low specific-activity portion of this waste stream will be separated and disposed in a grouted form in near-surface vaults at the site. The remaining wastes are to be solidified into borosilicate glass before being shipped to a commercial geologic repository. The DOE will build a plant for vitrifying these wastes. Current plans call for the plant to start operating in 1999. Encapsulated strontium and cesium wastes will also be sent to the commercial repository for disposal.

The DOE has identified preferred disposal alternatives for some waste forms while it has deferred its decision for other types of wastes (DOE87b, DOE89c). For example, the DOE currently plans to store, for the foreseeable future, wastes contained in single- and double-shell tanks (DOE87b, DOE89c). The potential disposal methods being considered include in-situ immobilization of the tanks and their contents, disposal of a fraction of the tank wastes (as low specific activity grout) in near-surface vaults, and waste treatment and vitrification for final disposal at the commercial HLW repository. Encapsulated strontium and cesium wastes are also to be sent to the commercial HLW repository once it becomes operational. Transuranic wastes buried at the Hanford facility will be eventually retrieved, processed, and repackaged prior to being sent to a dedicated transuranic waste disposal facility (DOE89a). For contaminated soils, DOE is evaluating several options which include in-situ stabilization and geologic disposal. In the interim, the DOE will continue maintenance activities of the sites where radioactive materials are presently buried. Current DOE schedules indicate that these disposal plans will be implemented over a 20-year period (DOE89c).

In response to Federal, State and local requirements, the DOE is conducting a comprehensive environmental monitoring program to assess the impact of facility operations in the vicinity of the Hanford Site (JAC88). The results of the environmental monitoring program indicate that on-site radionuclide ground-water concentrations were noted to be above the EPA Drinking Water Standards (DWS) and in some instances above the DOE's Derived Concentration Guides (DCG) (JAC88). The results of the monitoring program indicates that the following radionuclides are present in ground-water: tritium, cobalt-60, strontium-90, technetium-99, ruthenium-106, antimony-125, iodine-129, iodine-131, cesium-137, uranium-234, and uranium-238.

Certain chemicals regulated by the EPA and the State of Washington were also present in the ground-water near the operating areas. The primary source of the ground-water contamination is due to liquid wastes released into the ground by past and on-going site and facility operations. Waste disposal activities, at both active and inactive sites, have also contributed to the current levels of contamination.

Elevated tritium levels are present in all Hanford Site Areas, except in the 300-Areas. The highest concentrations were noted to be in the 200-Areas, in both East and West sections. The tritium plume in the 200-Areas is characterized with peak concentrations ranging from 1 million to nearly 14 million pCi/L. The peak tritium concentration in the 100-Areas was reported to be 1.3 million pCi/L. Other locations on the Hanford Site are characterized by tritium concentrations ranging from non-detectable levels (about 300 pCi/L) to a few hundred thousands pCi/L. In general, the tritium plumes are moving east and southeast following the movement of the ground water toward the Columbia River. The EPA DWS for tritium is 20,000 pCi/L and the DOE DCG limit is 2 million pCi/L (JAC88). The EPA DWS is based on an organ (whole body for tritium) annual dose limit of 4 mrem while the DOE DCG represents a committed effective dose equivalent of 100 mrem per year (JAC88).

Gross-beta radioactivity was found in wells throughout the Hanford Site. This radioactivity is associated with the presence of cobalt-60, strontium-90, technetium-99, antimony-125, cesium-137, uranium decay products (thorium-234 and protactinium-234), and to a certain extent to iodine-131 and iodine-129. The highest concentrations were noted to occur in the 100 and 200-Areas. The gross-beta radioactivity in ground-water samples is characterized with peak concentrations ranging from 1,000 to nearly 16,000 pCi/L. Other locations on the Hanford Site are characterized by total gross-beta radioactivity ranging from non-detectable levels (about 16 pCi/L) to a several hundreds pCi/L. The radionuclide distribution and ground-water concentrations were reported, in decreasing order, to be technetium-99 (up to 29,000 pCi/L), iodine-131 (up to 28,000 pCi/L), strontium-90 (up to 10,000 pCi/L), antimony-125 (up to 300 pCi/L), iodine-129 (up to 47 pCi/L), ruthenium-106(less than 30 pCi/L), cesium-137 (less than or equal to 22 pCi/L), and Co-60 (less than or equal to 20 pCi/L) (JAC88). As with tritium, these radionuclides are also moving east and southeast following the movement of the ground-water toward the Columbia River. Except for technetium and iodine, the ground-water plumes associated with these radionuclides are not as extensively dispersed as the one due to tritium. The reported concentrations for technetium, iodine, and strontium in several wells exceed the EPA DWS.

The presence of alpha-emitting radionuclides were detected in several wells located in the 100, 200, and 300-Areas. The total gross-alpha radioactivity is thought to be due to uranium since plutonium concentrations were noted to be below the limit of detection (about 0.1 pCi/L) (JAC88). The highest concentrations were noted to occur in the 200-Areas (West) while much lower concentrations were detected in the eastern sector of the 200-Areas. The peak concentrations in the 200-Areas (West) were reported to range from 100 to 10,500 pCi/L. Uranium has been also been noted in the vicinity and downgradient of the fuel fabrication facilities (300-Areas) and near inactive waste disposal sites. The average uranium concentrations were reported to range from 2 to 310 pCi/L, with peak concentrations ranging from 100 to nearly 12,000 pCi/L. Other locations on the Hanford

Table 4.3-1. Summary of waste sites, volume, and activity at the DOE Hanford Facility.[a]

Type of Wastes	Number of Sites/Tanks	Area (Ha)	Volume (m^3)[b]	TRU (Ci)[b]
Single-shell tanks	12/149	5.5	1.4E+5	6.1E+4
Double-shell tanks	5/28	1.2	9.7E+4	3.2E+5
Capsules[c]	1/--	0.01	minimal[d]	minimal[d]
Retrievable TRU wastes	7/--	5.0	2.6E+4	9.0E+4
Buried TRU wastes[e]	9/--	7.3	1.1E+5	3.0E+4
TRU contaminated soil	24/--	1.2	3.2E+4	2.0E+4

[a]Data from DOE87b, Volume 1, Table 3.1, page 3.6. See Chapter 3 for more details.
[b]Exponential notation, 1.4E+5 means 1.4×10^5 or 140,000.
[c]For this entry and the following ones, the wastes are not stored in tanks.
[d]Presence of TRU material is negligible, most of the activity is due to long-lived fission products totaling of about 203 million curies. The volume of the capsules is less than 0.002% of the waste volume to be treated and disposed.
[e]Wastes buried up to 1970.

Site are characterized by uranium concentrations ranging from non-detectable levels (0.5 pCi/L) to less than 100 pCi/L. The reported gross-alpha concentrations in several wells exceed the EPA DWS.

The Hanford radiological environmental surveillance program also routinely monitors other areas, at both on and off-site locations. These locations include three on-site ponds and one lake, soils at 38 different on and off-site locations, and at upstream and downstream points on the Columbia River.

Radionuclide concentrations in the three ponds and West Lake have been noted to vary (DOE87b). The 1987 survey results indicate that tritium is the dominant radionuclide with peak concentrations ranging from 160 to 9,500 pCi/L. The next predominant radionuclide is cesium-137 which was reported to range from 1.1 to 50 pCi/L. Strontium-90 was also detected in pond and lake water samples with peak concentrations ranging from 0.4 to 2.8 pCi/L. Total gross beta and alpha water sample activity revealed peak water concentrations of 490 and 267 pCi/L, respectively, for West Lake. The gross beta and alpha water activity in the three ponds were typically one to two orders of magnitude lower than those noted for West Lake.

Soil sample analyses at 15 on-site locations revealed four radionuclides have been routinely detected in measurable levels. Strontium-90 is known to be present in concentration

varying from 0.02 to 0.38 pCi/g with an average of 0.31 pCi/g. Cesium-137 has been measured at concentrations varying from 0.01 to 16 pCi/g with an average of 2.0 pCi/g. Plutonium-239 and 240 have also been measured at concentrations varying from 0.001 to 0.17 pCi/g with an average of 0.027 pCi/g. Finally, uranium was reported at concentrations ranging from 0.19 to 3.8 pCi/g with an average of 0.58 pCi/g. Typically, the average on-site measurements are higher than those noted off-site by factors ranging from about 2 to 5.

Analyses of water samples taken downstream in the Columbia River indicate that radionuclides identified with Hanford Site operations were noted at very low concentrations, typically well below the applicable drinking water standards (DOE87b). The water samples were taken at two different locations, one at the 300-Areas Water Intake and the other at the Richland Pumphouse located about 3 km downstream from the site boundary. The Richland Pumphouse is the first downstream point on the river where water is withdrawn for public use. Water sample analyses revealed that tritium is the most predominant radionuclide with a reported peak concentration of 200 pCi/L. Other radionuclides were also reported, including strontium-89 and -90 (0.2 and 0.15 pCi/L, respectively), total uranium (0.61 pCi/L), gross beta (2.8 pCi/L), and gross alpha (0.79 pCi/L). Other radionuclides, including plutonium-239 and -240 as well as other fission products, were reported at lower concentrations, typically ranging from 1.0×10^{-6} to 4.5×10^{-2} pCi/L.

Savannah River Plant -- At the Savannah River Plant, high-level wastes, in the form of alkaline liquids, alkaline sludges, and salt cake, are stored underground in high integrity, double walled, stainless-steel tanks. By 1993, hot operation of a waste processing facility to vitrify these wastes into borosilicate glass is scheduled to begin.

Idaho National Engineering Laboratory -- In Idaho, high-level wastes, in the form of acidic liquids, are first stored in underground tanks and later converted to calcine. Stainless steel tanks housed in concrete vaults are used to store liquid wastes, and stainless steel bins in concrete vaults are used for the calcine wastes. According to DOE plans, a facility for immobilizing newly generated wastes will start operations early in the next century. It will also process the stored calcine wastes. Evaluations of waste forms and immobilization processes are being pursued.

4.3.2 Transuranic Wastes and Defense Waste Programs

The research and development (R&D) efforts for defense wastes are divided into three major categories: 1) the immobilization of high-level wastes, 2) the preparation of transuranic wastes for shipment to the WIPP facility, and 3) investigations to demonstrate the performance of the WIPP site. Also these R&D efforts include the development of technology for in-place immobilization of wastes stored in tanks and evaluation of methods for immobilizing wastes stored at the Idaho National Engineering Laboratory.

- Waste Isolation Pilot Plant (WIPP)

The DOE is developing the Waste Isolation Pilot Plant (WIPP) in a bedded-salt formation near Carlsbad, New Mexico to demonstrate the disposal of defense TRU wastes. The WIPP project was authorized in 1980 by Public Law 96-164 to provide a research and

development facility for demonstrating the safe disposal of transuranic wastes produced by national defense activities. If testing proves satisfactory, the DOE is expected to open the site for the permanent disposal of TRU waste.

The WIPP site is in a sparsely populated area on land owned by the Federal government. The WIPP plant consists of surface facilities (mainly a waste-handling building), four access shafts, and underground facilities designed to emplace approximately 6.5 million cubic feet of TRU waste in a 100-acre repository. About 12 acres have also been set aside as an underground test area to conduct experiments and study the behavior and performance of the repository. The repository has been excavated in a bedded-salt formation (the Salado Formation) 2,150 feet beneath the surface.

By mid-1989, the initial major construction activities at the WIPP had been nearly completed (DOE89a). The surface facilities were essentially complete, and most of the underground rooms for experimentation and initial waste emplacement had been excavated. A five-year test phase is planned to develop data for incorporation into the performance assessment. All of the wastes will be retrievably emplaced should the site be declared unsuitable at the end of the test period. During this phase, the DOE will monitor the site and facility as part of the environmental monitoring programs it has been conducting since 1980.

For shipment to the WIPP site, TRU wastes will be contained in Type B shipping containers (TRUPACT-II) certified by the NRC and carried by truck. The DOE's purpose in using truck transportation for moving waste to the WIPP is to have greater accessibility to the site and greater control of the transportation system, routes, and speed. The proposed routes from the waste storage locations use the interstate highway system to the maximum extent possible (DOE89a). To ensure safe and efficient transport, the DOE will use a transportation tracking and communication system that will combine navigation, satellite communication, and computer network technologies to monitor the movements of TRU waste shipments to the WIPP.

All of the wastes received by the WIPP will have to meet acceptance criteria covering factors such as waste forms and characteristics, gas generation, immobilization, presence of toxic and corrosive substances, and thermal power. All incoming packages will be checked for surface contamination and external radiation exposure rates, and repackaged or repaired if necessary.

- Greater Confinement Disposal Facility

In 1981, the National Low-Level Waste Management Program and the DOE's Nevada Operations Office began a project to demonstrate the feasibility of "greater depth" burial in the alluvial sediments of the Nevada Test Site (REY83, EPA87). The purpose of the project, named Greater Confinement Disposal Test (GCDT), was to evaluate the feasibility of disposing of classified TRU wastes and high specific-activity low-level wastes at intermediate depths in large-diameter augered holes. These wastes originate from weapon facilities across the nation. The basic concept involves sinking a shaft 3 meters in diameter and nearly 40 meters deep. The shaft has a capacity of about 1,100 m^3. Wastes are then lowered into the hole and stacked up to depth of about 20 meters from the surface. At this point, the hole is

backfilled with soil all the way up to the surface. The goal of the GCDT program is collect and analyze data on radionuclide migration and to develop waste handling procedures and equipment. Plans are also being developed to retrieve these wastes after emplacement, if necessary.

4.4 POTENTIAL HOST ROCKS FOR GEOLOGIC REPOSITORIES

Many types of rocks are potentially suitable as host rocks for a repository, depending on the natural attributes of the rock and the geohydrologic setting. Ideally, the host rock should be suitable for the construction of the repository and for waste containment, and the surrounding rock formations should provide adequate isolation (DOE80b). Important natural attributes include thermal, mechanical, hydraulic, and chemical characteristics that affect the response of the host rock to heat, the movement and chemistry of ground water, and the ability to retard the migration of radionuclides. The desirable geohydrologic properties include low rates of ground-water flow, long path lengths to the accessible environment, and evidence of long-term stability (DOE80b).

In the United States, early plans for geologic disposal were based on bedded salt and salt domes. Salt was the rock investigated most extensively as part of a site screening program. Later, when the DOE began to study Federal lands dedicated to nuclear activities, several other host rocks came under investigation. They included argillaceous rocks and tuff in Nevada and basalt in the State of Washington. For the second repository, DOE began to study crystalline rock formations (DOE86a-g). Other rocks that have been considered are limestone, sandstone, anhydrite, chalk, and argillaceous rocks like shale (GON85). The sections that follow briefly review the properties of host rock media most studied in the United States.

4.4.1 Basalt

Basaltic rock masses are among the strongest of common rock types. In addition, basalt has moderate thermal conductivity and a high melting temperature, which enable it to withstand high thermal loads. The basaltic formation that had been investigated in the first repository program was a thick section, about 950 meters below the surface, near the middle of the extensive basalt flows of the Columbia Plateau. The basaltic rock in this section contains openings filled with alteration products (mainly clay minerals), and as a result the rock mass is of low permeability. On the other hand, the basalts of the Columbia Plateau commonly have columnar joints or rubbles that are potential channels for water flow. Water-bearing sedimentary interbeds within the basalt section are also common.

A potential site in basalt is located in the State of Washington. Thick basaltic formations also occur in the States of Idaho and Oregon.

4.4.2 Bedded Salt and Salt Domes

Of the nine sites identified as potentially acceptable for the first repository, seven were in salt: four sites in bedded-salt formations and three in salt domes (DOE85a).

Salt is suitable as a host rock because of its structural strength, radiation shielding capability, high plasticity (which enables fractures to heal or seal themselves at repository depths), low moisture content, and low permeability. In addition, salt deposits are abundant in the United States and are relatively easy to mine. Desirable features of many salt basins are their relatively simple structure and predictable stratigraphy over large areas.

Although salt deposits are widespread, the salt itself and the other deposits with which it is often associated (e.g., hydrocarbons or potash) could increase the probability of human intrusion into a repository. Furthermore, the solubility of salt is greater than that of any other potential host rock. The potential for this failure mode must be carefully assessed in analyzing the long-term performance of a repository sited in salt.

4.4.3 Granite and Related Crystalline Rocks

Granite and related crystalline igneous and metamorphic rocks, such as gneiss, are the most abundant rocks in the upper 10 kilometers of the Earth's continental crust. These rocks underlie virtually all of the United States; they occur at the surface in stable areas, in the cores of many mountain ranges, and beneath all of the younger sedimentary rocks. Their strength, structural and chemical stability, and low porosity make them attractive for geologic repositories. The water content of these rocks is low and is held mainly in fractures and in hydrous silicate minerals. The permeability of these rocks is largely dependent on the presence of fractures, and it is reduced considerably by the closure of fractures, which occurs at depths in excess of several hundred meters. The depth for a repository is likely to vary from region to region, depending on how the permeability is affected by the tectonic history of the region.

Granite as a potential host rock is being investigated in some European countries. In the United States, the DOE had conducted preliminary investigations of near-surface and exposed crystalline rock formations in 17 States in a search for sites for the second repository. However, the Amendments Act directed DOE to terminate site-specific activities for a second repository and limited such activities only to tuff.

4.4.4 Tuff

Tuff is the dominant component of the voluminous and widespread volcanic strata in the Basin and Range province of the western United States. The tuff formation at the Yucca Mountain site, located in southern Nevada, currently being characterized for the first repository, consists of a sequence of welded and non-welded tuffs.

The site selected as the potential host rock is moderately to densely welded and devitrified, with a minor number of cavities. This section of the rock formation has high density, low porosity and water content, good compressive strength, and the ability to withstand the heat generated by radioactive waste. However, the characteristics that affect the thermal and mechanical properties of tuff, such as porosity, degree of saturation, and stress state, are known to vary both laterally and vertically. Consequently, the thermal and mechanical properties are also likely to vary spatially.

Lying beneath the welded tuff are non-welded tuffs containing zeolite, a hydrous silicate. These tuffs are characterized by low density, moderate compressive strength, moderate thermal conductivity, and excellent capability for sorption. The latter is important to the waste isolation performance of a repository because it would allow these rocks to significantly retard the migration of radionuclides into the accessible environment.

4.5 INTERNATIONAL ACTIVITIES

Countries which are committed to use nuclear power or in which nuclear power already makes up a significant fraction of the total electrical generating capacity are establishing long-term programs for the safe management and disposal of spent fuel and high-level radioactive wastes. Such programs include adopting a national strategy, assigning the technical responsibility for research and development activities to designated agencies, selection of disposal technologies and geological media, and setting the appropriate regulatory standards to protect the public and environment.

Typically, the objective of a geological disposal program is to immobilize and isolate radioactive wastes from the environment for a sufficient period of time under conditions such that any radionuclide releases from the repository will not result in unacceptable radiological risks. For illustrative purposes, the disposal programs of eight countries are summarized below (NEA86, NEA88, SCH88, SCH91, IEAL87). These countries are Canada, the United Kingdom, France, the Federal Republic of Germany, Belgium, Switzerland, Sweden, and Japan. A summary of these countries' institutional and regulatory programs is also provided in Chapter 2, Section 2.3.

4.6.1 Canada

Atomic Energy of Canada Limited (AECL), a Crown corporation reporting to the Federal Minister for Energy, Mines and Resources, has been assigned the responsibility for the permanent disposal and isolation of radioactive wastes in Canada. Currently, the program considers only direct disposal of spent fuel without reprocessing, although the reprocessing option has not been completely ruled out. Until a repository is available, spent fuel will initially be stored at each reactor site and, later, possibly at a central facility. Under a joint agreement, Ontario Hydro (a provincially owned utility) has been mandated to develop the technologies needed for the interim storage and transportation of spent fuel.

The Canadian disposal concept considers siting a repository in a granitic formation located in the Canadian Shield. The repository will be located at depths of 500 to 1,000 meters. The spent fuel canisters will be inserted into floor cavities located in excavated disposal rooms. Once filled, the floor cavities and room excavations will be backfilled and sealed using engineered barriers. The AECL facility design is already well defined and the concept was submitted for public and regulatory review in 1988. AECL is now preparing a final Environmental Impact Statement which it will submit to a government-appointed Review Panel by mid-1993, after which public hearings will be held. The Panel is expected to present findings and recommendations to the government in early 1995; subsequently, the government will reach a finding on the acceptability of the concept. AECL estimates that

siting, licensing and construction of a disposal facility will take 25 to 30 years and that the facility could, therefore, be in operation by 2025.

In 1986, AECL established an underground research laboratory (URL) in undisturbed granitic rock at a depth of 240 meters at Lac du Bonnet in the Province of Manitoba. AECL has since deepened the facility to 440 meters. The purpose of the URL is to conduct large-scale, in-situ experiments in the type of rock envisioned under the Canadian disposal concept, demonstrating some of the components of the disposal concept (the facility is not a candidate repository site). The AECL is developing methodologies and analytical techniques to evaluate the geomechanical and geohydrological properties of granitic rock. Construction of the URL was completed in 1988.

4.6.2 United Kingdom

In the United Kingdom, the responsibility to develop a national strategy for radioactive waste management lies with the Department of the Environment. The organizations which produce the wastes have the direct responsibility for their safe management and funding. An industry consortium, the UK NIREX Ltd, has been established to develop and operate new low- and intermediate-level radioactive waste disposal facilities in England.

The United Kingdom's radioactive waste disposal program strategy has postponed the development of a disposal facility in deep geological media. Rather, the current plans call for reprocessing of spent fuel, solidification, and surface storage for about 50 years. The United Kingdom has also adopted a policy of monitoring the results of research activities being conducted by other countries. Depending on the outcome of research being conducted abroad, Britain would then identify a high-level waste disposal strategy and repository program development activities using concepts that best fit British needs.

However, some in-situ research has been conducted by the UK Atomic Energy Authority and UK NIREX Ltd in heat transfer properties of Cornish granite, statistical analysis of fracture occurrence, orientation, and aperture in granite, and fractured flow in Cornwall shale. Other research activities have included geohydrological and geophysical measurements, geochemistry, radionuclide migration and transport, integrated site characterization and model validation, and characterization of model parameters and measurement methods.

4.6.3 France

The French radioactive waste disposal program is based on a closed fuel cycle involving spent fuel reprocessing, interim storage, and recovery and re-use of plutonium in breeder and light-water reactors. The nuclear waste program has been entrusted to the National Radioactive Waste Management Agency (ANDRA), an arm of the French Atomic Energy Commission (CEA). Since 1969, short-lived radioactive wastes have been emplaced in engineered near-surface disposal facilities at Centre de la Manche, near Cherbourg on the English Channel. This facility will reach its design capacity in 1994; a new facility, Centre de l'Aube, began operation January 1992 about 100 miles southeast of Paris.

ANDRA was previously investigating four geological media for HLW disposal -- clay, salt, granite, and schist -- and had begun investigative work at a site in each medium. A URL was to be established at one or more of the candidate sites; if found suitable, one of these was to have been converted to an operating repository to receive TRU wastes by 2000 and HLW by 2010. However, in light of the serious public protests at three of the sites under investigation, former Prime Minister Michel Rocard declared a one-year moratorium on siting activities in February 1990 to allow a reassessment of the overall French waste management strategy. The Parliamentary Office for the Assessment of Technological Options published a report in January 1991 recommending major changes to the program, and the Parliament enacted a new Law on Radioactive Wastes on December 30, 1991.

The 1991 law allows the government to resume site selection efforts for underground laboratories. A waste "negotiator" will be appointed to discuss proposed investigations with local and regional officials, and the government is expected to select two sites to host laboratories. Only research quantities of waste may be emplaced in these laboratories. The law calls on the government to submit a report to Parliament within 15 years assessing the results of studies on partitioning and transmutation of actinides, use of test facilities for retrievable and permanent storage of HLW, and technologies for waste conditioning and surface storage. In addition, the government report to the Parliament must propose a bill to authorize an underground waste repository. The law does not establish a schedule for developing a HLW repository; Parliament will reassess the program based on the results of the 15-year research phase.

In preparation for the underground laboratory phase, the Institute for Nuclear Protection and Safety (IPSN) within CEA is independently preparing facilities to evaluate the long-term safety requirements of a HLW repository, on behalf of the French regulatory authority. IPSN operates two Methodological and Instrumental Laboratories in a granite formation near Limoges; it is preparing two similar facilities in clay and schist formations.

4.6.4 Germany

Germany sends spent fuel to foreign reprocessors and will receive vitrified HLW in return, which it intends to dispose in deep geological formations. The Federal government's Institute for Radiation Protection (BfS) is responsible for the design, construction and operation of waste disposal facilities. BfS intends to dispose of non-heat generating low- and intermediate-level wastes in the Konrad repository, an abandoned iron ore mine in Lower Saxony in the north central part of unified Germany. Vitrified HLW returned from foreign reprocessors and other heat-producing wastes will eventually be disposed at the Gorleben facility, a salt dome also located in Lower Saxony, if the site proves acceptable. Vitrified waste will be stored at Gorleben and another facility, Ahaus, until the repository is ready for operation.

A former salt mine at Asse, which served until 1978 as a repository for the disposal of 125,000 containers of low-level and smaller quantities (1,300 drums) of intermediate-level radioactive wastes, now serves as a URL.

The Gorleben facility will be situated at depths ranging from 250 to 3,000 meters. The geology of the site has been widely investigated by exploratory drilling and by geophysical measurements. In 1986, the construction of a URL was initiated. In 1987, all work was stopped for over one year because of a construction fatality. Pending completion of the site characterization studies in the mid-1990s, the construction of the repository could start at the turn of the century. Once opened, the facility is planned to be remain operational for as long as 60 years.

4.6.5 Belgium

The Belgian research and development program to establish a radioactive waste repository was initiated in 1974. A national agency, ONDRAF, was established to take the responsibility for implementing and managing a multi-year national program. The Belgian waste management program has included domestic spent fuel reprocessing in the past, but spent fuel is now either sent to France for reprocessing or stored in reactor pools. Long-term storage of solidified wastes is planned, followed by construction of a repository located in a deep clay formation at the Mol-Dessel site.

Investigation of the Mol-Dessel site as a candidate for the Belgian radioactive waste repository began in 1975. The site is situated in a deep clay formation and is the only suitable geological medium identified in Belgium. By 1980, a repository conceptual design was developed for a clay site, and by 1985 a URL at Mol-Dessel (Project HADES) was declared operational. The underground laboratory extends to a depth of 224 meters, and since 1987 a new experimental gallery has been added to the original facility. The purpose of the additional gallery is to conduct high specific-activity disposal experiments and pilot studies. The studies include experiments in corrosion properties of containers and engineered barriers, geochemistry and radionuclide migration, backfilling and sealing technology, and near-field effects of heat and radiation on clays. Based on the outcome of these studies, a larger underground facility will be constructed for a full-scale demonstration project.

Assuming that the results of investigations at Mol-Dessel are favorable, repository construction could begin around 2025 and operation around 2030.

4.6.6 Switzerland

The responsibility for establishing radioactive waste disposal facilities lies with the National Cooperative for the Storage of Radioactive Waste (NAGRA). NAGRA plans to begin construction of an intermediate-depth repository for low- and intermediate-level wastes by no later than 2000 and field studies have been conducted at four candidate sites (Bois de la Glaivaz, Oberbauenstock, Piz Pian Grand and Wellenberg). NAGRA plans to select the preferred site by mid-to-late 1993 for full characterization.

With regard to developing a deep geologic repository for HLW and TRU waste, NAGRA has performed extensive field work in crystalline rock formations and will synthesize all project work based on crystalline rock during 1992-1993. To support the crystalline rock studies, investigative techniques and equipment have been tested in an underground laboratory at the Grimsel Test Site (which is not a candidate repository site). In

addition, a broad survey of sedimentary formations under way since 1988 has resulted in the selection of clay and a variant of freshwater molasse for further study. NAGRA plans to choose between clay and molasse by the end of 1993 (NAGRA gives higher priority to clay) as the sedimentary medium for further study, and conduct field work in the selected medium by 1997. NAGRA must submit a program -- the Siting Feasibility Project -- for government approval by 2000 demonstrating the feasibility of siting a repository in one or more of the crystalline or sedimentary media under consideration, and intends to choose by 1997 which medium or media to present in that program. Commissioning of a repository will not occur before 2020 to allow a 40-year spent fuel/HLW cooling period. Participation in any international repository projects that may develop is also under consideration.

4.6.7 Sweden

Following a 1980 national referendum, the Swedish Parliament decided to phase out nuclear power plants by the year 2010. Consequently, Swedish utilities sold their contract rights to foreign reprocessing services. The Swedish Nuclear Fuel and Waste Management Company (SKB) began operating a centralized spent fuel storage facility (CLAB) in 1985 that will eventually hold all Swedish spent fuel (about 8,000 metric tons) for about 40 years. The facility is situated in an underground granite cavern at a depth of 30 meters, near an existing nuclear power plant (Oskarshamn). A repository for short-lived low- and intermediate-level wastes, SFR, began operating in 1988 near the Forsmark nuclear power plant.

SKB's reference disposal concept for spent fuel is to encapsulate it in high-integrity copper canisters and emplace the canisters in a repository built in crystalline rock at a depth of about 500 m, backfilling the deposition holes with highly-compacted bentonite and the tunnels and shafts with a mixture of sand and bentonite. SKB is evaluating alternative concepts such as deep boreholes and tunnel emplacement, as well as alternative canister designs. Three candidate repository sites are to be identified in 1993, followed by preliminary characterization of the sites, to be completed around 1996. Subsequently, two sites would be characterized in detail, beginning in 1997 and lasting about six years. SKB would file a license application for one site in 2003. Construction is anticipated to begin around 2010 and operation around 2020.

The international OECD/Nuclear Energy Agency conducted an international research project in a URL at Sweden's Stripa mine from 1980 to 1991. SKB has decided to build a second laboratory under the island of Äspö, 2 km north of Oskarshamn, as a means of preparing for site selection, site characterization and licensing the spent fuel repository. Construction of the Äspö Hard Rock Laboratory began in October 1990; the facility is scheduled to begin operation by the end of 1994 at a depth of 500 m.

4.6.8 Japan

Under 1985 plans for waste management published by the Atomic Energy Commission and an R&D plan announced by the Science and Technology Agency in 1986, the Power Reactor and Nuclear Fuel Development Corporation (PNC) has the lead responsibility for HLW disposal R&D, while the Japan Atomic Energy Research Institute (JAERI) and others share in the R&D work. The current waste management strategy includes spent fuel

reprocessing using domestic and foreign facilities, on-site spent fuel storage, waste solidification followed by long term storage (30-50 years), and eventual disposal in a suitable deep geological formation.

The site selection process for an HLW repository consists of four phases: 1) selection of effective formations (completed in 1984); 2) selection of a candidate disposal site (now underway); 3) demonstration of the disposal technology at the candidate site; and 4) construction, operation and closure of the disposal facility. The conclusion of the first phase was that HLW disposal should be possible in any geologic formation excluding unconsolidated media (e.g. soil and sand). The site selection phase currently in progress emphasizes generic R&D at sites that are not candidates to host the repository. The demonstration phase is expected to begin at a candidate repository site by 1995. Because of geological heterogeneities in Japan, geological characterization is expected to be difficult, causing uncertainties in predicting the performance of natural barriers. Thus, Japan is assigning a major role to the engineered barrier system, while defining a small number of critical natural characteristics for the site which are expected to be achievable in various geological settings.

PNC operates an underground test facility in the Tono Uranium Mine in central Japan, in both sedimentary and crystalline rock environments. Major experiments in the Tono Mine include a ground-water flow investigation, studies on the effects of excavation on the mechanical and hydraulic behavior of the repository, natural analogue studies and evaluations of the chemical durability of simulated waste glasses and the corrosion rates of candidate overpack materials. In addition, PNC is conducting tests in the Kamaishi iron ore mine in northern Honshu. Major investigations at Kamaishi have included detailed fracture mapping, cross-hole hydraulic and geophysical testing, drift excavation-effect studies and in-situ stress measurements, single-fracture flow tests and observations of seismic activity. Furthermore, PNC is conducting analogue studies on the stability of glass, iron, concrete and bentonite in natural settings.

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Chapter 5: RADIATION DOSIMETRY

5.1 INTRODUCTION

The setting of standards for radionuclides requires an assessment of the doses received by individuals who are exposed by coming into contact with radiation sources. Two forms of potential radiation exposures can occur from these sources -- 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 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 radionuclides.

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.

5.2 BASIC CONCEPTS

Radioactive materials produce radiation as their constituent radioactive nuclides undergo spontaneous radioactive decay. The mechanisms of emitting this energy are characteristic of the decay process and include energetic charged particles (alpha and beta particles) and photons (gamma rays 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 that they traverse. Beta particles are electrons or positrons emitted in radioactive decay. Their penetration power in material is greater than that of alpha particles. Gamma and x-rays are electromagnetic radiation and are distinguishable from alpha and beta particles by their greater penetrating power in material.

This section introduces some terminology used in Chapters 5 and 6 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 (NCRP71).

5.2.1 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

$$A_i(t) = -dN/dt = \overset{R}{\lambda_i} N , \tag{5-1}$$

where $\overset{R}{\lambda_i}$ 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), are also often used. The curie, which is defined as 3.7×10^{10} disintegrations per second, is the approximate activity of 1 gram (g) of radium-226.

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

$$A_i(t) = A_{oi} \exp(- \overset{R}{\lambda_i} t). \tag{5-2}$$

A_{oi} is the activity of nuclide i 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:

$$A(t) = \Sigma_i A_i(t) \tag{5-3}$$

5.2.2 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., the time t that $A_i(t) = A_{oi}/2$) is called its radioactive half-life, T_i^R, and is defined as:

$$T_i^R = (\ln 2)/ \lambda_i^R \tag{5-4}$$

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.

5.2.3 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 (^{90}Sr) decays by emitting a beta-particle, producing the daughter yttrium-90 (^{90}Y), which also decays by beta emission to form the stable atom zirconium-90 (^{90}Zr):

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

5.2.4 Biological Half-Life

The biological half-life of radionuclides is the time required for biological tissues to eliminate one-half of the activity by elimination processes. This time is the same for both stable and radioactive isotopes of any given element.

5.2.5 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 in 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 the emitted 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 in 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.

5.2.6 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

$$D = d\bar{\varepsilon}/dm = \lim_{\Delta m \to 0} (\Delta\bar{\varepsilon}/\Delta m). \qquad \text{(rad) (5-6)}$$

Internal and external exposures from radiation sources 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, $\dot{D}$

$$\dot{D} = dD/dt = \lim_{\Delta t \to 0} (\Delta D/\Delta t). \qquad \text{(mrad/y) (5-7)}$$

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 millirads per year (mrad/yr).

5.2.7 Linear Energy Transfer (LET)

Linear energy transfer, L_∞, is the loss of kinetic energy, by collision, by charged particles per unit length of an absorbing medium. The increment of the mean energy lost, ΔE, to tissue by a charged particle of specified energy in traversing a distance, ΔX:

$$L_\infty = dE/dX = \lim_{\Delta x \to 0} (\Delta E/\Delta X) \qquad (\text{keV } \mu\text{m}^{-1}) \quad (5\text{-}8)$$

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) imparts more energy per unit length of organ tissue than does low-LET radiation (x-rays, gamma rays, and beta particles). Consequently, the former are more effective per unit dose in causing biological damage.

5.2.8 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 that 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:

$$H = D\ Q\ N. \qquad (\text{rem}) \quad (5\text{-}9)$$

The quality factor is a dimensionless quantity that depends on the collision stopping power for charged particles. It accounts for the differences in biological effectiveness found among varying types of radiation. By definition, it is independent of tissue and biological endpoint. The generally accepted values for quality factors for high- and low-LET radiation, which are used by EPA, are given in Table 5-1. The product of all other modifying factors, N, such as dose rate, fractionation, etc., is taken as 1.

Table 5-1. Quality factors for various types of radiation (ICRP77).

Radiation Type	Quality Factors (Q)
x-rays, gamma rays, and electrons	1
alpha particles	20

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

$$H = dH/dt = \lim_{\Delta t \to 0} (\Delta H/\Delta t). \qquad \text{(mrem/yr) (5-10)}$$

5.2.9 Effective Dose Equivalent and Effective Dose Equivalent Rate

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

$$H_E = \Sigma_T \, w_T \, H_T, \qquad \text{(rem) (5-11)}$$

where H_T is the dose equivalent in tissue and w_T is the weighting factor, which represents the estimated proportion of the stochastic risk resulting from tissue, T, to the stochastic risk when the whole body is uniformly irradiated (ICRP77). The weighting factors recommended by the ICRP are listed in Table 5-2.

Table 5-2. Weighting factors recommended by the ICRP for stochastic risks (ICRP77).

Organ or Tissue	W_T
Gonads	0.25
Breast	0.15
Red Bone Marrow	0.12
Lung	0.12
Thyroid	0.03
Bone Surfaces	0.03
Remainder	0.30

The effective dose equivalent rate is the time rate of the delivery of the dose equivalent and is expressed as H_E, where:

$$\dot{H}_E = \Sigma_T \, w_T \, \dot{H}_T \, . \qquad \text{(mrem/yr) (5-12)}$$

5.2.10 Relationship of the Dose Equivalent and the Effective Dose Equivalent to Risk

The dose equivalent was introduced by the ICRP to allow one to combine and compare - on the basis of biological effects - absorbed doses of different types of radiation. Subsequently, the effective dose equivalent was introduced to provide a single-valued indicator of risk for dose equivalents distributed nonuniformly in the body. By convention, these concepts, in combination with the ICRP-recommended quality factors and organ-weighting factors, are widely used in radiation protection. These recommended factors, however, are based on dose response models that differ significantly from those used by EPA to estimate risk (see Chapter 6).

To calculate risk, EPA first calculates age-specific, high- and low-LET absorbed dose rates, by organ, for a uniform intake or external exposure rate. The risk from each year's dose is then calculated using the life table procedure in conjunction with age- and organ-specific risk models adapted from the BEIR III report (NAS80).

These models (see Chapter 6) assume a linear dose-response relationship and a lifetime relative risk projection for cancers other than bone cancer and leukemia, for which absolute risk projection is employed. Finally, the risks from each year's dose are summed to arrive at the risk from lifetime exposure.

In calculating dose equivalents and effective dose equivalents, the ICRP Publication 30 convention was employed, including the same quality factors and organ-weighting factors. Nevertheless, in calculating the risk from a given absorbed dose of alpha particle irradiation, RBEs of 8 and 2.7 were used for the induction of cancers and genetic effects, respectively (see Chapter 6). Since these RBEs are lower than the assumed alpha quality factor (Q=20), EPA's estimates of the risk per unit dose equivalent (mrem) will be lower for alpha particles than for x-rays or gamma rays. Likewise, the ICRP organ-weighting factors shown in Table 5-2 do not stand in the same proportion as the organ risks calculated using the EPA models for cancer induction or genetic mutations. Furthermore, EPA considers somatic and genetic risks separately. Thus, even if attention was restricted to low-LET radiation, the estimated risk from a given effective dose equivalent will vary, depending on how the absorbed dose is distributed within the body.

To summarize, because EPA risk models differ from those underlying the ICRP recommendations, the risks calculated directly by EPA are not strictly proportional to the effective dose equivalents derived using ICRP quality factors and organ weighting factors.

5.2.11 Working Levels and Working Level Months

The working level is a unit that has been used as a measure of the radon decay-product activity in air. It is defined as any combination of short-lived radon daughters (through polonium-214) per liter of air that will result in the ultimate emission of 1.3×10^5 million electron-volts (MeV) of alpha energy. An activity concentration of 100 pCi/L of radon-222 in equilibrium with its short-lived daughters gives rise to a potential alpha-energy concentration of approximately 1 WL. The WL unit could also be used for thoron daughters. The potential alpha energy exposure is commonly expressed in units of working level month (WLM). One WLM corresponds to an exposure to a concentration of 1 WL for the commonly used reference period of 170 hours.

5.2.12 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 5-3. While the SI radiological units are almost universally used in other countries for radiation protection regulation, the United States has not yet officially adopted their use for such purposes.

Table 5-3. Comparison of customary and SI special units for radiation quantities.

	Customary Unit		Special SI Unit	
Quantity	Name	Definition	SI Unit	Definition
Activity (A)	Curie (Ci)	3.7×10^{10} s^{-1}	becquerel (Bq)	1.0 s^{-1}
Absorbed dose (D)	rad	10^{-2} J kg^{-1}	gray (Gy)	1.0 J kg^{-1}
Dose equivalent (H)	rem	10^{-2} J kg^{-1}	sievert (Sv)	1.0 J kg^{-1}
Linear energy transfer (L_∞)	keV μm^{-1} (kiloelectron volts per micrometer)	1.602×10^{-10} J m^{-1}		

5.3 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. In most cases, the EPA 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).

5.3.1 Internal Dose Models

EPA implements contemporary models to estimate absorbed dose rates as a function of time to specified organs in the body. Estimates of the doses resulting from 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).

5.3.1.1 Generalized Scheme for Estimating Organ Absorbed Dose Rates

5.3.1.1.1 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 region of the 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 5-4. 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 5-1 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 body. 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 contrasts the ICRP assumption that daughters behave exactly as the parent.

Table 5-4. Target organs and tissues used for calculating the ICRP effective dose equivalent and the EPA cancer risk.

ICRP effective dose equivalent	EPA cancer risk
Ovaries	
Testes	
Breast[a]	Breast
Red marrow	Red marrow
Lungs[b]	Pulmonary lung[c]
Thyroid	Thyroid
Bone surface	Bone surface (endosteum)
Stomach wall	Stomach wall
Small intestine wall	Intestine[d]
Upper large intestine wall	
Lower large intestine wall	
Kidneys	Kidneys
Liver	Liver
Pancreas	Pancreas[e]
Brain	
Spleen	
Thymus	
Uterus	
Adrenals	
Bladder wall	

a) Dose to breast is assumed to equal dose to muscle.

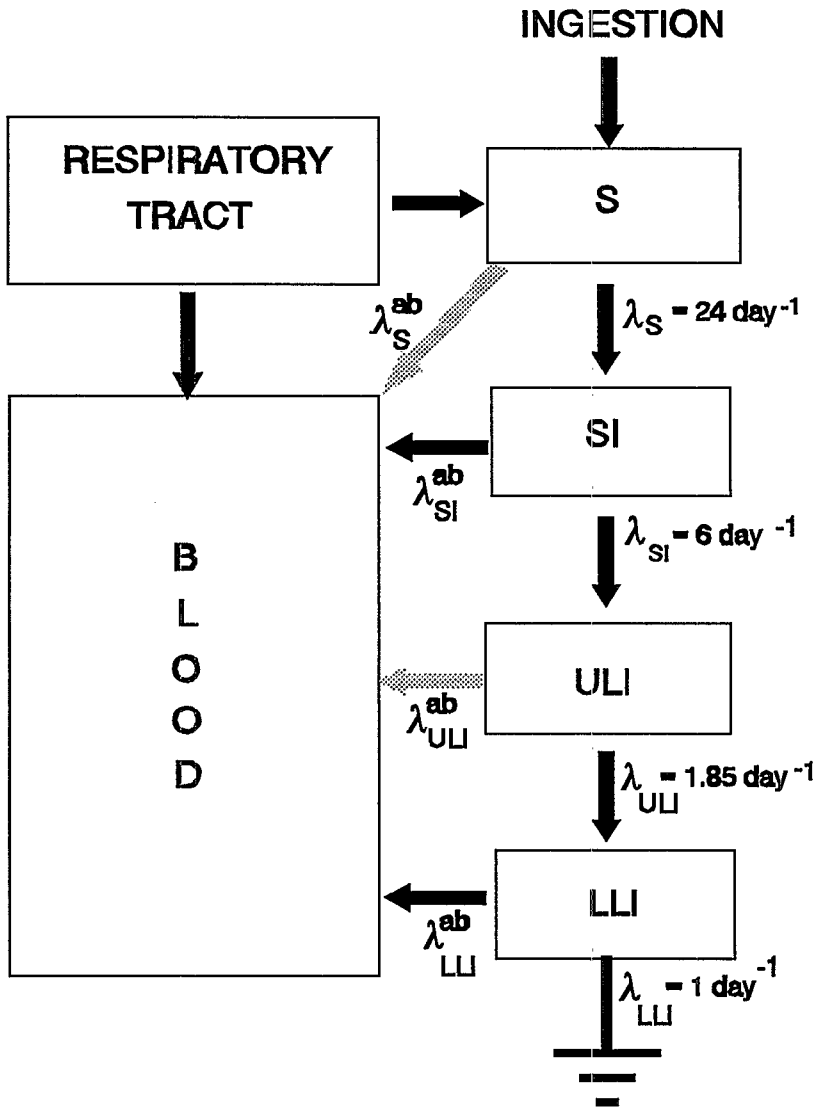
b) The ICRP considers the lungs to be a composite of the tracheo-bronchial region, pulmonary region, and the pulmonary lymph nodes with a combined mass of 1,000 g (ICRP79).

c) The EPA calculates lung cancer risk on the basis of the dose to the pulmonary lung. The mass of this region, which does not include venous or arterial blood, is considered to be 570 g.

d) The EPA averages the values for the small, upper large, and lower large intestine using weights of 0.2, 0.4, and 0.4 respectively for calculating the risk of bowel cancer.

e) The pancreas is also used as a surrogate organ for calculating the cancer risk for all other organs and tissues.

Figure 5-1. A schematic representation of radioactivity movement among respiratory tract, gastrointestinal tract, and blood.



S = stomach
SI = small intestine
ULI = upper large intestine
LLI = lower large intestine
λ = elimination rate constant

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:

$$A_{ilk} = -(\lambda_i^R + \lambda_{ilk}^B)A_{ilk} + C_{ilk}(\lambda_i^R \ \Sigma_{j=1}^{i-1} B_{ij} \ \Sigma_{r=1}^{L_{jk}} A_{jr} + p_{ik})$$
$$l = 1.....L_{ik} \qquad (5\text{-}14)$$

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 of species i;

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

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

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

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

c_{ik} = 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:

$$R_{ik}(t) = \Sigma_{l-1}^{L_{ik}} \ c_{ilk} \ \exp[-(\lambda_i^R + \lambda_{ilk}^B)t] \qquad (5\text{-}15)$$

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."

5.3.1.1.2 <u>Dose Rates to Target Organs</u>

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, $\dot{D}_i$ (X;t) to target organ X at time t due to radionuclide species i in source organs $Y_1, Y_2, ..., Y_M$ is estimated by the following equation:

$$D_i(X;t) = \Sigma_{k=1}^{M} \; D_i(X \leftarrow Y_k;t) \tag{5-16}$$

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:

$$S_i(X \leftarrow Y_k) \;=\; c\ \Sigma_m\ f_m\ E_m\ \phi_m(X \leftarrow Y_k) \tag{5-17}$$

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 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. 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 type of 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 ... Y_k:

$$D(X;t) = \Sigma_k\ \Sigma_m\ A_{ik}(t)\ S_{im}(X \leftarrow Y_k) \tag{5-18}$$

The corresponding dose equivalent rate, $H_i(X;t)$, can be estimated by inclusion of the quality factor, Q_m, and the modifying factor, $N_m(Y_k)$:

$$H_i(X;t) = \Sigma_k \Sigma_m \; A_{ik}(t) \; Q_m N_m(Y_k) \; S_{im}(X \leftarrow Y_k) \qquad (5\text{-}19)$$

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 by beta particles among skeletal tissues were taken from ICRP Publication 3 (ICRP80). 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).

5.3.1.1.3 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).

5.3.1.1.4 Effects of Decay Products

In calculating doses from internal and external exposures, the in-growth 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.

5.3.1.2 Inhalation Dosimetry - ICRP Respiratory Tract Model

As stated earlier, individuals immersed in contaminated air will breathe radioactive aerosols or particulates which can lead to doses to the lung and other organs in the body. The total internal dose caused by inhalation of these aerosols can depend on a variety of factors, such as breathing rates, particle sizes, and physical activity. Estimating the total dose to individuals over a specific time period requires specifying the distribution of particle depositions in the respiratory tract and the mathematical characteristics of the clearance parameters. The EPA currently uses assumptions established 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.

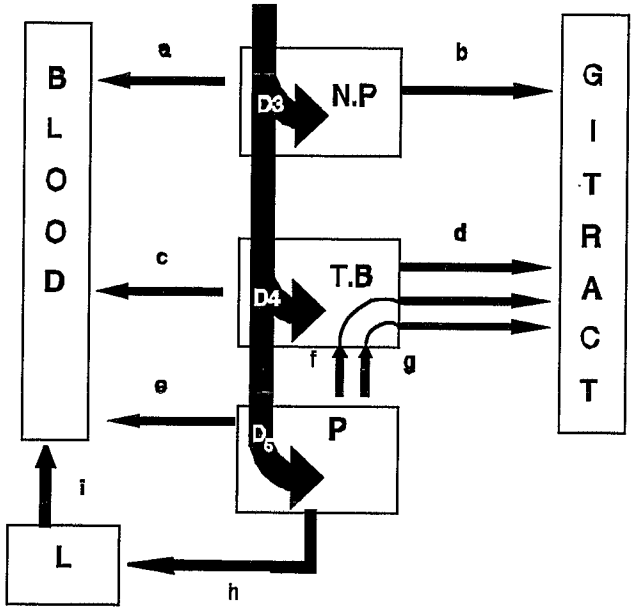
The basic features of the ICRP lung compartmental model are shown in Figure 5-2. According to this model, the respiratory tract is divided into four regions: naso-pharyngeal (N-P), tracheo-bronchial (T-B), pulmonary (P), and lymphatic tissues.

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. For purposes of risk calculations, EPA uses AMADs 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.

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 (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 22,000 liters. This value was determined by averaging the ICRP-23 adult male and female values based on 8 hours of working "light activity," 8 hours of nonoccupational activity, and 8 hours of resting.

Figure 5-2. The ICRP Task Group lung model for particulates

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
(D_5 - 0.25)	g	n.a.	n.a.	50	0.4	500	0.4
	h	0.5	0.2	50	0.05	500	0.15
L	i	0.5	1.0	50	1.0	1000	0.9



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 retension function. The values shown for D_3, D_4, and D_5 correspond to activity median aerodynamic diameter, AMAD = μm, and represent the fraction of the inspired material depositing in the lung regions.

5.3.1.3 Ingestion Dosimetry - 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 5-1. It is assumed that absorption into the blood occurs only from the small intestine (SI).

This model postulates that radioactive material entering 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:

$$f_1 = \lambda_{SI}^{ab} / (\lambda_{SI}^{ab} + \lambda_{SI}) \tag{5-20}$$

where

λ_{SI}^{ab} = the absorption coefficient (s^{-1})

λ_{SI} = the transfer coefficient from the small intestine to the large intestine (s^{-1})

Figure 5-1 graphically presents the role of these coefficients in the gastrointestinal model. The kinetic model, as formulated by the ICRP, does not permit total absorption of a nuclide ($f_1 = 1$).

5.3.1.4 Dose Rate Conversion Factors

EPA uses the computer code RADRISK (Du80) for calculating radiation doses and risks to individuals resulting from a unit intake of a radionuclide, at a constant rate, for a lifetime exposure (50-yr dose commitment). These calculations are done for the inhalation and ingestion pathways to 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 assumed 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.

5.3.1.5 Special Radionuclides

The following paragraphs briefly summarize some of the special considerations for particular elements and radionuclides.

5.3.1.5.1 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 to be rapidly mixed 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.

5.3.1.5.2 Noble Gases

Retention of noble gases in the lungs is treated according to the approach described by Dunning et al. (Du79). The inhaled gas is assumed to remain in the lungs until lost by radiological decay or respiratory exchange. Translocation of the noble gas to systemic organs is not considered, but doses due to translocated decay products produced in the lungs are calculated. The inhalation of the short-lived decay products of radon is assessed using a potential alpha energy exposure model (see Chapter 6) rather than by calculating the doses to lung tissues from these radionuclides.

5.3.1.5.3 Uranium and Transuranics

The metabolic models for transuranics elements (polonium, neptunium, plutonium, americium, and curium) are consistent with those used for the EPA transuranic guidance (EPA77). 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 (Hu73, Sp73).

5.3.1.6 Uncertainties in Internal Dose Estimates

Estimates of radiation dose in risk assessment studies have traditionally been based on dosimetric models derived in the context of radiation protection for adult workers. Despite the obvious differences between risk assessment and radiation protection, the dosimetric formulations of the latter have been generally adopted, often with no modifications, in risk assessment activities. This approach permits use of a substantial body of information assembled by international experts from the occupational setting and provides models that avoid the complex problems encountered in biokinetic modeling of radionuclides for the general public in an age-dependent sense. However, the continued use in risk assessment of dosimetric data derived for workers, which neglects organ-specific biokinetics and age dependence, is becoming increasingly difficult to justify. One major limitation of the current ad hoc dosimetric formulations is the great difficulty in making informed estimates of the uncertainties in the estimated dose.

All dosimetry models are inherently uncertain. At best, these models can only approximate real situations in organs and tissues in humans. Consequently, without extensive human data, the

uncertainties associated with their use for risk assessment purposes is extremely difficult, and in some cases impossible, to quantify. However, consideration of their limitations in estimating doses to an average member of the general population is essential.

In applying the dosimetric models in current use, as discussed in the previous sections, the primary sources of uncertainty are attributed to ICRP model formulation and parameter variability produced by measurement error or natural variation. The purpose of this section is to provide a general but limited discussion of these sources and to introduce an uncertainty scheme for classifying radionuclides. The authors gratefully acknowledge Dr. Keith Eckerman of Oak Ridge Laboratory for discussions with respect to implementation of ICRP models and for guidance regarding the magnitude of uncertainties. However, the conclusions presented here are those of the Agency.

5.3.1.6.1 Uncertainties Due to ICRP Model Formulation

Uncertainty in calculations based on ICRP models arises primarily from five sources: (1) the uncertainty in the Reference Man data; (2) the uncertainty in the lung and GI-tract model describing the translocation and absorption of inhaled or ingested activity into the blood; (3) the uncertainty associated with the formulation of the ICRP Publication 30 biokinetic models describing the distribution and retention of the activity among the various organs in the body; (4) the uncertainty in the dose models to calculate the absorbed dose to organs from that activity; and (5) the uncertainty in the model parameters.

5.3.1.6.2 Reference Man Concept

To establish a degree of consistency in occupational dosimetry calculations, the ICRP developed the concept of Reference Man (ICRP75). Reference Man is a conceptual individual who has the anatomical and physiological characteristics of a healthy 20 to 30 year old male with a total body mass of 70-kg. The anatomical and physiological data of Reference Man have been embedded in many computational models for estimating organ doses and applied in radiation protection and in some calculations for medicine.

Although these data have been extensively applied in calculating doses, the approach in which Reference Man data is used to represent average individuals in a specific population introduces bias from the outset. The uncertainties in this approach are primarily due to age- and sex- specific differences in the anatomical and physiologic parameters. Biological and ethnic variability also contribute. In addition, the Reference Man data do not always represent data for a 70-kg man. Many of the data found in ICRP Publication 23 were from adults who had anatomical or physiological characteristics significantly different from those of a 70-kg man.

Due to the many parameters involved and the quality of the data available to define the numerical values, it is very difficult to establish the level of uncertainty in using Reference Man data to estimate doses to the average individual in the U.S. population. Furthermore, the Reference Man concept was not formulated so as to facilitate a quantitative analysis of the uncertainty in the dose estimates. Finally, Reference Man is not intended to be representative of the U.S. population.

5.3.1.6.3 ICRP Respiratory Tract Model

When individuals inhale radioactive aerosols, the dose to the lungs and other organs in the body depends primarily on how the aerosols are deposited in and cleared from the airways of the respiratory tract. Mechanisms involved in the deposition of inhaled aerosols and gases are affected by physical

and chemical properties, including aerosol size distribution, density, shape, surface area, electrostatic charge, chemical composition and gas diffusivity and solubility. Deposition is also affected by respiratory physiology, morphometrics and pathology.

The ICRP modeling system assumes that deposition rates for aerosols in the respiratory tract are controlled primarily by three mechanisms: sedimentation, impaction and Brownian diffusion. The major uncertainties associated with the ICRP deposition models for the lungs are: (1) the uncertainty in the anatomical model of the respiratory tract, (2) the uncertainty in the effective aerodynamic diameter of the inhaled particles, (3) the uncertainty in the breathing patterns and rates, and (4) the questionable validity of the fluid dynamic models used for all exposure situations.

The number of particles deposited in the lung essentially depends on physiologic, morphometric and anatomical properties, such as airway dimensions and numbers, branching and gravitational angles of airways, and distances to the alveolar walls. The ICRP respiratory tract model (ICRP66) uses the anatomical model devised by Findeisen (Fi35) in its dosimetric calculations. This model assumes that lung airways are rigid tubes with symmetric dichotomous branching patterns and that their morphometric properties are those of an adult male. In reality, however, the airways have circular ridges or longitudinal grooves (FRC67), and many airways, like the trachea, are irregular in shape (Br52). In addition, airways change in diameter and length during inspiration and expiration (Ho75, Hu72, Th78), which affects gravitational and branching angles (Ph85). Since many of these properties depend on age and sex, using the anatomic and morphometric lung properties of an adult male for estimating doses to other members of the population is likely to introduce considerable bias.

Clearance of particles from the respiratory tract depends on many factors, such as site of deposition, chemical composition, physical properties of the deposited material, and mucociliary transport rates. The uncertainties associated with using the values provided by the ICRP are due primarily to the sparseness of data on lung clearance mechanisms, in general, and secondarily to age, activity levels and general health status of the individual at the time of exposure. Furthermore, as stated earlier, most of the lung deposition data and models are derived from studies of healthy adults. Studies have shown, however, that children's lungs differ from adults' with respect to anatomical, physiological, and morphological properties. As a consequence, particle deposition in the respiratory tract is expected to be higher in children than in adults.

5.3.1.6.4 <u>ICRP GI-Tract Model</u>

The ICRP GI-tract model assumes that ingested material (radionuclides) 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. The model has no provision for addressing endogenous secretion. In addition, it is assumed that radionuclides are absorbed into the blood from the small intestine (SI).

Uncertainties arise when applying these assumptions to the estimation of doses to average individuals. Although radionuclides transported through the GI tract are primarily absorbed into the blood stream from the SI, fractions can be absorbed from the other compartments. Furthermore, the removal rates, which are model parameters, vary among different individuals in the population. Considerable differences can exist depending on the type of radionuclide ingested, its chemical form, the amount and composition of food in the stomach at the time of intake and other factors which vary because of nutritional status, age, and the sex of the individual. The f_1 factor, which represents the fraction of material absorbed from the SI, generally contributes the largest uncertainty in the GI tract model. This parameter will be discussed in a later section.

5.3.1.6.5 ICRP 30 Biokinetic Models

The ICRP biokinetic models were chosen to represent adult male members of the population. Uncertainties are associated with the approach because they do not account for differences in the metabolic behavior of radionuclides, which vary depending on age, sex, and dietary intakes of an individual at the time of exposure. In addition, many of the models chosen for dosimetry calculations are based on very limited observational data that cannot be reliably applied across the population.

Below is a list of additional uncertainties associated with the ICRP biokinetic models:

a) The models have been constructed largely from animal data in such a way that extrapolation to humans has no strong logical or scientific support.

b) Doses to heterogeneously distributed radiosensitive tissues of an organ (e.g., skeletal and lung tissues) cannot be estimated accurately, since the actual movement of radionuclides in the body is not accurately tracked.

c) Some radionuclides are assigned the model of an apparently related nuclide (e.g., americium, curium, neptunium are assigned the plutonium model) although differences in metabolism are known.

d) The growth of radioactive daughters is often not handled realistically, and the format of the models makes it difficult to supply alternative assumptions.

e) The models often yield inaccurate estimates of excretion even for the average adult.

5.3.1.6.6 ICRP Dose Models

ICRP models estimate doses to organs of the body by considering the distribution of the radioactivity and the interaction of radiation with cells and tissues in these organs. Estimates of the absorbed dose in a region (referred to as the target region) depend upon the spatial relationships of that region to the regions containing the radionuclide (referred to as source regions) and how the activity is distributed in the source region. For organs other than bone, it is assumed that the radionuclides are uniformly distributed in the source regions and that the radiosensitive cells of interest are uniformly distributed in the target region. However, this assumption may bias the dose estimates because of the nonuniformity of the activity that is normally found in human organs.

5.3.1.6.7 Uncertainties Due to Parameter Value Variability

Most discussions concerning the uncertainties in dose estimates focus on the uncertainty associated with model parameter values. These discussions assume that the ICRP metabolic and dose models are correct. The most important parameters of concern for dose assessment calculations are: radionuclide intake rates, organ masses, blood transfer factors, organ uptake rates, and biological half-times of radionuclides. Although parameter value variability can be attributed to measurement and sampling errors and natural biological variation, in many cases, age is the largest source of variability.

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, even though the body is able to discriminate somewhat against strontium in favor of calcium after the first few weeks of life.

Given the importance of age as a contributor to parameter variability in dose estimates, the possible age dependence in thyroid dose for chronic ingestion of a fixed iodine-131 concentration in milk is examined in more detail below. Some other examples of parameter variability will also be noted.

A simple model that can be used to relate the absorbed dose rate to a target organ due to radioactivity located in that organ can be expressed as follows :

$$\dot{D}(t) = c\ I\ f_1\ f_2'\ E\ [1-\exp(-\lambda t)]/m\lambda \qquad (5\text{-}21)$$

where:

$\dot{D}(t)$ = 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 = elimination constant (day^{-1}) = $0.693/T_{1/2}$, where $T_{1/2}$ is the effective half-time, including the effects of both biological removal and radioactive decay.

E = energy absorbed by the target organ for each radioactive transformation.

c = proportionality constant (51.2×10^6g 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:

$$\dot{D}(t) = c\ I\ f_1\ f_2'\ E/m\lambda \qquad (5\text{-}22)$$

In addition, it is assumed that the parameters remain constant throughout the period of investigation and are independent of each other.

Equation 5-22 is a simplified form of the model 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.

For this illustration, the chronic intake of iodine-131 is considered 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. Two papers have used published data to model the absorbed dose from radioiodine. In the first (Du81), the authors compiled and evaluated the variability in three of the principal biological parameters contained in Equation 5-22: m, λ, and f_2. In the second (Br69), the author provided age-specific values for most of the same model parameters. Differences in these data illustrate how parameter variability, when used in the same model, can affect absorbed dose rate estimates for members of the general population.

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 containing such material, which, in turn, 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 analyses 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 analysis, the daily intake rate of milk by children (under 1 yr) was twice that for an adult (25 to 29 yr) male. The intake rates for milk used in the models are 0.7 L/day and 0.5 L/day for the child and adult, respectively.

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 (Cr83). For some radionuclides, the f_1 value appears to decrease rapidly 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 (Cr83). 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.

For some essential elements, such as potassium and chemically similar radioelements, such as rubidium and cesium, 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 conditions. For example, f_1 values for uranium were found to range from 0.005 to 0.05 for industrial workers, but a higher average value of 0.2 (0.12 to 0.31) 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; hence the f_1 value is taken as 1 for all ages, which is the value used in this analysis.

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 5-2, 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.

Based on data reviewed by Dunning and Schwarz (Du81), the mass of an adult thyroid ranges from 2 to 62 g. It is expected that this parameter variability would be reflected in large dosimetric variability among 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. For this illustration, the same values are used as 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).

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 slightly after 24 hours. The uptake of iodine-131 by the thyroid is similar to that of stable iodine in the diet 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. This analysis uses f_2 values of .35 and .15 for a child and adult, respectively.

Effective Half-Life, $T_{1/2}$

Some data suggest 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 faster elimination rates and greater uptakes (Ro58). For iodine, a range of biological half-lives of 21 to 200 days for adults has been 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; for children in the age group from .5 to 2 yr, the range was 4 to 39 days.

In light of the possible inverse relation between the biological half-life and the f_2 value, this analysis uses biological half-lives of 24 and 129 days, respectively, for children and adults, based on the paper by Bryant (Br69). Including the effect of radioactive decay, these values imply an effective half-life of 6 days in adults and 8 days in children.

Effective Energy per Disintegration, E

The effective energy per disintegration (MeV/dis) of a radionuclide within an organ depends on 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 for most radionuclides, some information has been published for iodine-131 and cesium-137. Considering the differences between the child and the adult thyroid, Bryant (Br69) estimates E to be 0.18 MeV/dis for the child and 0.19 MeV/dis 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.

Taking into account all the age-dependent factors discussed above, this analysis indicates that, for a given concentration of I-131 in milk, the estimated dose rate to the thyroid of a 6-month-old child would be approximately 13 times that to an adult thyroid. In other words, use of adult parameters would underestimate the thyroid dose to the child by about a factor of 13.

5.3.1.6.8 Significance of Parameter Variabililty to EPA Dose and Risk Assessments

In its radiological risk assessments, EPA is generally interested in estimating the risk to an average individual due to chronic lifetime exposures. Variation in dosimetric parameter values among people and age groups is of reduced importance in this context because such variation gets averaged over a population and/or over a lifetime. Nevertheless, it should be kept in mind that some individuals in a population are going to be at higher risk from a given exposure. Furthermore, despite such averaging, parameter value variability can contribute substantially to the uncertainty in the dose and risk estimates.

Parameter value variation among individuals contributes uncertainty to the models by causing random errors in any measured human data upon which the dosimetric models are based. To the extent that the subjects from whom such data are collected are atypical of the U.S. population (e.g., with respect to health status), parameter variation may also be a source of bias. In this respect, since the parameters contained in the dosimetric models were estimated for adult males, primarily, they may not provide an adequate basis for calculating the average dose or risk in cases where age- and sex-related variations in these parameters are large. This problem becomes more significant in light of the generally higher risks associated with a given dose for childhood exposures (see Chapter 6); if doses are also higher in childhood, the enhanced effect on risk will be compounded.

5.3.1.6.9 Past Approaches Used in Estimating Uncertainties in Calculated Organ Dose

As in any predictive exercise, it is useful to question the reliability of the predictions. Variations in environmental levels, dietary and life style preferences, and the variability of controlling physiological and metabolic processes contribute to the distribution of dose among members of the

exposed population. Superimposed on this variability is a component of uncertainty arising from limitations in the predictive ability of the dosimetric models themselves. Various approaches have been taken to understand and quantify these uncertainties.

It has recently become popular to estimate the uncertainty by computing the distribution of dose among exposed individuals. This approach consists of repeated solution of the dosimetric model using parameter values selected at random from a frequency distribution of potential values suggested in the literature. It is assumed that the dosimetric model has been properly formulated, although these models were developed to yield point estimates. Despite these and other difficulties, propagation of parameter uncertainty through the dosimetric equation can provide a measure of the model uncertainty. Application of these methods to the estimation of dose from iodine-131 and cesium-137 ingestion can be found in the literature (Du81, Sc82).

An alternative approach to assessing the potential variability is to consider that the observed frequency distribution of a measurable quantity is closely related to dose. Cuddihy and co-workers (Cu79) have investigated the variability of selected target organ deposition among test animals and some individuals exposed. However, they did not address differences in age, gender, magnitude or duration of exposure.

5.3.1.6.10 Uncertainty Classification of Radionuclides

In this section, radionuclides of interest are classified in terms of the uncertainties in estimated dose per unit intake. Nuclides are placed in broad groups, largely reflecting the general status of information on their biokinetic behavior in the body. It is assumed that the uncertainty associated with the calculation of the energy deposition in the target tissues is a minor contributor to the overall uncertainty.

Classification of Uncertainty in Radionuclide Dose

Establishing numerical values of uncertainty for model dose estimates of each of the many radionuclides, for each route of exposure, is a formidable task. Even if there is agreement on the definition of uncertainty, any quantification will be arbitrary to a degree. No model has been verified in man for any long-term exposure scenario; some of the models may be fundamentally wrong in their formulation. In addition, the data selected to establish the parameters used in the model may not be representative of the population being evaluated. Most risk assessors use some informed scientific judgment in estimating the level of uncertainty in a dose model.

A broad categorization of radionuclides reflecting the estimated magnitude of the dosimetric uncertainties is presented. Because of the problems cited above with respect to the development of models and model parameters, it is quite possible that the error in model estimates may be larger than indicated in some cases. Nevertheless, this exercise is useful since it provides some perspective on the magnitude of the uncertainties in light of current evidence and focuses attention on the largest gaps in knowledge. Ultimately, however, better quantification of dose estimates and their associated uncertainties can be obtained only through the development and verification of improved dosimetric models.

Radioisotopes behave biologically like their stable elements. The elements, in turn, can be broadly grouped as: (1) essential elements and their analogs, (2) inert gases, (3) well-studied toxic metals and (4) others. Uncertainties for each of these categories will be expressed as multiplicative

factors, which roughly estimate the 95% upper and lower confidence interval limits. [Since the interval is based on judgment, a preferable term would be "credibility interval" (NIH85).]

Group I - Essential Elements and Their Analogs

Essential elements are controlled by homeostatic mechanisms to within narrow tolerances. Usually, analogs of essential elements have distribution and deposition patterns similar to those of the essential element. The uncertainty expected in calculated dose for essential elements is a factor of two or less in major critical organs, perhaps 3 or less in other significant tissues and organs. The expected dose uncertainty for analogs of essential elements is perhaps a little greater, a factor of 3 or less in major organs and up to 5 or more in less significant tissues. Important radionuclides of essential elements include hydrogen-3, carbon-14, phosphorus-32, potassium-40, calsium-45, cobalt-60, iodine-129, and iodine-131; important analogs include strontium-89, strontium-90, cesium-134, cesium-137, radium-226, and radium-228.

Group II - Inert Gases

Uptake and retention of inhaled inert gases has been fairly well studied. The uncertainty in dose, particularly average whole body dose, is not expected to be large. However, the gases do not distribute uniformly in body tissues, and the effect of distribution on organ dose estimates has not been carefully addressed. The uncertainty in the calculated dose is expected to be about a factor of 2. This group includes, but is not limited to argon-41, krypton-85, xenon-133, and radon-222.

Group III - Well-Studied Toxic Metals

A number of elements have been extensively studied in animals with limited information available for man. Examples here include toxic elements encountered in industrial activities, e.g., mercury, cadmium, lead, and uranium, for which studies were carried out to help establish safe working conditions. Often the available information is not sufficiently complete to identify the dominant processes governing the biokinetic behavior or is simply fragmentary. For example, while much information exists on the biokinetics of uranium, considerable uncertainty remains associated with the absorption to blood from the small intestine. Uncertainties for dose estimates in this group of elements would be variable, ranging from 2 or less for lead up to about 5 or more for polonium, thorium, uranium, and the transuranics. Nuclides in this group include, but are not limited to lead-210, polonium-210, uranium-235, uranium-238, thorium-230, thorium-232, plutonium-239, plutonium-241, and americium-241.

Group IV - Other Elements

For a number of radionuclides information is largely limited to data from animal studies. While animal studies often are the major source of detailed information on the processes governing the biokinetics, the lack of a general framework for extrapolations to man and the limited information upon which to judge the reasonableness of the extrapolations suggest that the estimates must be considered to be potentially in error by at least an order of magnitude. Nuclides in this group include, but are not limited to cerium-144 and other rare earth elements, technetium-99, curium-244, californium-252, etc.

The groupings listed above represent the Agency's best judgment on the uncertainty of internal radionuclide dose estimates. The primary source of uncertainty is in the biokinetic modeling with little uncertainty in the physics. The magnitudes of the uncertainties posited for each group of radionuclides

should be regarded as only rough estimates; however, the qualitative breakdown between groups is fairly reliable.

Specific Problems

Certain radioisotopes and aspects of dosimetry pose unique problems. While the effect of these problems may be to increase the uncertainty in dose estimates, the extent of such an increase has yet to be evaluated.

Long-Lived Bone Seekers

Radioisotopes with effective half-lives that are short compared to the average life span are expected to be in dynamic equilibrium. However, some bone seekers have long effective half-lives; therefore, they do not reach dynamic equilibrium during a life span. Since the relevant human biokinetic data are quite limited, dose estimates for such radionuclides are more uncertain.

Nonuniformity of Distribution

The distribution of an element within an organ may not be uniform; in particular, the distribution may be nonuniform with respect to biological targets of interest. This can be a serious problem with respect to the estimation of relevant doses from internally deposited alpha emitters, given the short range of alpha particles in matter. For example, where an alpha emitter is distributed nonuniformly in bone, the calculation of doses to sensitive cells in the bone and the bone marrow will be difficult. Another example is the uncertainty in estimating doses to cells lining the GI tract from ingested alpha emitters passing through the tract. In some cases, the mucus lining may effectively shield the target cells from irradiation.

5.3.2 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.

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 were calculated with the DOSFACTOR code (Ko81a,b). 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, the ingrowth of its decay product, metastable barium-137, which is a photon emitter, must first be calculated.

5.3.2.1 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. First, the dose rate factor (the dose rate for a unit concentration) in air is calculated for a source of photons with energy E_γ. At all points in an infinite uniform source, conservation of energy considerations

require 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

$$DRF_{\gamma}^{a}\ (E_{\gamma}) = 1/2k\ E_{\gamma}/\rho \tag{5-23}$$

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

= 1.62E-13 (J/MeV) x 3.7E+10 (dis/s-Ci) x 1.0E+3 (g/kg) x 100 (rad kg/J)

= 5.93E+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.

5.3.2.2 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 consider explicitly 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. Trubey (Tr66) has provided a useful and reasonably accurate expression to approximate this buildup:

$$B_{en}^{a}\ (\mu_a r) = 1 + C_a\ \mu_a\ r\ \exp(D_a \mu_a r) \tag{5-24}$$

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

$$DRF_{\gamma}^{a}(z,E_{\gamma}) = 1/2k(E_{\gamma}/\rho)(\mu_{en}/\rho)_a\{E_1(\mu_a z) + \tag{5-25}$$

$C_a/(1-D_a)\exp[-(1-D_a)\mu_a z]\}$

where $(u_{en}/\rho)_a$ 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.,

$$E_1(x) = \int_x^\infty \frac{\exp(-u)}{u}\, du \qquad (5\text{-}26)$$

C_a and D_a are the buildup coefficients in air at energy E_γ; and
$k=5.93\times10^2$ (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.

5.3.2.3 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

$$\boldsymbol{DRF}_\gamma^k(E_\gamma) = G^k(E_\gamma)\ \boldsymbol{DRF}_\gamma^a(E_\gamma) \qquad (5\text{-}27)$$

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 has used the same set of $G^k(E_\gamma)$ values for calculating organ dose rate factors for both types of exposure (Ko81).

5.3.2.4 Uncertainty Considerations in External Dose Rate Factors

In computing the immersion dose rate factor in air, the factor of 1/2 in Equation 5-27, 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, and 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]."

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Chapter 6: ESTIMATING THE RISK OF HEALTH EFFECTS RESULTING FROM EXPOSURE TO LOW LEVELS OF IONIZING RADIATION

6.1 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 significant.

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 is known, 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 in an individual or will cause one in the future.

Beginning nearly with the discovery of X-rays in 1895 but especially since World War II, an enormous amount of research has been 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, e.g., a cell or a tissue. This aspect (dosimetry), which may involve consideration of physiological, metabolic, and other factors, is discussed more fully in Chapter 5.

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 - is it possible to estimate risks more reliably.

Nevertheless, there are 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, probably due to the limited numbers in the exposed cohort providing inadequate power to detect a dose-response. Likewise, no statistically significant excess of cancers has been demonstrated below about 5 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 estimates of cancer and genetic risks used here are based largely on the results of a National Academy of Sciences (NAS) study done by its Committee on the Biological Effects of Ionizing Radiation (BEIR) as reported in the BEIR III 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, the various assumptions made in calculating radiation risks based on the 1980 NAS report are outlined and these risk estimates are compared 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 (UNSC77, 82, 86, 88), and the National Radiological Protection Board of the United Kingdom (St88). Because information on radiation risks is incomplete, estimates of risk based on the various models may not be highly accurate. This discussion identifies some of the deficiencies in the available data base and points out possible sources of bias in current risk estimates. Nevertheless, the risk estimates made by EPA are believed to be reasonable in light of current evidence.

Sections 6.2 to 6.2.6 consider the cancer risk resulting from whole-body exposure to low-LET (see Chapter 5) 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 in Sections 6.2.7 to 6.2.9. Sections 6.2.10 to 6.2.12 summarize recent developments in radiation risk estimation and discuss the uncertainties in the estimates.

Organ doses can also result from high-LET radiation, such as that associated with alpha particles. The cancer risks when high-LET radiation is distributed more or less uniformly within a body organ is the third situation considered (Section 6.3). 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, Po-218, Pb-214, and Po-214. For these radionuclides, cancer risk estimates are based 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 6.4).

Section 6.5 reviews and quantifies the risk of deleterious genetic effects from radiation and the effects of exposure in utero on the developing fetus. Finally, in Section 6.6, cancer and genetic risks from background radiation are calculated using the models described in this chapter.

6.2 CANCER RISK ESTIMATES FOR LOW-LET RADIATION

6.2.1 Basis for Risk Estimates

There are extensive human epidemiological data upon which to base risk estimates for radiation-induced cancers. Most of the observations of radiation-induced carcinogenesis in humans are of groups exposed to low-LET radiations. These groups include the Japanese atomic-bomb (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 (UNSC77, 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 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 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 new dosimetry system, termed the Dosimetry System 1986 (DS86), is now nearly complete, and preliminary analyses of the risk based on DS86 have been published (Pr87,88; Sh87).

At present, the "BEIR V Committee" of the National Academy of Sciences is preparing a report on radiation risks in light of DS86 and other new information. A detailed

reevaluation of EPA's current risk estimates is indicated when this report is issued. A brief discussion of the new dosimetry and its likely effect on risk estimates is included.

To derive risk estimates for environmental exposures of the general U.S. population from epidemiological studies of irradiated populations requires some extrapolation. First, much of the useful epidemiological data pertain to acute doses of 50 rad or higher, whereas we are concerned with small chronic doses incremental to the natural background level of about 100 millirad per year (mrad/year). Second, epidemiological follow-up of the irradiated study cohorts is incomplete; hence, obtaining lifetime risk estimates involves some projection of risk beyond the period of follow-up. Third, an extrapolation must be made from a study population to the U.S. population. In general, these populations will differ in various respects, for example, with respect to organ-specific, base-line cancer rates.

Data pertaining to each of these three extrapolations exist but in no case are they definitive. Hence, uncertainty in our risk estimates is associated with each of them. These uncertainties are in addition to statistical uncertainties in the epidemiological data (sampling variations) and errors in dose determinations. Generally speaking, it is the former, modeling uncertainties, which are more important.

6.2.2 Dose Response Functions

Radiogenic cancers in humans have been observed, for the most part, only following doses of ionizing radiation that are relatively high compared to those likely to result from a 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. 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 40s, 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 and dose rate extrapolation is often large.

The 1980 NAS report (NAS80) examined 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) quadratic, where the risk varies as the square of the dose at all dose levels.

The 1980 NAS BEIR Committee considered only the Japanese mortality data in its 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 the subsequent reassessment of the A-bomb dose 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; Pr87; Sh87). Therefore, it is likely that most of the response attributed to neutrons was caused by the gamma dose, not the dose from neutrons.

Under the revised DS86 dosimetry, the A-bomb survivor data is more consistent with a linear dose response than under T65. Indeed, the linear coefficient obtained by fitting a linear-quadratic function to the data for either leukemia or solid tumors differs only slightly from the respective proportionality constant obtained by fitting a simple linear function (Sh88). Thus, the linear and linear-quadratic functions derived from statistical fits to the Japanese DS86 data yield very similar predictions at low doses. Other human data, particularly that relating to induction of breast cancer (NAS80, NIH85), also lend support to a linear dose response for radiogenic human cancers.

On the other hand, there is extensive laboratory evidence on irradiated animals and cellular preparations which indicates that the effectiveness of low-LET radiation is substantially reduced at low doses and low dose rates. Guided by those observations, as well as by the Japanese data interpreted according to the T65 dosimetry system, the BEIR III committee expressed preference for a linear-quadratic dose response model for low-LET radiations.

For low-LET radiations, the BEIR III Committee preferred the <u>linear-quadratic dose response</u> model. In this model, the risk from an acute dose, D, of low-LET radiation is assumed to be of the form $\alpha D + \beta D^2$. The BEIR III Committee assumed that the linear and quadratic terms were equal at 116 rads, leading to a linear coefficient α, which was about 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 III linear-quadratic dose response model are only about 40 percent of those based on the BEIR III linear model.

Building on earlier work by Lea (Lea62), 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 and cellular 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 (To65, 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.

The quadratic model was put forward in the BEIR III Report, in large part, to account for observed differences in solid tumor induction between Hiroshima and Nagasaki. In Hiroshima, the dose-response appeared linear, but in Nagasaki it appeared quadratic. Rossi suggested that the cancers in Hiroshima were mostly due to neutron doses, while in Nagasaki neutrons were largely absent, so the observed quadratic dose-response there reflected the "true" response to gamma-rays (NAS80). With the revisions in A-bomb dosimetry, this rationale is lost. Preliminary analyses based on DS86 dosimetry indicate that the quadratic model generally provides a poorer fit to the data than do the other two models (Sh88). Some laboratory evidence also suggests that the risk in humans may increase linearly with dose at

low doses (Gr85). Thus, though a quadratic dose-response at low doses (or even a threshold) cannot now be definitively ruled out, EPA does not consider such models suitable for radiation risk assessment.

Finally, "supralinear models," in which the risk coefficient decreases with increasing dose (downward bending, or convex, dose response curve) should be mentioned. 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." The A-bomb survivor data, upon which most of these 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. 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.

6.2.3 The Possible Effects of Dose Rate on Radiocarcinogenesis

The BEIR III 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 its 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; indeed, human studies where there are sufficient data to develop a dose-response function for organ exposure seem to contradict them. Highly fractionated small doses to human breast tissue are apparently as carcinogenic as large acute doses (NAS80, La80). Small acute doses (less than 10 rads) to the thyroid have been found to be as effective per rad as much larger doses in initiating thyroid cancer (UNSC77, NAS80). Also relevant in this connection, perhaps, is the finding that a radiation-induced mutation increased linearly with dose, and independently of dose rate, in human cells but not in rodent cells (Gr85).

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[1] 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 (UNSC77). That choice of a DREF is fully consistent with and equivalent to the reduction of risk at low doses obtained by substituting the BEIR III 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).

6.2.4 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

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[1] Risk assessments require choosing 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.

(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 and 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 more 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) have reached similar conclusions on the trend in excess cancer with time among the irradiated spondylitic patients. More recent analysis of the spondylitic data does show evidence of a fall-off in relative risk after 25 years post-exposure, but the decrease is not yet statistically significant (Da86).

Although 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 risks 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 very 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 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 III 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 I report (NAS72), the BEIR III 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, lifetime cancer risk estimates for a lifetime exposure of equal annual doses are presented. 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.

6.2.5 EPA Assumptions about Cancer Risks Resulting from Low-LET Radiation

The EPA estimates of radiation risks, presented in Section 6.2.6, are based on a presumed linear dose response function. Except for leukemia and bone cancer, where a 25-year expression period for radiogenic cancer is used, a lifetime expression period is used, as in the NAS report (NAS80). Because the most recent Life Span Study Report (Ka82) indicates that 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.

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 a relative risk projection model (EPA84), a 10-year minimum induction period, and the remaining balance of an exposed person's lifetime as the expression period.

6.2.6 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 details of the life table analysis are 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 III report. For relative risk estimates, EPA has used age-specific cancer mortality data also provided by NCHS (NCHS73). The EPA computer program used for the life table analysis was furnished

to the NAS BEIR III 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.

Both absolute and relative risk models have been considered to project the observed risks of most solid radiogenic cancers beyond the period of current observation. 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 been tested in some detail only for lung and breast cancer (La78), based on current evidence, it appears to be the better projection model for solid cancers. Therefore, it has been adopted for risk estimates in this report. Previously, EPA 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, the analysis uses relative risk projections (the BEIR III L-L model) for solid cancers and the absolute risk projection for leukemia and bone cancer (the BEIR III 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 (10^6) person-rad (for the BEIR III linear-quadratic model, a low-LET whole-body dose would yield an estimated lifetime risk of about 160 fatalities per 10^6 person-rad).

BEIR III also presented estimates of excess soft tissue cancer incidence risk coefficients for specific sites, as a function of age at exposure, in its Table V-14. By summing the site-specific risks, it 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 1.7 and 2.1 times their counterparts for the BEIR III $\overline{\text{L-L}}$ model and 4.2 and 5.2 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 this risk estimate is apportioned for whole-body exposure when considering the risks following the exposure of specific organs.

6.2.7 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.

To determine the probability that fatal cancer occurs at a particular site, EPA has performed life table analyses for each cancer type using the information on cancer incidence and mortality in NAS80. NAS80 published incidence risk coefficients (NAS80 Table V-14) and mortality to incidence ratios (NAS80 Table V-15). The data in Tables 6-1 and 6-2 are from these tables with the exception of the mortality to incidence ratios for thyroid and lung cancer. Since not all forms of thyroid cancer can be induced by radiation and since, for those that are, a more reasonable mortality to incidence ratio would be 0.1 (NRC85), EPA has used that value in its calculations. Lung cancer incidence and mortality have both shown an increasing trend between 1970 and 1980. Since incidence leads mortality, an uncorrected mortality to incidence ratio gives a low estimate of the fraction of those persons who, having been diagnosed with lung cancer, will die of that disease. Therefore, a mortality to incidence ratio of 0.94, based on long-term survival studies by the National Cancer Institute for lung cancer (J. Horn, private communication), has been used.

Risk coefficients for a site-specific relative risk model were calculated as follows:

1. Mortality risk coefficients for an absolute risk model were calculated using the data in Tables 6-1 and 6-2.

2. Following the procedure used in NAS80, absolute risks at an absorbed dose rate of 1 mrad/y were calculated for each site for males and females in each age group. A 10-year minimum latency and a 20-year plateau, i.e., a 30-year follow up, was used for these calculations.

3. The relative risk coefficients (1/rad) for each age group providing the same 30-year projected risk were then calculated. Following the NAS80 convention, the values calculated for ages 10-19 were used for ages 0-9. For consistency, this report uses this convention for all cancers including lung and breast, for which the NAS80 absolute risk coefficients are zero in the first decade. For calculating thyroid risks, the relevant age-specific mortality rate was considered to be one-tenth of the corresponding incidence rate.

4. Male and female risks for lifetime expression of risk at 1 mrad/y were then calculated and combined to obtain estimates for the general population.

EPA used the NCHS 1970 life table and mortality data for all these calculations. Male and female cohort results were combined presuming a male:female sex ratio at birth of 1.0511, consistent with the expected lifetimes at birth for the 1970 male, female, and general cohort life tables.

The average risk for a uniform dose to all tissues was calculated to be 542×10^{-6}, 806×10^{-6}, and 678×10^{-6} per rad for males, females, and the general population, respectively.

It is generally accepted that the risk estimates for the individual sites are less certain than are the risk estimates for all sites combined. Table 6-3 summarizes the relative risk calculations for the BEIR III $\overline{L}$-$\overline{L}$model. The calculational procedure was the same as that outlined above.

Table 6-1. Site-specific incidence risk coefficients (10^{-6} per rad-y).

Site	Age at Exposure 0-9	10-19	20-34	35-50	50+
Male					
Thyroid	2.20	2.20	2.20	2.20	2.20
Breast	0.00	0.00	0.00	0.00	0.00
Lung	0.00	0.54	2.45	5.10	6.79
Esophagus	0.07	0.07	0.13	0.21	0.56
Stomach	0.40	0.40	0.77	1.27	3.35
Intestine	0.26	0.26	0.52	0.84	2.23
Liver	0.70	0.70	0.70	0.70	0.70
Pancreas	0.24	0.24	0.45	0.75	1.97
Urinary	0.04	0.23	0.50	0.92	1.62
Lymphoma	0.27	0.27	0.27	0.27	0.27
Other	0.62	0.38	1.12	1.40	2.90
All Sites	4.80	5.29	9.11	13.66	22.59
Female					
Thyroid	5.80	5.80	5.80	5.80	5.80
Breast	0.00	7.30	6.60	6.60	6.60
Lung	0.00	0.54	2.45	5.10	6.79
Esophagus	0.07	0.07	0.13	0.21	0.56
Stomach	0.40	0.40	0.77	1.27	3.35
Intestine	0.26	0.26	0.52	0.84	2.23
Liver	0.70	0.70	0.70	0.70	0.70
Pancreas	0.24	0.24	0.45	0.75	1.97
Urinary	0.04	0.23	0.50	0.92	1.62
Lymphoma	0.27	0.27	0.27	0.27	0.27
Other	0.62	0.38	1.12	1.40	2.90
All Sites	8.40	16.19	19.31	23.86	32.79

Source: NAS80, Table V-14

Table 6-2. Site-specific mortality-to-incidence risk ratios.

Site	Male	Female
Thyroid	0.10	0.10
Breast	----	0.39
Lung	0.94	0.94
Esophagus	1.00	1.00
Stomach	0.75	0.78
Intestine	0.52	0.55
Liver	1.00	1.00
Pancreas	0.91	0.90
Urinary	0.37	0.46
Lymphoma	0.73	0.75
Other	0.65	0.50

Source: NAS80, Table V-15, except thyroid and lung (see text).

The risks tabulated in Table 6-3 are slightly different from those in NAS80. These differences reflect a correction in the exposure interval data for each age group and the use of final rather than preliminary 1970 mortality data. NAS80 also combined male and female risk estimates presuming a sex ratio at birth of 1:1 which is not consistent with natality data.

Since the total risk for all sites is considered more certain than the risk for each site individually, the lifetime risks calculated for the $\overline{L\text{-}L}$ model have been used as a constraint for the sum of the individual site estimates. The relative risk coefficient for each site shown in Table 6-4 has been calculated by multiplying the coefficient for the unconstrained model for each sex by the quotient of the average risk for all age groups for the $\overline{L\text{-}L}$ unconstrained site-specific model. The constrained risk coefficients are about one-half of the unconstrained values.

The L-L absolute risk model coefficients for leukemia and bone cancer are shown in Table 6-5. The risk coefficient for bone was obtained by dividing the value for alpha particles (high-LET) in NAS80 Table A-27 by an RBE of 8 to obtain a low-LET value of 1.25×10^{-7} per rad-year. The risk coefficients for leukemia were obtained by subtracting the risk coefficients for bone from the risk coefficients for leukemia and bone from NAS80 Table V-17. The EPA has followed the BEIR III Committee's practice of using the absolute risk model projections for leukemia and bone cancer with the relative risk projection for all other cancers. Since the expression period for leukemia and bone cancer is 27 years, there is no difference between the number of cancers projected for a 30-year and a lifetime follow-up period.

Table 6-3. BEIR III L-L model for excess fatal cancers other than leukemia and bone cancer.

Group	Age at Exposure 0-9	10-19	20-34	35-49	50+	All
Risk Coefficients (10^{-6} per rad-y) for Absolute Risk Model*						
Male	1.92	1.457	4.327	5.921	8.808	
Female	2.567	1.955	5.807	7.1021	1.823	
Risk Coefficients (10^{-3} per rad) for Relative Risk Model						
Male	4.458	4.458	2.793	1.007	0.861	
Female	4.748	4.748	3.875	1.902	1.586	
General	4.586	4.586	3.322	1.447	1.257	
Cohort Deaths at 10^{-3} rad/y for Relative Risk Model						
Male	0.612	0.609	0.563	0.181	0.112	2.076
Female	0.689	0.686	0.824	0.357	0.268	2.823
General	0.649	0.647	0.690	0.267	0.188	2.440
Risk per Unit Dose (10^{-6} per rad) for Relative Risk Model						
Male	627	629	397	134	56	310
Female	702	703	568	252	101	378
General	664	665	481	193	81	345

* Source: NAS80, Table V-20

Table 6-4. Mortality risk coefficients (10^{-3} per rad) for the constrained relative risk model.

	Age at Exposure				
Site	0-9	10-19	20-34	35-50	50+
Male					
Thyroid	52.74	52.74	38.00	28.63	22.43
Breast	0.00	0.00	0.00	0.00	0.00
Lung	2.99	2.99	2.15	1.34	1.18
Esophagus	6.15	6.15	1.44	0.71	1.15
Stomach	11.71	11.71	4.20	1.76	1.70
Intestine	3.35	3.35	1.28	0.48	0.46
Liver	120.37	120.37	25.19	7.23	4.24
Pancreas	67.81	7.81	2.49	1.12	1.37
Urinary	4.14	4.14	1.38	0.59	0.39
Lymphoma	4.41	4.41	1.28	0.42	0.21
Other	1.12	1.12	1.02	0.44	0.47
Female					
Thyroid	35.30	35.30	35.96	34.81	29.53
Breast	10.52	10.52	2.80	1.52	1.02
Lung	6.36	6.36	6.27	6.10	6.12
Esophagus	13.30	13.30	3.90	2.31	3.17
Stomach	14.15	14.15	7.08	3.19	2.60
Intestine	2.63	2.63	1.06	0.45	0.42
Liver	142.77	142.77	46.62	16.29	7.80
Pancreas	11.81	11.81	3.61	1.50	1.59
Urinary	8.10	8.10	3.41	1.63	0.96
Lymphoma	6.28	6.28	1.60	0.50	0.25
Other	0.53	0.50	0.47	0.24	0.27
General					
Thyroid	40.01	40.18	6.67	33.15	28.10
Breast	10.57	10.57	2.82	1.54	1.07
Lung	3.61	3.61	2.91	2.19	2.15
Esophagus	8.01	8.01	2.08	1.14	1.77
Stomach	12.63	12.63	5.37	2.34	2.10
Intestine	2.95	2.95	1.16	0.47	0.44
Liver	126.87	126.84	32.42	10.37	5.70
Pancreas	9.66	9.66	3.00	1.30	1.48
Urinary	5.48	5.48	2.08	0.95	0.61
Lymphoma	5.28	5.28	1.43	0.45	0.23
Other	0.76	0.76	0.69	0.32	0.34

Table 6-5. BEIR III L-L model for excess incidence of (and mortality from) leukemia and bone cancer (absolute risk model).

	Age at Exposure					
Site	0-9	10-19	20-34	35-50	50+	All
Risk Coefficient (10^{-6} /rad-y)*						
Male						
Leukemia	3.852	1.724	2.471	1.796	4.194	
Bone	0.125	0.125	0.125	0.125	0.125	
Female						
Leukemia	2.417	1.067	1.541	1.112	2.631	
Bone	0.125	0.125	0.125	0.125	0.125	
General						
Leukemia	3.147	1.399	1.005	1.439	3.277	
Bone	0.125	0.125	0.125	0.125	0.125	
Cohort Deaths at 10^{-3} rad/y						
Male						
Leukemia	.0923	.0405	.0829	.0508	.0968	.3634
Bone	.0030	.0029	.0042	.0035	.0029	.0165
Total	.0953	.0435	.0871	.0543	.0997	.3799
Female						
Leukemia	.0588	.0257	.0543	.0357	.0932	.2677
Bone	.0030	.0030	.0044	.0040	.0044	.0189
Total	.0618	.0287	.0587	.0398	.0976	.2866
General						
Leukemia	.0760	.0333	.0689	.0435	.0950	.3167
Bone	.0030	.0030	.0043	.0038	.0036	.0177
Total	.0790	.0363	.0732	.0472	.0987	.3344
Risk per Unit Dose (10^{-6} per rad)						
Male						
Leukemia	94.7	41.9	58.5	37.5	48.6	54.2
Bone	3.1	3.0	3.0	2.6	1.4	2.5
Total	97.8	44.9	61.4	40.1	50.1	56.7
Female						
Leukemia	59.9	26.3	37.4	25.3	35.3	35.9
Bone	3.1	3.1	3.0	2.8	1.7	2.5
Total	63.0	29.4	40.4	28.1	36.9	38.4

Table 6-5. (continued) BEIR III L-L model for excess incidence of (and mortality from) leukemia and bone cancer (absolute risk model).

	Age at Exposure					
Site	0-9	10-19	20-34	35-50	50+	All
Risk per Unit Dose (10^{-6} per rad)						
General						
Leukemia	77.7	34.3	48.1	31.4	41.2	44.8
Bone	3.1	3.1	3.0	2.7	1.6	2.5
Total	80.8	37.4	51.1	34.1	42.8	47.3

* Source: NAS80, Table V-17

Table 6-6 shows the average mortality risks per unit absorbed dose for the combined leukemia/bone and constrained relative risk models. The risk, in general, decreases with increasing age at exposure. For a constant, uniform absorbed dose rate to all organs and tissues, about 60 percent of the risk is conferred by the exposures in the first 20 years of life.

The mortality-to-incidence ratios in Table 6-2 were used to convert the mortality risk estimates in Table 6-6 to incidence risk estimates. For leukemia and bone cancer, the incidence risks are considered to be equal as in NAS80. The resultant incidence risks are shown in Table 6-7.

6.2.8 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). BEIR III reported estimates of factors of 10-80 times reduction for iodine-131 compared to X-rays and noted the estimates were derived primarily from animal experiments (NAS80). However, one study on rats reported that iodine-131 was just as effective as X-rays in inducing thyroid cancer which led an NRC review group to select one-third as the minimum ratio of iodine-131-to-X-ray effects that is compatible with both old and new data (NRC85).

It would be prudent to use this factor until further information from animal studies or some human data are developed. In this document, EPA has employed a thyroid cancer risk coefficient for internal exposures to iodine-131 and I-129 which is one-third that used for gamma-rays or beta radiations from other radionuclides.

6.2.9 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

Table 6-6. Site-specific mortality risk per unit dose (10^{-6} per rad) for combined leukemia-bone and constrained relative risk model.

Site	Age at Exposure 0-9	10-19	20-34	35-50	50+	All
Male						
Leukemia	94.68	41.86	58.46	37.52	48.64	54.19
Bone	3.07	3.04	2.96	2.61	1.45	2.47
Thyroid	8.25	8.25	5.08	2.69	0.80	4.32
Breast	0.00	0.00	0.00	0.00	0.00	0.00
Lung	145.90	146.95	107.22	61.40	22.55	84.21
Esophagus	25.57	25.76	6.13	2.82	2.03	9.91
Stomach	110.95	111.72	40.63	16.4	9.36	46.95
Intestine	53.49	53.83	20.89	7.60	4.30	22.78
Liver	168.01	168.24	35.40	9.48	2.50	58.87
Pancreas	74.36	74.90	24.21	10.34	10.34	30.78
Urinary	40.73	40.99	13.85	5.79	2.22	16.60
Lymphoma	33.43	33.28	9.62	2.88	0.71	12.49
Other	37.48	37.23	33.72	13.09	6.93	22.66
Total	796.43	746.05	358.15	172.65	108.06	366.25
Female						
Leukemia	59.93	26.35	37.39	25.27	35.27	35.86
Bone	3.10	3.09	3.03	2.84	1.67	2.53
Thyroid	15.85	14.54	11.46	7.46	2.24	8.42
Breast	309.33	310.52	81.01	36.93	10.30	107.63
Lung	78.57	78.89	77.09	64.70	24.96	56.72
Esophagus	21.47	21.57	6.32	3.46	2.26	8.33
Stomach	102.64	103.05	51.49	22.38	10.73	45.00
Intestine	57.15	57.38	23.07	9.57	5.01	23.08
Liver	115.94	115.25	36.97	11.95	2.80	40.74
Pancreas	103.00	103.48	31.71	12.70	7.11	38.15
Urinary	46.40	46.54	19.64	9.08	3.06	18.80
Lymphoma	45.71	45.66	11.54	3.35	0.79	15.13
Other	27.69	27.65	24.48	11.27	5.80	16.20
Total	986.78	953.96	415.21	220.95	112.01	416.59

Table 6-6. (continued) Site-specific mortality risk per unit dose (10^{-6} per rad) for combined leukemia-bone and constrained relative risk model.

	Age at Exposure					
Site	0-9	10-19	20-34	35-50	50+	All
General						
Leukemia	77.69	34.26	48.06	31.39	41.20	44.76
Bone	3.09	3.06	2.99	2.72	1.58	2.50
Thyroid	12.22	11.33	8.23	5.07	1.61	6.43
Breast	151.21	152.03	39.95	18.40	5.75	55.36
Lung	112.98	113.63	92.34	63.00	23.91	70.07
Esophagus	23.56	23.71	6.22	3.14	2.16	9.09
Stomach	106.89	107.48	45.98	19.37	10.13	45.95
Intestine	55.28	55.57	21.96	8.58	4.70	22.94
Liver	142.55	142.30	36.17	10.71	2.67	49.55
Pancreas	88.36	88.89	27.90	11.51	6.87	34.57
Urinary	43.50	43.71	16.70	7.43	2.69	17.73
Lymphoma	39.44	39.34	10.56	3.11	0.76	13.85
Other	32.69	32.54	29.16	12.18	6.30	19.34
Total	889.49	847.84	386.21	196.60	110.32	392.14

Table 6-7. Site-specific incidence risk per unit dose (10^{-6} per rad) for combined leukemia-bone and constrained relative risk model.

	Age at Exposure					
Site	0-9	10-19	20-34	35-50	50+	All
Male						
Leukemia	94.68	41.86	58.46	37.52	48.64	54.19
Bone	3.07	3.04	2.96	2.61	1.45	2.47
Thyroid	87.59	82.5	50.84	26.92	8.04	43.23
Breast	0.00	0.00	0.00	0.00	0.00	0.00
Lung	155.21	156.33	114.07	65.31	23.99	89.58
Esophagus	25.57	25.76	6.13	2.82	2.03	9.91
Stomach	147.94	148.97	54.18	21.87	12.48	62.61
Intestine	102.87	103.52	40.16	14.63	8.28	43.81
Liver	168.01	168.24	35.40	9.48	2.50	58.87
Pancreas	81.71	82.31	26.60	11.37	7.20	33.83
Urinary	110.08	110.79	37.44	15.65	6.01	44.87
Lymphoma	45.80	45.58	13.17	3.94	0.98	17.12
Other	57.66	57.27	51.88	20.15	10.65	34.86
Total	1080.20	1026.20	491.27	232.28	132.25	495.35
Female						
Leukemia	59.93	26.35	37.39	25.27	35.27	35.86
Bone	3.10	3.09	3.03	2.84	1.67	2.53
Thyroid	158.45	145.42	114.59	74.60	22.38	84.16
Breast	793.16	796.20	207.73	94.69	26.40	275.97
Lung	83.59	83.93	82.01	68.83	26.56	60.34
Esophagus	21.47	21.57	6.32	3.46	2.26	8.33
Stomach	131.59	132.11	66.01	28.69	13.75	57.70
Intestine	103.90	104.34	41.94	17.40	9.11	41.96
Liver	115.94	115.25	36.97	11.95	2.80	40.74
Pancreas	114.44	114.98	35.23	14.11	7.91	42.39
Urinary	100.88	101.16	42.70	19.74	6.66	40.88
Lymphoma	60.95	60.88	15.38	4.47	1.06	20.18
Other	55.38	55.30	48.97	22.54	11.61	32.40
Total	1802.80	1760.60	738.28	388.58	167.42	743.44

Table 6-7. (continued) Site-specific incidence risk per unit dose (10^{-6} per rad) for combined leukemia-bone and constrained relative risk model.

	Age at Exposure					
Site	0-9	10-19	20-34	35-50	50+	All
General						
Leukemia	77.69	34.26	48.06	31.39	41.20	44.76
Bone	3.09	3.06	2.99	2.72	1.58	2.50
Thyroid	22.24	113.32	82.26	50.66	16.05	64.28
Breast	387.78	389.82	102.42	47.18	14.74	141.95
Lung	120.19	120.88	98.24	67.02	25.43	74.54
Esophagus	23.56	23.71	6.22	3.14	2.16	9.09
Stomach	139.95	140.71	60.00	25.25	13.20	60.08
Intestine	103.38	103.92	41.03	16.00	8.74	42.86
Liver	142.55	142.30	36.17	10.71	2.67	49.55
Pancreas	97.71	98.30	30.85	12.73	7.60	38.23
Urinary	105.58	106.08	40.02	17.68	6.37	42.28
Lymphoma	53.21	53.07	14.26	4.20	1.02	18.69
Other	56.55	56.31	50.43	21.33	11.19	33.60
Total	1433.50	1385.70	612.96	310.01	151.96	622.96

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.

6.2.10 Effect on Risk Estimates of Recent Information Regarding A-Bomb Survivors

Since publication of the BEIR III report, there has been further epidemiological follow-up of the Japanese A-bomb survivors. As discussed above, the results have lent support to the relative risk projection model for solid tumors, which has been utilized here.

The additional data provided by the follow-up reduces statistical uncertainties in the risk coefficients and fills in important gaps pertaining to some organ-specific risks, particularly with respect to childhood irradiation (Pr88).

Subsequent to BEIR III, there has also been a major reassessment of doses assigned to the A-bomb survivors, the effect of which, in general, will be to increase the risk of low-LET radiation calculated according to a particular model.

Investigators from Oak Ridge National Laboratory carried out careful state-of-the art evaluation of the dose to A-bomb survivors in the early 1960s (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.

Serious inadequacies in the T65 dosimetry system were discovered in the late 1970s. A comprehensive reevaluation of the doses to survivors was carried our under the auspices of the U.S.-Japan Joint Committee for Reassessment of Atomic Bomb Dosimetry in Hiroshima and Nagasaki. In 1986, this committee provided results to the Radiation Effects Research Foundation (RERF) from which a revised dosimetry system, termed "DS86," was developed. Although work in the DS86 is largely complete, small adjustments in dose estimates are anticipated over the next few years (Pr87). In addition, about 1,000 survivors from Nagasaki, who were shielded by terrain or were in factories, have so far been excluded from the analysis because of difficulties in estimating their doses. It is anticipated that dose estimates for some of these survivors will be forthcoming in the near future (Pr87).

The major differences between T65 and DS86 are: (1) the neutron dose in DS86 is decreased to 10 percent of its former value in Hiroshima and 30 percent in Nagasaki (as a result, neutrons now contribute relatively little to the estimated excess of cancers in the two cities); (2) the DS86 free-in-air gamma dose increases somewhat in Hiroshima but decreases in Nagasaki relative to T65; (3) transmission of gamma-rays through wooden structures is decreased by about a factor of 2 in DS86; and (4) transmission of gamma-rays through the body to internal organs is generally increased, partially nullifying the change associated with the decreased transmission through structures (Pr87, Sh87).

Analysis of the A-bomb survivor data using the DS86 dosimetry is continuing. Preliminary indications are that risk estimates corresponding to a given dose-response model (linear or linear-quadratic) will be increased by more than a factor of 2 as compared to BEIR III estimates. This increase arises not only from changes in dosimetry, but also from further epidemiological follow-up and new statistical procedures employed (Pr87, Pr88). A

preliminary estimate of low-LET radiation risk to the Japanese population based on DS86 dosimetry and the linear, relative risk model is 1.2 X 10^{-3} fatal cancers per rad (Pr88) - approximately 3 times the corresponding BEIR III estimate. Recent publications by UNSCEAR (UNSC88) and the British NRPB (St88) obtained similar estimates for the Japanese and United Kingdom populations, respectively.

It appears that either a linear or linear-quadratic dose response is consistent with the survivor data, analyzed according to DS86 (Pr87). However, as noted above, linear and linear-quadratic best fits to the data differ only slightly in their predictions at low doses. It would also appear that the residual difference in risk per unit dose between Hiroshima and Nagasaki is no longer statistically significant under DS86 dosimetry (Sh87).

6.2.11 Comparison of Risk Estimates for Low-LET Radiation

Table 6-8 summarizes various estimates of risk from low level, low-LET exposures of the general population. As discussed above, the highest risk estimates are obtained by assuming a linear dose response (for purposes here, equivalent to a DREF = 1.0) and a relative risk projection model. The EPA's current risk estimate of 392 x 10^{-6}/rad corresponds to that obtained by the BEIR III committee (NAS80) using these "conservative" assumptions. However, this estimate was not derived from the most recent Japanese data; recent calculations based on similar assumptions but revised data yield about three times higher risk (see Pr88 in Table 6-8). Thus, as illustrated by a comparison with the UNSC88 and St88 entries in Table 6-8, the EPA89 estimate is in good agreement with the new data if one assumes that the risks projected from a linear fit to the epidemiological data should be reduced by a factor of about three when extrapolating to chronic low dose conditions. Such an assumption is reasonable in view of supportive laboratory data and the apparent decreased effectiveness of iodine-131 in causing thyroid cancer in humans relative to X-rays (NCRP77). However, it should be noted that while the current estimate 392 x 10^{-6}/rad is reasonable, and well within the range of uncertainty, it can no longer be regarded as conservative, in the sense of providing an extra margin of public health protection. The EPA plans to reevaluate its risk models, including the choice of DREF, in light of the UNSC88 and NAS BEIR V reports.

It is expected that this review will also lead to revisions in the distribution of fatal cancer risk among organs. To assign organ risks, evidence on the Japanese A-bomb survivors has to be integrated with that from other epidemiological studies. As an indicator of the possible impact that the new Japanese data may have on EPA's organ-specific risk estimates, Table 6-9 compares EPA's current organ risk estimates with those recently published by the NRPB for the general U.K. population (St88), which take into account recent changes in the Japanese data. Two model estimates are presented from the NRPB publication: (a) one based on a linear extrapolation of high dose epidemiological data and (b) one based on an assumed DREF of two for breast cancer induction and three for all other sites. Both sets of model estimates assume a relative risk protection for cancers other than bone cancer and leukemia. Thus the model assumptions underlying the first NRPB set of organ risk estimates closely parallel those employed by EPA. The difference in the risk estimates largely reflect changes in the Japanese data. The second set of NRPB risk estimates, which the authors preferred to use at low environmental doses and dose rates, are, for the most part, in reasonable agreement with EPA's current model estimates (to within about a factor of two).

6.2.12 Sources of Uncertainty in Low-LET Risk Estimates

The most important uncertainties in estimating risk from whole body, low-LET radiation appear to: (1) the extrapolation of risks observed in populations exposed to relatively high doses, delivered acutely, to populations receiving relatively low-dose chronic exposures and (2) the projection of risk over a full lifespan - most critically, the extent to which high relative risks seen over a limited follow-up period among individuals exposed as children carry over into later years of life when baseline cancer incidence rates are high.

Another significant uncertainty relates to the extrapolation of risk estimates from one population to another, e.g., from the Japanese A-bomb survivors to the U.S. general population. This source of uncertainty is regarded as important for estimating risk of radiogenic cancer in specific organs for which the baseline incidence rates differ markedly in the two populations.

In addition to the model uncertainties alluded to above, errors in dosimetry and random statistical variations will contribute to the uncertainty in the risk estimates. The errors in T65 dosimetry were discussed in Section 6.2.10. The residual error of DS86 dosimetry is estimated to be a relatively minor contributor to the overall uncertainty (see below). Statistical variability will be the most important where relatively few excess cancers have so far been observed, e.g., with respect to specific cancer sites or with respect to childhood irradiation in the A-bomb survivors.

6.2.12.1 Low Dose Extrapolation

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 upon a review of this literature, the National Council on Radiation Protection (NCRP80) has concluded that "linear interpolation from high doses (150 to 350 rads) and dose rates (>5 rads per minute) may overestimate the effects of either low doses (0 to 20 rads) or of any dose delivered at dose rates of five rads per year or less by a factor of two to ten." Judged solely from laboratory experiments, therefore, about a factor of ten 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 seems to argue against such a large DREF from human cancer induction, however. 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 on thyroid cancer induction is equivocal: medical X-ray data suggest a linear dose response (NAS80, NIH85); on the other hand, irradiation by iodine-131 appears to be at least three times less effective than an equal dose of X-rays in inducing human thyroid cancer. One plausible explanation for which is a reduced effectiveness at low dose rates. (NCRP77)

Table 6-8. Comparison of general population risk estimates for fatal cancers due to low level, whole-body, low-LET radiation.

Source of estimate	Fatalities per 10^6 person-rad	Risk projection model	DREF[a]
NAS72[b]	117	Absolute	1.0
NAS72[b]	621	Relative	1.0
NAS80	158	Absolute	1.0
NAS80	403	Relative[c]	1.0
NAS80	67	Absolute	2.48[d]
NAS80	169	Relative[c]	2.48[d]
EPA84	280	Ave.(Rel.& Abs.)	1.0
EPA89[e]	392	Relative[c]	1.0
UNSC77	75-175	-----	2.5
Pr88[f]	1200	Relative[c]	1.0
UNSC88[f]	110-550	Relative[c]	2-10
St88[f]	450	Relative[c]	3.0[g]

[a]Factor by which risk estimate is reduced from that obtained by linear extrapolation of high dose epidemiological results.

[b]As revised in NAS80.

[c]For all cancers other than leukemia and bone cancer.

[d]Based on comparison of linear coefficients for linear and linear-quadratic models used to calculate radiogenic cancers other than leukemia and bone cancer; the corresponding DREF is 2.26 for these two sites.

[e]Refers to this document.

[f]From analyses of A-bomb survivor data using DS86 dosimetry.

[g]Except breast - a DREF of 2 is assumed for that site.

Table 6-9. Site-specific mortality risk per 10^6 person-rad from low level, low-LET radiation exposures of the general population.

Cancer	EPA	NRPB[a]	NRPB[b]
Leukemia	44.8	84	28
Bone	2.5	15	5
Thyroid	6.4 (2.1)[c]	7.5	2.5
Breast	55.4	110	55
Lung	70.1	350	120
Stomach	46.0	73	24
Intestine	22.9	110	37
Liver	49.6	45	15
Pancreas	34.6	---	---
Urinary	17.7	---	---
Other	42.3	500	163
Total	392	1290	450

[a] Relative risk model recommended by authors for use only at high dose rates. Use at low dose rates would be equivalent to adopting a DREF of 1. (St88).

[b] Preferred relative risk model projection for use at low dose rates; assumes DREF=2 for breast and DREF=3 for all other sites.

[c] Value in parentheses represents estimate for important case of iodine-131 (or iodine-129) exposure.

The BEIR III Committee's analysis of the A-bomb survivor data based on T65 dosimetry, suggested a quadratic component to the dose response function. After removing the estimated neutron-induced leukemia, the Committee's linear-quadratic fit to the data yielded a linear coefficient that was a factor of 2.3 times lower than the coefficient obtained from a simple linear fit (NAS80). Thus, the analysis suggested a 2.3 times lower risk at low doses (and dose rates) than estimated by linear extrapolation of the high dose data. Results of the curve fitting for solid tumors were too unstable to estimate a shape for the dose response; for simplicity, the Committee assumed that the shape of the linear-quadratic fit for solid tumors was identical to that derived for leukemia. At low doses, the linear-quadratic model predicts about 2.5 times fewer solid tumors than the corresponding linear model. However, the DS86 data appear to be more consistent with a simple linear dose response for both leukemia and solid tumors. Reflecting this finding, low dose extrapolations of the linear and linear-quadratic fits to the DS86 data apparently differ from one another by less than a factor of 2 (Sh88, Pi89). Thus, if one posits a linear-quadratic dose response model, the available

human data would suggest that linear extrapolation from high doses and dose rates overestimates risks at low doses and dose rates by about a factor of 2 or less.

6.2.12.2 Time and Age Dependent Factors

Because epidemiological follow-up of exposed population is generally incomplete, a risk projection model must be used in estimating lifetime risks due to a given exposure. For leukemia and bone cancer, where the expression time is limited to 25 years, absolute and relative risk projection models yield the same number of radiogenic cancers. For other cancers, the BEIR III 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 lifetime risks 2-3 times larger than an absolute risk projection. Recent follow-up of the A-bomb survivor population strongly suggests that the relative risk projection model better describes the variation 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. For other cancers, the relative risk may have been roughly constant for the current period of follow-up but may eventually decrease over time. The uncertainty relating to risk projection will naturally decrease with further follow-up of irradiated study cohorts, but in view of the continuing increase in attributable risk with age in the A-bomb survivors, it would appear that the relative risk projection model does not overestimate the lifetime task in the general population by more than about a factor of 2.

Similarly, there is yet insufficient information on radiosensitivity as a function of the age at exposure, particularly on the ultimate effects of exposure during childhood. As the A-bomb survivor population ages, more information will become available on the cancer mortality of persons irradiated when they were young. Recent follow-up studies support the view that relative risks are highest in those aged 0-9 years at exposure. Full inclusion of the projected effects on this group was a major contributor to the increase in risk found with the recent analysis based on DS86 dosimetry (Pr87, Pr88).

6.2.12.3 Extrapolation of Risk Estimates to U.S. Population

There is also an uncertainty associated with applying the results of an epidemiological study on a population 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 has called this the "transportation problem," a helpful designation because it is often confused with the risk projection problem described above. 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 6.4 for the case of risk estimated for radon progeny.

The BEIR III 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 was 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 that 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 the "transportation problem" was taken by the 1972 NAS BEIR-I 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 United States 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 appears that the absolute risk should be transported for breast cancer. Evidence is lacking regarding the other cancer sites, 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, EPA believes that using the absolute risk "transportation model" is unlikely to cause large errors 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.

6.2.12.4 Dosimetry and Sampling Errors

As discussed in Section 6.2.10, there were systematic biases in the T65 dosimetry system for the Japanese A-bomb survivors leading to a significant downward bias in the estimates of risk due to low-LET radiation. Under DS86 dosimetry, systematic errors are believed to be no more than about ± 15%, i.e., one standard deviation (1 SD) (Ka89). Random errors in the individual dose estimates are estimated to be ± 28% (1 SD), with an overall uncertainty in individual doses of about ± 32% (Ka89). The random errors in dosimetry will tend to cancel, but they are expected to bias the slope of the dose response curve downward, thus reducing the estimate of risk (Ma59, Da75, Gi84). The magnitude of this bias has been estimated to be roughly 10% (Pi89).

The precision of risk estimates are also limited by statistical fluctuations due to finite sample size. The uncertainty in the low-LET risk coefficient for leukemia or all cancers due to this cause is about ± 20% (90% confidence interval) (Sh89). Uncertainties due to sampling error are larger where data are sparse, e.g., with respect to risks for specific age groups or specific cancer sites (Sh88). Finally, there will be some error in ascertaining cancer cases, most often an under-reporting of cases or mislabeling of cancer type. The latter type of error

would not be expected to greatly affect the estimates of whole-body risk from ionizing radiation. The former would tend to bias risk estimates downward somewhat but it would be difficult to quantify this effect.

6.2.12.5 Summary and Conclusions Regarding Uncertainties in Low-LET Cancer Risk Estimates

Uncertainties in low-LET risk estimates arise both from data uncertainties pertaining to ascertainment of radiation doses and cancer cases and from uncertainties in the proper choice of model assumptions. The data uncertainties include both systematic errors (biases) and random errors. Generally speaking, the modeling uncertainties are larger but random sampling errors may be a very important contributor to the uncertainty in risk for certain types of radiogenic cancers or for certain irradiated subpopulations.

The EPA central estimate of average lifetime risk, approximately 400 fatal cancers per 10^6 person-rad, is taken from the NAS BEIR III Committee report (NAS80), incorporating the most conservative model assumptions utilized by the Committee, i.e., a linear-dose-response and age-specific relative risks projected over a lifetime for solid tumors (L-RR model). For reasons discussed above, it would now appear that estimates of average lifetime risk based on the L-RR model assumptions must be revised upwards - to roughly 1,200 fatal cancers/10^6 person-rad. Although further analysis of the A-bomb survivor data may increase this estimate, the conservatism inherent in the model's assumptions supports the view that the 1,200/10^6 value is an upper bound, pending release of the NAS BEIR V report now in preparation.

Animal data would suggest that the linear dose response may overestimate risk by roughly a factor of 3. Likewise, while the epidemiological data clearly indicate an increase in risk with age at expression, the (age-specific) constant relative risk projection may overstate lifetime risk by about a factor of 2. Allowing even for the additional sources of uncertainty discussed above, it would appear that the upper bound (L-RR) model estimate may be high by a factor of 5 to 10. Therefore, as a lower bound estimate of the average lifetime risk, a value which is one-tenth the upper bound, or 120 fatal cancers/10^6 person-rad, has been adopted.

The L-RR model estimate from BEIR III, about 400 fatal cancers/10^6 person-rad, falls near the geometric mean of what tentatively appears to be a reasonable range for the estimate of risk, based on current information. The EPA has chosen the BEIR III, L-RR model value as its "central estimate." It should be emphasized that this estimate cannot be regarded as "conservative" in the sense of providing any significant margin of safety with respect to public health protection. The decision by EPA to employ the central estimate of 400 fatalities/10^6 person-rad and a range of 120-1,200 fatalities/10^6 person-rad was reviewed and approved by a special panel set up by the Agency's outside Radiation Advisory Committee, and by the Committee itself, as an interim measure for this proposed rulemaking.

The uncertainty in risks for specific cancer sites may be substantially larger than the uncertainty in the whole-body risk. One reason is that the epidemiological data pertaining to some sites may be very sparse. In addition, the uncertainty in projecting risk from one

population to another, e.g., Japanese to U.S., is important at sites for which incidence rates differ markedly between populations.

6.3 FATAL CANCER RISK RESULTING FROM HIGH-LET RADIATION

This section explains 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 kiloelectron-volts (keV) per micrometer (μm) in tissue, 5-MeV alpha particles result in energy deposition of more than 100 keV per μm. 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.

6.3.1 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 a well-defined endpoint, a quality factor is based on an 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 absorbed 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. 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 its decision to estimate the risk of low-LET radiations, in occupational situations, on the assumption that biological effects were reduced at low doses and dose rates. There is evidence that the risks from high-LET radiation are linear with dose and independent of dose rate (for low to moderate doses). 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 employ a DREF; therefore, in order to avoid an artificial inflation in high-LET risk estimates, EPA has assumed an RBE of 8 (20/2.5) for calculating the risks from alpha particles (see Section 6.3.3).

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."

6.3.2 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 maximal at low doses (Ar81, Ho81, Wh83); in addition, some studies with animals show the same response (Ch81, Ul82). The EPA agrees with the NAS BEIR III 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 EPA believes 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 Ra-226 (NAS80). These data are consistent with a dose-squared response (Ro78). Consequently, the NAS BEIR III 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 inconsistent with the linear response model. 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 area of active current research. There is good empirical evidence, from both human and animal studies, that repeated exposures to radium-224 alpha particles are 5 times more effective in inducing bone sarcomas than a single exposure that 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.

6.3.3 Assumptions Made by EPA for Evaluating the Risk from Alpha-Particle Emitters

EPA has 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. As in the case of low-LET radiations, EPA risk estimates for high-LET radiations are based on a linear dose response function. For bone cancer and leukemia, EPA uses the absolute risk projection model described in the previous section. For other cancers, the Agency uses relative risk projections.

Lifetime risk estimates for alpha doses, as a function of age, sex, and cancer site, are easily obtained by multiplying the appropriate entry in Table 6-6 or 6-7 by a factor of 8. The whole-body risks from lifetime exposure of the general population are then calculated to be 3.1×10^{-3}/rad (mortality) and 5.0×10^{-3}/rad (incidence).

As outlined above, the risk estimate for bone cancer in the BEIR III report is based directly on data for high-LET (alpha) radiation. Some readers may note that the EPA high-LET risk estimate, 20 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, 25 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.

6.3.4 Uncertainties in Risks from Alpha-Particle Emitters

The uncertainties in risk associated with internally deposited alpha emitters are often greater than for low-LET radiation. Human epidemiological data on the risks from alpha emitter are largely confined to: (1) lung cancer induced by radon decay products (see below); (2) bone cancer induced by radium; and (3) liver cancer induced by injected thorotrast (thorium). Many of the risk estimates presented here for alpha irradiation assume an RBE of 8, as determined from high-dose experiments on animals. The available evidence on cells, animals, and humans points to a linear dose-response relationship for the risk from alpha emitters (NAS88). The extrapolation to low doses is therefore considered to be less important as a source of uncertainty for alpha irradiation than for low-LET irradiation. There is, however, considerable variability in the RBE determined from animal studies; the extrapolation of these results to humans is also problematic.

For many alpha-emitting radionuclides, the most important source of uncertainty in the risk estimate is the uncertainty in the dose to target cells. Contributing to this uncertainty are uncertainty in the location of these cells, ignorance regarding the metabolism of the radionuclide, nonuniformity of radionuclide deposition in an organ, and the short range of alpha particles in tissue (see Chapter 5).

In the case of alpha irradiation of the lung by radon decay products, there are human epidemiological data that allow direct estimation of the risk per unit exposure. Knowledge of RBE and the actual dose to target cells is therefore not important except as the dose per unit exposure might differ between mine and indoor environments. As a consequence, the estimated uncertainty in average radon risk estimates is similar to that for low-LET radiation. [As discussed in Section 6.4.5, the EPA is employing a central risk estimate for excess radon exposure of 360 fatal lung cancers/10^6 WLM and an uncertainty range of 140-720 fatal lung cancers/10^6 WLM.]

As discussed in Section 6.2, recent analyses of the Japanese A-bomb survivor data indicate that risk estimates for whole-body, low-LET radiation predicated on the linear, relative risk model will have to be increased approximately three-fold, although individual

organ risks will generally change by differing factors. Since the organ specific, high-LET risk estimates used here are 8 times those calculated for low-LET radiation, one would expect a corresponding 3-fold increase in high-LET risk estimates. Moreover, application of a DREF to the calculation of low-LET risks would not affect this conclusion, since, as discussed above, this would imply a compensating increase in the RBE. Consequently, it might be argued that current EPA estimates of risk due to alpha irradiation are too low.

While EPA intends to conduct a comprehensive review of both its low- and high-LET risk estimates after the BEIR V report becomes available, we do not believe that current high-LET risk estimates are biased low in a serious way. It should be noted, in this connection, that the doses from internally deposited alpha emitters are usually concentrated in certain organs - especially bone, bone marrow, and lung. Risks of bone cancer caused by bone-seeking radionuclides (NAS80; NAS88) or lung cancers caused by inhaled radon decay products (see Section 6.4) are derived directly from epidemiological data on high-LET radiation; consequently, these risk estimates will not be affected by changes in the Japanese data. Epidemiological evidence indicates that the risk of radiogenic leukemia induced by alpha emitters deposited in the bone is lower than would be estimated from the gamma-ray risk after adjusting for alpha RBE (NAS88); possibly this discrepancy relates to difficulty in estimating dose to target cells in the bone marrow due to alpha particles originating in the mineral phase of the bone. The EPA's estimates of risk from alpha emitters deposited in the lung in the form of insoluble particles are also conservative. Alpha radiation emitted from such particles, for the most part, irradiate the pulmonary region of the lung (the alveoli). The risk of lung cancer is calculated, in this case, by multiplying the pulmonary region dose by the risk factor for the whole lung. Using the pulmonary dose as an effective lung dose will bias the risk estimate high by an unknown but possibly large factor, especially since the great majority of human lung cancers seem to originate in the tracheobronchial region of the lung.

The next section describes how EPA estimates the risk due to inhalation of alpha-emitting radon progeny, a situation where the organ dose is highly nonuniform.

6.4 ESTIMATING THE RISK FROM LIFETIME POPULATION EXPOSURES FROM RADON-222 PROGENY

The Agency's estimates of the risk of lung cancer due to inhaled radon progeny do not use a dosimetric approach, 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. These decay products, commonly referred to as "progeny" or "daughters," readily attach to inhalable aerosol 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. A wealth of data indicate 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, and in some members of the general public. Recently the National Academy of

Sciences, BEIR IV Committee, and the International Commission on Radiological Protection reviewed the question of radon risks and reported their conclusions (NAS88, ICRP87).

Although considerable progress has been made in modeling the deposition of radon daughters in the lung, it is not yet possible to characterize adequately the bronchial dose delivered by alpha radiation from inhaled radon-222 progeny (NAS88). This is in part due to the uncertainty concerning the kinds of cells in which bronchial cancer is initiated and the depth of these cells in the bronchial epithelium.

Aside from the uncertainties in the dose calculations, a purely dosimetric approach to radon risk estimation appears untenable. Such an approach relates the risk from a given absorbed dose to the lung resulting from radon progeny exposure to that from gamma or X-ray exposure. This approach ignores the extensive epidemiological data on radon exposed miners and bases risk estimates indirectly on epidemiological studies of populations exposed to low-LET radiation. It must also, therefore, make use of an RBE for alpha particles estimated from animal studies. Given the uncertainties in the latter epidemiological studies and in the RBE, there would seem to be no advantage to this approach. Consequently, EPA agrees with the BEIR IV Committee conclusion that radon decay product dosimetry in the lung is only useful for extrapolating radon risk estimates from one exposure situation to another (NAS88).

6.4.1 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 value was chosen because it is the alpha energy released from the total decay of the short-lived radon progeny at radioactive equilibrium with 100 pCi/L of radon-222. The WL 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 hours. Because the results of epidemiological studies are expressed in units of WL and WLM, the following outlines how they can be interpreted for members of the general population exposed to radon progeny.

There are age- and sex-specific respiratory rate and volume differences, as well as differences in duration of exposure, in a general population as compared to a mining population. In earlier reports, EPA used an "exposure equivalent," a modified WLM in which adjustments were made for age-specific differences in airway dimensions and surface area, respiratory frequency, and tidal volume. These factors were expected to influence aerosol deposition and, therefore, radiation dose from radon daughters. This approach to quantifying exposure, correcting for differences in these factors, was recommended by Evans (Ev69) and is consistent with the original derivation of the working level (Ho57).

The BEIR IV Committee, however, concluded that the tracheo-bronchial "dose per WLM in homes, as compared to that in mines, differs by less than a factor of 2," and advised that the dose and risk per WLM exposure in residences and in mines should be considered to

be identical until better dosimetric estimates are developed (NAS88). The EPA will follow the lead of the BEIR IV Committee in this regard and will not use the "exposure equivalent" correction employed to compensate for age- and sex-specific tracheo-bronchial deposition in earlier EPA reports. In this report, exposure of any individual to one WL for 170 hours is one WLM and for one year is 51.56 WLM. This change puts EPA risk estimates in standard units generally used for this purpose, still without requiring dose calculations.

For indoor exposure, an occupancy factor of 0.75 is still employed. Discussion of the support for this estimate can be found in EPA86.

6.4.2 The EPA Model

The initial EPA method for calculating radon risks has been described in detail (EPA79, El79). As new data were reported, the EPA revised its model to reflect changes, as contained in consecutive reports (EPA79, EPA82, EPA83a, EPA83b, EPA84, EPA85, and EPA86). The Agency initially projected radon lung cancer deaths for both absolute and relative risk models, but, since 1978, EPA 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. A life table analysis has been 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 increased the age-specific risk of lung cancer by 3 percent over the age-specific rate in the U.S. population as a whole (EPA79). In the most recent documents, lifetime risks were calculated for a range of risk coefficients from 1 percent to 4 percent per WLM (EPA86).

Although occupational exposures to pollutants other than radon-222 progeny are probably not important factors in the observed lung cancer risk for underground miners (El79, Th82, Mu83, Ra84, Se88), 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 for children and women. While for most estimates, it is assumed that the risk per unit dose received by children is no higher than that received by adults, this assumption may not be correct.

The A-bomb survivor data indicate that, in general, the risk from childhood exposure to low-LET radiation is greater than from adult exposure and continues for at least 33 years, the time over which A-bomb survivors have been observed (Ka82). There are not, as yet, adequate age-specific data on occurrence of lung cancer in those under 10 years of age at the time of exposure (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.

6.4.3 Comparison of Earlier Risk Estimates

Several estimates of the risk due to radon progeny have been published since the original EPA model was developed. These risk estimates were reviewed recently in a number of EPA reports (EPA84, EPA85, and EPA86).

The recent EPA risk estimates for lifetime exposure of 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 6-10. The AECB estimate for lifetime exposure to Canadian males is 830 fatalities per 10^6 person-WLM (Th82). In Table 6-10, this estimate has been adjusted for the U.S. 1970 male and female population.

The National Institute for Occupational Safety and Health reviewed published data on miner studies used as a basis for estimated risk coefficients and pointed out some of the strengths and limitations of selected studies (NIOSH87).

The occupational exposure groups that constitute the epidemiological database for the risk estimates are as follows:

1. U.S. Uranium Miners (NIOSH87)

   (a) Strengths: A large, clearly defined, well-traced cohort with some smoking histories and exposure records on the same persons. Standard sampling techniques were used to make measurements.

   (b) Limitations: There were few measurements in small mines, work histories were self-reported, exposures were high, and potential error due to excursions in exposure levels is high.

   (c) Follow-up: 19 years in 1977.

2. Czechoslovakian Uranium Miners (NIOSH87)

   (a) Strengths: Extensive exposure data with a large number of low level exposures and limited exposure to other underground mining. Many possible confounding factors have been investigated and eliminated.

   (b) Limitations: Exposure estimates prior to 1960 based on radon gas measurements. Person years at risk not determined in standard manner. Smoking effect neglected. Elevated levels of arsenic in ore.

   (c) Follow-up: 26 years in 1975.

Table 6-10. Risk estimate for exposures to radon progeny.

Organization	Model	Fatalities per 10^6 person-WLM	Exposure period	Expression period
EPA	Rel.	760 (460)[a]	Lifetime	Lifetime
NAS*	A-S Abs.	730 (440)[a]	Lifetime	Lifetime
AECB [b]	Rel.	600 (300)[a]	Lifetime	Lifetime
ICRP	-	150-450	Working Lifetime	30 years
UNSCEAR	-	200-450	Lifetime	40 years
NCRP [c]	Dec. Abs.	130	Lifetime	Lifetime

*BEIR III

[a] 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.

[b] Adjusted for U.S. General Population: see text.

[c] NCRP84: Table 10.2; assumes risk diminishes exponentially with a 20-year halftime, and no lung cancer risk is expressed before age 40.

Sources: EPA83b; NAS80; Th82; ICRP81; EPA86; UNSC77; NCRP84; USRPC80.

Models: Rel. - Relative Risk Projection
A-S Abs. - Age-Specific Absolute Risk Projection
Dec. Abs. - Decaying Absolute Risk Projection

3. Ontario Uranium Miners (NIOSH87)

(a) Strengths: Miners received low mean cumulative exposures. Prior mining experience was carefully traced. Exposures prior to 1967 may be disputed.

(b) Limitations: Median age of the cohort was 39 years in 1977. Thoron and gamma exposures may have been high but not accounted for. Smoking history is limited.

(c) Follow-up: 18 years in 1977.

4. Malmberget Iron Miners (NIOSH87)

(a) Strengths: Low exposure levels, long follow-up and stability of work force. Complete ascertainment of vital status and confirmation of diagnosis. Risk from confounders was examined and ruled out.

(b) Limitations: Relatively small cohort with limited exposure data and an unclear cohort definition.

(c) Follow-up: 44 years in 1976

5. Eldorado - Uranium Miners (NAS88)

(a) Strengths: Very low exposure rates, miners screened for prior mining experience, roughly equal groups of surface only and underground only miners, silica and diesel exhaust exposures low. Potential confounders investigated.

(b) Limitations: Exposure estimates are disputed. Sixteen percent of the miners excluded for incorrect or missing data. Average age in 1980 was 43 years.

(c) Follow-up: 14 years in 1980.

6.4.4 Recent Radon Risk Estimates

6.4.4.1 BEIR IV

At the beginning of 1988, the National Academy of Sciences released the BEIR IV Committee report, reviewing information on the risks from radon and other alpha-emitting radionuclides (NAS88). With the cooperation of the principal investigators, BEIR IV examined in detail the mortality experience of four cohorts of underground miners (the U.S., Ontario, and Eldorado uranium miners and the Malmberget iron miners) and how the mortality related to radon daughter exposure. The Committee calculated the relationship of age-specific relative risk to exposure level and time-since-exposure (TSE) in two analyses. The first used internal cohort comparisons and was a grouped-data analog of a Cox relative-risk regression (NAS88). The second analysis compared the cohorts with external rates and was a generalization of standard SMR methods. Separate parallel analyses were carried out to establish a single combined value for each parameter.

The mathematical form of the Committee's preferred TSE model for the radon related age-specific mortality rate at age a is

$$r(a) = r_o(a)[1 + 0.025\,\gamma(a)(W_1 + 0.5W_2)] \qquad (6\text{-}1)$$

where

$r_o(a)$ = age-specific lung cancer mortality rate

$\gamma(a)$ = 1.2, if a is less than 55 years
1.0, if a is between 55 and 64 years
0.4, if a is greater than 64 years

W_1 = WLM incurred between 5 and 15 years prior to age a

W_2 = WLM incurred more than 15 years prior to age a

The Committee model is, therefore, an age-specific, relative-risk projection model with a 5-year latent period prior to expression of risk.

The BEIR IV Committee also estimated what the lung cancer risk coefficient would be for an age-constant, relative-risk model. The results of this analysis are summarized in Table 6-11.

Table 6-11. BEIR IV committee estimate of lung cancer risk coefficient for age-constant, relative-risk model.

Cohort	Excess Risk per WLM	95% Confidence Limits
U.S.	0.6	0.3 - 1.3
Ontario	1.4	0.6 - 3.3
Eldorado	2.6	1.3 - 6.0
Malmberget	1.4	0.3 - 8.9
Combined	1.34	0.8 - 2.3

In its analysis, the BEIR IV Committee identified two major areas of uncertainty affecting its conclusions: (1) uncertainty related to the Committee's analysis of cohort data and (2) uncertainty related to projection of the risk to other groups. The Committee's TSE model uses risk coefficients derived from analysis of data from four miner cohorts. Random or systematic errors, particularly systematic errors, could bias the conclusions. Sources of error in addition to basic sampling variation include: (1) errors in exposure estimates, particularly since the magnitude of error may differ among the studies; (2) errors of misclassification of cause of death; (3) errors in smoking status of individual miners, and (4) modeling uncertainty--i.e., does the model properly address all parameters that are determinants of risk?

Having developed the TSE model for miners, the Committee anticipated the following sources of uncertainty in projecting the model across other groups: (1) effect of gender (miner data all for males); (2) effect of age (miner data contain no information on exposures before about age 20); (3) effect of smoking (miner data contain poor information on smoking status); (4) temporal expression of risk (not enough miners have died to establish accurately the

pattern of lifetime risk from radon exposure), and (5) extrapolation from mining to indoor environments (what are significant differences in the air in mines compared to air indoors?). After reviewing the various sources of uncertainty, the BEIR IV Committee concluded [p. 42], "...The imprecision that results from sampling variation can be readily quantified, but other sources of variation cannot be estimated in a quantitative fashion." Therefore, the Committee chose not to combine the various uncertainties into a single numerical value" (NAS88).

The question of errors in exposure estimates is particularly interesting since the modeling is strongly influenced by the U.S. uranium miner data. In fact, the model risk estimates would be 33 percent higher if the U.S. cohort was removed. Exposure in the U.S. cohort is poorly known: cumulative WLM (CWLM) are calculated from measured radon levels for only 10.3 percent of the miners, varying amounts of estimation are required for about 36.1 percent of the miners, and guesswork is used for about 53.6 percent of the miners (NAS88, Lu71). Only 26.1 percent of the U.S. uranium miner exposure data are based on measured values (Lu71).

The Ontario cohort exposure estimates also are not well founded. Upper and lower estimates were developed: the lower from measured values, the upper based on engineering judgment (NAS88). Eldorado cohort estimates of CWLM were based almost entirely on measured values, while Malmberget cohort estimates were based on a reconstruction of past ventilation conditions (NAS88). Of the four cohorts, the United States has one of the poorest bases for CWLM estimates. One serious problem is the potential error due to large excursions in radon daughter concentrations (NIOSH87). The uncertainties in exposure estimates are particularly significant in view of the rather large impact the U.S. cohort has on the form of the model.

When the BEIR IV model is run with the 1980 lifetable and vital statistics at an exposure level of 0.001 WLM per year, the reference risk can be calculated (see Table 6-12).

Table 6-12. BEIR IV Risk Model - Lifetime Exposure and Lifetime Risk.

Group	Risk (10^{-6}/WLM)
Male	530
Female	185
Combined	350

6.4.4.2 ICRP 50

The International Commission on Radiological Protection, in its Publication 50, addressed the question of lung cancer risk from indoor radon daughter exposures. The ICRP Task Group took a direction quite different from the BEIR Committee. The Task Group reviewed published data on three miner cohorts: U.S., Ontario, and Czech uranium miners. The estimated risk coefficients by cohort are presented in Table 6-13.

Table 6-13. Estimated lung cancer risk coefficients from radon progeny exposure for three miner cohorts.

Cohort	Follow-up	Relative model	Absolute model
U.S.	1950-1977	0.3%-1.0%	2-8 cases/10^6 PWLMY
Czech	1948-1975	1.0%-2.0%	10-25 cases/10^6 PWLMY
Ontario	1958-1981	0.5%-1.3%	3-7 cases/10^6 PWLMY
Average		1%	10 cases/10^6 PWLMY

Source: ICRP87.

The relative risk model then developed for a constant exposure rate is:

$$\lambda(t) = \lambda_o(t)[1 + \int_0^{t-\tau} r(t_e)\, \dot{E}(t_e)\, dt_e] \qquad (6\text{-}2)$$

= the mortality rate at age t

where:

$\lambda_o(t)$ = the age-specific lung cancer rate at age t

$r(t_e)$ = risk coefficient at age of exposure t_e

$\dot{E}(t_e)$ = age-dependent exposure rate

τ = time lag (minimal latency) = 10 years

In the case of a constant exposure rate or constant annual exposure, the equation collapses to:

$$\lambda(t) = \lambda_o(t)[1 + \bar{r}\, E(t - \tau)] \qquad (6\text{-}3)$$

where:

$\bar{r}$ = age averaged relative risk coefficient

$E(t - \tau) = \dot{E}[t - \tau]$

= cumulative exposure to radon daughters to age $t-\tau$

Since ICRP recommends the use of the relative risk model, the ICRP-50 absolute risk model will not be addressed further in this document.

To adapt the relative risk model derived from studies of underground miners to the general population, the ICRP Task Group introduced several adjustments. The first was to correct for co-carcinogenic influences in mines. To account for unidentified, unproven carcinogens that might be present in mine environments but not elsewhere, only 80 percent of the risk was attributed to radon. The second adjustment was for dosimetric corrections. The dose to bronchial epithelium used by the Task Group for persons indoors was estimated to be only 80 percent as large as that for persons in mines; therefore, the risk to the public from radon was considered to be 80 percent of the risk of miners.

Adjusting the average relative risk coefficient of one percent per WLM by these two factors gives a risk coefficient of 0.64 percent per WLM:

$$1.0\% \times 0.8 \times 0.8 = 0.64\%. \qquad (6\text{-}4)$$

The third adjustment made by the Task Group is related to age. Since reports of Japanese A-bomb survivors and some other radiation-exposed groups support an elevated estimate of risk in children compared to adults, the Task Group increased the risk coefficient of persons between birth and age 20 by a factor of 3.

The final relative risk coefficients in the ICRP 50 model are: 1.9 percent per WLM if the age at time of exposure is between birth and 20 years, and 0.64 percent per WLM if age at time of exposure exceeds 20 years.

When the ICRP-50 relative risk model is run with 1980 U.S. lifetable and vital statistics at an exposure level of 0.001 WLM per year, the reference risk calculated is:

Group	Risk (10^{-6}/WLM)
Male	610
Female	205
Combined	420

6.4.5 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 III, 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 was believed to be outside the lower bound. With this procedure, the EPA arrived at relative risk coefficients of 1.2 percent to 2.8 percent per WLM exposure equivalent (300 to 700 fatalities per 10^6 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 did 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, was 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, Ed84) - indicated a relative risk coefficient greater than 3 percent per WLM, perhaps as large as 3.6 percent.

The EPA therefore increased the upper limit of its estimated range of relative risk coefficients. To estimate the risk due to radon-222 progeny, the EPA used 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 yielded, 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 with 1980 vital statistics, 4.6×10^{-4}/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 6-7).

However, in light of the two recently published consensus-based reports, BEIR IV and ICRP 50, and a recent report on the Czech miner groups (Se88), the Agency has reviewed its basis for radon risk estimation. Comparable relative risk coefficients for miners (age-constant relative risk) yield a coefficient of around 1 percent in ICRP 50, 1.34 percent in BEIR IV, and 1.5 percent in the Czechs. This suggests that the range, 1 percent to 4 percent, used by EPA may be too wide. Nevertheless, note that only 5 of the 20 or so studies for which there are some data are included in these estimates.

The BEIR IV Committee noted and modeled a drop in relative risk with increasing time of exposure and a decreasing relative risk with increasing age after exposure (NAS88). The Czech miners show a similar response pattern (Se88). Though the Committee did note a dose rate effect in the U.S. uranium miner cohort, i.e., a decrease in risk per unit exposure at high dose rates, it was not included in the model (NAS88). The possibility of a similar dose-rate effect was found recently in a study on Port Radium uranium miners (Ho87).

The ICRP 50 Task Group worked from a different database and developed a simpler model with fewer age- and time-dependent parameters. The Task Group provided a 3 times higher risk for exposure between birth and 20 years of age than after 20 years of age (ICRP87). The finding in the recent Czech report that risk prior to age 30 is 2 to 2.5 times greater than after age 30 lends some support to the ICRP conclusions (Se88).

Both BEIR IV and ICRP 50 models treat radon and smoking risks as multiplicative. This conclusion is based primarily on data from the U.S. uranium miner cohort. Although apparently based on weaker evidence, the report on Malmberget miners and the recent report on Czech miners both concluded that the interaction of smoking and radon exposure is small (Ra84, Se88). The attributable risk per unit exposure in smokers and non-smokers was essentially the same (Se88). The

true interaction of radon and cigarette smoking is controversial. Both antagonistic (Ax78, Lu79, Ax80) and multiplicative (Lu69, Wh83) interactions have been reported in man and animal studies can be found to justify any position (Ch81, Ch85, Cr78). In prior calculations, EPA has always treated the interaction between radon daughters and cigarette smoke as multiplicative. The EPA will continue to treat the radon daughter-smoke interaction as multiplicative at this time.

Important unresolved issues pertaining to the risks from inhaled radon progeny remain. At the advice of the Radiation Advisory Committee of EPA's Science Advisory Board, EPA will continue to use relative risk models but shall include both BEIR IV and ICRP 50 model calculations to illustrate the difference in results from the two models. The ICRP 50 model will be slightly modified. The risk reduction factor of 0.8 to compensate for differences in dosimetry will be removed to place the ICRP 50 model and BEIR IV model on a comparative basis. Calculations in the ICRP 50 model will be made using risk coefficients of 2.4 percent per WLM from birth to age 20 and 0.8 percent per WLM for ages greater than 20 years, yielding the estimates listed in Table 6-14.

Table 6-14 summarizes risk estimates based on the BEIR IV and the ICRP 50 model, modified as described above. For the calculations in this document, both models were adjusted for the effect of background radon exposure (see section below).

The ICRP Task Group concluded that, all things considered, the range of variation of the mean relative risk coefficient is from about 0.3 up to 2 times the value stated (ICRP87). The range of risk cited in Table 6-14 for the ICRP model reflects this uncertainty in the risk coefficient. Since the BEIR IV Committee did not provide a numerical range of uncertainty, no range is given for that model.

6.4.6 Correction of Radon Risk Estimates for the Effect of Background Exposure

A relative risk model for radon-induced lung cancer generally assumes the excess risk, λ_r, from a given exposure, is proportional to the observed baseline risk of lung cancer in the population, λ_o. Thus, for a constant exposure rate, w, the excess risk at age, a, attributable to previous exposure can be written:

$$\lambda_r(w,a) = \lambda_o(a)\ \beta(a)f(w,a) \qquad (6\text{-}5)$$

For example, in the case of an age-constant relative risk model with a 10-yr minimum latency:

$$\beta(a) = \beta = \text{constant} \qquad (6\text{-}6)$$

$$f(w,a) = (a-10)w \qquad (6\text{-}7)$$

Although λ_r is commonly assumed to be proportional to λ_o, a more consistent (and biologically plausible) way to formulate a relative risk model is to assume that the radon risk, λ_r, is proportional to λ'_o, the lung cancer rate that would prevail in the absence of any radon exposure (Pu88):

$$\lambda_r(w,a) = \lambda'_o\ (a)\beta(a)f(w,a). \qquad (6\text{-}8)$$

Table 6-14. Lifetime risk of lung cancer death from radon daughter exposure (per 10^6 WLM).

Group	Model	
	BEIR IV	ICRP 50
Men	530	760
Women	185	255
Combined Population	350	500
(Range)	-	(170-840)

Presuming that the risk model can be used to relate $\lambda_o(a)$ to $\lambda'_o(a)$, then

$$\lambda_o(a) = \lambda'_o(a)\ [1 + \beta(a)f(\bar{w},a)] \tag{6-9}$$

where $\bar{w}$ is the <u>average</u> exposure rate in the population. It follows from the previous equation that

$$\lambda'_o(a) = \lambda_o(a)/[1 + \beta(a)f(\bar{w},a)] \tag{6-10}$$

The inferred baseline rate without radon exposure depends, of course, on both the risk model and the presumed average background exposure rate. The excess risk associated with an arbitrary exposure situation can be calculated using standard life table methodology.

The ICRP 50 committee did correct the baseline rate in this way in calculating lifetime population risks, assuming an average exposure rate of 0.2 WLM/yr. The BEIR IV Committee did not incorporate the correction, noting that it would be small (see NAS88, p. 53). In arriving at a final estimate based on the ICRP 50 and BEIR IV models (see Table 6-15), EPA has incorporated a model-specific baseline correction, calculated on the assumption of a 0.25 WLM/yr average radon exposure rate (Pu88). As seen in Tables 6-14 and 6-15, this correction results in roughly a 15 percent reduction in each of the estimates of lifetime risk for the general population.

Table 6-15. Lifetime risk from excess radon daughter exposure (adjusted for a background exposure of 0.25 WLM/yr).

	Risk of Excess Lung Cancer Deaths per 10^6 WLM		
Group	BEIR IV	ICRP 50	Average
Men	460	640	550
Women	160	215	190
Population Combined	305	420	360
(Range)		(140-720)	(140-720)

6.4.7 Summary of Baseline Corrected Radon Risk Estimates

Consistent with the recommendations of the Agency's Radiation Advisory Committee, EPA has here averaged the risk estimates derived from the BEIR IV and ICRP 50 models. These estimates are based on 1980 U.S. vital statistics and are adjusted for an assumed background exposure rate of 0.25 WLM/yr. Thus, as shown in Table 6-15, the excess lifetime risk in the general population due to a constant, low-level, lifetime exposure is estimated to be 360 excess lung cancer deaths per 10^6 WLM, with a range of 140 to 720 excess lung cancer deaths per 10^6 WLM. (At lifetime exposures above about 100 WLM, numerical estimates would be reduced because of "competing risk" considerations.)

The BEIR IV and ICRP models differ substantially with respect to their dependence on age and time since exposure. Hence, in evaluating exposures at different ages or time periods it is instructive to consider the predictions made by each model. Illustrative examples of such calculations are given in Tables 6-16 and 6-17.

6.5 OTHER RADIATION-INDUCED HEALTH EFFECTS

The earliest report of radiation-induced health effects was in 1896 (Mo67); it dealt with acute effects in skin generally caused by very large X-ray exposures. Within the six-year period following, 170 radiation-related 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 routine radionuclide emissions.

Radiation-induced carcinogenesis was the first delayed health effect described. The first case was reported in 1902 (Vo02); by 1911, 94 cases of skin cancer and 5 of leukemia were reported (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 human 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 (UNSC69). 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).

6.5.1 Types of Genetic Harm and Duration of Expression

Genetic harm (or the genetic effects) of radiation exposure is defined as stable, heritable changes induced in the germ cells (eggs or sperm) of exposed individuals, which are transmitted to and expressed only in their progeny and in future generations.

Table 6-16. Lifetime risk for varying age at first exposure and duration of exposure (Background = 0.25 WLM/yr).

Lifetime Risk of Lung Cancer per 10^6 WLM

Age(yr)	Exposure Duration(yr)	Male BEIR IV	Male ICRP 50	Female BEIR IV	Female ICRP 50
Birth	1	476	1382	184	511
	10	480	1394	185	515
	Lifetime	459	638	159	213
10	1	481	1398	186	516
	10	483	1402	186	517
20	1	486	470	188	173
	10	495	474	190	173
30	1	509	477	195	172
	10	535	472	205	168
40	1	572	461	217	161
	10	592	435	217	148
50	1	602	392	208	130
	10	516	335	170	109
60	1	378	253	114	79
	10	331	182	95	58
70	1	251	96	69	34
	10	182	57	52	22
80	1	88	15	32	8
	10	55	8	21	4
90	1	12	1	7	-
	10	8	1	4	-
100	1	2	-	1	-
	10	1	-	-	-

Table 6-17. Lifetime risk for varying age at first exposure and duration of exposure (Background = 0.25 WLM/yr).

		Excess Lung Cancer Deaths per 10^6 Persons Exposed at 1 WLM/yr			
		Male		Female	
Exposure Age (yr)	Duration (yr)	BEIR IV	ICRP 50	BEIR IV	ICRP 50
Birth	1	472	1372	183	508
	10	4723	13725	1828	5085
	Lifetime	32171	44859	12352	16545
10	1	481	1398	186	516
	10	4814	13984	1857	5159
20	1	486	470	187	172
	10	4902	4691	1891	1721
30	1	508	476	195	172
	10	5299	46788	2041	1676
40	1	571	461	217	161
	10	5804	4267	2142	1468
50	1	600	391	208	129
	10	4909	3187	1652	1051
60	1	374	251	114	79
	10	2949	1623	895	546
70	1	246	94	68	34
	10	1406	439	456	192
80	1	84	14	31	8
	10	323	45	146	30
90	1	11	1	7	-
	10	30	2	19	1
100	1	2	-	-	-
	10	2	-	2	-

Of the possible consequences of radiation exposure, the genetic risk is more subtle than the somatic risk, since it affects not 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 (nearly 1,000 years) 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 chromosomal aberration, is transmitted to, and may be expressed in, a child conceived after the radiation exposure. However, the damage may also be expressed in subsequent generations or only after many generations. Alternatively, it may never be expressed because of failure to reproduce or failure of the chance to reproduce.

EPA treats genetic risk independently from 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 in 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 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, e.g., genetic conditions with complex causes, and constitutional and degenerative diseases).

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 defined as 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. The EPA assessment of risks of genetic effects includes both first generation estimates and total genetic burden estimates.

In the EPA Background Information Document for Radionuclides (EPA84), direct and indirect methods for obtaining genetic risk coefficients are described, and some recent estimates based on these methods are tabulated. Briefly, the direct method takes the frequency of mutation or occurrence of a heritable defect per unit exposure observed in animal studies and extrapolates to what is expected for humans. Direct estimates are usually used for first generation effects estimates. The indirect method, on the other hand, uses animal data in a different way. The estimated human spontaneous mutation rate per gene site is divided by the average radiation-induced mutation rate per gene observed in mouse studies to obtain the relative radiation-mutation risk in humans. The inverse of this relative radiation

mutation risk is the expected "doubling dose" for radiation-induced mutations in man. The doubling dose is the exposure in rads which will double the current genetic malformation level in man and usually is used to estimate equilibrium effects or all future generation effects.

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 III Committee geneticists estimated that one-sixth of the total genetic burden of x-linked mutations would be expressed in the first generation and five-sixths across all subsequent generations. The EPA assessment of risks of genetic effects includes both first generation estimates and total genetic burden estimates.

The 1986 UNSCEAR report (UNSC86) reviewed data on genetic effects. While there was much new information, changes in direct estimates of first generation risk were minimal, reflecting primarily changes in estimates of survival of reciprocal translocations. There was however, an appreciable change in the doubling dose estimate of genetic risk. Because of Hungarian studies the birth prevalences of isolated and multiple congenital anomalies in man was estimated to be 597.4 per 10^4 live births (UNSC86). The UNSCEAR Committee also estimated congenital anomalies and other multifactorial disorders to have a spontaneous prevalence of 600,000 per 10^6 live births. The UNSCEAR Committee however, made no estimate of the genetic radiation risk coefficients for these types of conditions (UNSC86). The 1988 UNSCEAR Committee also reviewed genetic risks (UNSC88) and confirmed the conclusions of the 1986 UNSCEAR Committee (Table 6-18).

The Agency concluded that the "spontaneous prevalence" of multifactorial disorders described by the UNSCEAR Committees were not all "disorders and traits that cause a serious handicap at sometime during lifetime." Since the multifactorial disorders compose a large fraction of the genetic risk in the BEIR III report, the BEIR III risk estimates will be used until the relevance of the Hungarian studies can be evaluated. The Agency also has concluded estimates of detriment (years of life lost or impaired) as made by several UNSCEAR Committees (UNSC82, 86, 88) should not be used to evaluate genetic risk at this time. As these changes in genetic risk assessment mature, the Agency will review their applicability.

6.5.2 Estimates of Genetic Harm Resulting from Low-LET Radiations

A number of committees have addressed the question of genetic risk coefficient (NAS72, 80, 88; UNSC58, 62, 66, 72, 77, 82, 86, 88; Of80). The detailed estimates of the BEIR III Committee (NAS80) are listed in Table 6-19, those of UNSCEAR (UNSC88) are listed in Table 6-18, and a summary of estimates of the various committees is listed in Table 6-20.

Table 6-18. UNSCEAR 1988 risks of genetic disease per 10^6 live births in a population exposed to a genetically significant dose of one rad per generation of low-dose-rate, low-dose, low-LET irradiation.

(100-rad doubling dose)

Type of genetic disorder	Current incidence per 10^6 liveborn	Effects of one rad per generation	
		First Generation	Equilibrium
Autosomal dominant and x-linked	10,000	15	100
Autosomal recessive diseases	25,000		
-Homozygous effects		no increase	11
-Partnership effects		negligible	4
Chromosomal diseases due to structural anomalies	400	2.4	4
Sub-total (rounded)	13,000	18	115
Early acting dominants	unknown	not estimated	
Congenital anomalies	60,000	not estimated	
Other multifactorial diseases*	600,000	not estimated	
Heritable tumors	unknown	not estimated	
Chromosomal diseases due to numerical anomalies	3,400	not estimated	

* prevalence up to age 70

Source: UNSC88

Although all of the reports cited 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 (see Table 6-20). The same uncertainties exist for the latter (central estimates) as for the former.

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 6-20 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, UNSC72, 77). However, factors of 0.5 to 0.1 have also been used in estimates of specific types of genetic damage (UNSC72, 77, 82).

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 (UNSC82). At present, the RBE for genetic endpoints due to beta particles is taken as 1 (UNSC77, 82).

6.5.3 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 (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.

In the Background Information Document for Radionuclides (EPA84), an RBE of 20 was assigned to high-LET radiation when estimating genetic effects. It was noted that 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, UNSC77, 82). Neutron RBE, determined from cytogenetic studies in mice, also ranged from about 4 to 50 (UNSC82, Gr83a, Ga82). However, an RBE of 4 for plutonium-239 compared to chronic gamma radiation was reported for specific locus mutations observed in neonate mice (NAS80).

Table 6-19. BEIR III estimates of genetic effects of an average population exposure of one 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	1,000	Very few	Very slow increases
Chromosomal aberrations	6,000	Fewer than 10	Increases only slightly
Total	107,000	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.

Most recently, the NAS BEIR IV Committee reviewed the effects of alpha-emitting radionuclides and estimated the genetic effects (See Table 6-21). The BEIR IV genetic risk estimates for alpha-emitters were based on the low-LET estimates given in Table IV-2 in the 1980 BEIR III report, applying an RBE of 15 for chromosome aberrations and 2.5 for all other effects.

These risk estimates, to a first approximation, give an average RBE of about 2.7 relative to the BEIR III low-LET estimates. This is numerically similar to the dose rate effectiveness factor for high dose rate. Therefore, for simplicity, it would be possible to use the same genetic risk coefficients per rad of high dose-rate, low-LET and per rad of high-LET radiation.

Table 6-20. Summary of genetic risk estimates per 10^6 liveborn of low-dose rate, low-LET radiation in a 30-yr generation.

Source	Serious hereditary effects: First generation	Serious hereditary effects: Equilibrium (all generations)
BEAR, 1956 (NAS72)	-	500
BEIR I, 1972 (NAS72)	49[a] (12-200)[b]	300[a](60-1500)
UNSCEAR, 1972 (UNSC72)	9[a] (6-15)	300
UNSCEAR, 1977 (UNSC77)	63	185
ICRP, 1980 (Of80)	89	320
BEIR III, 1980 (NAS80)	19[a] (5-75)	260[a](60-1100)
UNSCEAR, 1982 (UNSC82)	22	149
UNSCEAR, 1986 (UNSC86)	17	104
UNSCEAR, 1988 (UNSC86)	18	115

[a] Geometric mean of the lower and upper bounds of the estimates. The geometric mean of two numbers is the square root of their product.

[b] Numbers in parentheses are the range of estimates.

6.5.4 Uncertainty in Estimates of Radiogenic Harm

Chromosomal damage and mutations have been demonstrated in cells in culture, in plants, in insects, and in mammals (UNSC72,77,82), and in peripheral blood lymphocytes of persons exposed to radiation (UNSC82, Ev79, Po78). However, they cannot be used for predicting genetic risk in progeny of exposed persons. Some believe such changes to be a direct expression of damage analogous to that induced by radiation in germ cells. At least, aberrations in peripheral lymphocytes show that radiation-induced chromosome damage can occur *in vivo* in humans.

Table 6-21. Genetic risk estimates per 10^6 liveborn for an average population exposure of one rad of high-LET radiation in a 30-year generation.

	Serious Hereditary Effects	
	First Generation	Equilibrium (all generations)
Range	28 - 298	165 - 2885
Geometric Mean	91	690

Source: NAS88

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 genetic risk estimates are based on extrapolations from animal data.

Data below (Table 6-22), collected by Van Buul (Va80), on induction of reciprocal translocations in spermatogonia in various species, indicate that animal-based estimates for this type of genetic effect may be within a factor of four of the human value. The 1986 UNSCEAR Committee (UNSC86) did report on radiation induction of reciprocal translocations in other primates but the range of responses and conclusions remain the same. However, if there were no human data on this genetic injury, in the majority of cases, assuming that animal results and human results would be similar would underestimate the risk in humans.

Table 6-22. Radiation-induced reciprocal translocations in several species.

Species	Translocations (10^{-4} per rad)
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

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 or not the sites that have been examined are representative of all sites and all gene loci, with developing evidence that the mouse 7-locus system is more sensitive to radiation than other members of the mouse genome (Ne88). Current research is focused on transposable genetic elements and the relevance of "mobile-genetic-element-mediated spontaneous mutations" to assumptions in the doubling-dose method (UNSC86). The Agency will review its position as new evidence develops.

There is some uncertainty as to which hereditary conditions would be doubled by a doubling dose; future studies on genetic conditions and diseases can apparently 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 that have a genetic component and hence may be increased by exposure to ionizing radiations.

6.5.4.1 Observations on Human Populations

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 they should be interpreted (Ne56).

This study addressed a number of endpoints, including sex ratio, malformations, perinatal data, and anthropometric data; subsequent studies have addressed other endpoints. 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 were 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.

The most recent report evaluated the following possible genetic effects: (1) untoward pregnancy outcomes, (2) all causes of early mortality, (3) balanced chromosomal exchanges, (4) sex-chromosome aneuploids, (5) early onset cancer, and (6) protein mutations. On the basis of the findings of the study, the authors concluded that the gametic doubling dose measured in humans for acute penetrating radiation exposure from atomic bombs is 150 rem to 190 rem (Ne88).

The EPA is using the geometric mean of the BEIR III range of doubling doses: about 110 rads. The EPA believes this estimate of doubling dose probably overstates the risk; however, it is compatible with both human and mouse data and should not be changed at this time. The EPA estimates of genetic risks will be reviewed and revised, if necessary, when more complete reports on the Japanese A-bomb survivors are published.

6.5.4.2 Ranges of Estimates Provided by Various Models

Following recommendations of the 1980 BEIR III and earlier committees, EPA has continued to use a linear nonthreshold model for estimating genetic effects, although some data on specific genetic endpoints obtained with acute low-LET exposures are equally 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 6.2.2), 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).

Even though genetic risk estimates made by different committees based on the linear non-threshold model vary, the agreement is reasonably good. Some of the committees made estimates in terms of a range. These ranges are expressed as a single value by taking the geometric mean of the range. This method was recommended and first used by UNSCEAR (UNSC58) for purposes of expressing genetic risk estimates. 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 6-20). For the most recent, more comparable estimates, the range is a factor of 2 to 4 (see ICRP, BEIR III, and UNSC 1982 in Table 6-17).

6.5.5 <u>The EPA Genetic Risk Estimates</u>

The EPA has used the estimates from BEIR III (NAS80) based on a "doubling dose" range with a lower bound of 50 rem and an upper bound of 250 rem. The reasons are as follows: mutation rates for all gene loci affected by ionizing radiation are not known nor have all loci associated with "serious" genetic conditions been identified. Because the risk estimated by the direct method is incomplete, even for the subject animal species, and does not include the same types of damage estimated by doubling doses, EPA does not consider it further. Moreover, the BEIR III genetic risk estimates provide a better estimate of uncertainty than the UNSCEAR 1982 and ICRP estimates because the BEIR III 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).

The BEIR III 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 (UNSC58) 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 weighted RBE for high-LET radiation as estimated in BEIR IV is about 3, which is numerically the same as the dose rate factor noted above.

Genetic risk estimates used by EPA for high- and low-LET radiations are listed in Table 6-23. As noted above (Section 6.5.1), EPA uses the dose received before age 30 in assessing genetic risks.

The EPA estimates in Table 6-23 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 largest source of human data, the Japanese A-bomb survivors, appears at best to provide an estimate of the doubling dose for calculating the genetic risk in man which is not statistically significant (Ne88).

Table 6-23. Estimated frequency of genetic disorders in a birth cohort due to exposure of the parents to 1 rad per generation.

	Serious heritable disorders (Cases per 10^6 liveborn)	
Radiation	First generation	All generations
Low Dose Rate, Low-LET	20	260
High Dose Rate, Low-LET	60	780
High-LET	90	690

In developing the average mutation rate for the two sexes used in the calculation of the relative mutation risk, the BEIR III Committee postulated that the induced mutation rate in females was about 40 percent of that in males (NAS80). Studies by Dobson, et al., show that the basis for 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 (Do83, Do84, Do88). Recently, Dobson, et al. have shown that mouse oocytes are very sensitive to radiation, doses of 4 to 12 rads killing 50 percent of the immature mouse oocytes. (Do88) Immature oocytes in women are not so easily killed. Dobson, et al. have also shown the existence of a special, hypersensitive, non-DNA lethality target (apparently the plasma membrane) in immature mouse oocytes. (Do88) Irradiation with low-energy neutrons, whose recoil protons have track lengths less than a cell diameter, induces genetic effects in immature mouse oocytes and yields effects similar to those observed in other cells (Do88). Immature human oocytes do not have the same hypersensitive target as mouse oocytes and so should be as susceptible as spermatogonia to genetic effects of radiation.

Unfortunately, BEIR III and, since it is based on BEIR III, BEIR IV have embedded sex-sensitivity differences in their risk estimates. In BEIR III: (1) autosomal dominants and X-linked effects are based on a lower estimate where the oocyte has zero sensitivity and an upper estimate where the oocyte is 44 percent as sensitive as spermatogonia (p. 118); (2) irregularly inherited effects are based on an estimate where the oocyte is 44 percent as sensitive as spermatogonia (pp. 114 and 110); and (3) chromosomal aberrations estimates are based on oocytes and spermatogonia of equal sensitivity (p. 123, NAS80).

Since the sex-specific differences are in both BEIR III and BEIR IV, no attempt is made at this time to correct them. Based upon genetic risk estimates in BEIR V , EPA is considering a review of its genetic risk estimates and may revise them. (NAS90)

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 irregularly inherited mutations which would be eliminated slowly. They may include mild mutations which are but slightly detrimental in their heterozygous state. However, they may be sustained by advances in medical science, thus persisting and accumulating for generations. To what extent this occurs will depend on medical practices in the future.

6.5.6 Effects of Multigeneration Exposures

As noted earlier, while the somatic effects (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 years 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 6-23, it is estimated that exposure to one 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 6-23 is equal to the BEIR III "equilibrium" estimate in Table 6-20. 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 re-expressed as an estimate of the effects expected for a given change in the level of background radiation (Table 6-24). Since these calculations are based both on the background-level mutations and the doubling dose, changes in either must be reflected in new calculations.

Table 6-24. Increase in background or 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	2.1
1.0	8.0	21.2
10.0	80	212

6.5.7 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 6-25. Uncertainties listed in this table 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.

6.5.8 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. (St21, Mu29, Go29). However, the irradiation exposures were high.

In 1930, Murphy exposed rats to X-rays at doses of 200 roentgens (R) to 1,600 R. Of 120 exposed females, 34 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).

Table 6-25. 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/÷ factor of 4
Selection of species and loci to use in developing a doubling dose +indeterminate [a]	-100% to estimate
Use of - division by a factor of 3 - to convert acute, high dose, low-LET estimates to chronic, low-LET estimates	x/÷ factor of 3
Sensitivity of oogonia compared to spermatogonia as described in BEIR-III	-44% to 56%
Background rate selected for use with a doubling dose	x/÷, 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	x/÷ a factor of 2 [b]

[a] The risk estimate cannot go below zero, -100%; but it may not be possible to determine the upper bound, indeterminate.

[b] 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 2 times greater than the doubling dose estimate derived from the mouse.

Much of the work done after World War II used mice (Ru50, Ru54, Ru56) or 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 exposures causing them. The lowest reported exposure was 12.5 R for structural defects and one R for functional defects. He also suggested human exposure between ovulation and about seven weeks gestational age could lead to structural defects, and exposures from about six 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, some data support this theory (Ot83, Ot84).

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 one 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 eighth 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 that are most sensitive at that time, higher doses may induce additional abnormalities in components that 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 (Do73). 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, Do81).

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 mid-gestation 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, Do81). As a consequence, it is unreasonable to compare a newborn rat's brain, which has not begun to myelinate, with that of a newborn human or a newborn guinea pig in which myelination has been completed (Do79, Do81).

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 necessary only 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, Do81). 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.

6.5.8.1 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 (Pl52). Twelve children with microcephaly, of which ten 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 a number of causes, e.g., genetic conditions, infections, physiologic conditions, 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 through the 15th week of gestation.

The ICRP published an excellent review of biology and possible mechanisms of occurrence of radiation-induced brain damage, in utero (ICRP86). ICRP estimates: (1) for exposures from the 8th through the 15th week after conception, the risk of severe mental retardation is 4×10^{-3} per rad, with a confidence interval of 2.5×10^{-3} to 5.5×10^{-3}, and (2) for exposures from the 16th through the 25th week after conception, the risk of severe mental retardation is 1×10^{-3} per rad. However, a threshold below 50 rad cannot be excluded (ICRP86).

The 1986 UNSCEAR Committee also reviewed biology and possible mechanisms (UNSC86). Although increased external granular layer pyknosis had been found in rats after exposures of three rads and degraded behavioral performance had been reported in rats after four one-rad doses, the UNSCEAR Committee concluded that ". . . no effects having clearly pathological connotations have been reported for doses in the brain structures lower than 0.1 Gy (10 rad) low-LET radiation." (UNSC86).

If the ICRP 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 (EPA84).

6.5.8.2 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.

Studies of the Japanese survivors 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 to exist, it has not been quantified.

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 during the critical period.

6.5.8.3 Other Teratologic Effects

Effects other than mental retardation and microcephaly have been noted in the Japanese A-bomb survivors. Schull, et al., reported that in individuals exposed prenatally between weeks 8 and 25 of gestation there is a progressive shift downward in IQ score with increasing exposure and that the most sensitive group is between 8 and 15 weeks gestational age at time of exposure (Sc99). Much the same pattern was reported for average school performance, especially in the earliest years of schooling (Ot88). Finally, a linear-nonthreshold relationship between exposure and incidence of unprovoked seizures in later life has been demonstrated to be consistent with the data for individuals exposed between 8 and 15 weeks gestational age (Du88).

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 (UNSC77) made a very tentative estimate based

on animal studies that the increased incidence of structural abnormalities in animals may be 0.005 cases per R per live born but stated that projection to humans was unwarranted. In 1986, UNSCEAR assumed the risk of an absolute increase of malformed fetuses of the order of 5×10^{-3} per rad seen in animals might apply to the human species as well, for exposure over the period from two to eight weeks post-conception (UNSC86). In any event, the available human data cannot show whether the risk estimates derived from high-dose animal data overestimate the risk in humans or if a threshold can be excluded.

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." (UNSC77).

6.5.9 Nonstochastic Effects

Nonstochastic effects, those effects that increase in severity with increasing dose and have a threshold, were reviewed in the 1982 UNSCEAR report (UNSC82). Further, nonstochastic effects following in utero exposure were reviewed in the 1986 UNSCEAR report (UNSC86). In general, acute doses of 10 rads low-LET radiation and higher are required to induce these effects in animals. 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 (UNSC77) or per rad (UNSC86) following radiation exposure soon after fertilization. However, there are no data to address the question of similar effects in humans. Usually, nonstochastic effects are not expected at environmental levels of radiation exposure.

In 1986, the UNSCEAR also reviewed the question of mental retardation as a part of the overall review of the biological effects of prenatal radiation exposure (UNSC86). The UNSCEAR, like the ICRP, concluded there was a risk of severe mental retardation of 4×10^{-3} per rad over the period of eight to 15 weeks after conception and of 1×10^{-3} per rad over the period of 16 to 25 weeks after conception (UNSC86). The UNSCEAR also estimated: (1) a pre-implantation loss of 1×10^{-2} per rad during the first two weeks after conception; (2) a malformation risk of 5×10^{-3} per rad during weeks two to eight after conception, and (3) a risk of leukemia and solid tumors expressed during the first 10 years of life of 2×10^{-4} per rad (UNSC86).

The British National Radiation Protection Board (NRPB) reviewed available information, including the 1988 UNSCEAR report to develop new health effects models (St88). The NRPB estimated a mental retardation risk of 4.5×10^{-3} cases per rad of exposure during weeks eight to 15 of gestation. The NRPB also estimated a cancer risk of 2.5×10^{-4} cases of leukemia and 3.5×10^{-4} cases of solid tumors per rad of in utero exposure (St88).

EPA has adopted similar risk coefficients for estimating prenatal carcinogenic, teratologic, and nonstochastic effects in man (see Table 6-26).

6.6 SUMMARY OF EPA'S RADIATION RISK FACTORS - A PERSPECTIVE

Table 6-27 summarizes EPA's estimate of risk from lifetime whole-body exposures to high- and low-LET radiation and to radon decay products. The nominal risk factors reflect EPA's best judgment as to the relationship between dose and risk based on review of all

Table 6-26. Possible effects of in utero radiation exposure.

Type of Risk Conceptus	Risk per Rad	Risk per Event in a 100 mrad per Year Background
Fatal Cancer	6.0×10^{-4}	4.5×10^{-5}
Mental Retardation (exposure at 8 - 15 weeks)	4×10^{-3}	6.0×10^{-5}
Mental Retardation (exposure at 16 - 25 weeks)	1×10^{-3}	1.5×10^{-5}
Malformation (exposure at 2 - 8 weeks)	5×10^{-3}	5.8×10^{-5}
Pre-implantation Loss (exposure at 0 - 2 weeks)	1×10^{-2}	3.8×10^{-5}

relevant information available to the Agency. Likewise, the cited ranges reflect EPA's current best judgment as to the uncertainties in these risk factors.

To provide a perspective on the risk of fatal radiogenic cancers and the hereditary damage due to radiation, EPA has calculated the risk from background radiation to the U.S. population using the risk factors summarized in Table 6-23. The risk from background radiation provides 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 absorbed dose rate from low-LET background radiation has three major components: (1) cosmic radiation, which averages about 28 mrad/yr in the United States; (2) terrestrial sources, such as radium in soil, which contribute an average of 28 mrad/yr (NCRP87); and (3) 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 (NCRP87). Other minor radiation sources such as fallout from nuclear weapons tests, cosmogenic radionuclides, naturally occurring radioactive materials in buildings, airline travel, and consumer products, contribute about another 7 mrad for a total low-LET whole-body dose of about 87 mrad/yr. The lung and bone receive somewhat larger doses, not included in the 87 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 6.2, the BEIR III linear, relative-risk models yield, for lifetime exposure to low-LET radiation, an average lifetime risk of fatal radiogenic cancer of 3.9×10^{-4} per rad. Note that this average is for a group having the age- and sex-specific mortality rates of the 1970 U.S. population. This risk estimate can be used 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 90 mrad/yr, the average lifetime dose is 6.4 rads. The risk of fatal cancer per person in this group is:

$$(3.9 \times 10^{-4}\ \text{rad}^{-1})\ (8.7 \times 10^{-3}\ \text{rad/y})\ (70.7\ \text{y}) = 2.4 \times 10^{-3} \qquad (6\text{-}11)$$

or about 0.24 percent of all deaths. The vital statistics used in EPA's 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 0.24 percent result for the BEIR III linear dose response model indicates that about 1.5 percent of all U.S. cancer is due to low-LET background radiation. The BEIR III 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 6-11 indicates a risk of $5.6 \times 10^{-4}\ \text{rad}^{-1}$ for alpha emitters in lung tissue. UNSCEAR estimated that in "normal" areas the annual absorbed dose in the lungs from alpha emitters other than radon decay products would be about 0.51 mrad (UNSC77). The individual lifetime cancer risk from this exposure is:

$$(5.6 \times 10^{-4}\ \text{rad}^{-1})\ (5.1 \times 10^{-4}\ \text{rad/y})\ (70.7\text{y}) = 2.0 \times 10^{-5}, \qquad (6\text{-}12)$$

which is about 1/100 of the risk due to low-LET background radiation calculated by means of the BEIR III linear model.

The 1982 UNSCEAR report indicates that the average annual absorbed dose to the endosteal surfaces of bone due to naturally occurring, high-LET alpha radiation is about 6 mrad/yr, based on a quality factor of 20 and an absorbed dose equivalent of 120 mrem/yr (UNSC82). Table 6-11 indicates that the individual lifetime risk of fatal bone cancer due to this portion of the naturally occurring radiation background is:

$$(2.0 \times 10^{-5}\ \text{rad}^{-1})\ (6 \times 10^{-3}\ \text{rad/y})\ (70.7/\text{y}) = 8.5 \times 10^{-6}. \qquad (6\text{-}13)$$

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 United States an indoor concentration of about 0.004 working levels (15 Bq/m^3) (UNSC82). 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 UNSC82, the national collective exposure may not be too dependent on exceptions to the mean concentration. The UNSCEAR estimate

Table 6-27. Summary of EPA's radiation risk factors.

Risk	Significant Exposure Period	Risk Factor Nominal	Risk Factor Range
Low LET		(10^{-6} rad^{-1})	
Teratological:			
Severe mental retardation	Weeks 8 to 10 of gestation	4,000	2,500 - 5,500
Genetic:			
Severe hereditary defects, all generations	30-year reproductive generation	260	60 - 1,100
Somatic:[a]			
Fatal cancers	Lifetime	390	120 - 1,200
All cancers	Lifetime	620	190 - 1,900
Fatal cancers	In utero	600	80 - 1,800
High LET		(10^{-6} rad^{-1})	
Genetic:			
Severe hereditary defects, all generations	30-year reproductive generation	690	160 - 2,900
Somatic:			
Fatal cancers	Lifetime	3,100	960 - 9,600
All cancers	Lifetime	5,000	1,500 - 15,000
Radon decay products		(10^{-6} WLM^{-1})	
Fatal lung cancer	Lifetime	360	140 - 720

[a] The range assumes a linear, non-threshold dose response. However, it is plausible that a threshold may exist for this effect.

for the United States 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 this country, the mean lifetime exposure, indoors, is about 18 WLM. Based on the geometric mean lifetime risk coefficient from Section 6.4.5, 360 cases/10^6 WLM, a lifetime risk of 0.64 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 of eight lung cancer deaths may be attributable to background radon exposure. This would correspond to about 4 percent of all cancer deaths. This is 2.5 times the 1.61 percent of all cancer fatalities estimated above for low-LET background radiation. The reader is cautioned, however, that this risk estimate applies only to the United States population taken as a whole, i.e., men and women, smokers and nonsmokers. Since the vast majority of the 1980 lung cancer mortality occurred in male smokers, this risk estimate cannot be applied indiscriminately to women or nonsmokers (see Section 6.4).

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, UNSC82). The low-LET background radiation dose of about 87 mrad/year in soft tissue results in a genetically significant dose of 2.6 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 at 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 about 690 genetic effects per 10^6 live births (see Section 6.5). This result indicates that about 0.6 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 7: INDIVIDUAL DOSE ASSESSMENT OF DISPOSAL OF TRANSURANIC WASTE IN MINED GEOLOGIC REPOSITORIES

7.1 INTRODUCTION

This chapter deals with the disposal of transuranic (TRU) waste in geologic repositories. Most TRU waste is from the production of nuclear weapons. It consists of materials contaminated mainly with radioactive isotopes of plutonium and americium but also contains other transuranic isotopes. The waste form varies widely but most of the waste can be described as contaminated plastic, wood, rubber, metal, cloth, paper, and laboratory trash.

Risk assessments for spent nuclear fuel, high-level, and TRU radioactive wastes were conducted in support of the disposal standards proposed in 1982. Review of the risk assessment by an EPA Science Advisory Board committee produced a number of recommendations which called for the analyses to be less conservative, that is, to use parameter values that were considered more likely and that would tend to produce lower estimates of population risks. In addition, DOE had developed extensive data on the nine specific locations to be evaluated for the first disposal facility. These data were available for use in risk assessments for the final rule.

An important consideration in repromulgating disposal standards has been the assessment of risks associated with the disposal of these wastes in mined geologic repositories. The risk assessments carried out in support of the development of the standards are intended to be "generic" in nature. In developing the assessments, the Agency considered a wide range of geologic environments and other related parameters. In the early stages of the EPA work, the DOE, which is responsible for developing a geologic disposal facility, had not yet developed extensive data associated with its principal candidate sites. Therefore, the risk assessments conducted in support of the 1982 proposed standards used data from the general literature on potential waste disposal environments as well as the limited data that had been obtained by DOE up to about 1980. Individual dose assessments in the current effort utilize recent data from DOE efforts to develop mined geologic repositories for high-level and TRU radioactive wastes.

The performance of the generic TRU waste disposal facility has been evaluated using the same methodology as the risk assessments for the spent nuclear fuel repositories presented in 1985 (EPA85) and updated in 1992 (RAE92). In this chapter, performance is quantified in terms of risks to an individual consuming contaminated ground water near the disposal facility. Radionuclide releases are assumed to occur through normal ground-water flow and gaseous transport, if applicable.

The risk assessment for the TRU waste disposal system is based upon the same conceptual models and data used in the spent nuclear fuel repository analyses. The data used in the TRU risk assessment are presented in detail later in this chapter. Because of the generic nature of the assessment, the results of the risk calculations are not intended to project actual risks expected at particular sites; such projections will be possible only after the potential sites are more fully characterized. Instead, the assessments are

intended to provide estimates of potential disposal facility performance. The Agency has attempted to ensure that its generic calculations are based upon reasonable parameter values.

7.2 TIME FRAME

Recommendations in the published literature vary widely on the time frame over which radioactive waste disposal alternatives should be evaluated or compared. If the future could be predicted with accuracy, then a very long time frame would provide a more complete evaluation than a short time frame and, ultimately, a more complete comparison among alternative disposal systems. For the containment requirements now in effect under 40 CFR Part 191, compliance must be demonstrated over a period of 10,000 years. That demonstration requires an analysis of the movement of radionuclides out of the repository into the environment. Such an analysis is also a first step in demonstrating compliance with the individual dose or ground-water protection requirements. For these requirements, the second step in the analysis involves following the radionuclides through the environment via pathways by which an individual could be exposed to radiation.

In the course of performing numerous risk assessments of radioactive waste disposal systems, the Agency has concluded that the risks identified over relatively short time spans, such as a few hundred to one thousand years, do not adequately portray important differences among alternative sites or disposal systems. This is because the ground-water travel times would probably be sufficiently long at most sites that no significant radionuclide releases would be predicted over this time period. If the analyses were carried further into the future, there could be substantial differences among the sites because of their different hydrologic or geochemical characteristics. With these considerations in mind, the risk assessments carried out in support of this rulemaking have been based upon a time frame of 10,000 years. This time frame appears long enough to identify important differences among sites and among other aspects of the disposal systems. Many of the computer simulations have been extended to 100,000 years in order to provide better insight on the long-term performance of disposal alternatives but this does not imply that the Agency believes that, at this time, technology and data would allow this time frame to be used to reasonably predict performance at any particular site.

Part of the risk assessment is concerned with the uncertainties in the calculated results. There are several sources of uncertainty, including spatial and temporal variations in site parameter values, an incomplete knowledge of the natural site characteristics, and the prediction of possible future events at the disposal site. Since site conditions far in the future are more difficult to reliably predict, the uncertainties in modeling system performance may increase with the length of the simulation period. Uncertainties may also increase as the radionuclide transport distance increases. In other words, it is easier to reliably predict the transport of radionuclides over a short distance than over a long distance. The variabilities in transport parameter values over a long flow path are generally greater than those for a shorter flow path. Thus, the uncertainties depend both on the distance to the accessible environment and the length of the simulation period.

7.3 MEASURES OF RISK

In examining the long-term effects of a radioactive waste repository, the Agency has considered both population and individual doses. These two risk measures provide very different perspectives. For example, a relatively unproductive ground-water supply could be contaminated with radionuclides released from a repository at some point in the future, but because of the limited availability of water from this supply only a few individuals would be exposed to the radioactivity. In this case, the individual doses might be high although the overall effect on the population could still be quite low. On the other hand, a future release from a radioactive waste repository could lead to very low level contamination of water supplies that serve a large population. In this case, the dose to any individual in that population might be small while the cumulative population dose could be substantial in terms of total cancers and genetic effects in the population. Because of these differences, the Agency has developed and applied techniques that can estimate both population and individual risks.

This analysis addresses only individual risks and ground-water protection as the population risks are addressed in the now reinstated containment requirements. An estimate of an individual's risk is determined by estimating the annual radiation dose from consuming two liters per day of ground water contaminated with radionuclides from the repository. This exposure is presented as a function of time after disposal for an individual using ground water at a particular distance (typically two kilometers) directly down gradient from the edge of the repository. This report uses the maximum individual dose from ingestion of drinking water as a measure to compare the effectiveness of various types of geologies and engineered designs.

Ground-water protection is evaluated in terms of the concentrations of radionuclides down gradient from the repository. The concentrations are calculated for Ra-226, total alpha-emitters, and beta- and gamma-emitting radionuclides.

7.4 COMPUTER CODE UTILIZED

The computer code used in this assessment is NEFTRAN-S. This code was developed by Sandia National Laboratories under contract to the NRC. The model is described and documented in SAND90. NEFTRAN-S calculates cumulative radionuclide releases, ground-water concentrations, and individual doses. Because it calculates radionuclide decay and ingrowth during transport, it is useful for assessment periods in excess of 10,000 years. The code uses the distributed velocity method to calculate radionuclide transport in a network of one-dimensional legs. The flow network is designed to represent ground-water flow in the vicinity of a repository.

The distributed velocity method used in NEFTRAN-S is a method for modeling the dispersive or diffusive transport of radionuclides in porous media. Rather than using a single transport velocity, a range of transport velocities is used. The radionuclide inventory is partitioned and each portion of the inventory is transported at a different velocity. This simulates the effect of dispersion or diffusion by allowing portions of the inventory to be transported at higher or lower velocities than the centroid of the contaminant plume. The distributed velocity method has some computational advantages

over other numerical approximation methods when applied to dispersive radionuclide transport. More detail on the mathematical basis of the distributed velocity method is given in reference SAND90.

One of the most useful aspects of the NEFTRAN-S code is its capacity to perform probabilistic analyses. The code uses a Monte Carlo sampling method to randomly select input values from probability distributions. For each random selection of input values, the transport model is executed and the results from each such sample are saved. After a number of samples have been evaluated, the results can be expressed as a probability distribution. This procedure is used in the risk assessment to evaluate the effects of parameter uncertainties.

7.5 GENERIC DISPOSAL SYSTEM FOR TRANSURANIC RADIOACTIVE WASTE

7.5.1 System Models

The waste disposal system considered in this risk assessment is based on national plans to develop mined geological repositories for disposal of TRU radioactive wastes. Such repositories consist of underground mines or excavations with working levels between 300 and 1,000 meters below the surface. Like the spent fuel repository assessment presented in 1985, the TRU waste disposal facility assessment will focus on disposal in four different host rock types.

Transuranic radioactive wastes differ from spent nuclear fuel and high-level radioactive waste in both radionuclide content and waste form. Transuranic wastes consist of a variety of waste forms, including plastic, rubber, wood, glass, cloth, and metal. The waste is generated from reprocessing, fabrication, and research at DOE facilities. The principal radioactive constituents of TRU waste are plutonium and americium. Present plans call for disposal in a mined geologic facility with the wastes packaged in metal drums or boxes and stacked in the mined waste disposal rooms. After emplacement of the wastes, the disposal facility would be backfilled to enhance its mechanical stability and to retard the movement of fluids. The shafts and boreholes which connect the disposal facility to the surface would be backfilled and sealed.

The structure of the analysis can be represented as shown in Figure 7.5-1. The components of the system to the right of the vertical dotted lines represent the "accessible environment." The components on the left side of the diagram represent the release and transport mechanisms from the repository to the accessible environment.

In order for radionuclides to reach the accessible environment they must be released from the waste form. Since much of the radioactivity in TRU waste is present as surface contamination, the waste form is not expected to significantly limit the radionuclide release rates. The release rates are likely to be controlled by radionuclide solubility. Upon leaving the waste form, the radionuclides enter the backfilled openings of the disposal facility. In general, radionuclides may travel from the disposal facility to the accessible environment in three ways: 1) direct pathways to the land surface, 2) vertical migration in slowly moving ground water to an aquifer and then to the surface,

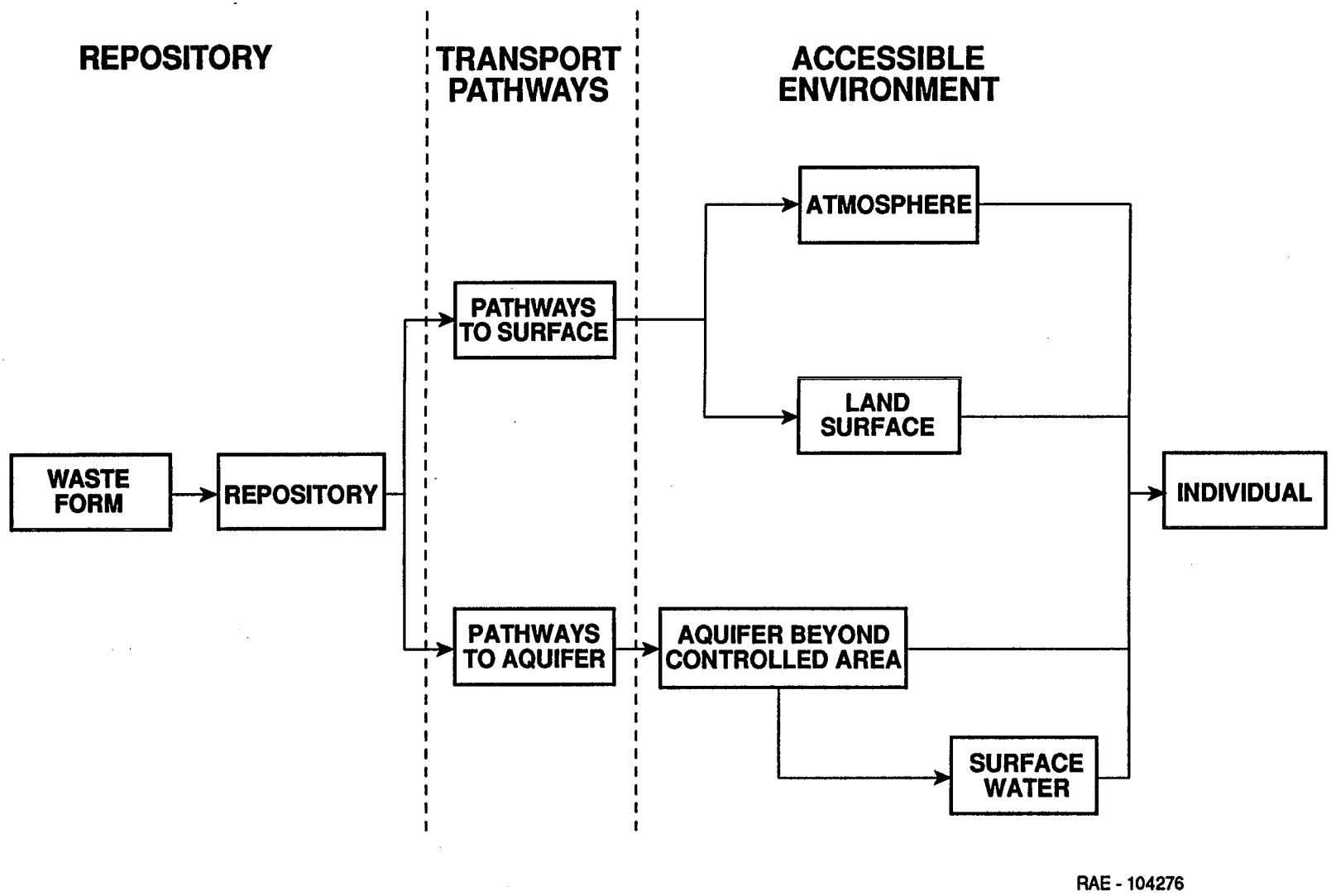

Figure 7.5-1. Components in the risk assessment of radioactive waste releases.

and, 3) transport of radioactive gases to the ground surface from a disposal facility in the unsaturated zone.

Direct pathways to the surface, caused by unusual events or human intrusion, are not evaluated in the current risk assessment. Gaseous transport is also excluded from the current analysis because TRU waste does not have a significant potential to produce radioactive gases.

Release pathways involving ground water present the possibility that individuals may be exposed to radioactivity by ingesting contaminated water from the aquifer. The possibility of such a scenario could be minimized by siting the disposal facility in an area where human use of ground water is unlikely. It is assumed in these analyses that the disposal facility is sited in a remote area where ground water is not used to support the needs of a large population.

The movement of radionuclides from the waste form, through the disposal facility, and ultimately to the accessible environment depends on a number of possible scenarios that might alter the conditions of the underground environment. The risk analyses reported here consider only normal ground-water flow.

7.5.2 Disposal Facility Parameters

Certain assumptions need to be made regarding the geometry and physical characteristics of the disposal facility. However, an examination of the Agency's risk analysis models indicates that they are not highly sensitive to these engineering assumptions. The layout of the generic TRU waste disposal facility is based on the design of the Waste Isolation Pilot Plant (WIPP). These parameters, which are summarized in Table 7.5-1, are taken from La89.

Two general categories of TRU waste are planned to be disposed in the facility. Contact-handled (CH) wastes are those with surface dose rates less than 200 mrem per hour. These wastes, which account for the majority of TRU waste, will be disposed in steel drums. The drums will be placed in the disposal rooms and stacked in three tiers. Waste packages with surface dose rates exceeding 200 mrem per hour are classified as remote-handled (RH) wastes. These wastes cannot be handled directly because of the high radiation levels. Remote-handled wastes will be disposed in special canisters which are placed in boreholes in the walls of the disposal rooms or drifts. Although the surface radiation levels of some TRU wastes are high, there is relatively little heat generated by the radioactive decay so no special facility design features are assumed necessary to dissipate the small anticipated thermal loading.

Each waste room measures about 92 meters long, 10 meters wide, and 4 meters high. Seven disposal rooms make up one panel. There are a total of ten panels in the underground disposal facility. The ten panels are enclosed in an approximately square area of 490,000 square meters. The total excavated area is about 110,000 square meters. After each panel is filled with waste and backfill, the panel is sealed to isolate it hydrologically from the other waste panels. When all waste is emplaced in the facility, the vertical access shafts to the waste horizon will be backfilled and sealed.

Table 7.5-1. Repository parameters used in TRU waste risk assessment.

Parameter	Value
Dimensions of repository	
Length	700 m
Width	700 m
Height	4 m
Repository area	490,000 m^2
Excavated area	110,000 m^2
Total mined-out volume	440,000 m^3
Number of waste drums	583,000
Number of waste panels	10
Number of waste rooms per panel	7
Waste room dimensions	
Length	92 m
Width	10 m
Height	4 m

Source: La89

A total controlled area of 100 square kilometers at the disposal site will provide a distance of approximately five kilometers from the center of the site to the accessible environment. For consistency with the repository risk assessments performed for HLW repositories, the TRU waste risk assessment uses a distance of two kilometers from the edge of the disposal facility to the accessible environment.

The disposal standard allows a distance of up to five kilometers to the accessible environment. The analyses presented here are based on a conservative distance of two kilometers. If an implementing agency chooses to use a five-kilometer distance instead, the performance assessment would show a greater margin of protection than for a two-kilometer distance. The analyses presented here show that the standard is technically feasible for a two kilometer distance.

For modeling purposes, the cross-sectional area of the ground-water flow path in the aquifer is defined by the length of the disposal facility (perpendicular to the flow path) multiplied by the thickness of the uppermost aquifer. Since the generic TRU waste disposal facility is approximately square, the orientation of the facility relative to the aquifer flow direction is not important.

The mined volume of the facility, as well as the porosity of the backfill, must be considered in calculating the amount of radionuclides that might dissolve in the ground water that could gradually seep into the disposal facility after its closure. Because such dissolution might be limited by solubility factors, this water volume is significant to some models.

7.5.3 Waste Form Parameters

The principal radionuclides in TRU waste presently planned to be emplaced into the disposal facility are plutonium (Pu), americium (Am), and uranium (U). The waste also contains smaller amounts of short-lived radionuclides such as strontium (Sr) and cesium (Cs). The radionuclide inventory used in the generic risk assessment is taken from recent projections of the waste inventory for the WIPP site. The estimates are documented in the Environmental Impact Statement for the WIPP facility (DOE89) and in the System Analysis of the WIPP (La89). The estimates include existing waste in storage at DOE facilities and waste expected to be generated through the year 2013.

About 96 percent of the TRU waste volume is classified as CH waste. The CH waste contains about 95 percent of the total radioactivity. The principal radionuclides in CH waste are, in order of decreasing activity, Pu-238, Pu-241, Am-241, Pu-239, and Pu-240. The RH waste accounts for about 4 percent of the volume and 5 percent of the total activity. The principal radionuclides in RH waste are, in order of decreasing activity, Pu-241, Pu-239, Pu-238, Sr-90, Cs-137, and Pu-240. The total combined inventory of CH and RH wastes is shown in Table 7.5-2.

It is estimated that the TRU waste inventory will be contained in 385,000 drums and 19,500 boxes (La89). Of the boxes, 13,500 are Standard Waste Boxes, each with a volume of 1.78 cubic meters. The remaining 6,000 boxes are of the "old" type. Each "old"

Table 7-5-2. Radionuclide inventory in the generic TRU waste repository.

Radionuclide	Initial Quantity in Repository (curies)[a]	Half-Life (years)	Ingestion Dose Conversion Factor[b] (mrem/Ci)
Sr-90	70,400	29	1.30E+08
Cs-137	59,600	30	4.61E+07
Th-232	0.274	1.4E+10	4.77E+08
U-233	7,800	159,000	1.06E+09
U-235	42	7.04E+8	1.00E+09
U-238	22.3	4.47E+9	9.46E+08
Np-237	8.02	2,140,000	4.01E+09
Pu-238	3,980,000	88	3.85E+09
Pu-239	519,000	24,400	4.46E+08
Pu-240	136,000	6,540	4.45E+08
Pu-242	23.3	376,000	4.24E+08
Am-241	782,000	432	4.43E+09
Cm-248	0.188	348,000	1.60E+10

NOTE: For convenience, some radionuclides which were found to be very small contributors to the total risks were omitted. Omissions are described in the text.

[a]Source: La89.
[b]Source: EPA89.

box measures 4 x 4 x 7 feet and has a volume of 3.2 cubic meters. Most of the old boxes (4,500) are wooden and the rest are metal. The total waste volume is equivalent to the volume of 583,000 drums.

The waste containers for the majority of the TRU waste are not designed to provide radionuclide containment for a long period of time. Under the conditions likely to be present in the disposal facility after closure, the steel drums would be subject to rapid corrosion. The wooden boxes would also degrade rapidly because of the presence of moisture or through bacterial decomposition. In modeling the long-term performance of the disposal facility, no credit is taken for the integrity of the waste containers. At the time of disposal, all of the TRU waste inventory is assumed to be in contact with moisture and to begin being released from the waste form.

The TRU waste is such that the present waste form is not expected to limit the releases of radionuclides. Radionuclide release is assumed to be controlled by solubility. At the bedded salt site, for example, the brine in the pore spaces of the salt can be highly corrosive (La89), so radionuclide solubilities in the brine could be quite high. The solubility is assumed to be the same for all radionuclides. A solubility of 10^{-6} mole/liter is used in the base case analysis (La89) for all sites and other values are evaluated in the sensitivity study.

Some of the radionuclides included in the waste inventories in DOE89 and La89 were not included in the risk assessment because of their short half-lives. On the basis of half-life, cobalt (Co)-60, ruthenium (Ru)-106, antimony (Sb)-125, cerium (Ce)-144, and Eu-155 have been eliminated from the risk assessment. The remaining short-lived radionuclides, Pu-241, curium (Cm)-244, and californium (Cf)-252, decay to long-lived radionuclides that must be included in the assessment.

The inventory shown in the table was used in NEFTRAN-S computer calculations. The NEFTRAN-S code has the capability to perform the ingrowth calculations.

7.5.4 Release Mechanism

In this analysis, only undisturbed normal ground-water flow is analyzed. The results are individual dose rates and ground-water contamination levels.

7.5.4.1 Normal Ground-Water Flow

All scenarios involving ground water are modeled using a Darcian flow system. The ground-water transport pathways all involve a vertical and a horizontal leg. The vertical leg is from the disposal facility vertically to an aquifer. The horizontal leg is the distance from the edge of the repository to the accessible environment. The five values needed to predict Darcian flow for each leg are distance, hydraulic conductivity, porosity, hydraulic gradient, and cross-sectional area. The first four are used to find travel time by the expression:

$$T = (d \cdot p)/(i \cdot K)$$

where:

T is the fluid travel time (years)
d is the length of the leg (meters)
p is the effective porosity
i is the hydraulic gradient and,
K is the hydraulic conductivity (meters/years)

Volumetric flow of water is found by:

$$V = K \cdot i \cdot A$$

where:

V is the volumetric flow (cubic meters/year)
i is the hydraulic gradient, and
A is the cross-sectional area of the pathway (square meters)

Additional equations used to implement the conceptual models are discussed in the computer code manual for NEFTRAN-S (SAND90). Specific parameter values characterizing the disposal facility are presented later in this section.

Unlike disposal facilities in basalt, granite, and tuff, a disposal facility in salt will have no normal ground-water flow through the undisturbed host rock. During the construction and operation of the disposal facility, water in the surrounding rock would be expected to gradually drain so that the rock will enter an unsaturated condition near the openings. After the end of the operational period and sealing of the disposal facility, water is expected to gradually seep back into pores and fractures in the rock. The creep closure of the salt will cause the hydraulic conductivity near the waste horizon to gradually decrease back to its value before excavation of the waste facility. This is assumed to effectively prevent any flow of water from the disposal facility, except in cases where inadvertent human intrusion or faults provide high permeability flow paths.

7.5.4.2 Gaseous Releases of Radionuclides

Some waste disposal sites present the possibility that radionuclides may be released in gaseous form. For gaseous release to occur, the geologic medium must be porous and unsaturated. The presence of air-filled pore spaces allows diffusive and/or advective transport of the radioactive gases. Decomposition of organic material in TRU waste may produce gases such as methane, hydrogen, or hydrogen sulfide. However, these gases are not likely to contain radioactive isotopes. Since TRU radionuclides are not likely to exist in gaseous form, the transport of radioactive gases from the disposal facility is not considered a viable release mechanism.

7.5.5 Generic Site Media Analyzed

Generic site models have been developed for four geological media found in the United States: bedded salt, basalt flows, unsaturated volcanic tuff formations, and granite. Three of the four generic site models were developed based on actual sites determined to be representative of the media: (1) the bedded salt deposits in the Palo

Duro Basin in Texas and the Paradox Formation in Utah, (2) the basalt flows on the Hanford reservation in Washington, and (3) the unsaturated volcanic tuff formations at Yucca Mountain in Nevada. No specific granitic formations were used as the basis for the granite model but rather the granite model presented here attempts to roughly represent geological and hydrological conditions that might be encountered in certain regions, viz., the north-central and New England areas (EPA85).

The four generic conceptual site models are configured similarly (Figure 7.5-2). A number of parameters are used to describe the geologic and hydrologic conditions assumed for each of the model repository sites. These are included in the description of the analysis performed for each generic site medium. The conceptual framework of the lithology for the basalt and salt sites is that the repository horizon is situated between an "upper aquifer" and a "lower aquifer." The tuff model assumes unsaturated conditions at the repository level with only a lower aquifer. Since it is assumed that in the granite model the repository is located within a granitic plutonic body, there is only an upper aquifer in the granite model.

To simulate conditions present at a real site, the aquifers do not represent single hydrostratographic units but rather they represent "synthetic aquifers" whose properties are defined to approximate the combined properties of a number of transmissive units above and below the repository horizon. For example, if a number of such transmissive units are present above the repository at a particular site and if the application of a generic model described here is intended to represent conditions similar to those at the site, then one can calculate the combined volumetric flows in the upper units and define appropriate hydrologic parameters so that the equivalent aquifer conveys the same total flow. Similarly, by varying one or more additional parameters, it is possible to simulate the effective fluid velocity in any one of the actual units. This will be illustrated in subsequent sections when specific lithologies are discussed.

The ground-water pathway in the generic risk analyses is modeled with the NEFTRAN-S code. At saturated sites, the upper aquifer is assumed to be the aqueous pathway of radionuclide transport. An upward gradient is assumed to exist between the repository horizon and the upper aquifer. Thus, greater emphasis is generally placed on the properties of the upper aquifer. At potential repository sites, however, the hydrogeologic environment may be different from that assumed in the generic model. For example, there may be no significant aquifer below the repository (as in a number of crystalline rock sites), or above the repository (as in the case of a repository in the unsaturated zone), or there may be a prevailing gradient that is downward from the upper aquifer, in which case the lower aquifer would appear to be the more likely release pathway. These cases can all be accommodated within the modeling of NEFTRAN-S.

The four generic conceptual site models are discussed in the following four sections. Each section presents a description of the conceptual site model in terms of the parameters used to evaluate the model through the NEFTRAN-S code. The results of the individual dose and ground water protection assessments and sensitivity analyses are also presented in each section. The results of the four sets of analyses are compared in Section 7.5.6.

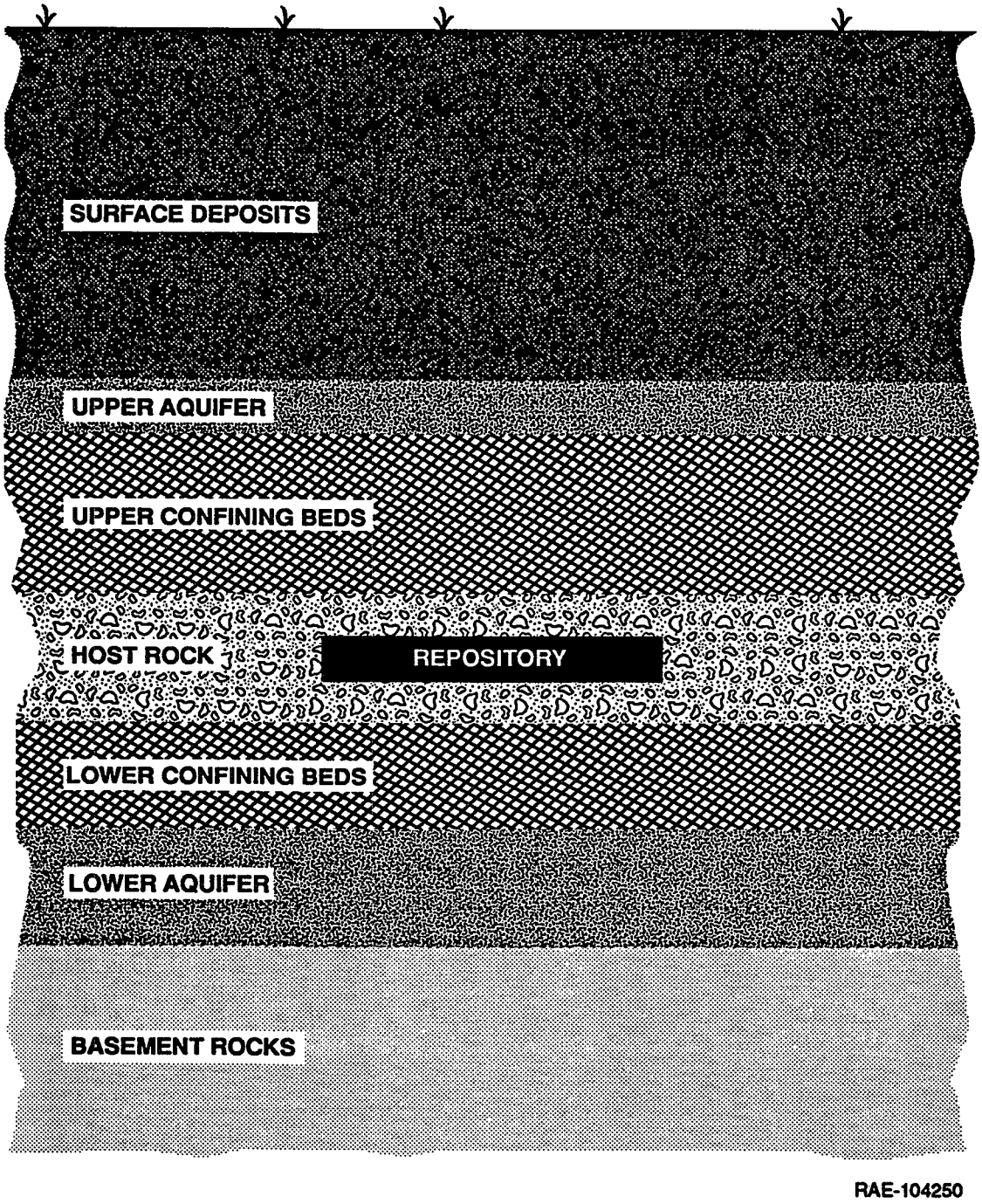

Figure 7.5-2. General cross-sectional structure for risk analysis.

7.5.5.1 Site Analysis - Basalt

7.5.5.1.1 Introduction

Basalt deposits in the Pacific Northwest have been under investigation for a number of years as potential host formations for a nuclear waste repository. Basalt is a dense, dark, fine-grained rock formed by the solidification of volcanic lava. The basalt deposits in the Northwest are flood basalts. They were extruded over extremely large areas and formed a layered structure of individual flows tens to hundreds of feet thick, separated by relatively minor sedimentary deposits and fractured or highly porous zones between the basalt flows. The dense interiors of the basalt flows are the potential repository host rocks evaluated in this section. Basalt deposits are permeated by fractures, but at the depths being considered for a repository, these fractures are expected to be quite tightly closed, thereby restricting the volume and the velocities of any ground-water movement. Nevertheless, there is expected to be some ground-water migration through a basalt repository and it is possible this might be accelerated by repository-induced effects. The layered structure of the basalt deposits provides for horizontal ground-water movement through relatively permeable zones between flows, In addition, the fracturing in a basalt deposit is expected to be somewhat greater than that in a well-chosen repository site in granite. This does not mean that such fracturing would necessarily lead to unacceptable repository performance, but only that it must be an important consideration in choosing a site and in estimating the performance of a repository at that site.

Before 1987, the DOE had investigated the possibility of siting a repository in basalt at the Hanford Reservation in southeastern Washington State. The relatively advanced stage of the DOE investigations at Hanford has provided considerable data on the characteristics of potential sites and repository host flows. However, much of the work carried out at the Hanford Reservation had been the subject of severe criticism by the NRC and others, and therefore the Agency incorporated into its analyses input not only from the DOE and its contractors but also from technical professionals from other organizations. Based on such data, the Agency believes that it is possible to define conceptual models of a basalt repository that should be adequate to make rough approximations of the potential performance that might be expected from such repositories and to identify some of the parameters that are most critical in determining that performance.

Section 7.5.5.1.2 discusses the important input parameters that have been used in the Agency's risk analyses for basalt. The data are based primarily on the Hanford site. Since most of those data can best be presented in the form of tables and figures, there is a minimum of text discussing additional details in this section. Also presented are data from the EPA population risk assessment (EPA82). Section 7.5.5.1.3 provides the results of the "base case" analyses of individual risks and ground-water contamination. Section 7.5.5.1.4 provides the results of the sensitivity and uncertainty analyses.

7.5.5.1.2 Input Parameters for Basalt

The conceptual model developed to support the evaluation of TRU waste disposal at a generic basalt site is depicted in Figure 7.5-3. A conceptual model for a generic basalt site was originally described in "Population Risks from Disposal of High-Level Radioactive Wastes in Geologic Repositories" (EPA82). The original conceptual model was modified based on characteristics of the Hanford site, as described in Appendix A of "Risk Assessment of Disposal of High-Level Radioactive Wastes in Geologic Repositories" (EPA85).

The geologic and hydrogeologic parameters which define the model are given in Table 7.5-3. The table provides the base-case values for the parameters which are required as input to the NEFTRAN-S code. The second column in the table gives the parameter values used in the EPA evaluation of population risks (EPA82). The table then gives the parameter values obtained from the description of the Hanford site (EPA85). Finally, the table gives the parameter values used in the current evaluation.

The aquifer parameters were taken from the previous generic analyses performed in 1980 (EPA82). The parameter values which differ from those in the 1980 risk assessment include the conductivity, porosity, and vertical gradient in the host rock. These values differ significantly from the values used in the 1980 assessments and are based on site characterization data collected by DOE at the Hanford reservation.

Geochemical parameters are also necessary to evaluate the transport of radionuclides through geologic media. For each radionuclide in the waste inventory, retardation values are required. These values are dependent on the geologic medium in which the waste is disposed. The retardation values used in the analysis of the basalt site are given in Table 7.5-4.

Releases from the source were characterized in terms of radionuclide solubility in ground water. The solubilities used for the basalt assessments are shown in Table 7.5-5.

A single set of waste form and repository configuration parameters was assumed for all sites modeled. These parameters include the radionuclide inventory and the dimensions and capacity of the underground repository facility. These parameters are discussed for all sites in Sections 7.5.2 and 7.5.3.

Analyses were conducted to evaluate sensitivities and uncertainties in the parameter values. In the sensitivity studies, single parameters were varied discretely from the base case values. In the uncertainty analysis, statistical distributions were defined for the key input parameters and those parameters were varied in a Monte Carlo analysis. Three key parameters were identified for the sensitivity and uncertainty analyses. The parameters characterize the release from the waste form and the rate of transport through the ground-water system. The specific parameters selected for the analyses are the radionuclide solubilities, the vertical hydraulic conductivity in the host rock, and the radionuclide retardation factors. While other related parameters could have been included in the sensitivity and uncertainty analyses, those identified represent the key parameters for characterizing the magnitude of the radionuclide releases and the transport through the host rock and aquifer.

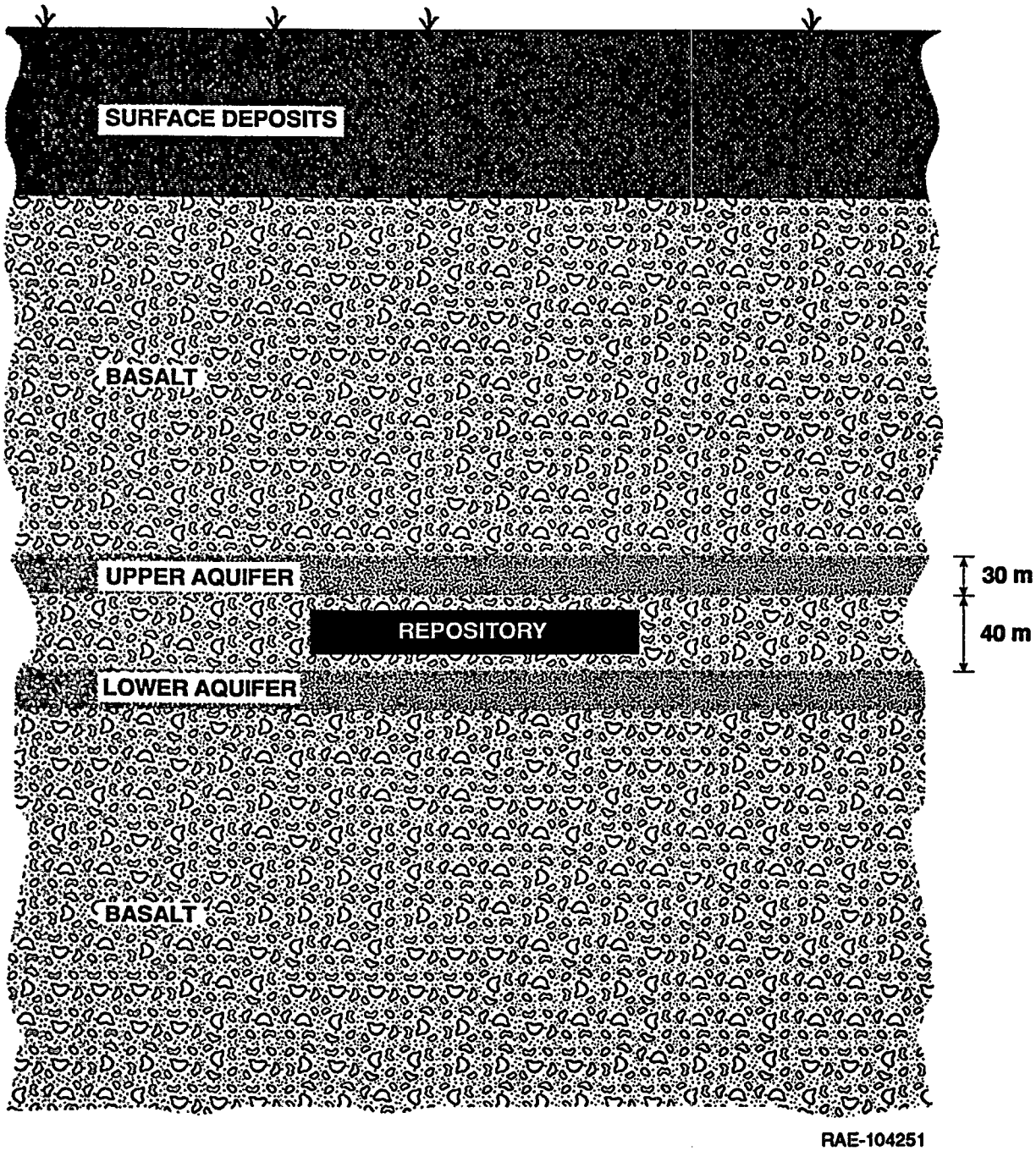

Figure 7.5-3. Cross-sectional structure for basalt repository (not to scale).

Table 7.5-3. Site parameters used in the individual dose and groundwater protection assessment of basalt.

Parameter	Early Generic EPA Model[a]	Hanford Site[b]	Current Model
Average porosity of backfill in repository	0.2	Not available	0.2
Distance from repository to overlying aquifer (meters)	100	20	20
Hydraulic conductivity of the host rock between the repository and the aquifer, after thermal effects (meters/yr)	0.00032	0.032	0.032
Porosity of the host rock between the repository and aquifer	0.0001	0.0001	0.0001
Hydraulic gradient between the repository and aquifer	0.025	0.014	0.014
Thickness of aquifer (meters)	30	30	30
Hydraulic conductivity of the aquifer (meters/yr)	31.5	315	31.5
Porosity of the aquifer	0.15	0.01	0.15
Horizontal gradient in aquifer	0.01	0.0003	0.01
Horizontal distance along the aquifer to the accessible environment (meters)	1600	2000	2000

[a]EPA-520/3-80-006 (EPA82)
[b]EPA-520/1-85-028 (EPA85)

Table 7.5-4. Radionuclide retardation factors for basalt.

Element	Range of Retardation Factors[a]		
	Low	"Base Case"	High
Strontium	50	200	2,000
Cesium	100	1,000	10,000
Lead	20	50	500
Radium	50	500	5,000
Actinium[b]	20	50	1,000
Thorium	500	5,000	10,000
Protactinium[b]	20	50	1,000
Uranium	20	50	1,000
Neptunium	10	100	500
Plutonium	100	500	5,000
Americium	60	500	50,000
Curium	100	500	10,000

[a]From 1983 WISP report (NAS83).
[b]Because values were not given in WISP report, values of uranium were used based on chemical similarities.

Table 7.5-5. Radionuclide solubilities for basalt.[a]

Nuclide	Solubility (Ci/m^3)
Ac-227	1.64E+01
Am-241	8.28E-01
Cm-248	1.06E-03
Cs-137	1.19E+01
Np-237	1.67E-04
Pa-231	1.09E-02
Pb-210	1.60E+01
Pu-238	4.08E+00
Pu-239	1.49E-02
Pu-240	5.47E-02
Pu-242	9.51E-04
Ra-226	2.24E-01
Sr-90	1.23E+01
Th-229	4.87E-02
Th-230	4.65E-03
Th-232	2.55E-08
U-233	2.26E-03
U-234	1.46E-03
U-235	5.08E-07
U-236	1.53E-05
U-238	8.01E-08

[a]Based on 1.0E-06 mole/liter (La89).

Ranges of retardation factors are given in NAS83. In the risk assessments, these ranges have been extended to include minimum retardation factors of one. A retardation factor of one represents a limiting case in which the site geochemistry is such that the repository host formation and the aquifer provide no radionuclide retardation. These conditions provide a bounding analysis of disposal system performance and indicate the importance of retardation in EPA's modeling of generic repository performance.

The parameter-value ranges are shown in Table 7.5-6. The ranges encompass the values used in previous Agency assessments. The probability distributions are given for use in the NEFTRAN-S uncertainty analysis. Due to the wide range of values, log uniform distributions were used for all of the parameters. This is preferable to using uniform distributions because the use of log uniform distributions causes the median values of the parameters to be closer to their base case values and is therefore more appropriate for parameters that vary over several orders of magnitude.

7.5.5.1.3 Base Case Results from the Assessment of the Generic Basalt Site

Figure 7.5-4 shows the results of the deterministic assessment of individual dose versus time using the NEFTRAN-S computer code. The analysis assumes undisturbed ground-water flow vertically through the repository horizon to the upper aquifer and then laterally through the aquifer. The assessment assumes an individual drinking water consumption of 2 liters per day at a point 2000 meters down gradient. Sensitivity of individual dose to solubility, retardation, and hydraulic conductivity are discussed in Section 7.5.5.1.4.

No radionuclides reach the 2000-meter boundary prior to approximately year 50,000. Thus, individual dose prior to year 50,000 is zero. At approximately year 50,000, the most mobile radionuclides, with retardation factors of 50, reach the 2000-meter boundary. Also, some of the radioactive decay products arrive at this time. Dose increases abruptly to approximately 1080 mrem/yr. The rapid increase is due to the relatively low dispersivity used in the model. A higher dispersivity would have led to a more gradual increase in the dose. Major contributing radionuclides include U-233 (650 mrem/yr), protactinium (Pa)-231 (150 mrem/yr), actinium (Ac)-227 (130 mrem/yr) and U-234 (100 mrem/yr). Dose remains fairly constant until year 93,000, when the arrival of Np-237 abruptly increases the dose to 17,000 mrem/yr.

Ground-water contamination was evaluated through three measures. First is the concentration of Ra-226. Second is the total concentration of all alpha-emitting radionuclides, excluding radon. Third is the drinking-water dose resulting from all beta- and gamma-emitting radionuclides. Each of these measures was evaluated through the NEFTRAN-S analysis.

Ra-226 is part of the Pu-238 decay series. Figure 7.5-5 shows the concentration of Ra-226, as a function of time, calculated 2000 meters down gradient. Radium first appears at approximately 50,000 years. Its concentration increases sharply to approximately 0.3 pCi/liter and remains fairly constant for the remainder of the 100,000-year simulation period.

Table 7.5-6. Parameter ranges and distributions for basalt.

Parameter	Minimum	Maximum	Distribution Type
Solubility (mole/liter)	1.0E-09	1.0E-03	Log Uniform
Vertical hydraulic conductivity (m/yr)	3.2E-04	3.2E-01	Log Uniform
Retardation factors	(a)	(b)	Log Uniform

[a]The sensitivity and uncertainty analyses used retardation factors of one, as well as the "low" values from Table 7.5-4.
[b]See Table 7.5-4.

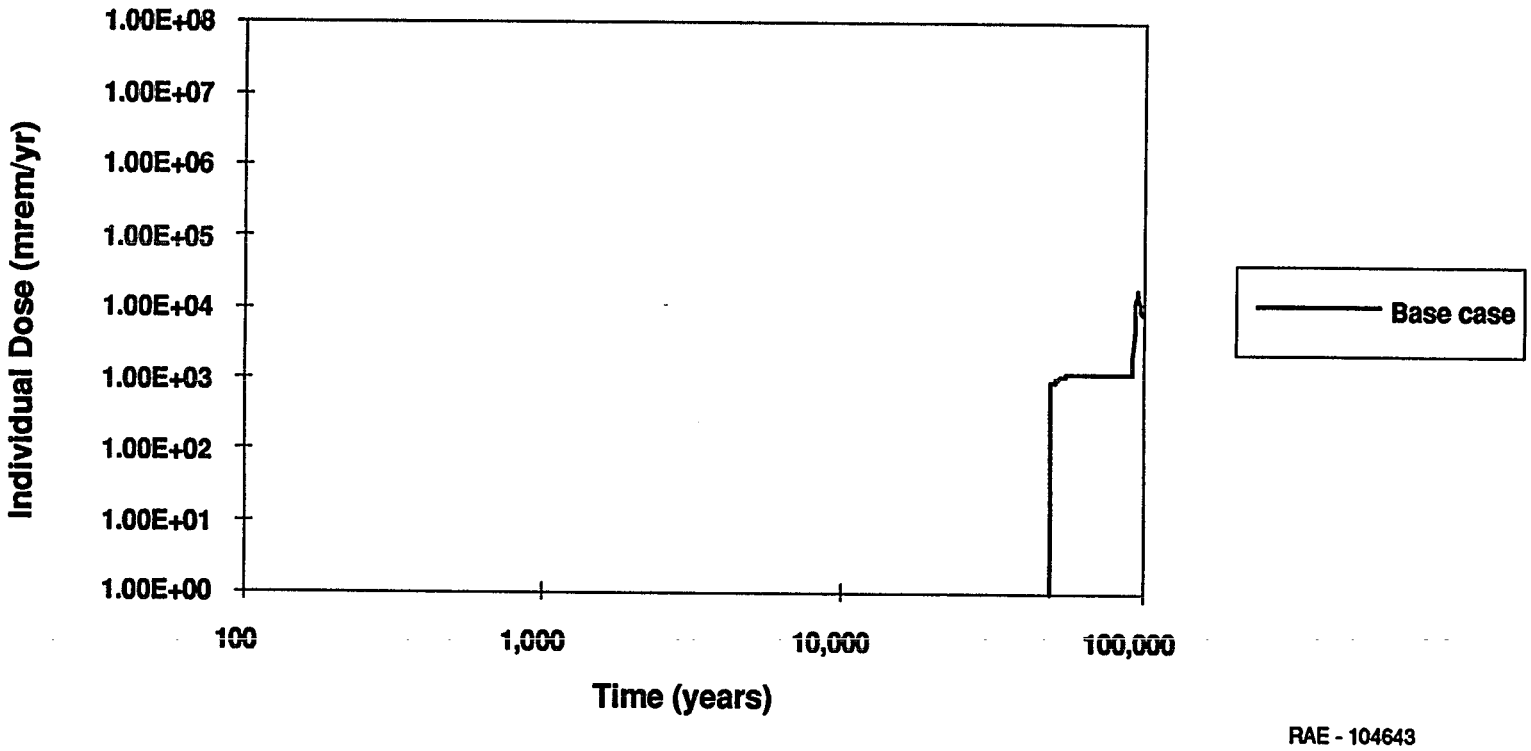

Figure 7.5-4. Individual dose for basalt.

Figure 7.5-6 shows the total concentration of alpha-emitting radionuclides, as a function of time, calculated 2000 meters down gradient. Concentration is zero until 50,000 years, when it rises sharply to almost 1100 pCi/liter. Major contributors to the total concentration are U-233 (866 pCi/liter), U-234 (132 pCi/liter), Pa-231 (54 pCi/liter) and U-236 (21 pCi/liter). Concentration remains fairly constant until year 93,000, when the arrival of Np-237 begins to increase the total concentration to 6600 pCi/liter at year 96,000.

The total concentration of beta-emitting radionuclides is measured in terms of the dose which would result from the consumption of two liters per day of the contaminated ground water. There are four beta-emitting radionuclides: Sr-90, Cs-137, Ac-227 (generated through the decay of Pu-239 and U-235), and lead (Pb)-210 (generated through the decay of Pu-238). As shown in Figure 7.5-7, the total dose is zero until 50,000 years. The dose then increases and remains at 100 mrem/yr to 200 mrem/yr. The dose results mainly from the concentration of Ac-227, although Pb-210 contributes somewhat. Sr-90 and Cs-137 do not contribute to the dose because of their short half-lives. The dose from beta-emitting radionuclides is less than 20 percent of the dose from all radionuclides shown in Figure 7.5-4.

7.5.5.1.4 Sensitivity and Uncertainty Analyses for the Generic Basalt Assessments

The previous section discussed the results of evaluating individual doses and concentrations in ground water using the base-case parameter values given in Table 7.5-3. This section discusses the sensitivity of individual dose and concentrations in ground water to variations in radionuclide solubility, hydraulic conductivity in the vertical transport leg, and radionuclide retardation factors.

Individual Dose - Radionuclide solubility controls the rate at which radionuclides enter into the ground water flow. Higher solubilities result in higher concentrations of radionuclides per unit of water. Figure 7.5-8 shows the sensitivity of individual dose to variations in solubilities. The base-case solubility was 10^{-6} mole/liter. Individual doses were calculated with higher (10^{-3} mole/liter) and lower (10^{-9} mole/liter) solubilities. Varying the solubility does not effect the time of arrival of the first measured dose. It does, however, significantly affect the magnitude of the dose. Increased solubility results in a much greater and sharper initial dose. The magnitude of this peak dose is approximately 2×10^{5} mrem/yr. At high solubility the dose falls off more rapidly with time due to depletion of the inventory.

Figure 7.5-9 shows the effect of varying the hydraulic conductivity of the basalt in the vertical transport leg. Increasing the vertical hydraulic conductivity increases the volume of flow through a given cross-sectional area and decreases the travel time. Since the vertical distance from the repository to the aquifer is only 20 meters, the decrease in the travel time is negligible. The increased flow, however, results in a greater release of radioactivity from the repository, and thus an increased dose. Decreasing the hydraulic conductivity several orders of magnitude has a more significant effect on the time of the initial dose, as well as on the magnitude of the dose.

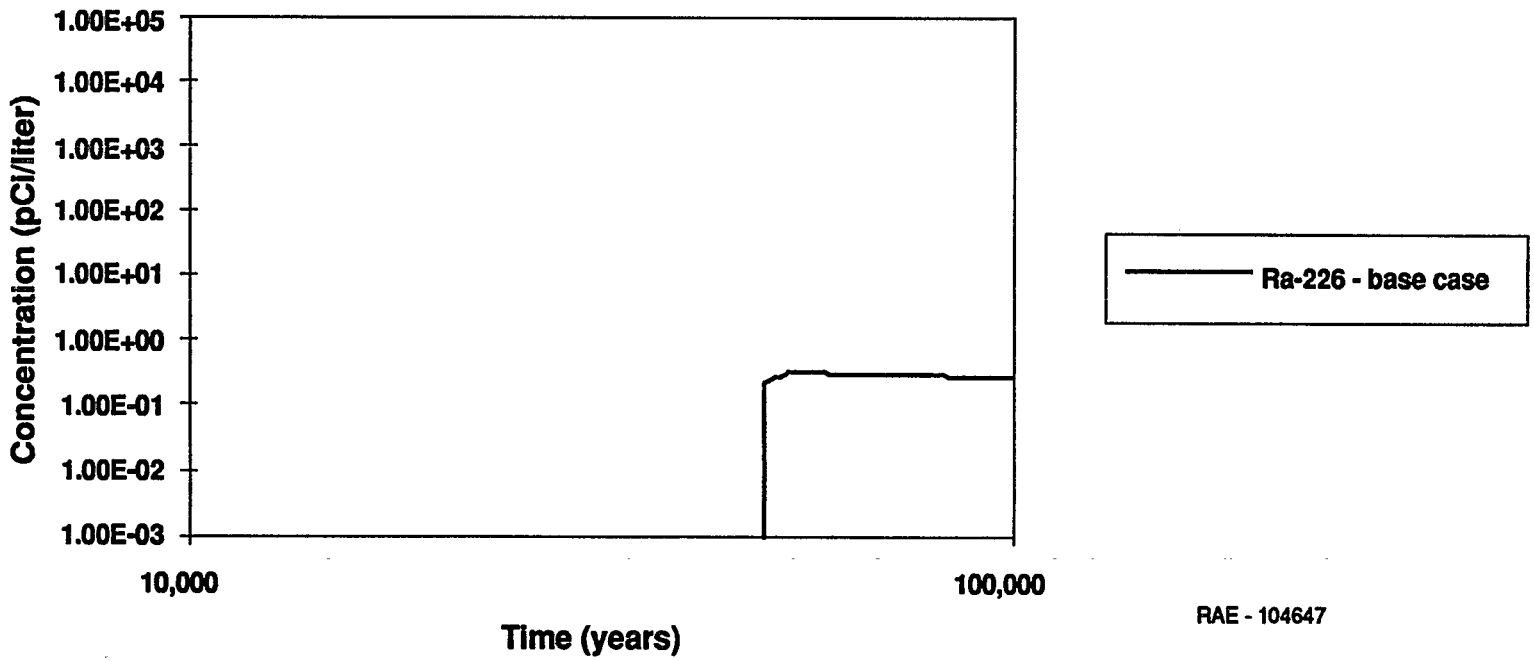

Figure 7.5-5. Ra-226 groundwater concentrations - base case basalt.

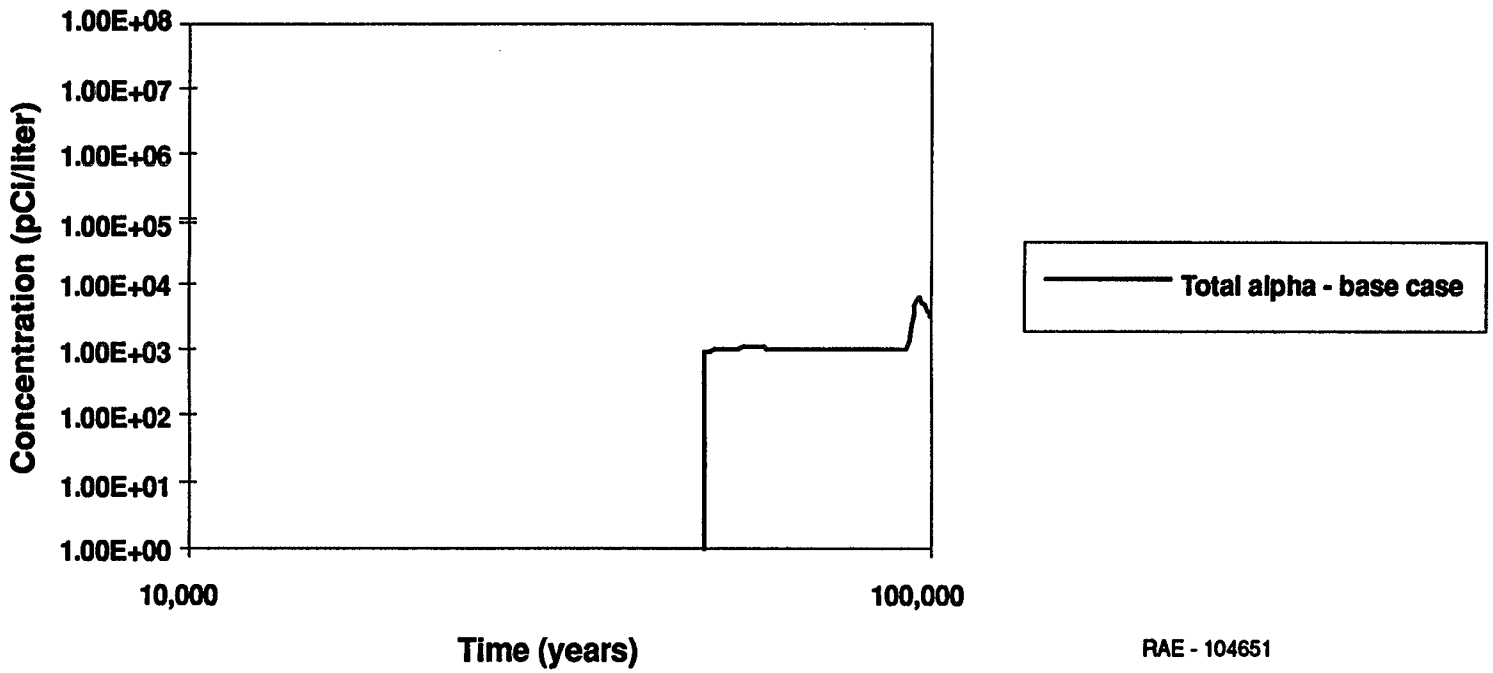

Figure 7.5-6. Total alpha groundwater concentrations - base case basalt.

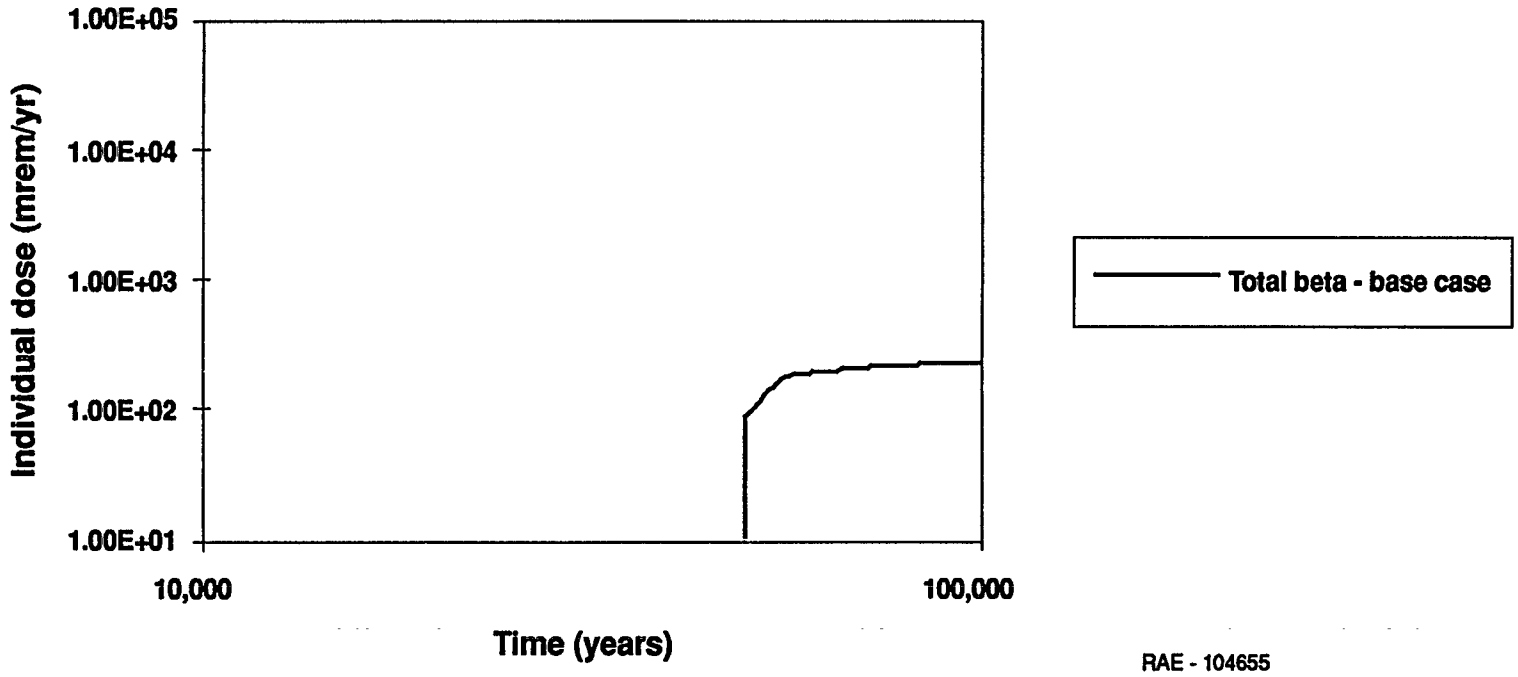

Figure 7.5-7. Total beta groundwater dose - base case basalt.

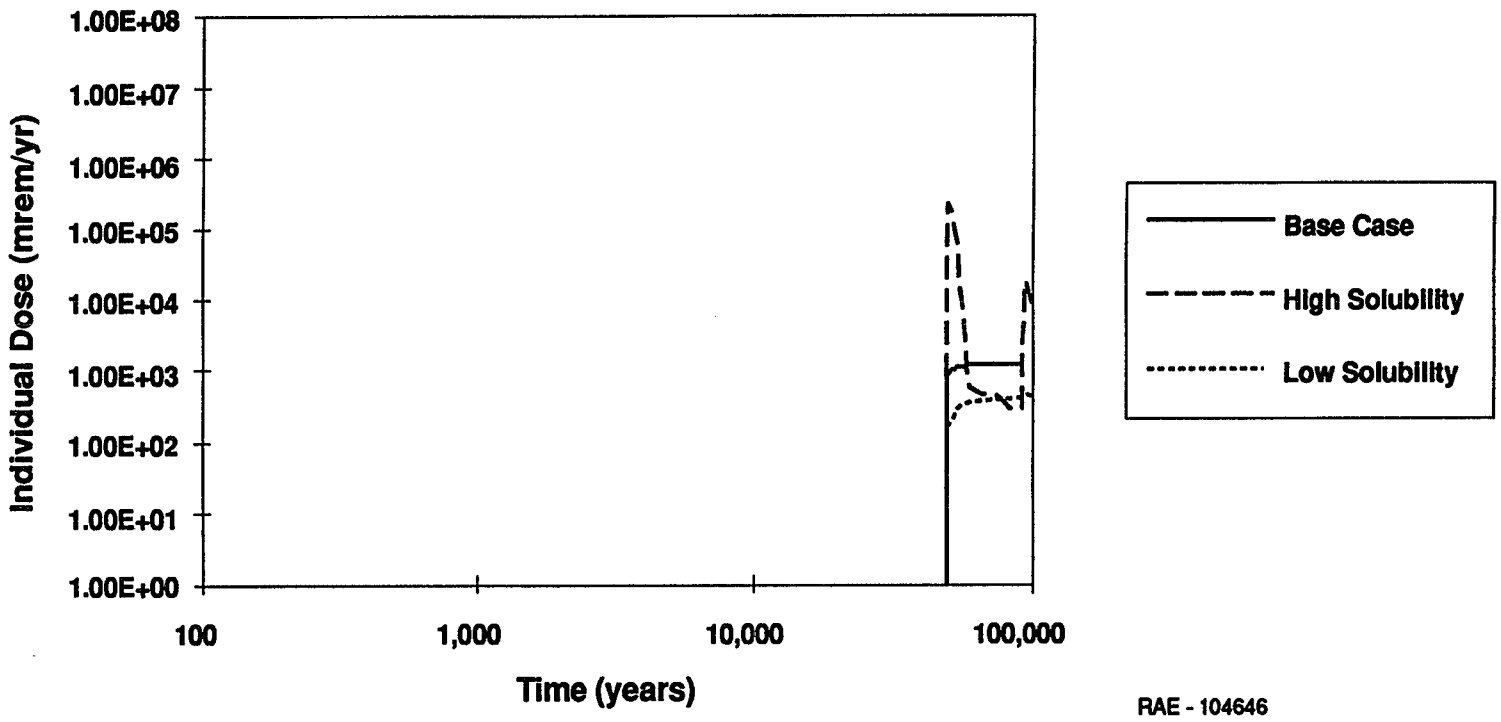

Figure 7.5-8. Sensitivity of dose to solubility - basalt.

Variations in radionuclide retardation have the greatest effect on individual dose (Figure 7.5-10). Decreasing the retardation factors increases the mobility of the radionuclides and thus decreases the travel time to the 2000-meter boundary. Thus, with lower retardation, the Np-237 peak arrives much earlier, i.e., at 11,000 years. All the Np-237 has passed by year 40,000. The retardations, however, are not low enough to see the arrival of plutonium, americium, or curium in 100,000 years. With no retardation, however, all radionuclides are transported at the same velocity. Thus, doses arrive much earlier, at 1,000 years, and doses are much greater due to the contributions from plutonium, americium, and curium.

In addition to the deterministic sensitivity studies of individual dose, a probabilistic uncertainty study was conducted. For the three parameters of interest - solubility, vertical hydraulic conductivity, and retardation - parameter-value ranges were assigned instead of single values. The ranges used for the three parameters are given in Table 7.5-6. Two analyses were made: one using the low retardation values (Table 7.5-4) as a minimum and a second assuming no retardation by using a retardation value of one. Using the Monte Carlo sampling routine of the NEFTRAN-S code, peak doses were calculated for the 10,000-year period following disposal. No dose was reported in the low-retardation analysis. In actuality, this means there is a very low probability of dose in 10,000 years, given the parameter-value uncertainty. The results of the analysis using a minimum retardation value of one are shown by the histogram in Figure 7.5-11. Considering the parameter-value uncertainty as represented by the input parameter-value ranges, there is a 0.12 probability of zero dose in 10,000 years. However, these results, which include no retardation, represent bounding conditions for repository performance. They are included to represent the importance of retardation in EPA's model of generic repository performance.

Ra-226 Concentrations - The sensitivities of Ra-226 concentrations in ground water 2000-meters down gradient are similar to those described for individual doses. Figure 7.5-12 shows that decreased solubility results in decreased concentrations, while increased solubility results in increased concentrations. At high solubility, the Ra-226 concentration falls off rapidly due to depletion of the source. As shown in Figure 7.5-13, increasing the hydraulic conductivity in the vertical leg has little effect on the initial arrival time but a significant effect on the magnitude of the concentration of Ra-226 due to increased flow. Decreasing the hydraulic conductivity has a more significant effect on the arrival time. Finally, varying the retardation of all radionuclides has a significant effect on both the arrival time and the magnitude of Ra-226 concentrations, as shown in Figure 7.5-14.

Total Alpha Concentrations - The sensitivities of concentrations of alpha-emitting radionuclides in ground water 2000 meters down gradient are also similar to those described for individual doses. Figure 7.5-15 shows that decreased solubility results in decreased concentrations, while increased solubility results in increased concentrations. Figure 7.5-16 shows that increasing the hydraulic conductivity in the vertical leg has little effect on the initial arrival time but a significant effect on the magnitude of the concentration, due to increased flow through the repository. Decreasing the hydraulic conductivity has a more significant effect on the arrival time. Finally, varying the retardation of all radionuclides has a significant effect on both the arrival time and the magnitude of the concentration of alpha-emitting radionuclides (shown in Figure 7.5-14).

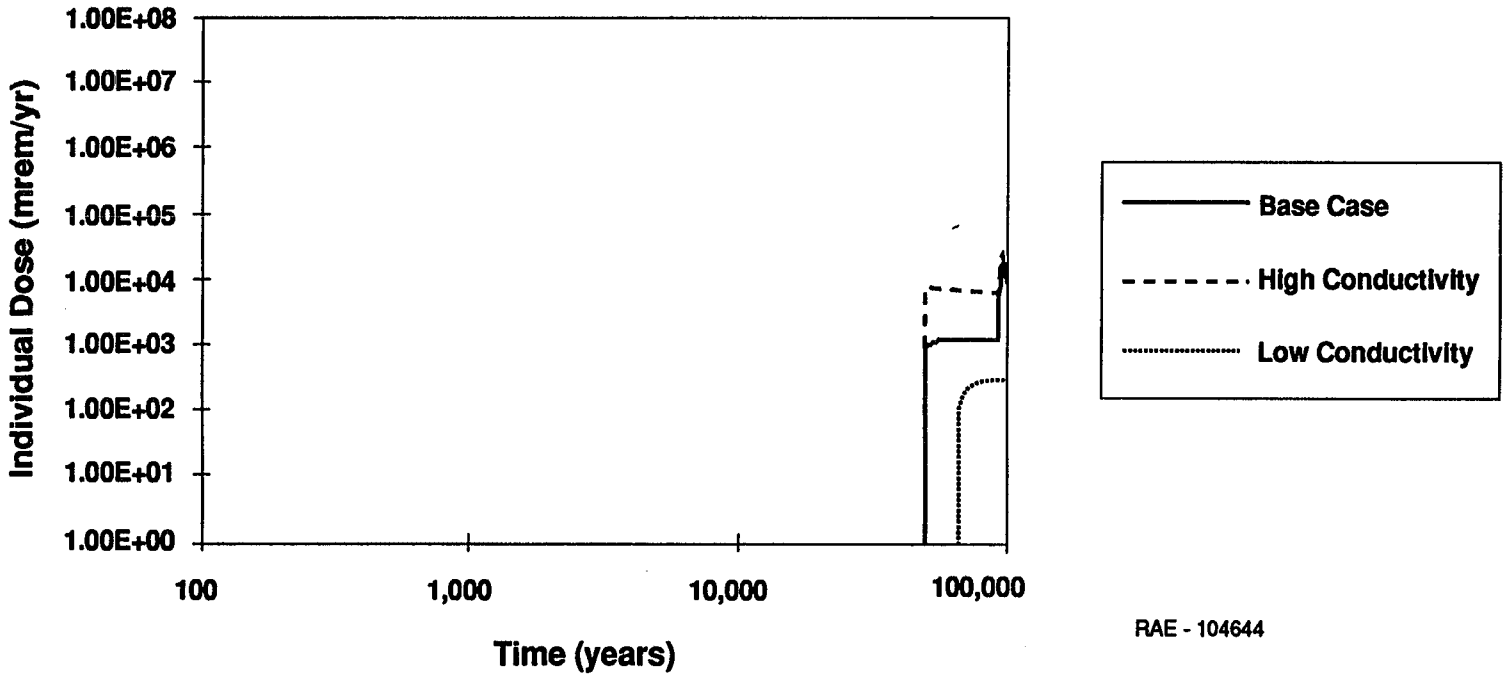

Figure 7.5-9. Sensitivity of dose to vertical hydraulic conductivity - basalt.

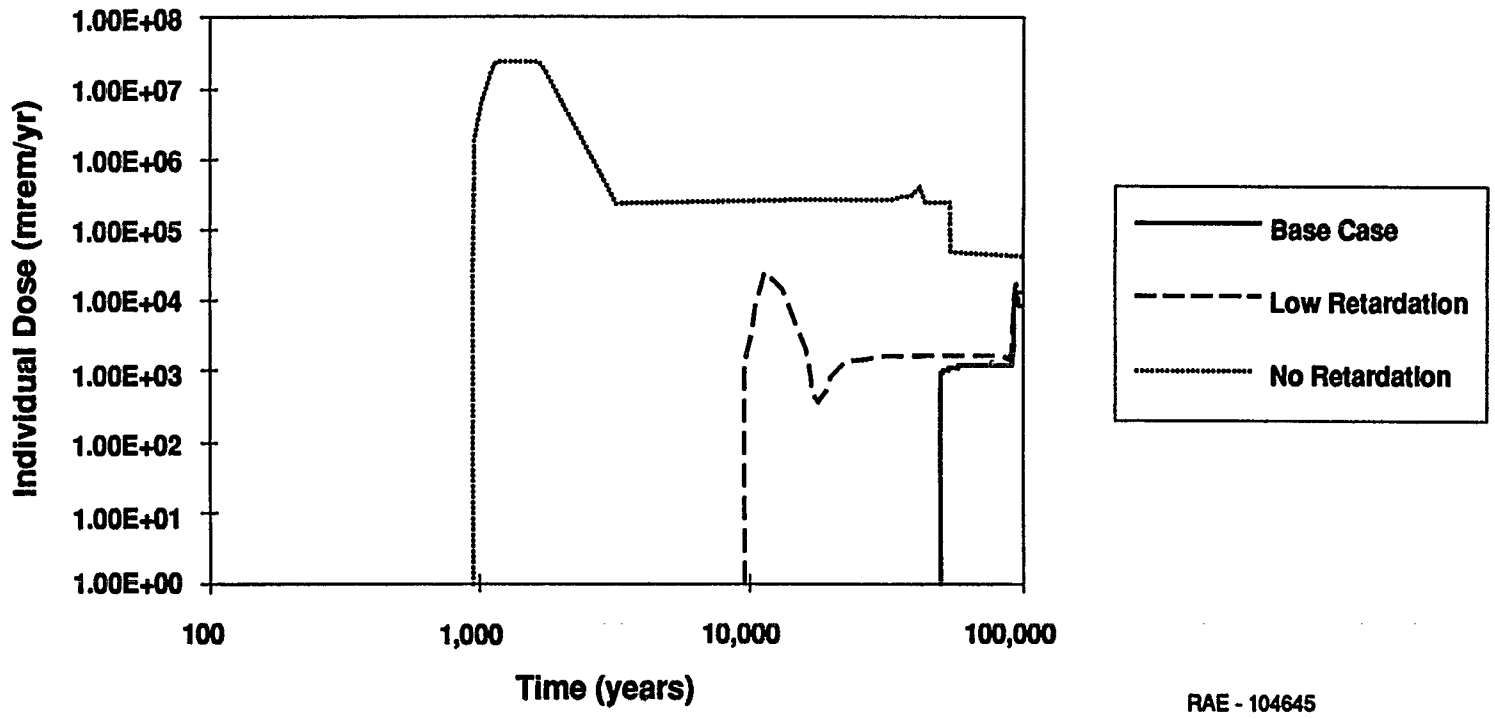

Figure 7.5-10. Sensitivity of dose to retardation - basalt.

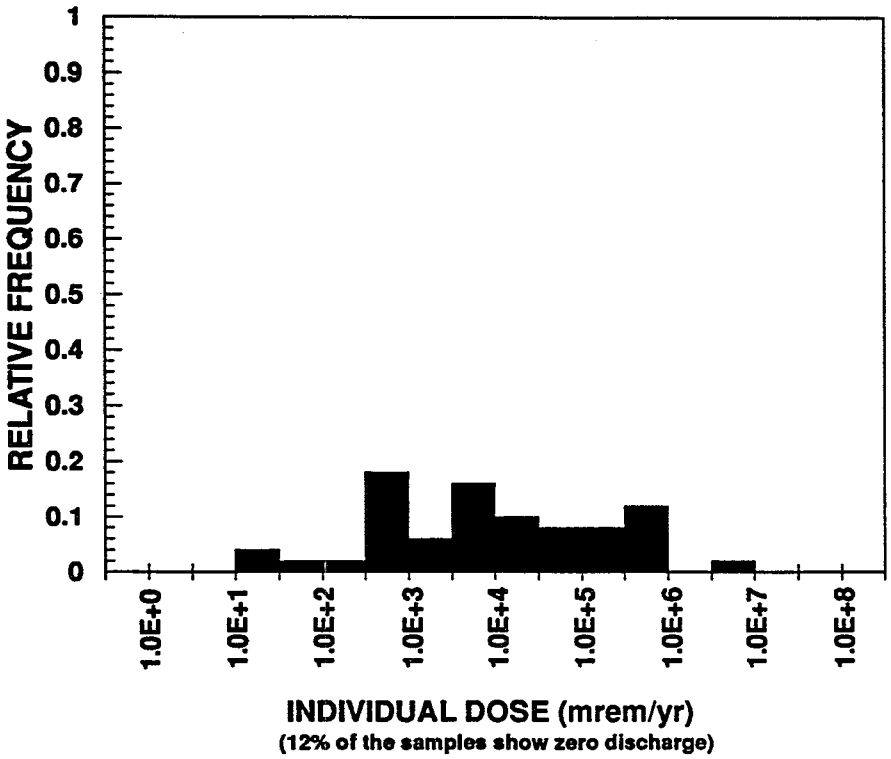

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Figure 7.5-11. Distribution of individual dose due to parameter value uncertainty (minimum retardation value of one) - basalt.

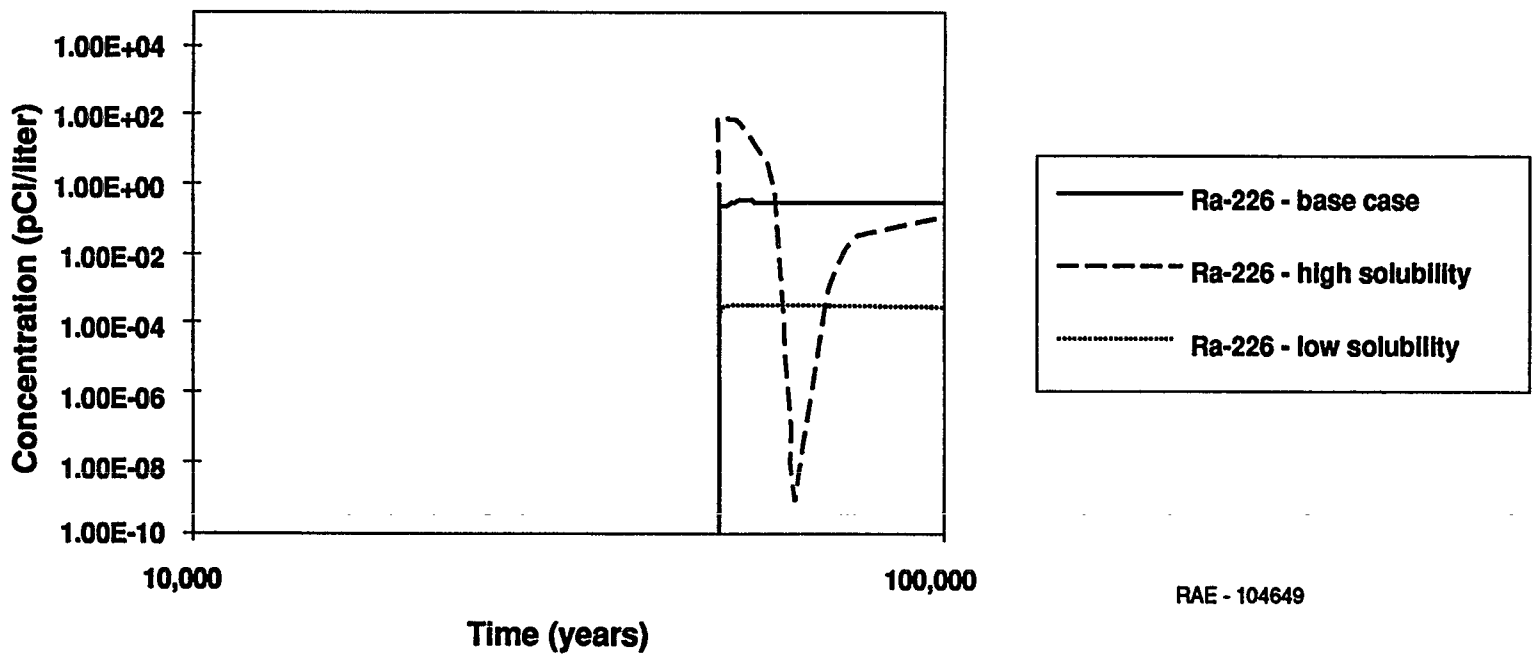

Figure 7.5-12. Sensitivity of Ra-226 groundwater concentrations to solubility - basalt.

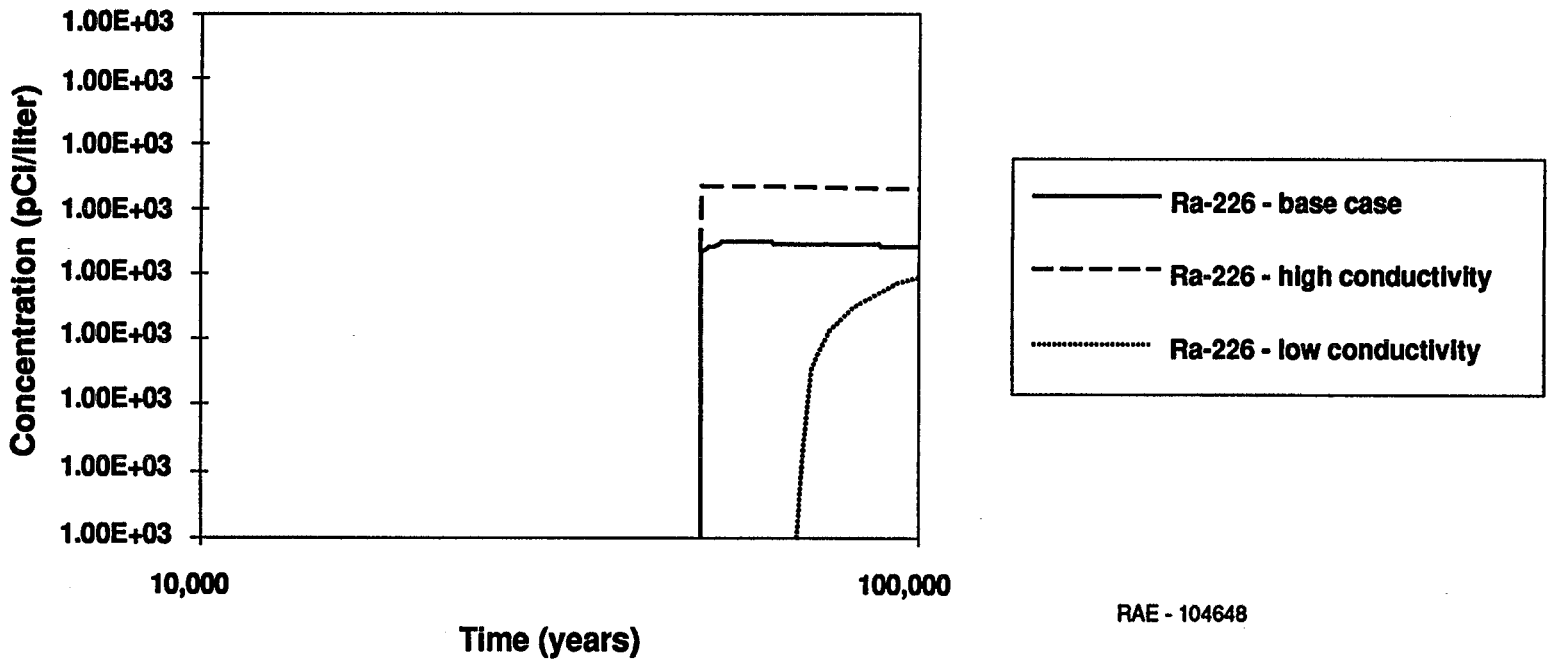

Figure 7.5-13. Sensitivity of Ra-226 groundwater concentrations to vertical hydraulic conductivity - basalt.

Total Beta and Gamma Concentrations - The sensitivities of dose from beta- and gamma-emitting radionuclides 2000 meters down gradient are shown in Figures 7.5-18 through 7.5-20. The only radionuclides in this category are Ac-227 and Pb-210, both of which are decay products from other radionuclides in the inventory. Although Sr-90 and Cs-137 are beta-emitters, they do not contribute to the concentrations because of their short half-lives. Figure 7.5-18 shows that decreased solubility results in decreased concentrations and thus dose, while increased solubility results in increased concentrations and dose. As shown in Figure 7.5-19, increasing the hydraulic conductivity in the vertical leg has little effect on the initial arrival time but a significant effect on the magnitude of the dose, due to increased flow. Decreasing the hydraulic conductivity has a more significant effect on the arrival time. Finally, varying the retardation of all radionuclides has a significant effect on both the arrival time and the magnitude of the dose from beta-emitting radionuclides (Figure 7.5-20).

7.5.5.2 Site Analysis - Granite

7.5.5.2.1 Introduction

Granitic rocks are widely distributed throughout the United States and thus offer the possibility of being found in connection with other desirable characteristics for a repository site. At depth they can be extremely "tight", the naturally occurring fractures being kept almost completely closed by the high lithostatic pressure. Mined openings in granitic rock are expected to be highly stable for well-chosen sites and there is considerable experience in such underground excavations from various kinds of hard rock mines and tunnels. The likelihood of associated valuable resources is low; when present they are often limited to veins at the boundaries of the granitic bodies. Water wells are occasionally drilled into granitic rock but because of the general trend of decreasing permeability with depth, such wells rarely exceed several hundred feet. An important distinction between granitic rocks and most of the other host rocks being considered for a repository is that they are often found as a bedrock formation or as an intrusive plutonic body, and thus the possibility of extensive aquifers at a depth below the repository is much less likely. This decreases the possibility of a productive and high pressure source of water that could cause upward flow and carry radionuclides towards the surface. On the other hand, the certain presence of fractures and the water-saturated condition expected at depth virtually guarantee that there would be some ground-water movement through a repository in granite. It may occur at extremely low volumetric flow rates and velocities, but it would be present and must be taken into account in estimating the performance of a repository.

The DOE had previously carried out a screening of the entire United States and had identified the north-central and northeastern regions of the United States as the most likely to contain suitable repository environments in granitic rocks. Based on data collected to date by the DOE and others, it is possible to define certain idealized conceptual models of repository sites in each of the two regions so as to make first approximations of the potential performance of such repositories and to identify parameters that are most critical to long-term performance. This section contains a summary of models and parameters used by the Agency as "generic" sites that are based on simplified models of the general geologic and hydrologic conditions reported at

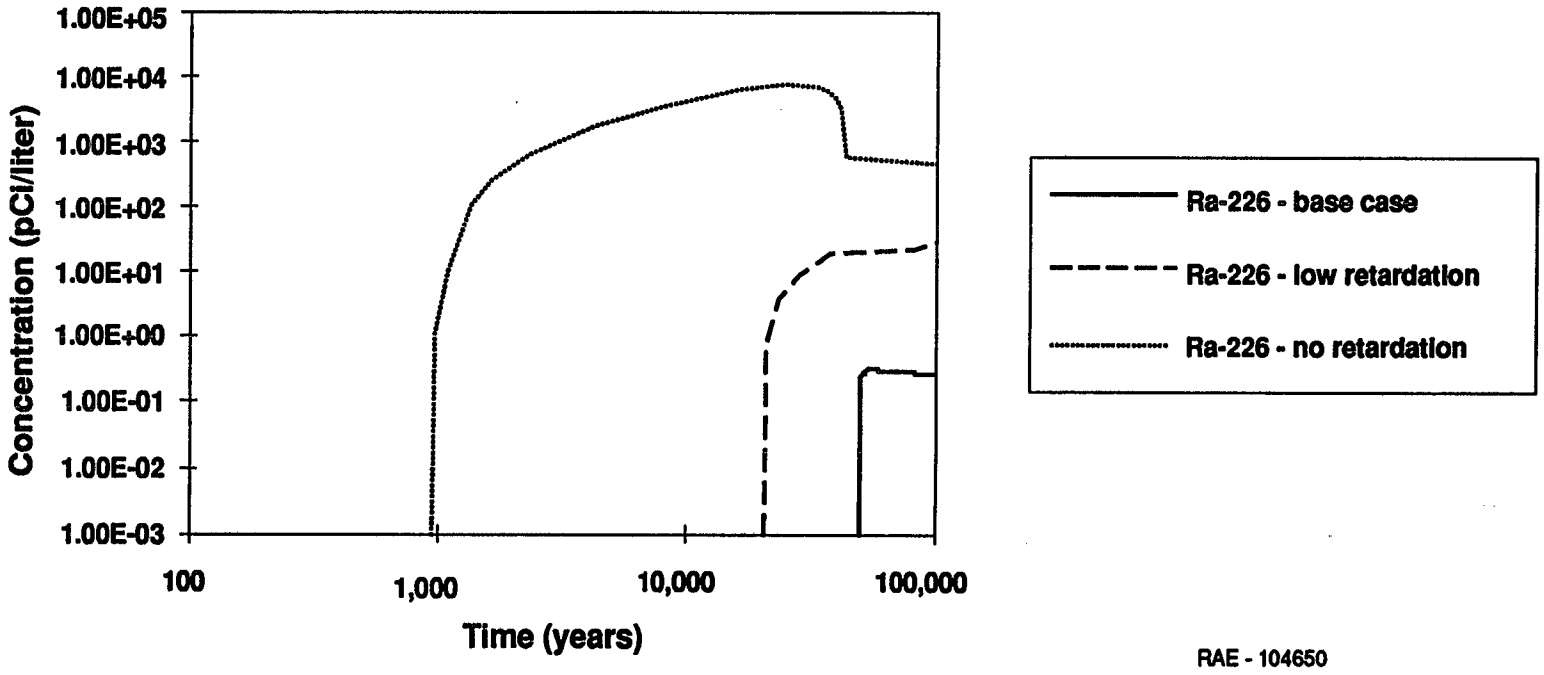

Figure 7.5-14. Sensitivity of Ra-226 groundwater concentrations to retardation - basalt.

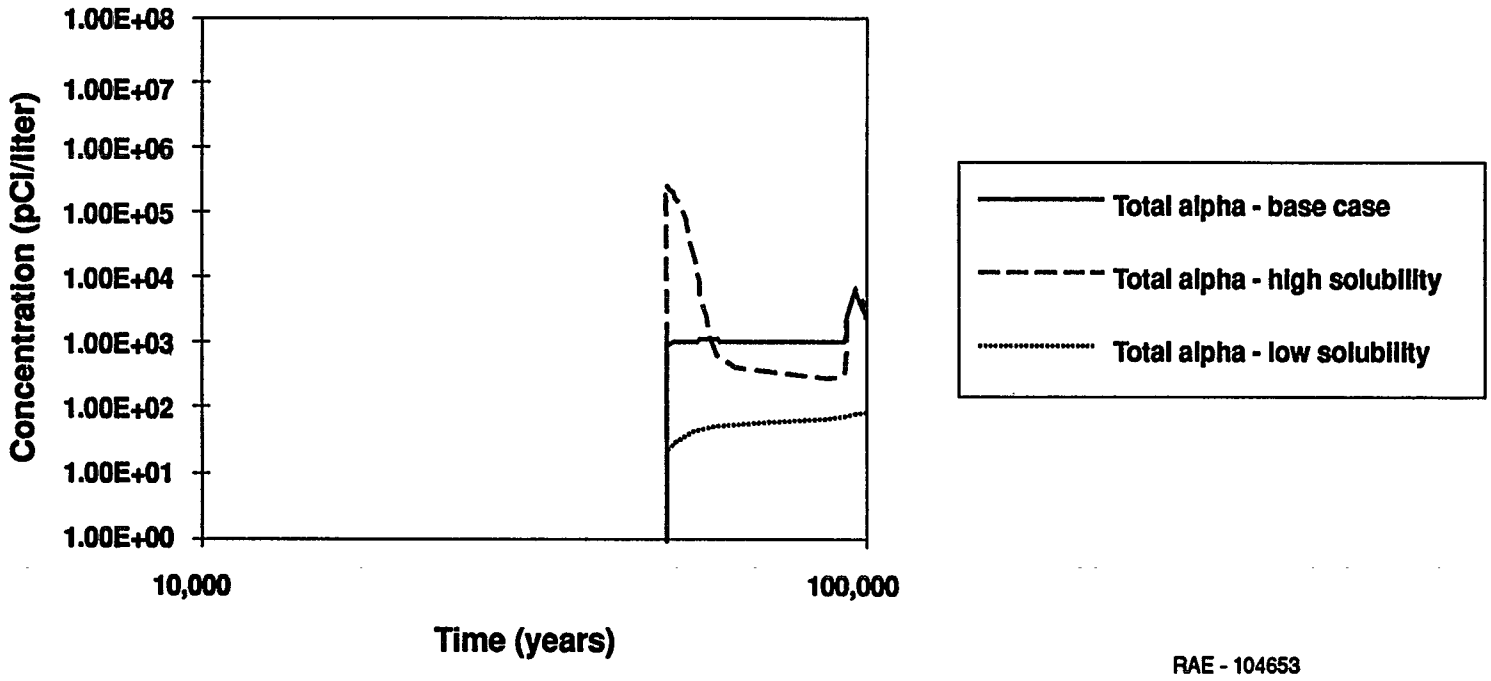

Figure 7.5-15. Sensitivity of total alpha groundwater concentrations to solubility - basalt.

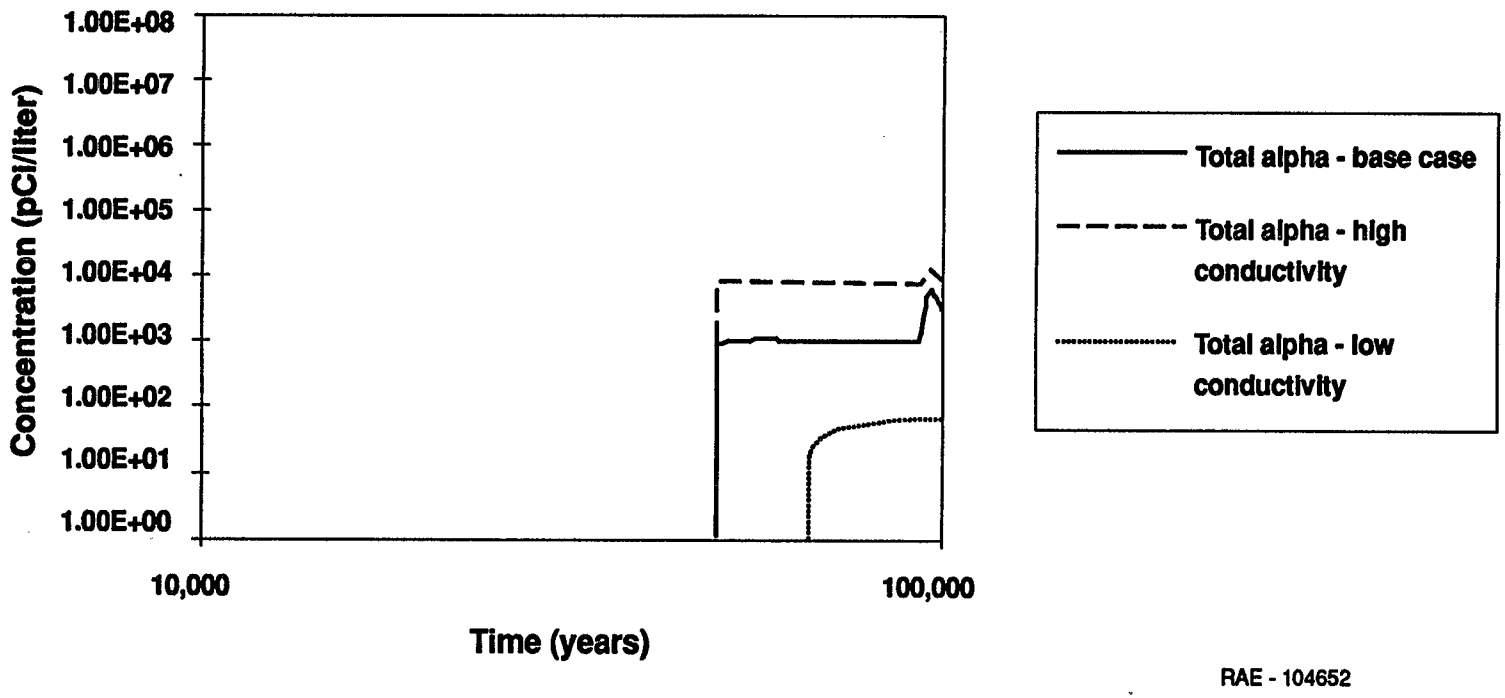

Figure 7.5-16. Sensitivity of total alpha groundwater concentrations to vertical hydraulic conductivity - basalt.

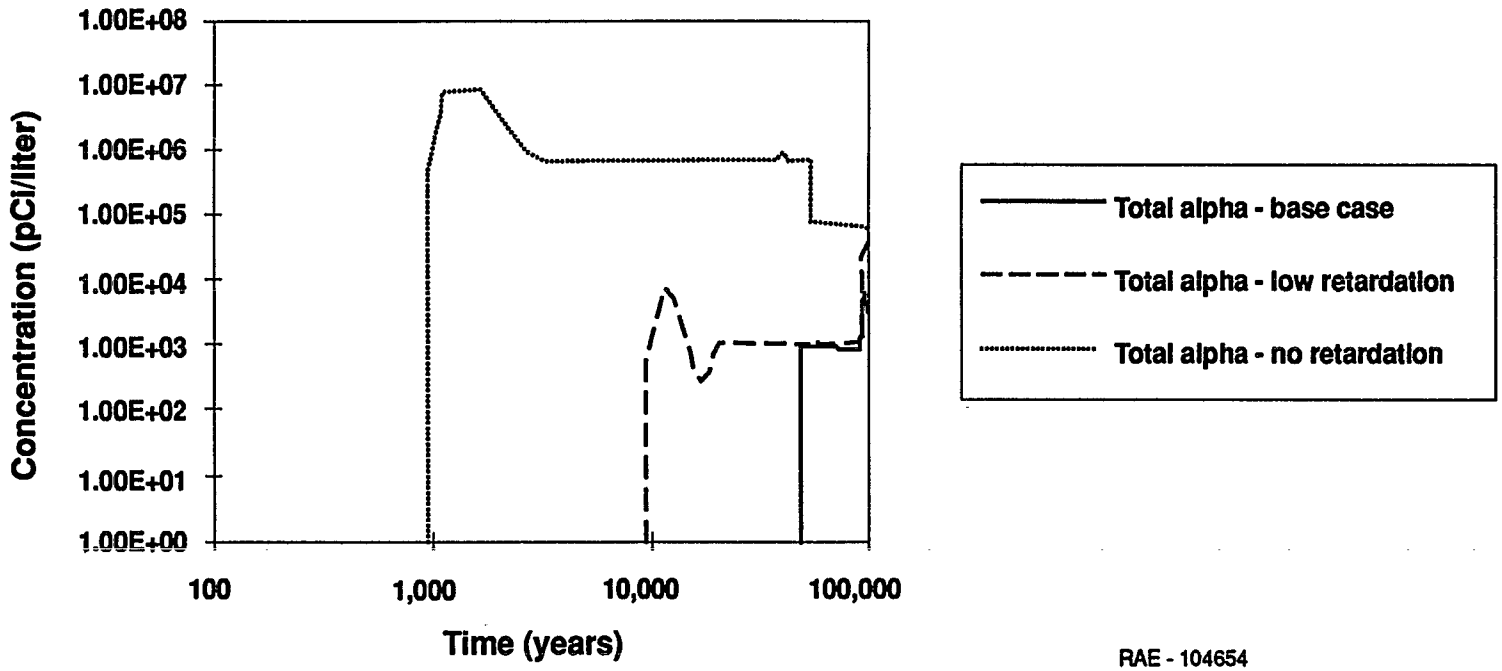

Figure 7.5-17. Sensitivity of total alpha groundwater concentration to retardation - basalt.

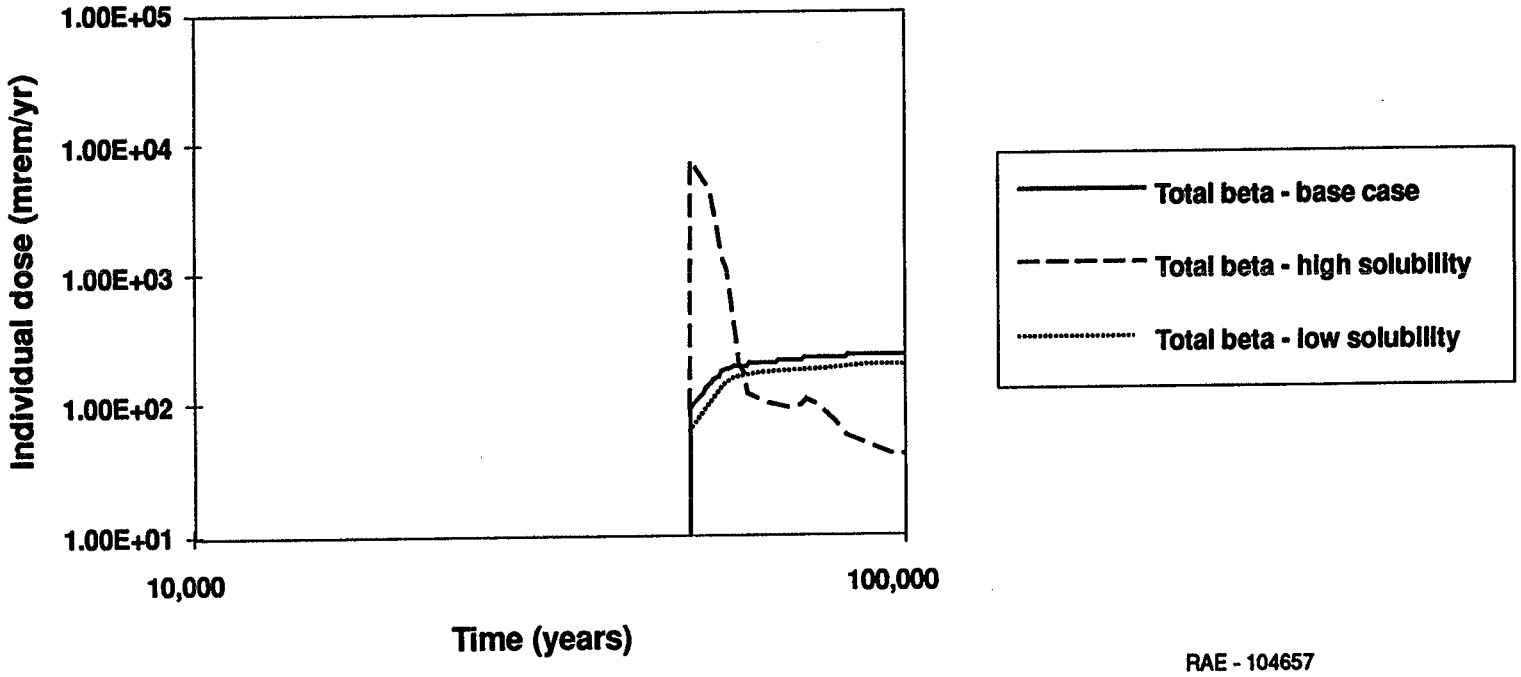

Figure 7.5-18. Sensitivity of total beta groundwater dose to solubility - basalt.

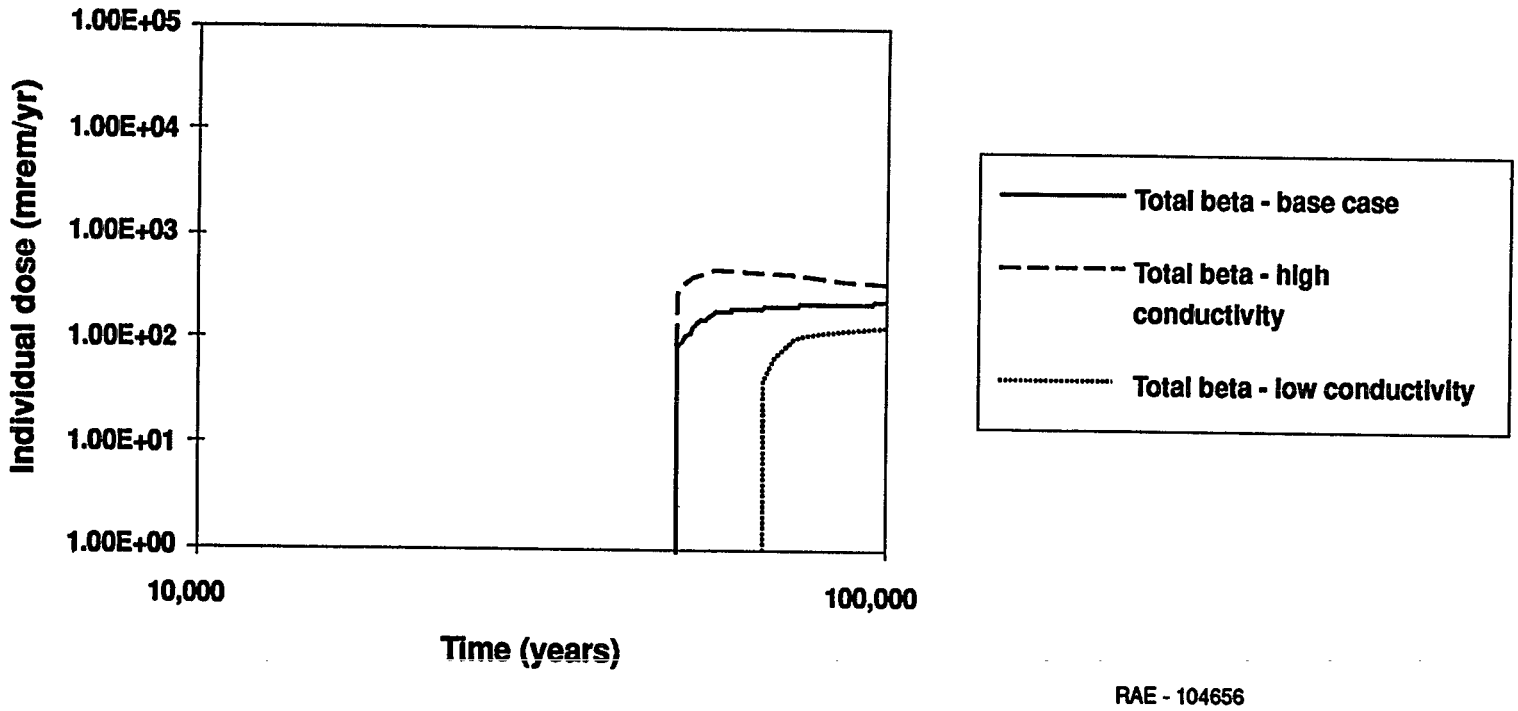

Figure 7.5-19. Sensitivity of total beta groundwater dose to vertical hydraulic conductivity - basalt.

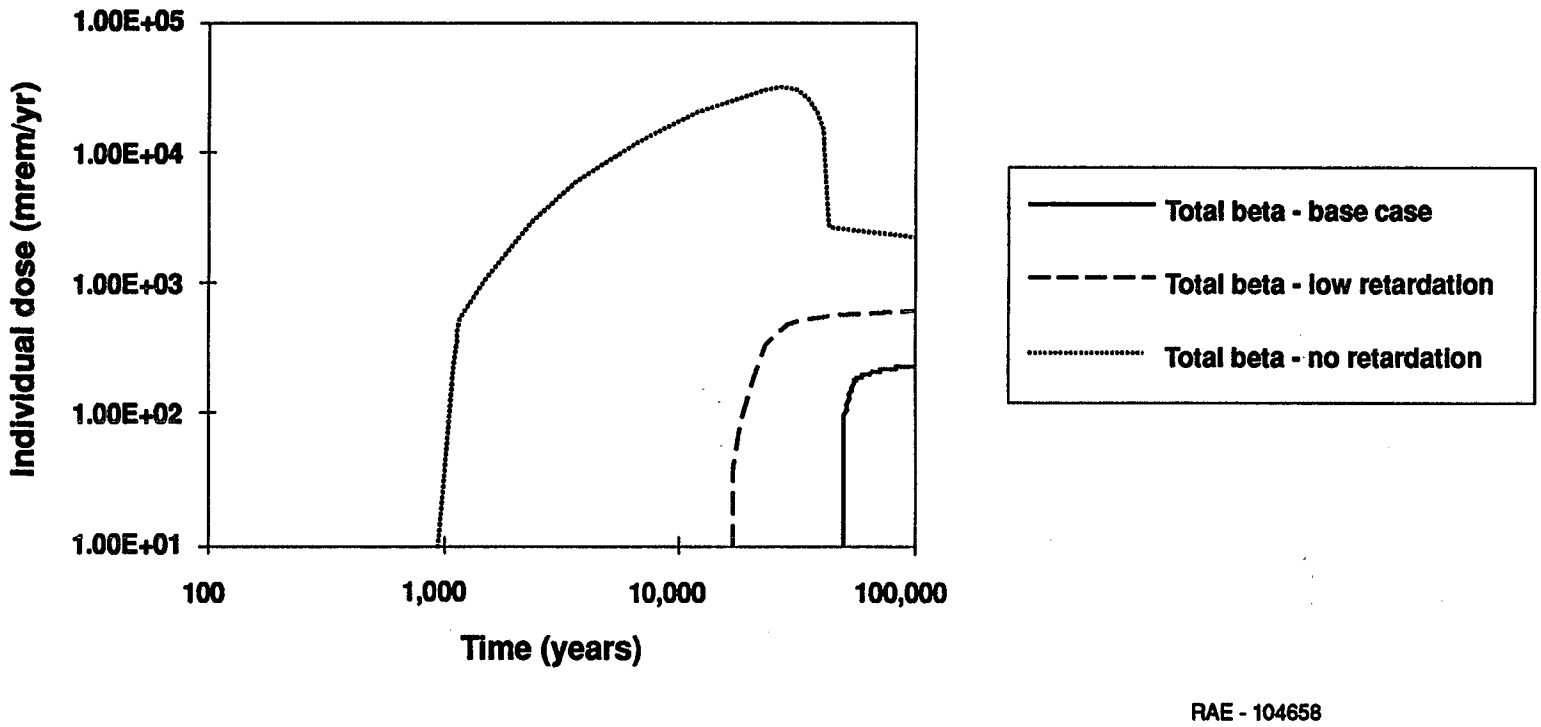

Figure 7.5-20. Sensitivity of total beta groundwater dose to retardation - basalt.

promising locations in each of these two regions. It is important to emphasize that these models are generic in nature, as are all the models of all media analyzed, and are not intended to represent performance at a particular site.

Section 7.5.5.2.2 discusses the important input parameters that have been used in the Agency's risk analysis for granite. These parameters are based on data from the generic north-central site and the generic northeastern site. Section 7.5.5.2.3 provides the results of the base case analyses. Section 7.5.5.2.4 provides the results of the sensitivity analyses and uncertainty analyses.

7.5.5.2.2 Input Parameters for Granite

The conceptual model developed to support the evaluation of TRU waste disposal at a generic granite site is depicted in Figure 7.5-21. A conceptual model for a generic granite site was originally described in "Population Risks from Disposal of High-Level Radioactive Wastes in Geologic Repositories" (EPA82). The original conceptual model was modified based on characteristics of the generic granite sites in the northeastern and north-central United States, as described in Appendix A of "Risk Assessment of Disposal of High-Level Radioactive Wastes in Geologic Repositories" (EPA85).

The geologic and hydrogeologic parameters which define the model are given in Table 7.5-7. The table provides the base-case values for the parameters which are required as input to the NEFTRAN-S code. The second column in the table gives the parameter values used in the EPA evaluation of population risks (EPA82). The table then gives the parameter values obtained from the descriptions of the generic granite sites in the north-central and northeastern United States (EPA85). Finally, the table gives the parameter values used in the current evaluation.

Two sources were used to compile the parameter information shown in the table. The aquifer parameters were taken from the previous generic analyses performed by the Agency in 1980 (EPA82). The aquifer parameters are similar for all sites evaluated, thus emphasizing the performance capabilities of the host rock. The host rock parameters, including the distance between the repository horizon and the aquifer, hydraulic conductivity, porosity, and vertical gradient, were based on a study of two actual representative granite sites in the northeastern and north-central United States (EPA85).

Geochemical parameters are also necessary to evaluate the transport of radionuclides through geologic media. For each radionuclide in the waste inventory, retardation values are required. These values are dependent on the geologic medium in which the waste is disposed. The retardation values used in the granite analysis are given in Table 7.5-8.

Releases from the source were characterized in terms of radionuclide solubility in ground water. The solubilities used for the granite assessments are shown in Table 7.5-9.

A single set of waste form and repository configuration parameters was assumed for all sites modeled. These parameters include the radionuclide inventory and the discussed for all sites in Sections 7.5.2 and 7.5.3.

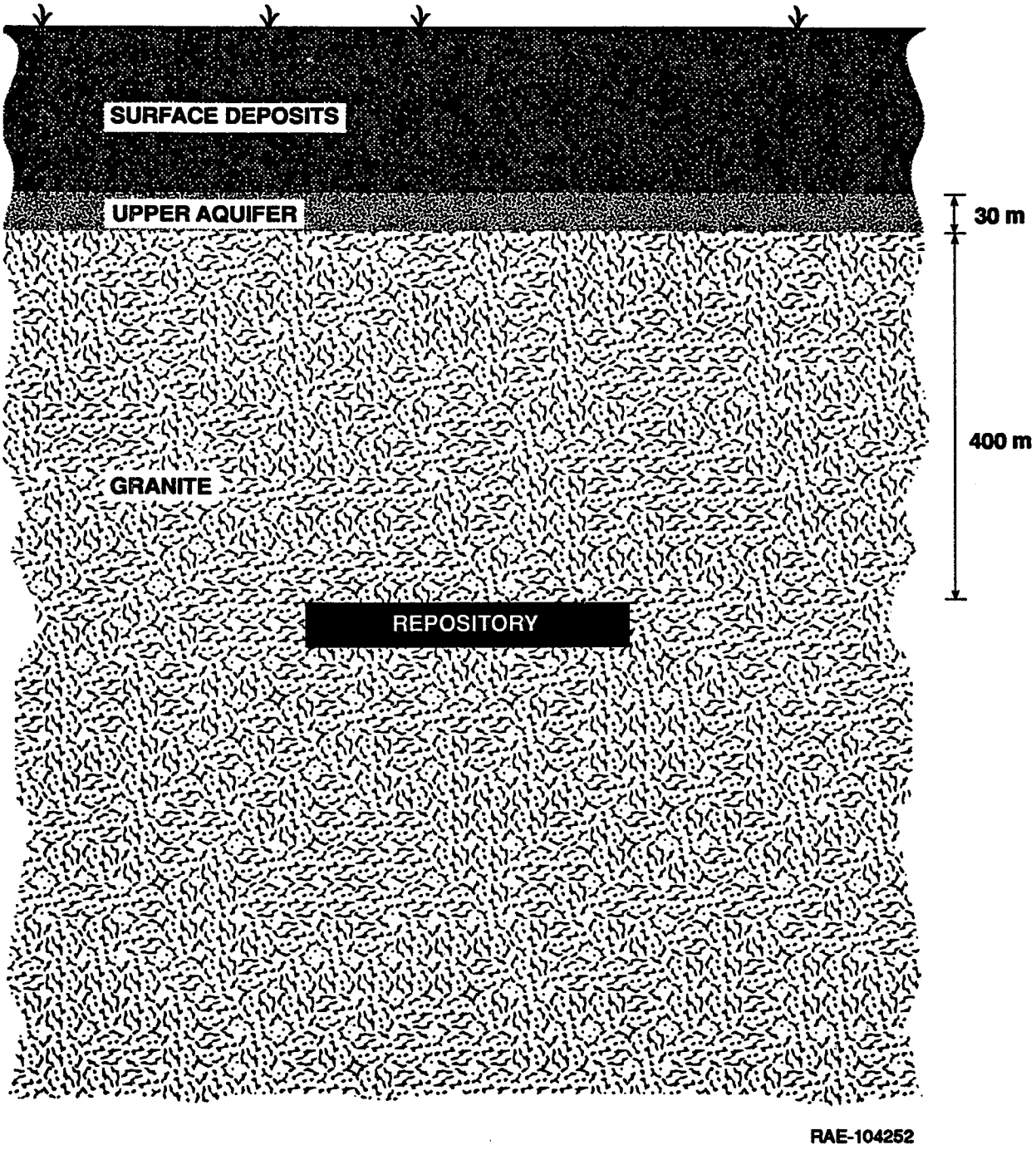

Figure 7.5-21. Cross-sectional structure of the model generic granite repository (not to scale).

Table 7.5-7. Site parameters used in the risk assessment of granite.

Input Parameter	Early Generic EPA Model[a]	Representative Granite Sites[b]		Current Model
		North Central	New England	
Average porosity of backfill in repository	0.2	Not Available	Not Available	0.2
Distance from repository to overlying aquifer (meters)	230	448	370	400
Hydraulic conductivity of the host rock between the repository and the aquifer, after thermal effects (meters/year)	3.20E-5	3.2E-2	3.2E-2	3.2E-2
Porosity of the host rock between the repository and aquifer	10^{-4}	10^{-4}	10^{-4}	10^{-4}
Hydraulic gradient between the repository and aquifer	0.1	0.01	0.01	0.01
Thickness of aquifer (meters)	30	10	80	30
Hydraulic conductivity of aquifer (meters/year)	31.5	5.7	315	31.5
Porosity of the aquifer	0.15	0.018	0.039	0.15
Horizontal gradient in aquifer	0.01	0.005	0.01	0.01
Horizontal distance along the aquifer to the accessible environment (meters)	1600	0[c]	2000	2000

[a]EPA-520/3-80-006 (EPA82)
[b]EPA-520/1-85-028 (EPA85)
[c]The model for this site assumes that surface water on the site is in contact with the ground water; therefore, there is no horizontal distance to the accessible environment, but only a vertical flow path.

Analyses were conducted to evaluate sensitivities and uncertainties in the parameter values. In the sensitivity studies, single parameters were varied discretely from the base case values. In the uncertainty analysis, statistical distributions of parameter values were defined for the key input parameters and those parameter values were varied in a Monte Carlo analysis. Three key parameters were identified for the sensitivity analysis. The parameters characterize the release from the waste form and the rate of transport through the ground-water system. The specific parameters selected for the analysis are the radionuclide solubilities, the vertical hydraulic conductivity in the granite host rock, and the radionuclide retardation factors. While other related parameters could have been included in the sensitivity and uncertainty analyses, those identified represent the key parameters for characterizing the magnitude of the radionuclide releases and the transport through the host rock and aquifer.

The parameter-value ranges for the granite analyses are shown in Table 7.5-10. The ranges encompass the values used in previous Agency assessments. The probability distributions are given for use in the NEFTRAN-S uncertainty analysis. Due to the wide range of values, log-uniform distributions were used for all of the parameters. This is preferable to using uniform distributions because a log-uniform distribution causes the median parameter value to be close to the base case value and is therefore more appropriate for parameters that vary over several orders of magnitude.

7.5.5.2.3 Base-Case Results from the Assessment of Generic Granite Sites

Figure 7.5-22 shows the results of the deterministic assessment of individual dose versus time for the granite site using the NEFTRAN-S computer code. The analysis assumes an undisturbed vertical ground-water flow through the repository horizon to the upper aquifer and then laterally through the aquifer. Dose was evaluated at a point 2000 meters down gradient. The assessment also assumes an individual drinking water consumption rate of 2 liters per day. Sensitivity of individual dose to solubility, retardation, and hydraulic conductivity are discussed in Section 7.5.5.2.4.

No radionuclides reach the 2000-meter boundary prior to approximately year 53,000. Thus, individual dose prior to year 53,000 is zero. At approximately year 53,000, the most mobile radionuclides, with retardation factors of 50, reach the 2000-meter boundary. Also, some of the radioactive decay products arrive at this time. Dose increases abruptly to approximately 570 mrem/yr. The rapid increase in dose is due to the relatively low dispersivity used as input to the model. A higher dispersivity would have led to a more gradual increase. From year 53,000 to year 61,000, dose increases to 900 mrem/yr. From year 61,000 to year 100,000, dose increases to 980 mrem/yr. Major contributing radionuclides at year 61,000 include U-233 (488 mrem/yr), Pa-231 (177 mrem/yr), Ac-227 (152 mrem/yr) and U-234 (68 mrem/yr).

Ground-water contamination was evaluated through three measures. First is the concentration of Ra-226. Second is the total concentration of all alpha-emitting radionuclides, excluding radon. Third is the drinking water dose resulting from all beta- and gamma-emitting radionuclides. Each of these measures was evaluated through the

Table 7.5-8. Radionuclide retardation factors for granite.

Element	Range of Retardation Factors[a]		
	Low	"Base Case"	High
Strontium	10	200	2,000
Cesium	100	1,000	10,000
Lead	10	50	200
Radium	50	500	5,000
Actinium[b]	10	50	500
Thorium	500	5,000	10,000
Protactinium[b]	10	50	500
Uranium	10	50	500
Neptunium	10	100	500
Plutonium	10	200	5,000
Americium	500	3,000	50,000
Curium	200	2,000	10,000

[a]From 1983 WISP report (NAS83).
[b]Because values were not given in WISP report, values of uranium were used based on chemical similarities.

Table 7.5-9. Radionuclide solubilities for granite.[a]

Nuclide	Solubility (Ci/m^3)
Ac-227	1.64E+01
Am-241	8.28E-01
Cm-248	1.06E-03
Cs-137	1.19E+01
Np-237	1.67E-04
Pa-231	1.09E-02
Pb-210	1.60E+01
Pu-238	4.08E+00
Pu-239	1.49E-02
Pu-240	5.47E-02
Pu-242	9.51E-04
Ra-226	2.24E-01
Sr-90	1.23E+01
Th-229	4.87E-02
Th-230	4.65E-03
Th-232	2.55E-08
U-233	2.26E-03
U-234	1.46E-03
U-235	5.08E-07
U-236	1.53E-05
U-238	8.01E-08

[a]Based on 1.0E-06 mole/liter (La89).

Table 7.5-10. Parameter ranges and distributions for granite.

Parameter	Minimum	Maximum	Distribution Type
Solubility (mole/liter)	1.0E-09	1.0E-03	Log Uniform
Vertical hydraulic conductivity (m/yr)	3.2E-05	3.2E-01	Log Uniform
Retardation factors	(a)	(b)	Log Uniform

[a]The sensitivity and uncertainty analyses used retardation factors of one, as well as the "low" values from Table 7.5-8.
[b]See Table 7.5-8.

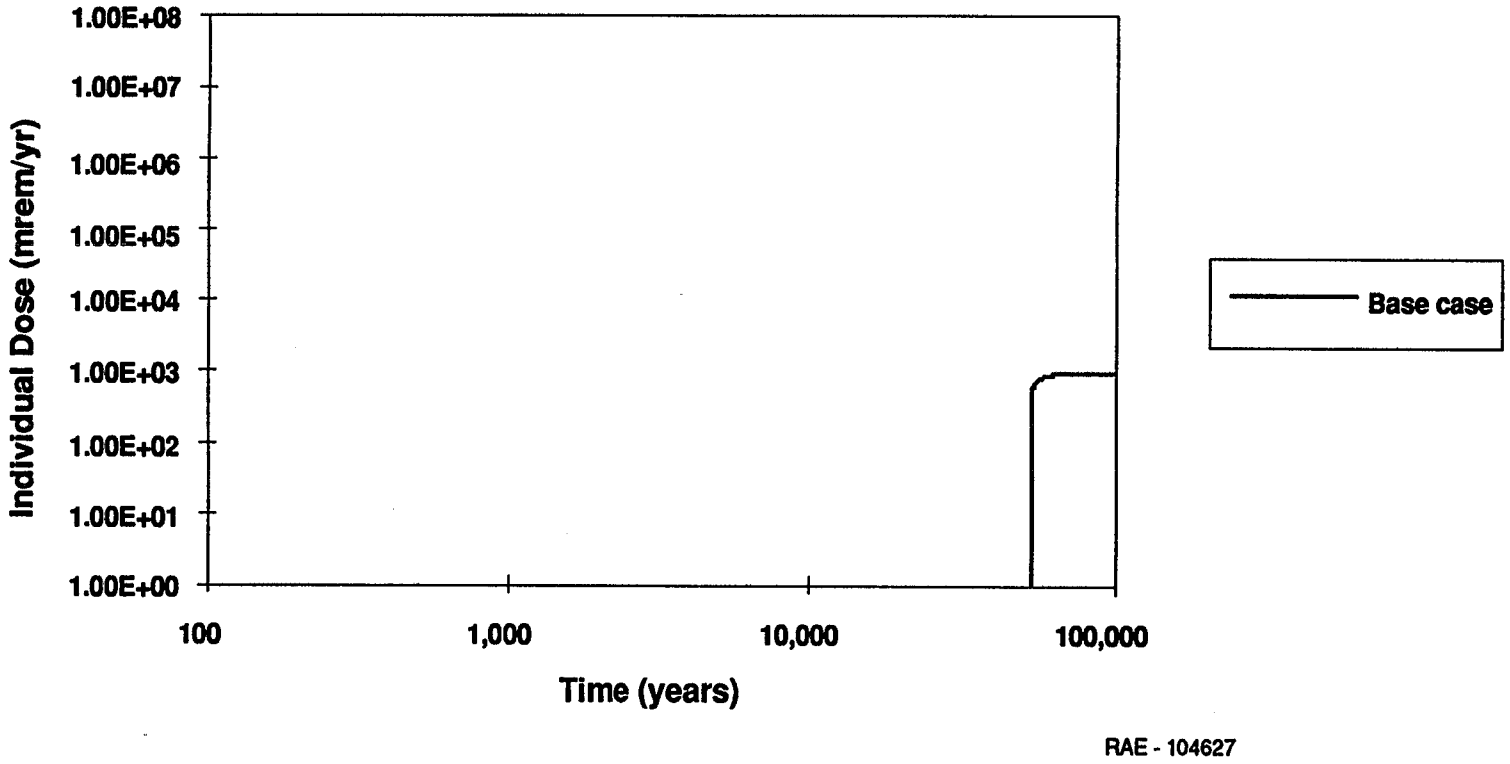

Figure 7.5-22. Individual dose for granite.

dimensions and capacity of the underground repository facility. These parameters are NEFTRAN-S analysis.

Ra-226 is part of the Pu-238 decay series. Figure 7.5-23 shows the concentration of Ra-226, as a function of time, calculated 2000 meters down gradient. Ra-226 first arrives at year 54,000, with a concentration in ground water of 10^{-7} pCi/liter. From 54,000 years to 60,000 years its concentration increases to 5.4×10^{-7} pCi/liter. Then its concentration increases sharply to approximately 2.0×10^{-2} pCi/liter at year 67,000, and then increases steadily to 0.7 pCi/liter at the end of the 100,000-year simulation period.

Figure 7.5-24 shows the total concentration of alpha-emitting radionuclides as a function of time, calculated 2000 meters down gradient. Concentration is zero until 53,000 years. It then rises sharply to almost 790 pCi/liter at year 61,000. The concentration then decreases slowly but steadily to a concentration of 750 pCi/liter at year 100,000. Major contributors to the total concentration at year 61,000 are U-233 (630 pCi/liter), U-234 (88 pCi/liter), Pa-231 (49 pCi/liter) and U-236 (16 pCi/liter).

The total concentration of beta-emitting radionuclides is measured in terms of the dose which would result from the consumption of two liters per day of the contaminated ground water. There are four beta-emitting radionuclides: Sr-90, Cs-137, Ac-227 (generated through the decay of Pu-239 and U-235), and Pb-210 (generated through the decay of Pu-238). As shown in Figure 7.5-25, the total dose is zero until 53,000 years. The dose then increases rapidly and then slowly, varying from 100 mrem/yr to 225 mrem/yr. The dose results mainly from the concentration of Ac-227, although Pb-210 contributes somewhat. Sr-90 and Cs-137 do not contribute to the dose because of their short half-lives. The dose from beta-emitting and gamma-emitting radionuclides is about 20 percent of the total dose from all radionuclides.

7.5.5.2.4 Sensitivity and Uncertainty Analyses in the Generic Granite Assessments

The previous section discussed the results of evaluating individual doses and concentrations in ground water using the base-case parameter values given in Table 7.5-7. This section discusses the sensitivity of individual dose and concentrations in ground water to variations in radionuclide solubility, hydraulic conductivity in the vertical transport leg, and radionuclide retardation factors.

Individual Dose - Radionuclide solubility controls the rate at which radionuclides enter into the ground water. Higher solubilities result in higher concentrations of radionuclides per unit of water. Figure 7.5-26 shows the sensitivity of individual dose to variations in solubilities. The base-case solubility was 10^{-6} mole/liter. Individual doses were calculated with higher (10^{-3} mole/liter) and lower (10^{-9} mole/liter) solubilities. Varying the solubility does not effect the time of arrival of the first measured dose. It does, however, significantly affect the magnitude of the dose. Increased solubility results in a much greater and sharper initial dose. The magnitude of this peak dose is approximately 21,000 mrem/yr. At high solubility, the dose falls off more rapidly with time due to depletion of the inventory.

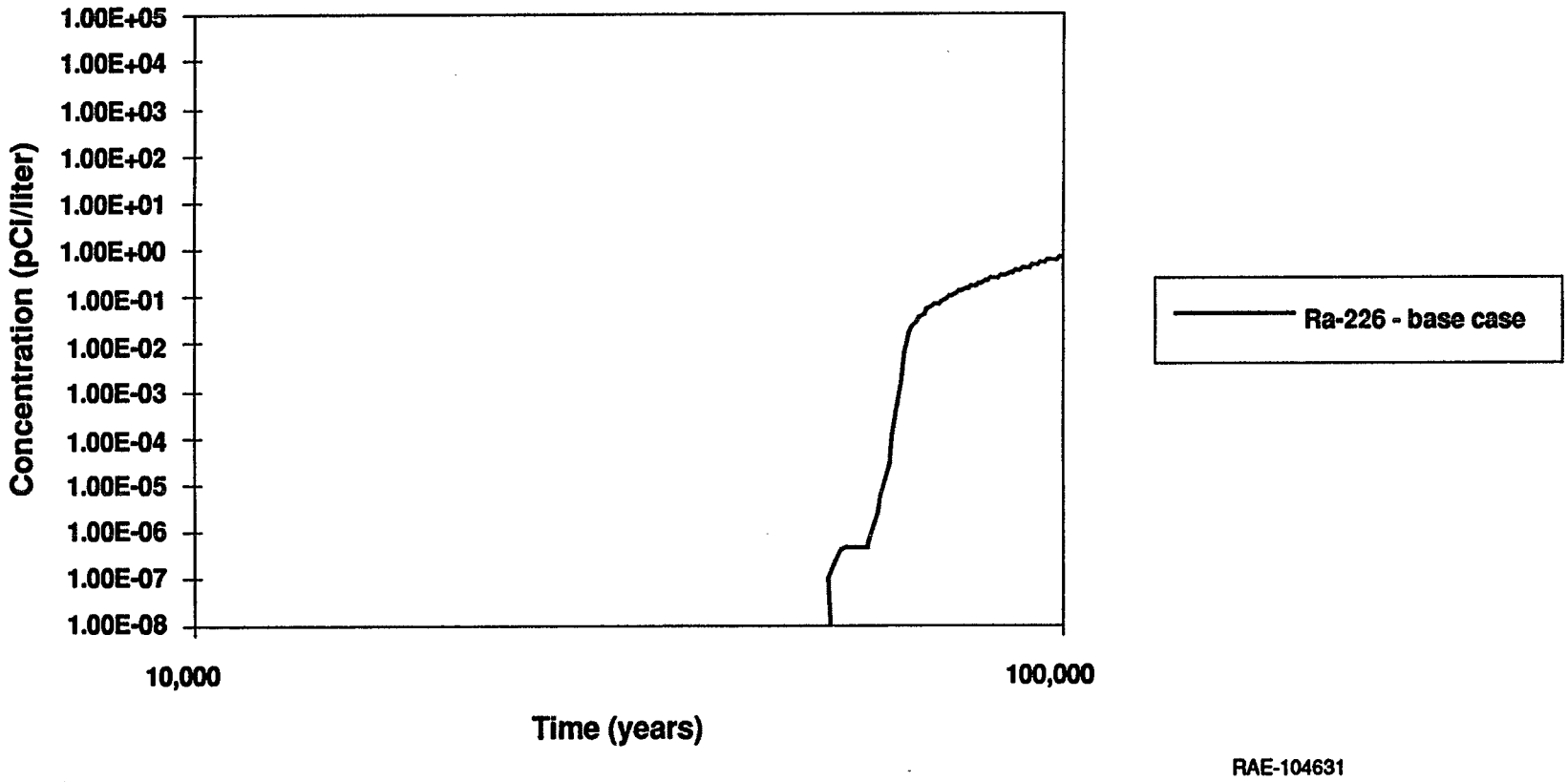

Figure 7.5-23. Ra-226 groundwater concentration - base case granite.

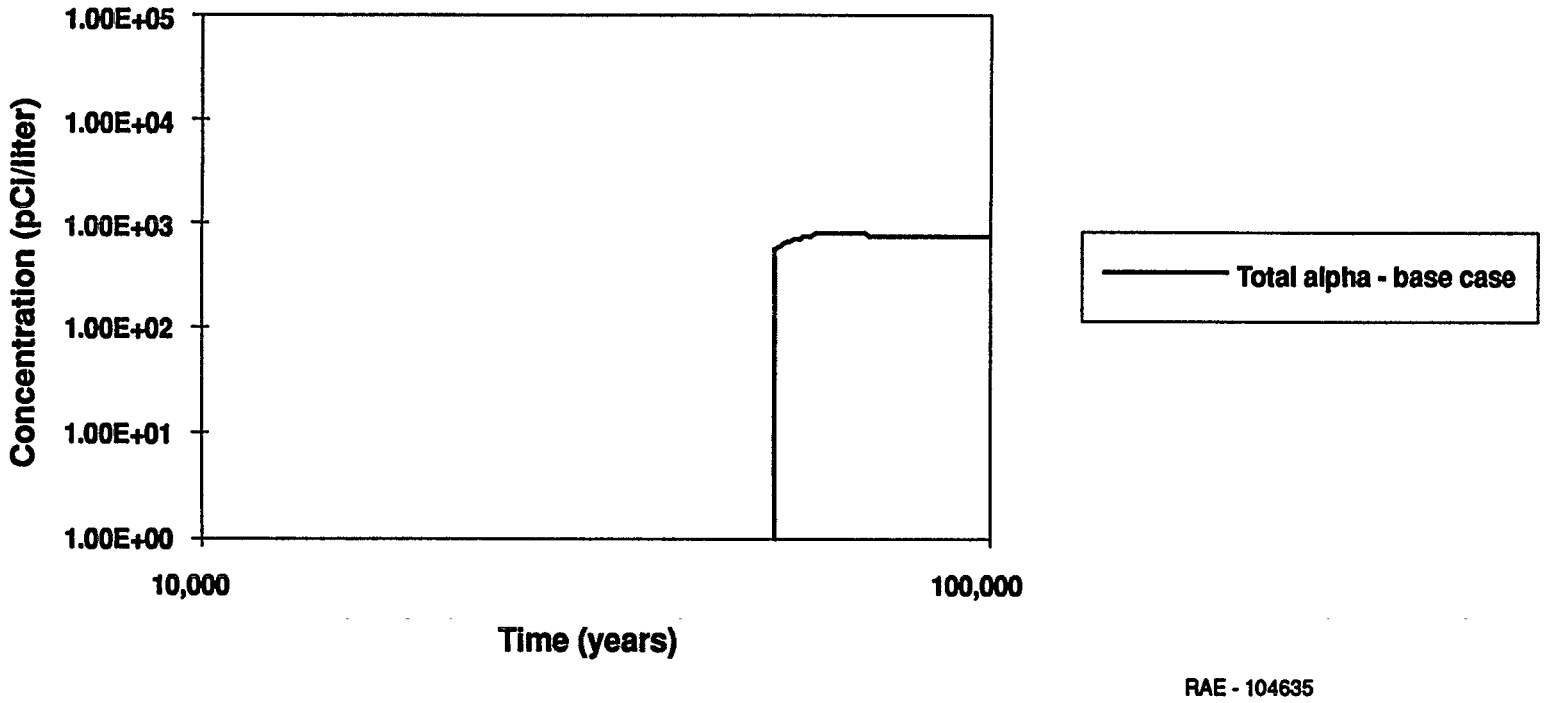

Figure 7.5-24. Total alpha groundwater concentrations - base case granite.

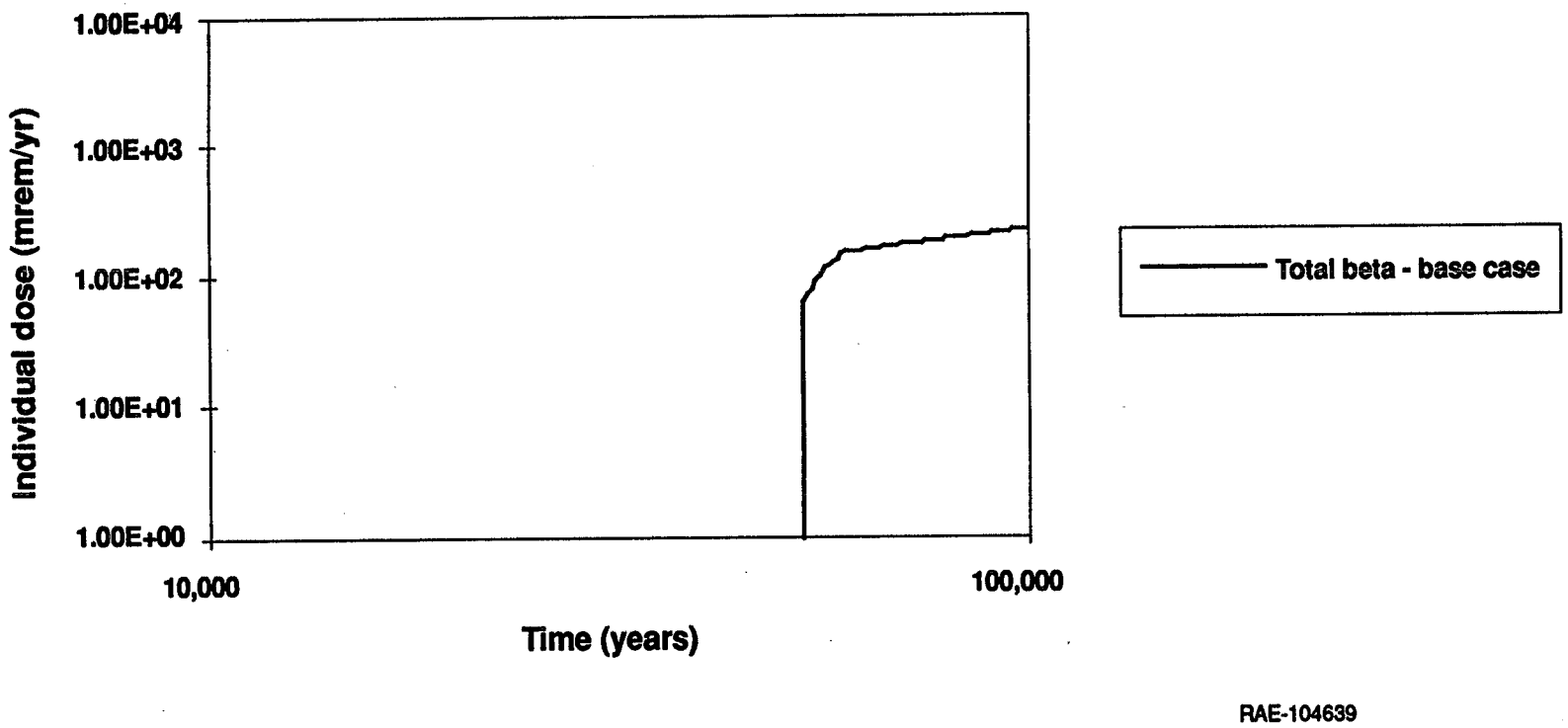

Figure 7.5-25. Total beta groundwater dose - base case granite.

Figure 7.5-27 shows the effect of varying the hydraulic conductivity of the granite in the vertical transport leg. Increasing the vertical hydraulic conductivity increases the volume of flow through a given cross-sectional area and decreases the travel time. Since the vertical distance from the repository to the aquifer is 400 meters, the decrease in the travel time due to an increase in vertical hydraulic conductivity is more apparent than in the basalt analysis. The increased flow through the repository horizon results in a greater release of radioactivity from the repository, and thus an increased dose. Decreasing the hydraulic conductivity several orders of magnitude to 3.2×10^{-5} m/yr resulted in zero dose during the 100,000-year assessment period.

Variations in radionuclide retardation have the greatest effect on individual dose (Figure 7.5-28). Decreasing the retardation factors increases the mobility of the radionuclides and thus decreases the travel time to the 2000-meter boundary. Thus, with lower retardations, the Np-237 peak arrives much earlier, at 10,500 years. All the Np-237 has passed by year 46,000. The plutonium radionuclides, with a lower retardation factor of 10, arrive with the uranium and neptunium at 10,500 years. The retardations are not low enough to see the arrival of americium or curium in 100,000 years. With no retardation, however, all radionuclides are transported at the same velocity. Thus, doses begin much earlier, at 1,000 years, and doses are much greater due to the contributions from plutonium, americium, and curium.

In addition to the deterministic sensitivity studies of individual dose, a probabilistic uncertainty study was conducted. For the three parameters of interest - solubility, vertical hydraulic conductivity, and retardation - parameter-value ranges were assigned instead of single values. The ranges used for the three parameters are given in Table 7.5-10. Two analyses were conducted: one using the low retardation values (Table 7.5-8) as a minimum and a second assuming no retardation by using one as a minimum retardation value. Using the Monte Carlo sampling routine of the NEFTRAN-S code, peak doses were calculated for the 10,000-year period following disposal. No dose was reported in the low-retardation analysis. In actuality, this means there is a very low probability of dose in 10,000 years, given the parameter uncertainty. The results of the analysis using a minimum retardation value of one are shown by the histogram in Figure 7.5-29. Considering the parameter uncertainty as represented by the input parameter ranges, there is a 0.62 probability of zero dose in 10,000 years. However, these results, which include no retardation, represent bounding conditions for repository performance. They are included to represent the importance of retardation in EPA's model of generic repository performance.

Ra-226 Concentrations - The sensitivities of Ra-226 concentrations in ground water 2000 meters down gradient are similar to those described for individual doses. Figure 7.5-30 shows that decreased solubility results in decreased concentrations, while increased solubility results in increased concentrations. As shown in Figure 7.5-31, increasing the hydraulic conductivity in the vertical leg has a minor effect on the initial arrival time but a significant effect on the magnitude of the concentration of Ra-226, due to increased flow. Decreasing the hydraulic conductivity results in no Ra-226 concentration during the 100,000-year assessment period. Finally, varying the retardation has a significant effect on both the arrival time and the magnitude of Ra-226 concentrations, as shown in Figure 7.5-32.

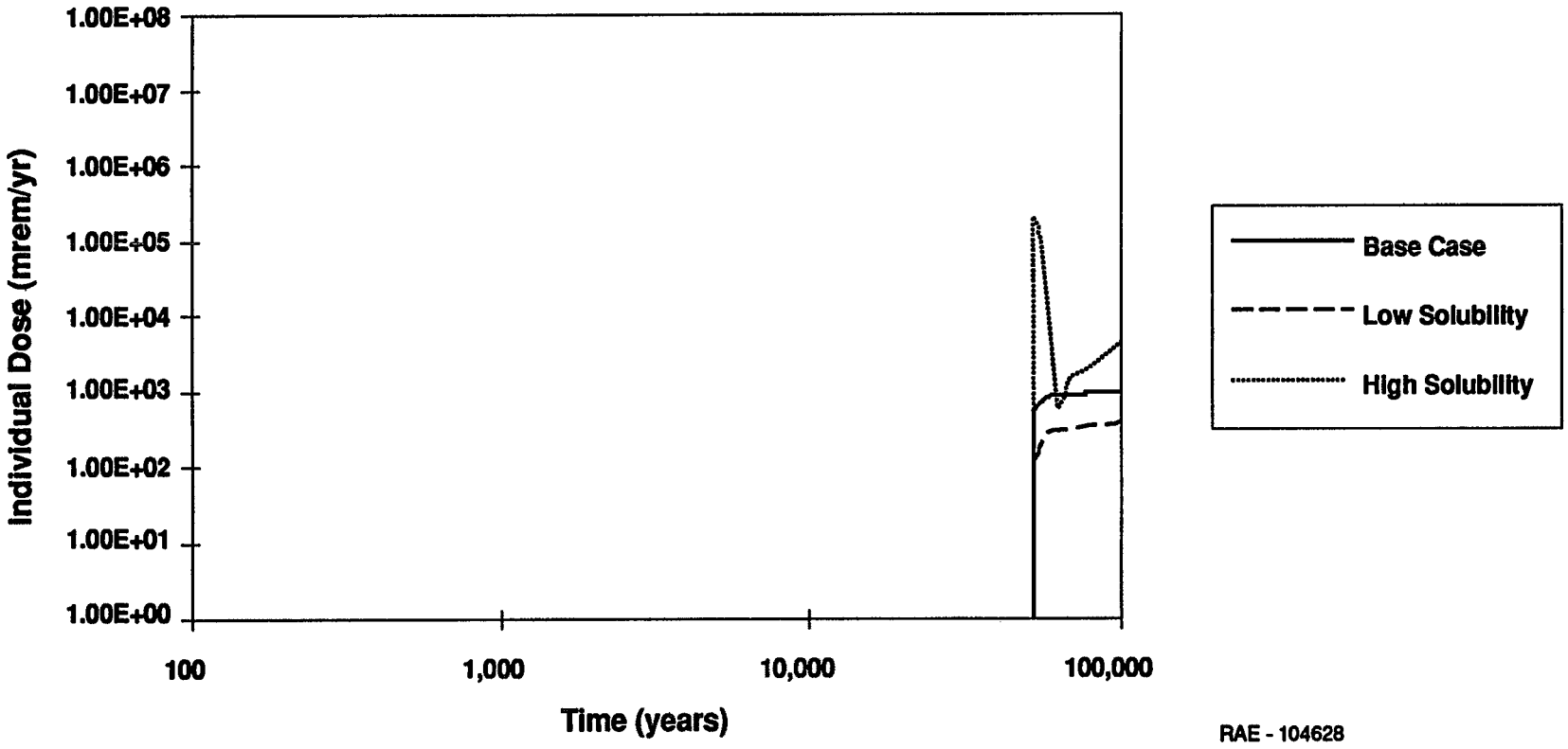

Figure 7.5-26. Sensitivity of dose to solubility - granite.

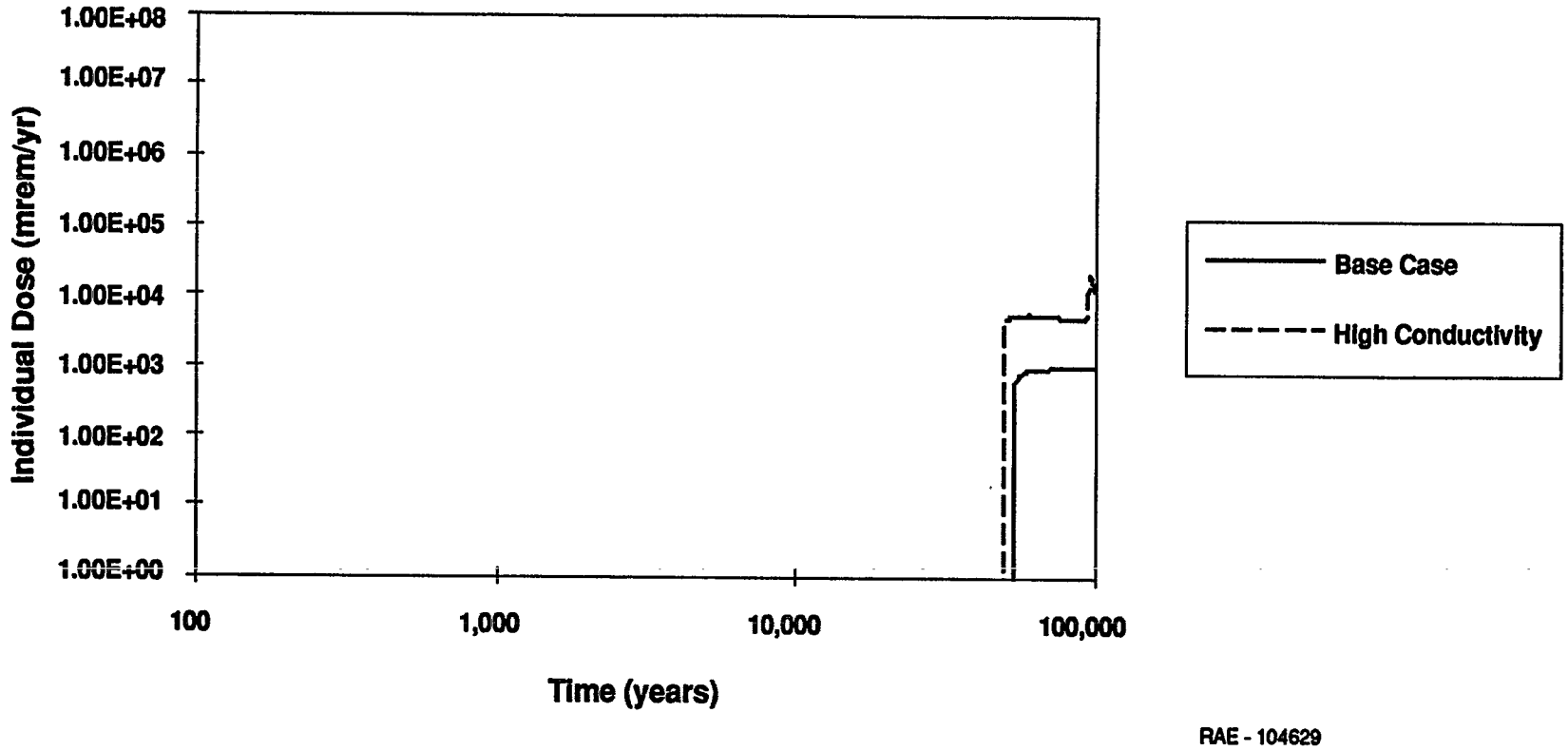

Figure 7.5-27. Sensitivity of dose to vertical hydraulic conductivity - granite.

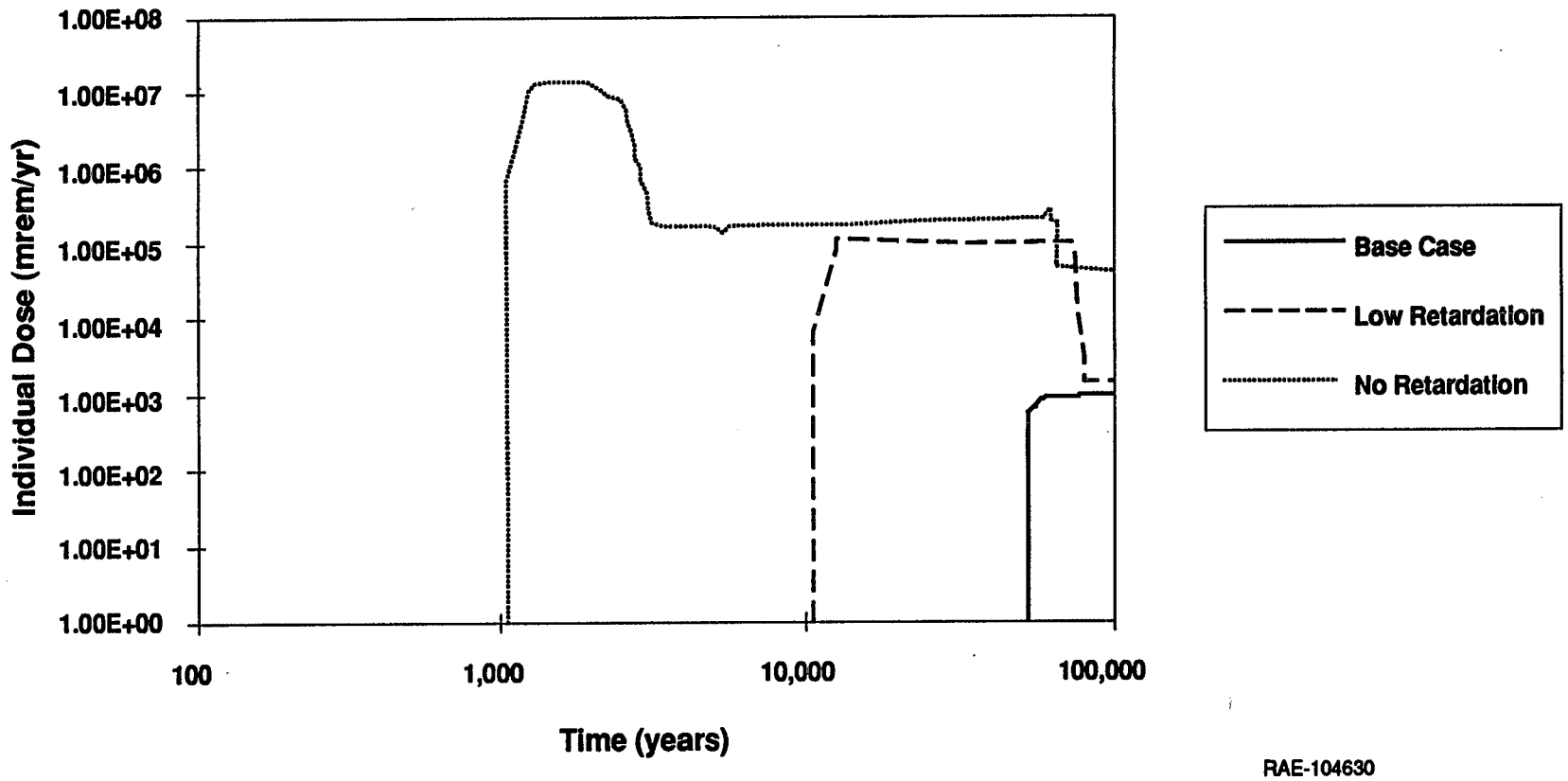

Figure 7.5-28. Sensitivity of dose to retardation - granite.

Total Alpha Concentrations - The sensitivities of concentrations of alpha-emitting radionuclides in ground water 2000 meters down gradient are also similar to those described for individual doses. Figure 7.5-33 shows that decreased solubility results in decreased concentrations, while increased solubility results in increased concentrations. Figure 7.5-34 shows that increasing the hydraulic conductivity in the vertical leg has a minor effect on the initial arrival time but a significant effect on the magnitude of the concentration, due to increased flow through the repository. Decreasing the hydraulic conductivity results in no alpha concentration during the 100,000-year assessment period. Finally, varying the retardation has a significant effect on both the arrival time and the magnitude of the concentration of alpha-emitting radionuclides, as shown in Figure 7.5-35.

Total Beta and Gamma Concentrations - The sensitivities of dose from beta- and gamma-emitting radionuclides 2000 meters down gradient are shown in Figures 7.5-36 through 7.5-38. Only Ac-227 and Pb-210 contribute to the concentrations. The other two beta-emitters, Sr-90 and Cs-137, do not arrive due to their short half-lives. Figure 7.5-36 shows that decreased solubility results in decreased concentrations and thus dose, while increased solubility results in increased concentrations and dose. As shown in Figure 7.5-37, increasing the hydraulic conductivity in the vertical leg has a minor effect on the initial arrival time but a significant effect on the magnitude of the dose, due to increased flow. Decreasing the hydraulic conductivity results in zero dose. Finally, varying the retardation has a significant effect on both the arrival time and the magnitude of the dose from beta-emitting radionuclides (Figure 7.5-38).

7.5.5.3 Site Analysis - Bedded Salt

7.5.5.3.1 Introduction

For almost 30 years, salt deposits have been considered prime candidates for a nuclear waste repository. There are a number of reasons for this. Salt deposits are common in several regions of the United States and they are found at depths considered to be suitable for a repository. By their very presence, they indicate relative geologic stability and hydrologic isolation, since if ground water had ready access to them the salt would have been dissolved and carried away. While it is the case that almost all known salt beds are undergoing gradual dissolution by ground water, the rates of such dissolution processes are generally so slow that these deposits are expected to remain substantially intact for millions of years. In addition, there is extensive experience in constructing underground mines in salt. Another advantage is that gradual creep of the salt will aid in the resealing and the reestablishment of total isolation of a repository placed in such an environment. On the other side, there is the disadvantage that if some unforeseen circumstances arise that bring ground water into contact with the salt near the repository, the effects might be severe with the relatively rapid dissolution of the salt. Also, salt deposits are located in sedimentary basins that often contain other valuable resources such as oil, gas, and potash. As a result, the adoption of a site for a nuclear waste repository may either preempt access to the resources present at the site or lead to future risks from efforts to obtain those resources.

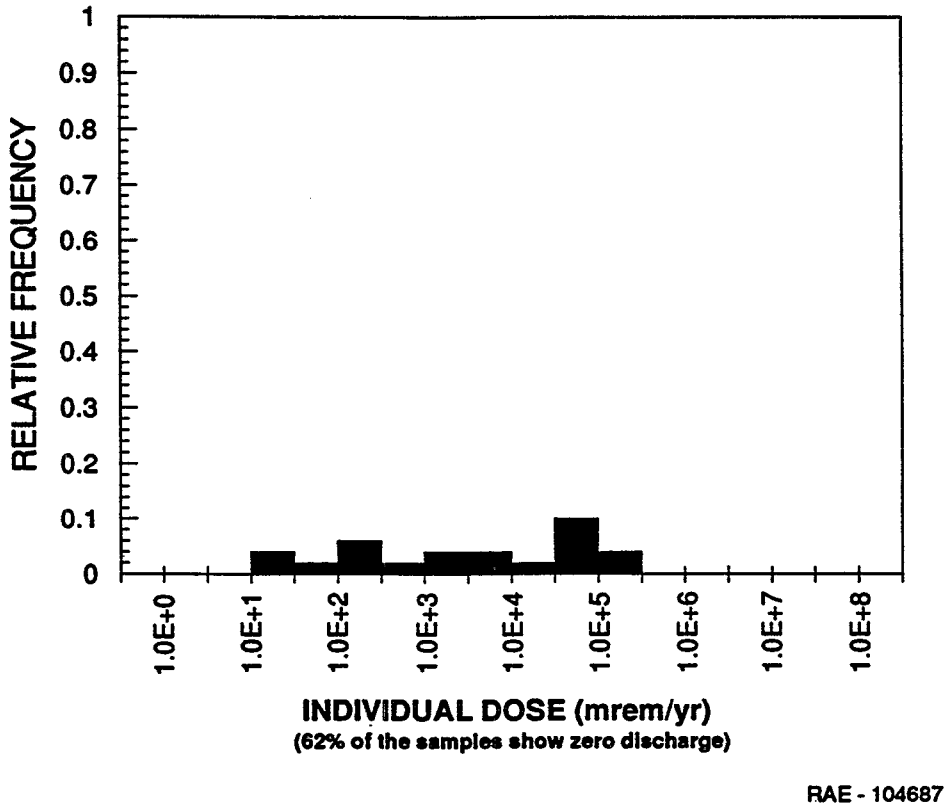

Figure 7.5-29. Distribution of individual dose due to parameter value uncertainty (minimum retardation value of one) - granite.

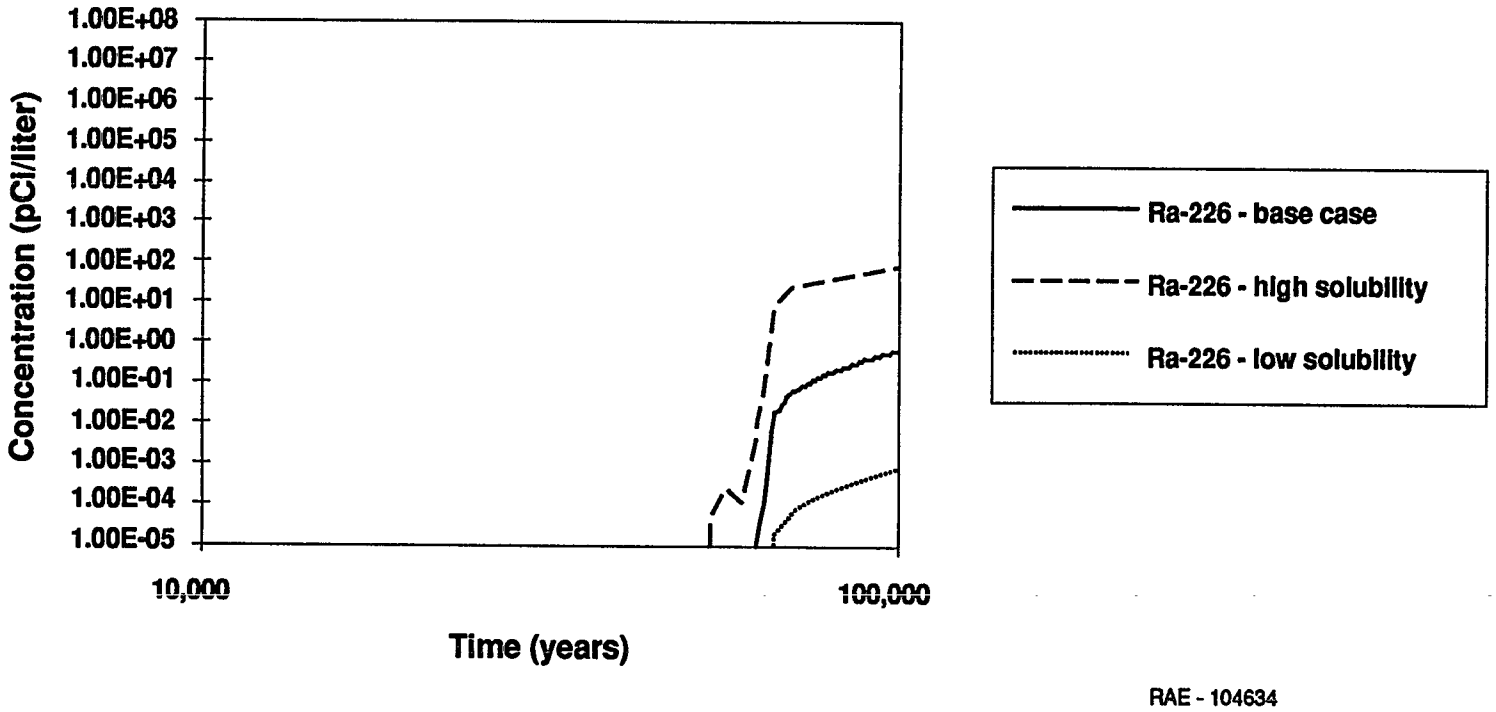

Figure 7.5-30. Sensitivity of Ra-226 groundwater concentrations to solubility - granite.

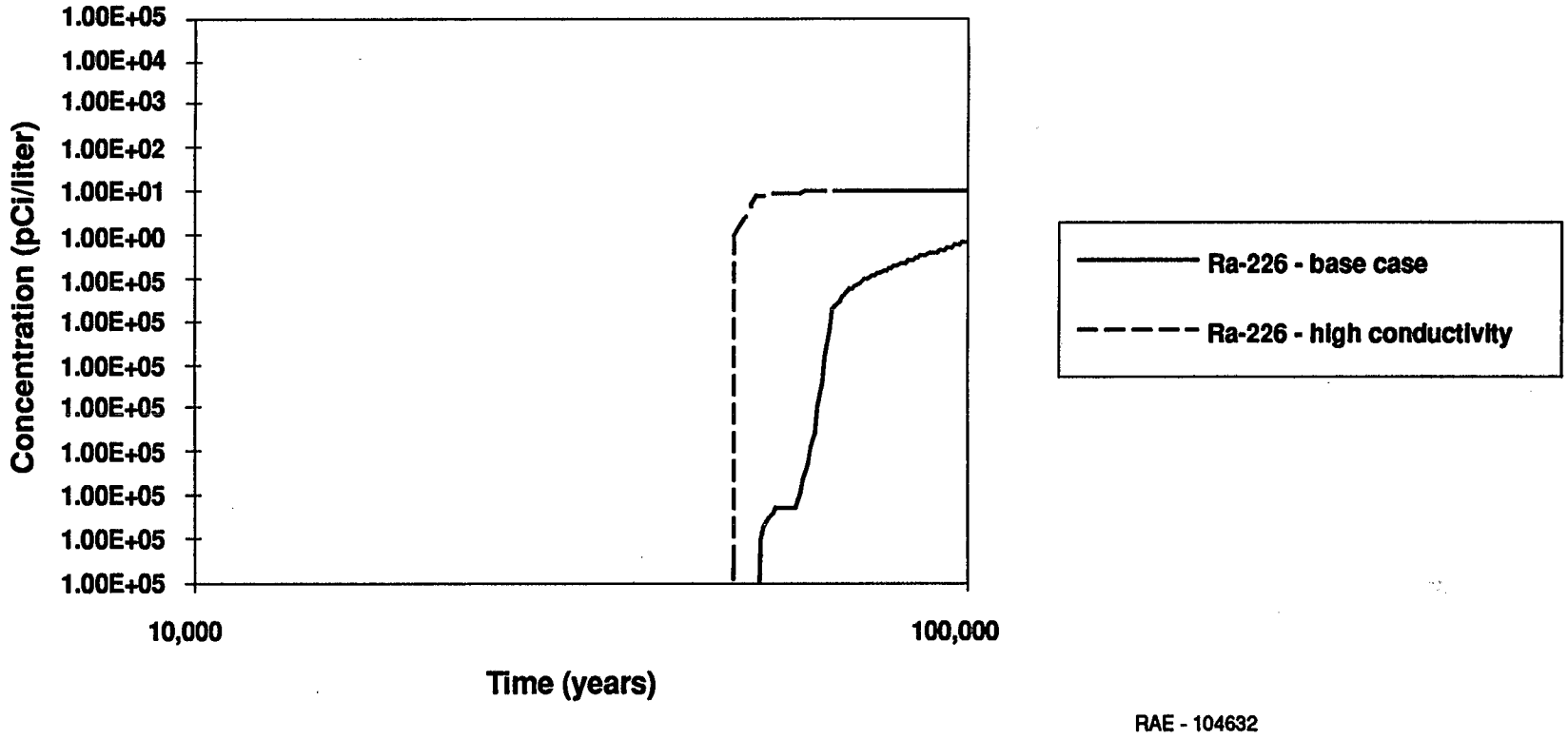

Figure 7.5-31. Sensitivity of Ra-226 groundwater concentrations to vertical hydraulic conductivity - granite.

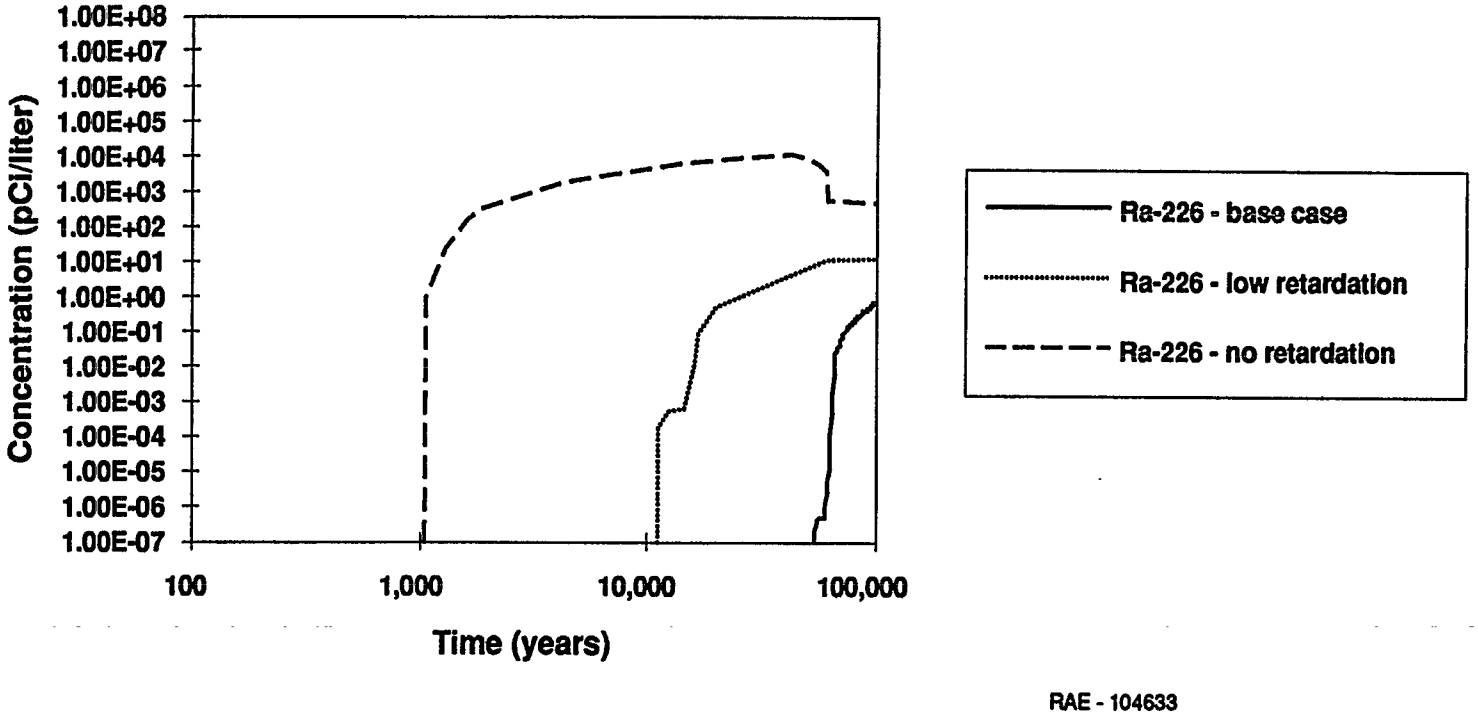

Figure 7.5-32. Sensitivity of Ra-226 groundwater concentrations to retardation - granite.

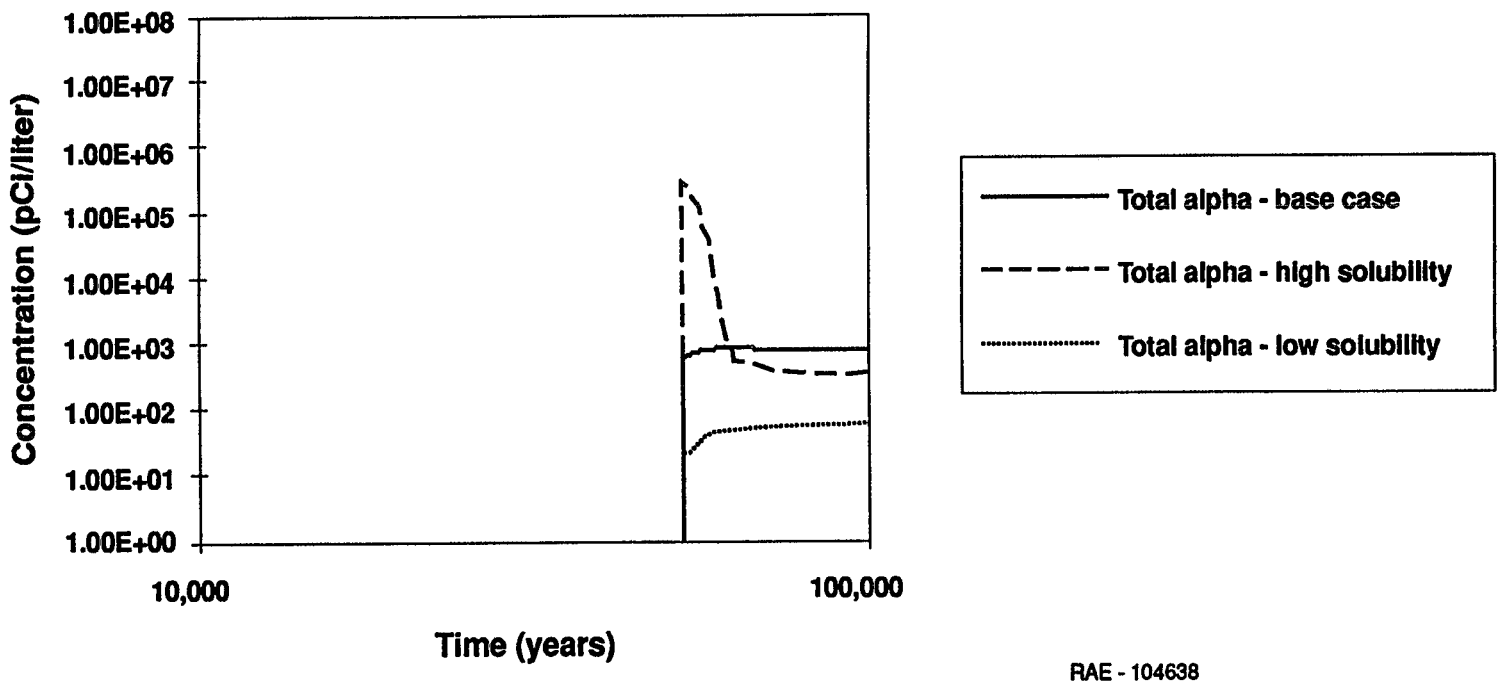

Figure 7.5-33. Sensitivity of total alpha groundwater concentrations to solubility - granite.

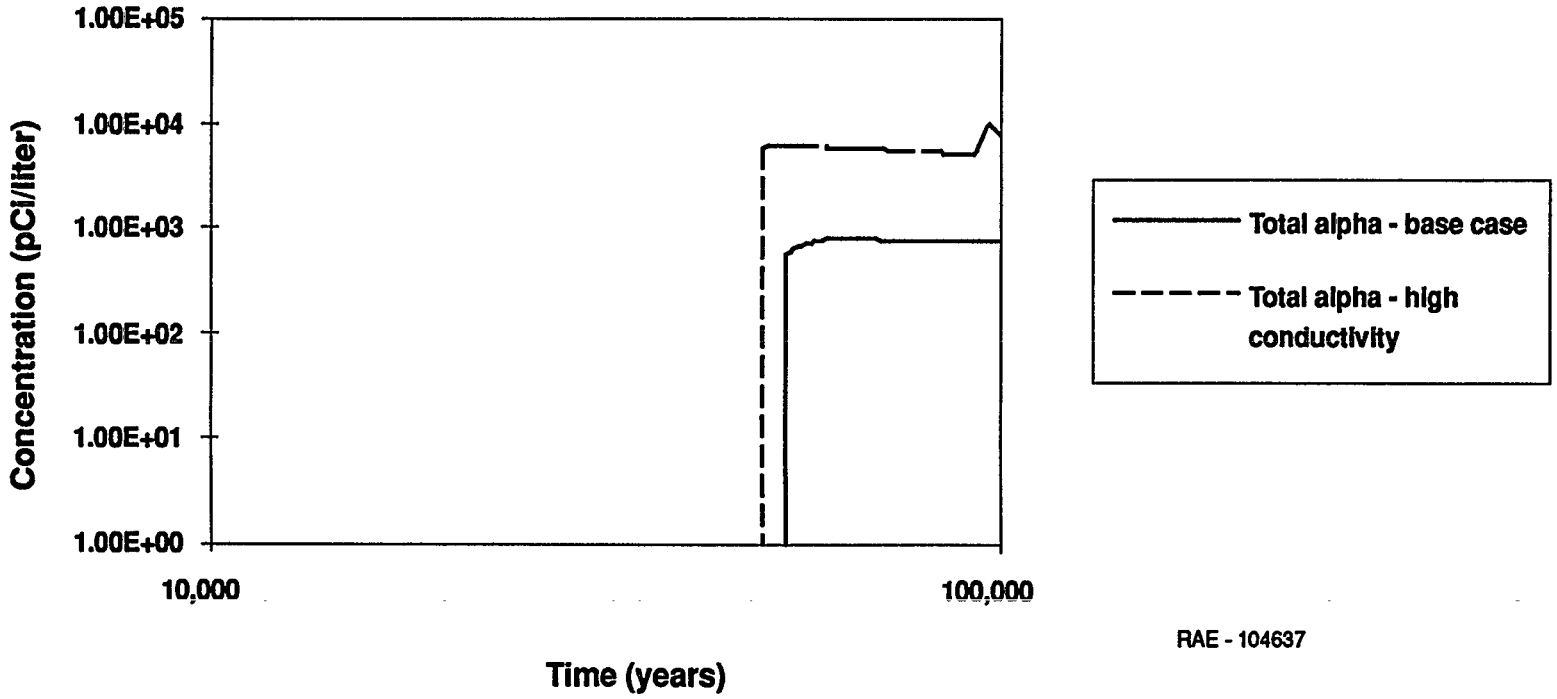

Figure 7.5-34. Sensitivity of total alpha groundwater concentratons to vertical hydraulic conductivity - granite.

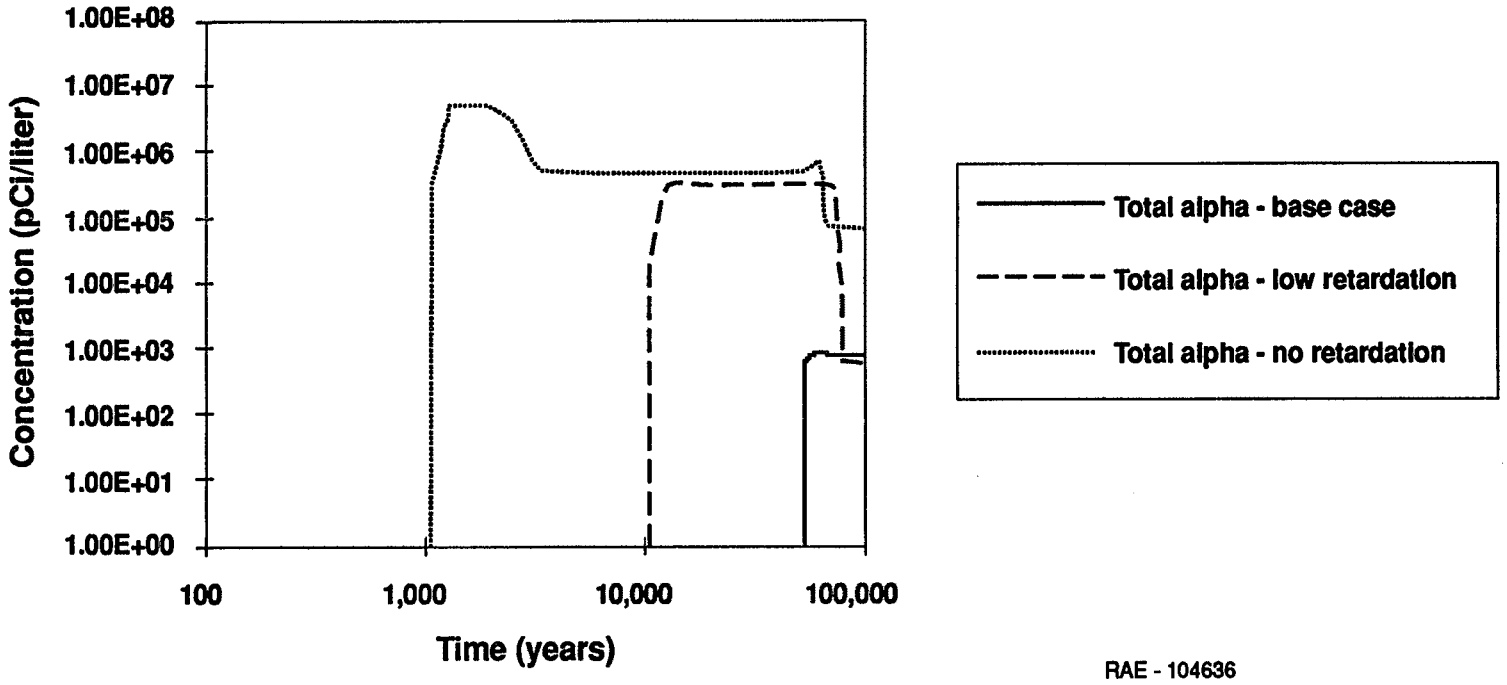

Figure7.5-35. Sensitivity of total alpha groundwater concentrations to retardation - granite.

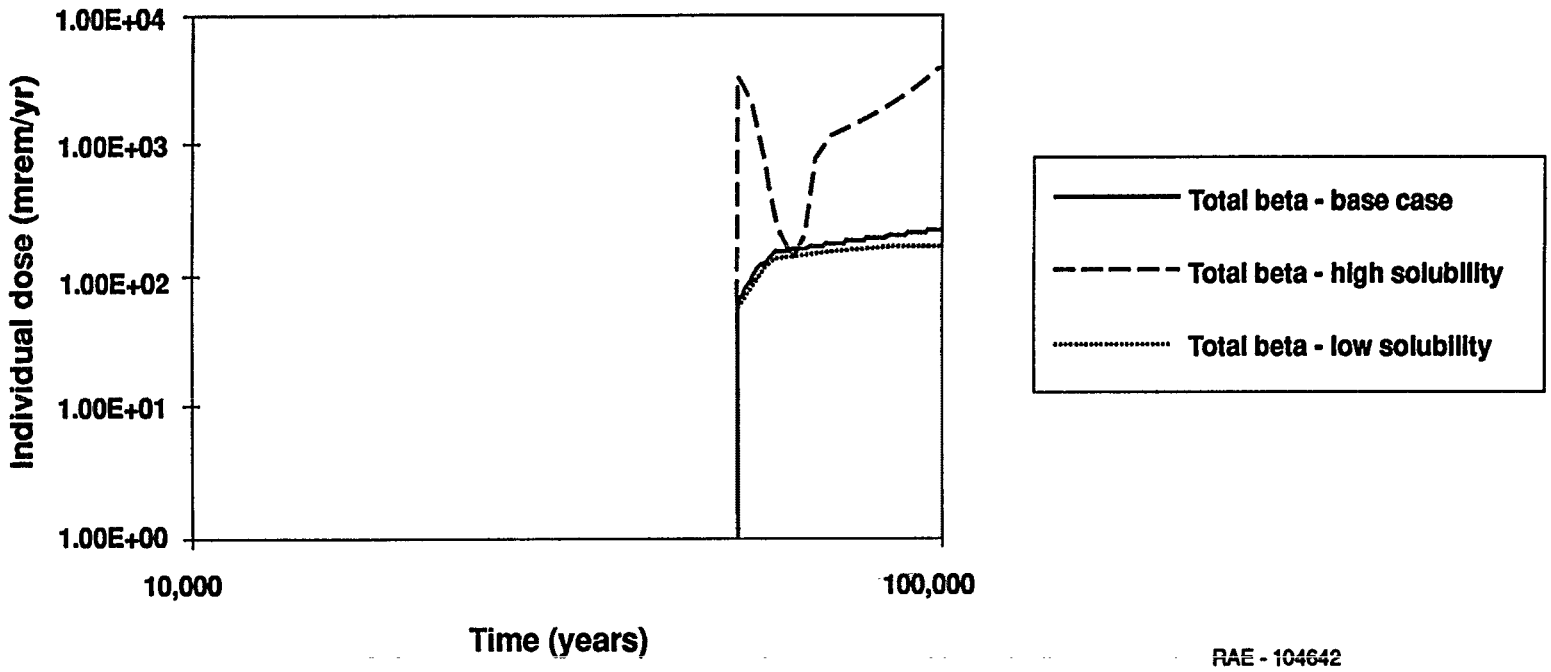

Figure 7.5-36. Sensitivity of total beta groundwater dose to solubility - granite.

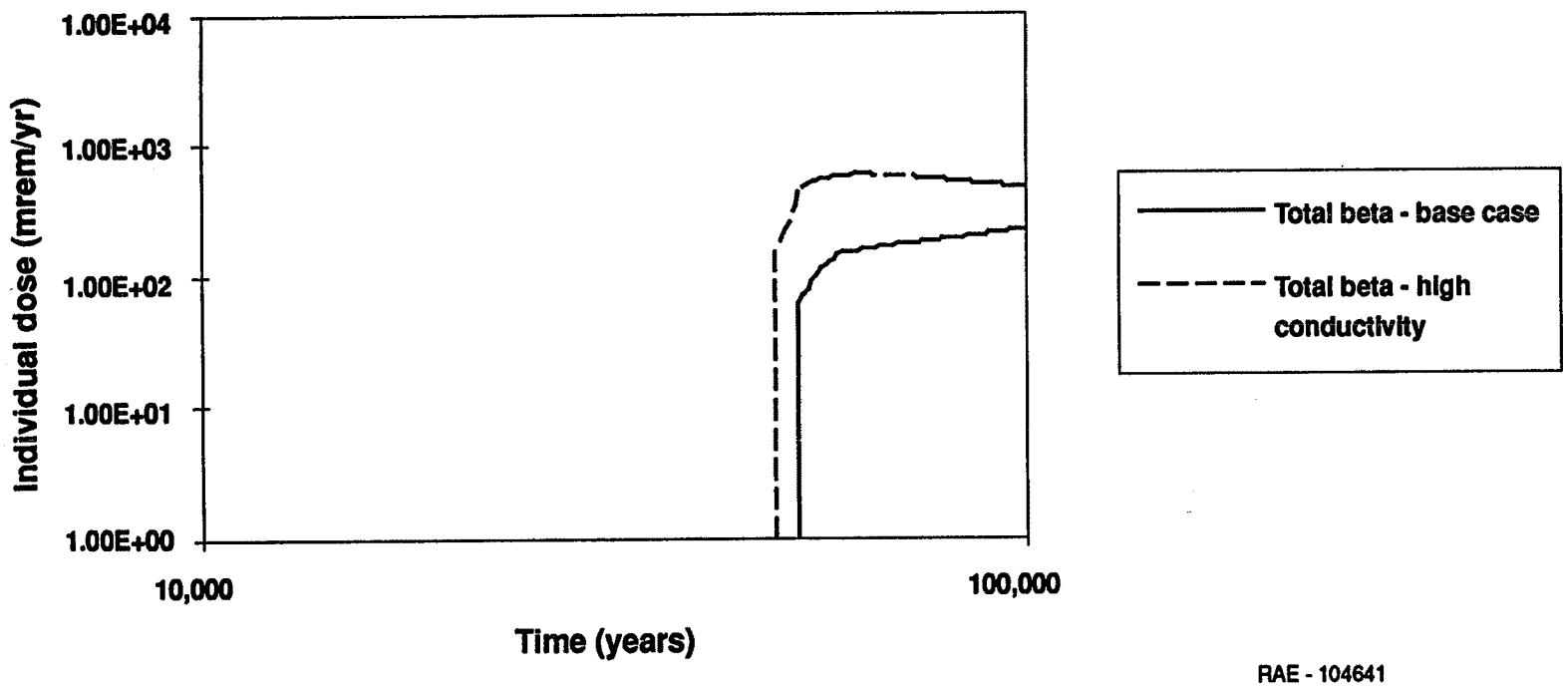

Figure 7.5-37. Sensitivity of total beta groundwater dose to vertical hydraulic conductivity - granite.

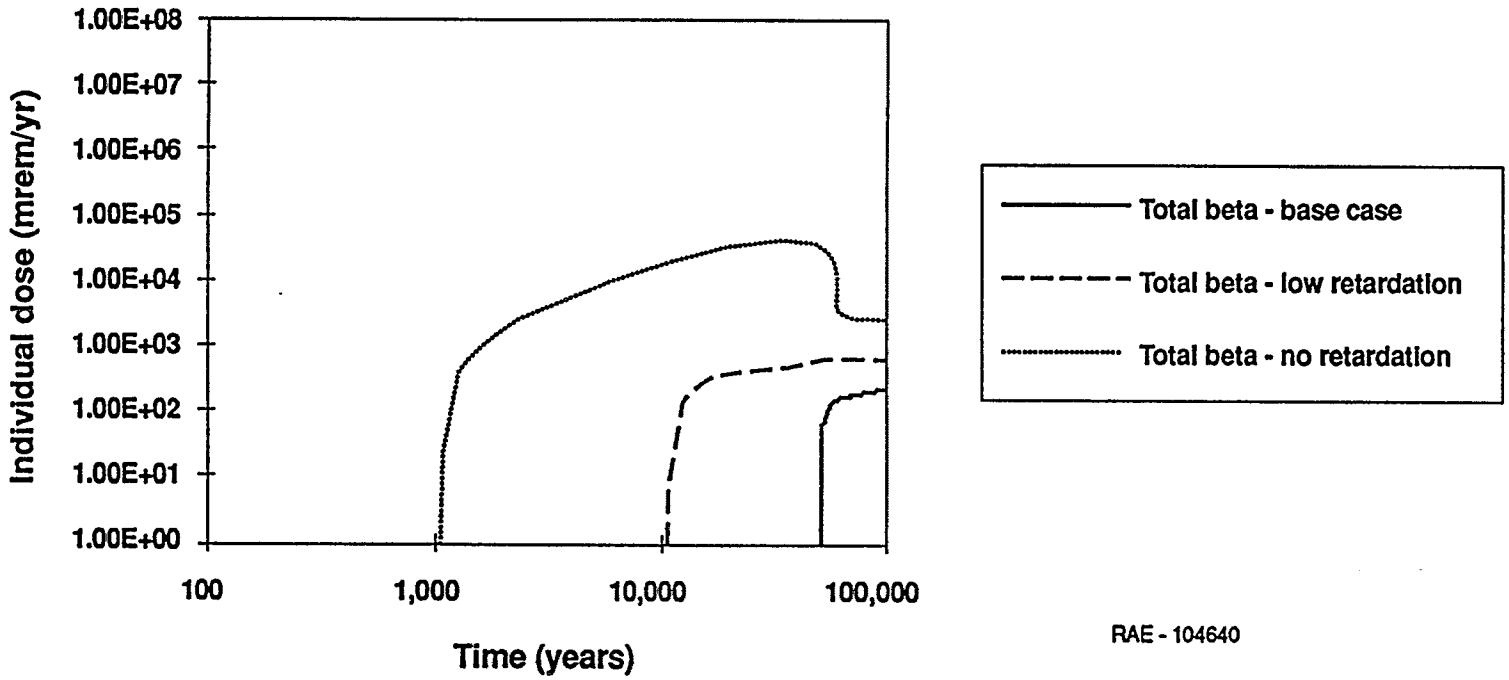

Figure7.5-38. Sensitivity of total beta groundwater dose to retardation - granite.

The DOE has investigated potential repository sites in bedded salt deposits in the Paradox Basin in Utah and in the Palo Duro Basin in Texas. Based on data collected by the Department and others, it is possible to define conceptual models of repository sites in each of the two basins so as to make rough first approximations of the potential performance of such repositories and to identify some of the parameters that are most critical in determining that performance. The models and parameters used by the Agency as the "generic" salt site are based on simplified models of the geologic and hydrologic systems in each of these two basins.

Section 7.5.5.3.2 discusses the important input parameters that have been used in the Agency's risk analyses for bedded salt. Since most of these data can best be presented in the form of tables and figures, there is minimal textual discussion of additional details. Also presented are data from the Agency's population risk assessment (EPA82). Section 7.5.5.3.3 provides the results of the base-case analyses. Section 7.5.5.3.4 provides the results of the sensitivity and uncertainty analyses.

7.5.5.3.2 Input Parameters for Bedded Salt

The conceptual model developed to support the evaluation of disposal at a generic salt site is depicted in Figure 7.5-39. The Agency's original conceptual model for a generic salt site was described in "Population Risks from Disposal of High-Level Radioactive Wastes in Geologic Repositories" (EPA82). The original conceptual model was modified based on characteristics of the Palo Duro Basin and Paradox Basin salt formations, as described in Appendix A of "Risk Assessment of Disposal of High-Level Radioactive Wastes in Geologic Repositories" (EPA85).

The geologic and hydrogeologic parameters, and their values, which define the generic salt model are given in Table 7.5-11. The table lists the parameter values used in the EPA evaluation of population risks (EPA82), the parameter values obtained from the descriptions of the Palo Duro Basin and Paradox Basin sites (EPA85), and the parameter values used in the current evaluation.

Two sources were used to compile the parameter information shown in the table. The aquifer parameters were taken from the previous generic analyses performed by the Agency in 1980 (EPA82). The aquifer parameters are similar for all sites evaluated, thus emphasizing the performance capabilities of the host rock. The host rock parameters, including the distance between the repository horizon and the aquifer, hydraulic conductivity, porosity, and vertical gradient, were based on a study of two representative salt sites in the Palo Duro Basin and the Paradox Basin (EPA85) and on the generic analysis.

Geochemical parameters are also necessary to evaluate the transport of radionuclides through geologic media. For each radionuclide in the waste inventory, retardation values are required. These values are dependent on the geologic medium in which the waste is disposed. The retardation values used in the salt site analysis are given in Table 7.5-12.

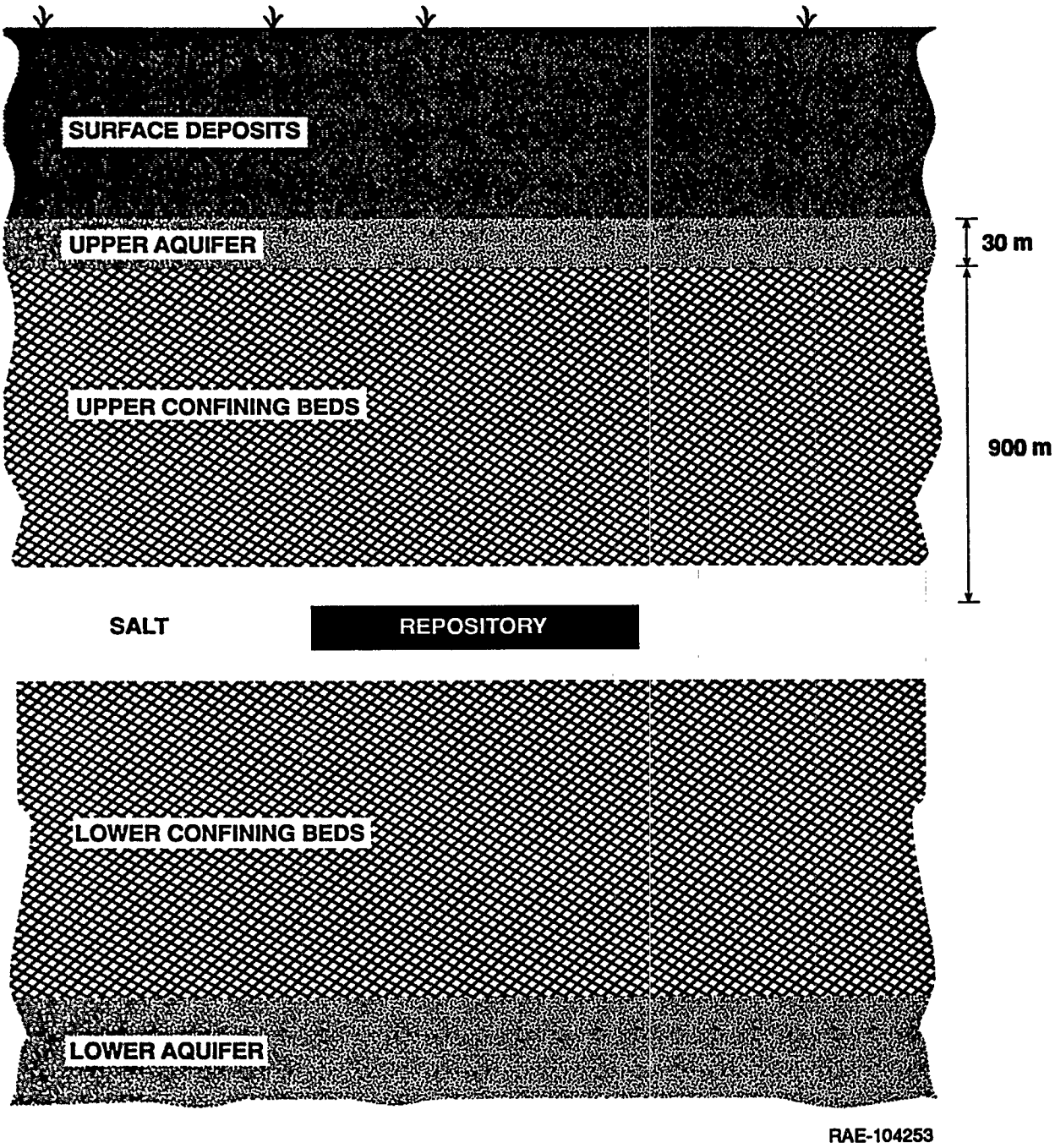

Figure 7.5-39. Cross-sectional structure of the model generic salt repository (not to scale).

Releases from the source were characterized in terms of radionuclide solubility in ground water. The solubilities used for the salt assessments are shown in Table 7.5-13.

A single set of waste form and repository configuration characteristics were assumed. Many of the parameters in the analysis are common to the generic repositories in all media analyzed by the Agency including, for example, the radionuclide inventory and the dimensions and capacity of the underground repository facility. All such parameters are discussed for all sites in Sections 7.5.2 and 7.5.3.

Analyses were conducted to evaluate sensitivities and uncertainties in the parameter values. In the sensitivity studies, single parameters were varied discretely from the base-case values. In the uncertainty analysis, statistical distributions were defined for the key input parameter values and those values were varied in a Monte Carlo analysis. Three key parameters were identified for the sensitivity analysis. The parameters characterize the release from the waste form and the rate of transport through the ground-water system. The specific parameters selected for the analysis are the radionuclide solubilities, the hydraulic conductivity in the vertical transport leg, and the radionuclide retardation factors. While other related parameters could have been included in the sensitivity and uncertainty analyses, those identified represent the key parameters for characterizing the magnitude and timing of the radionuclide releases and the transport through the host rock and aquifer.

The parameter-value ranges for the salt analyses are shown in Table 7.5-14. The ranges encompass the values used in previous Agency assessments. The probability distributions are given for use in the NEFTRAN-S uncertainty analysis. Due to the wide range of values, log-uniform distributions were used for all of the parameters. This is preferable to using uniform distributions since log-uniform distributions cause the median parameter values to be close to the base-case values and are therefore more appropriate for parameters that vary over several orders of magnitude.

7.5.5.3.3 Base Case Results from the Assessment of Generic Salt Sites

Individual doses and ground-water contamination were evaluated for undisturbed conditions only. Under undisturbed conditions, the hydraulic conductivity of salt is essentially zero, resulting in no ground-water flow. Therefore, under undisturbed conditions, there is no radionuclide release, no dose to individuals, and no contamination of ground water.

7.5.5.3.4 Sensitivity and Uncertainty Analyses for the Generic Salt Assessments

At a salt site, releases are controlled by the ground-water flow regime. Therefore, only variations in the hydraulic conductivity were considered. Neither radionuclide solubility nor retardation is of concern. Increasing the hydraulic conductivity from zero to 3.0×10^{-6} m/yr (La89, RAE92), a high value for salt, resulted in a vertical travel time of 30 million years; consequently, no dose or ground-water contamination occurs during the 100,000-year assessment period.

Table 7.5-11. Site parameters used in the risk assessment of bedded salt.

Input Parameter	Early Generic EPA Model[a]	Representative Bedded Salt Sites[b]		Current Model
		Palo Duro Basin	Paradox Basin	
Initial average porosity of backfill in repository	0.2	Not Available	Not Available	0.2
Distance from repository to overlying aquifer (meters)	100	1105	666	900
Hydraulic conductivity of the host rock between the repository and the aquifer, after thermal effects (meters/year)	0	0	0	0
Porosity of the host rock between the repository and aquifer	0.01	n/a	n/a	n/a
Hydraulic gradient between the repository and aquifer	0.1	0.26	0	0.1
Thickness of aquifer (meters)	30	300	18	30
Hydraulic conductivity of aquifer (meters/year)	31.5	1.6	7.6	10
Porosity of the aquifer	0.15	0.05	0.2	0.15
Horizontal gradient in aquifer	0.01	0.005	0.02	0.01
Horizontal distance along the aquifer to the accessible environment (meters)	1600	2000	2000	2000

[a]EPA-520/3-80-006 (EPA82).
[b]EPA-520/1-85-028 (EPA85).

Table 7.5-12. Radionuclide retardation factors for salt.

Element	Range of Retardation Factors[a]		
	Low	"Base Case"	High
Strontium	1	10	100
Cesium	1	10	2,000
Lead	5	20	100
Radium	5	50	500
Actinium[b]	10	20	60
Thorium	300	1,000	5,000
Protactinium[b]	10	20	60
Uranium	10	20	60
Neptunium	10	50	300
Plutonium	10	200	10,000
Americium	300	1,000	5,000
Curium	200	1,000	3,000

[a]From 1983 WISP report (NAS83).
[b]Because values were not given in WISP report, values of uranium were used based on chemical similarities.

Table 7.5-13. Radionuclide solubilities for salt.[a]

Nuclide	Solubility (Ci/m^3)
Ac-227	1.64E+01
Am-241	8.28E-01
Cm-248	1.06E-03
Cs-137	1.19E+01
Np-237	1.67E-04
Pa-231	1.09E-02
Pb-210	1.60E+01
Pu-238	4.08E+00
Pu-239	1.49E-02
Pu-240	5.47E-02
Pu-242	9.51E-04
Ra-226	2.24E-01
Sr-90	1.23E+01
Th-229	4.87E-02
Th-230	4.65E-03
Th-232	2.55E-08
U-233	2.26E-03
U-234	1.46E-03
U-235	5.08E-07
U-236	1.53E-05
U-238	8.01E-08

[a]Based on 1.0E-06 mole/liter (La89).

7.5.5.4 Site Analysis - Tuff

7.5.5.4.1 Introduction

Welded tuff has recently received increased attention as a potential host rock for a high-level waste repository. It is unique among geologic media considered in this analysis in that the repository horizon at the tuff site is assumed to be unsaturated. This is because the generic tuff site is modeled after the unsaturated tuff site in southern Nevada. The welded tuffs that serve as the generic host rock for this analysis consist of airborne volcanic debris fused into a mass with high porosity and low permeability. They appear to have the necessary engineering properties for repository construction. Because the tuff is composed of fragments of porous volcanic rock, the residence time of water moving through it is relatively long and the likely mineral assemblages are expected to provide favorable retardation. Tuff shares with granite and basalt a relatively low occurrence of oil, gas, or valuable minerals that might be exploited by future drilling to any considerable depth. Similarly, the depth of the water table is a deterrent to the drilling of water wells or the development of underlying aquifers.

Two distinctive and important features emerge from the analyses conducted to date by the Agency and Sandia National Laboratories (SAND84-1492) for a repository located above the water table in an unsaturated zone. First, unlike any other medium, upward aqueous flow is improbable as long as the rock remains unsaturated. Second, as long as infiltration at the ground surface is low enough to maintain an unsaturated condition, water in a flow path such as a fault zone or drill hole may preferentially move into the matrix pore space by capillary attraction rather than downward along the flow path (SAND84-1492).

The DOE is currently investigating the area including Yucca Mountain in southern Nevada as a possible candidate site for a high-level waste repository. Other tuff sites may be found but the relative abundance of hydrogeologic data for this location, coupled with the very low precipitation in the region, make it appropriate to use the general site characteristics to define a conceptual model of a repository in unsaturated tuff. Analysis of a tuff repository is a departure from the Agency's original risk analyses of generic repositories, which did not consider this lithology. However, a tuff repository was included in the analyses in 1985. The addition of tuff as a possible isolation medium was presented in 1985 on the basis of its apparent performance and the additional insight it brings to evaluating performance of a repository located in an unsaturated medium.

The preliminary tuff data used in the current analysis are largely derived from studies of the Yucca Mountain area but should be regarded as representative of a generic hypothetical site. It is the Agency's opinion that the parameter values used present a valid but conservative estimate of the performance of a repository in unsaturated tuff.

Section 7.5.5.4.2 discusses the important input parameters that have been used in the Agency's risk analyses for tuff. Since most of the input parameter data can best be presented in the form of tables and figures, there is minimal textual discussion of additional details. The section includes a discussion of the data used to characterize gaseous releases and transport, a release scenario unique to an unsaturated site. Section

Table 7.5-14. Parameter ranges and distributions for salt.

Parameter	Minimum	Maximum	Distribution Type
Solubility (mole/liter)	1.0E-09	1.0E-03	Log Uniform
Retardation factors	(a)	(b)	Log Uniform
Hydraulic Conductivity (m/yr)	0	3.0E-06	Log Uniform

[a]The sensitivity and uncertainty analyses used retardation factors of one, as well as the "low" values from Table 7.5-12.
[b]See Table 7.5-12.

7.5.5.4.3 provides the results of the base-case assessment. Section 7.5.5.4.4 presents the results of the sensitivity and uncertainty analyses.

The conceptual model for the generic tuff site assumes that the waste disposal horizon will be located above the water table in unsaturated rock. Therefore, the potential exists for gaseous transport of radionuclides upward through the unsaturated rock to the surface in addition to the aqueous transport of radionuclides downward to the aquifer. The generic analysis performed in support of the 1985 standard did not consider gaseous releases. It has since been realized that gaseous releases may be significant at an unsaturated site where spent nuclear fuel is disposed. The Agency has investigated gaseous releases from a tuff site (RAE92a) and has found that C-14 (as carbon dioxide) and possibly I-129 are the only radionuclides likely to be released in a gaseous state in significant quantities. C-14 and I-129 are not present in the initial TRU waste inventory and are not generated through the decay of any of the radionuclides in the inventory. Therefore, gaseous transport of these radionuclides from a TRU waste repository is not considered.

7.5.5.4.2 Input Parameters for Tuff

The NEFTRAN-S model for tuff assumes downward flow from the repository through the unsaturated zone to the underlying aquifer. As long as the water infiltration rate is less than the saturated hydraulic conductivity, flow is driven by gravity and a downward gradient of one is assumed. Between the repository and the saturated zone, natural variations in hydrologic properties are simplified to a single set of vertical leg parameters. Potential releases to the accessible environment are modeled through the uppermost aquifer, located about 200 meters below the repository.

Hydraulic conductivity is used in conjunction with Darcy's Law to estimate volumetric flow rates through various components, such as pathways from the repository down through the unsaturated zone to the aquifer and horizontally within the aquifer. For further elaboration on the mathematical equations, one may consult EPA82 and the references cited there. Only Darcian flow has been treated in the analyses and work by DOE at specific sites tends to confirm that this approach is adequate (SAND90a). The porosity is used to convert Darcian flow velocities into average effective fluid velocities in the direction of movement. In particular, the Darcian flow velocity is divided by the volumetric moisture content to obtain an effective fluid velocity. This is used to determine the time of arrival of contaminated ground water at the discharge point to the accessible environment.

Figure 7.5-40 shows the geologic cross section used to define the simplified model used in the Agency's analyses for tuff. Table 7.5-15 shows the geometric and hydrologic input parameters. The radionuclide retardation factors and solubilities are shown in Tables 7.5-16 and 7.5-17, respectively. The generic assessment of TRU waste disposal in tuff use the waste form and repository parameters presented in Sections 7.5.2 and 7.5.3.

The parameter values used in the risk assessment were varied in a sensitivity study. The parameters selected for the sensitivity and uncertainty analyses are listed in Table 7.5-18, along with the ranges and distribution types. As explained earlier, log

uniform distributions were selected for the radionuclide solubilities, the retardation factors, and the infiltration rate. The infiltration rate is analogous to the vertical hydraulic conductivity in the saturated site assessments because it determines the travel time from the repository to the aquifer.

7.5.5.4.3 Base-Case Results from the Assessment of the Generic Tuff Site

Assuming the base-case parameter values given in Table 7.5-15, the pore velocity in the vertical transport leg is approximately 0.013 meters/year. With a distance of 200 meters between the repository and the aquifer, the unretarded travel time in the vertical leg is approximately 15,600 years. The lowest base-case retardation factor is 40, for uranium, actinium and protactinium. These radionuclides would be the first to reach the aquifer, but only after 624,000 years. Thus, there are no doses or radionuclide concentrations 2000 meters down gradient during the 100,000-year assessment period.

7.5.5.4.4 Sensitivity and Uncertainty Analyses for the Generic Tuff Assessments

Sensitivity analyses were conducted, varying solubility, infiltration rates, and retardation. No radionuclides travel 2000 meters down gradient in the 100,000-year assessment period unless retardation factors are reduced from the base-case values.

The effect on individual dose of reducing retardation factors is shown in Figure 7.5-41. Lowering the retardation factors results in a dose of 35 to 45 mrem/yr from the arrival of radionuclides at year 76,000. The dose results mainly from U-233 (60%), Pa-231, Ac-227, and U-234. With no retardation, dose results as early as year 15,600. The dose rate increases sharply to 7700 mrem/yr. Americium, neptunium and curium are depleted early. After year 70,000, the dose rate drops as the plutonium radionuclides are depleted.

Using the same assumptions and analysis but with an aquifer 30 meters thick, which is the same as the other generic site analyses assume, lowering the retardation factor values results in a dose rate of 2800 to 3600 mrem/yr for the arrival of radionuclides at year 76,000. The dose results mainly from U-233 (60%), Pa-231, Ac-227, and U-234. When no retardation is assumed, doses begin occurring as early as year 15,600; dose increases sharply to 616 rem/yr. As before, americium, neptunium, and curium are depleted early and, after year 70,000, the dose rate drops as the plutonium radionuclides are depleted.

A probabilistic assessment of peak dose rate over 10,000 years was conducted using NEFTRAN-S and the parameter ranges shown in Table 7.5-18. Two analyses were conducted: one using the low retardation values (Table 7.5-16) as a minimum on the range of retardation and a second using a value of one as the minimum retardation value. Using the Monte Carlo sampling routine of the NEFTRAN-S code, peak dose rates were calculated for the 10,000-year period following disposal. The results of the low minimum retardation analysis are shown by the histogram in Figure 7.5-42. Given the parameter uncertainties, there is a 0.9 probability of zero dose during the 10,000-year assessment period. The results of the analysis using a minimum retardation value of one are shown by the histogram in Figure 7.5-43. Given the parameter uncertainties, there is a 0.62

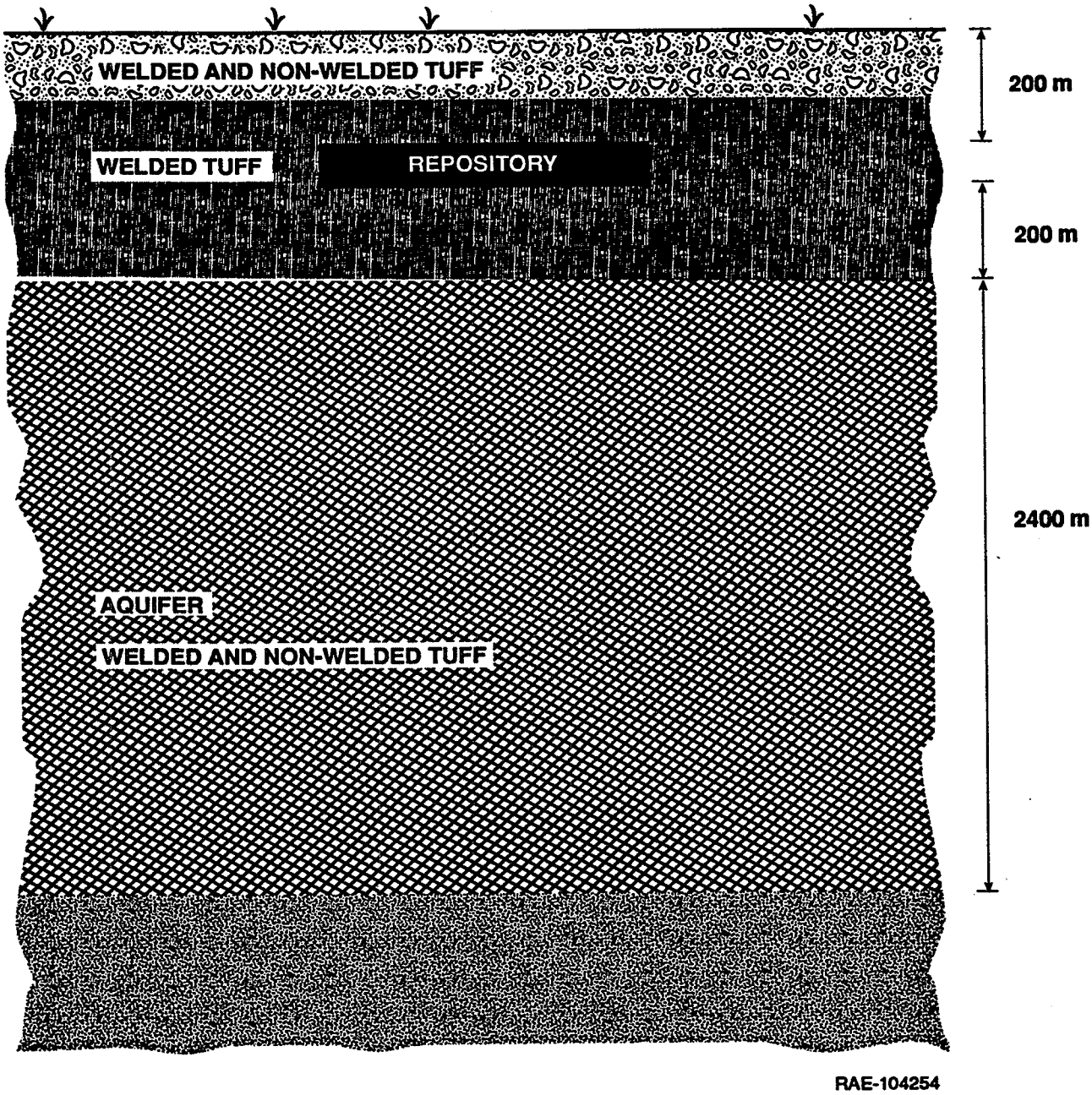

Figure 7.5-40. Cross-sectional structure of the model generic tuff repository (not to scale).

Table 7.5-15. Site parameters used in the assessment of tuff.

Input Parameter	Early Generic EPA Model[a]	Current Model
Average porosity of backfill in repository	0.2	0.2
Distance from repository to underlying aquifer (meters)	100	200
Saturated hydraulic conductivity of the host rock between the repository and the aquifer, after thermal effects (meters/year)	0.001	0.004
Infiltration rate (m/yr)	n/a	0.0005
Porosity of the host rock between the repository and aquifer	0.10	0.06
Unsaturated hydraulic gradient between the repository and aquifer	1	1
Thickness of aquifer (meters)	1,000	2,400
Hydraulic conductivity of aquifer (meters/year)	30	200
Porosity of the aquifer	0.002	0.002
Horizontal gradient in aquifer	0.00034	0.0004
Horizontal distance along the aquifer to the accessible environment (meters)	2,000	2,000
Distance from repository to ground surface (meters)	---	200

[a]EPA-520/3-85-028 (EPA85).

Table 7.5-16. Radionuclide retardation distributions for tuff.

Element	Range of Retardation Factors[a]		
	Low	"Base Case"	High
Strontium	20	200	10,000
Cesium	60	500	10,000
Lead	20	50	500
Radium	50	500	5,000
Actinium[b]	5	40	200
Thorium	500	5,000	10,000
Protactinum[b]	5	40	200
Uranium	5	40	200
Neptunium	10	100	500
Plutonium	50	200	5,000
Americium	300	1,000	50,000
Curium	100	500	10,000

[a]From 1983 WISP report.
[b]Values not given in WISP report; values of uranium were used based on chemical similarities.

Table 7.5-17. Radionuclide solubilities for tuff.

Nuclide	Solubility (Ci/m^3)
Ac-227	1.64E+01
Am-241	8.28E-01
Cm-248	1.06E-03
Cs-137	1.19E+01
Np-237	1.67E-04
Pa-231	1.09E-02
Pb-210	1.60E+01
Pu-238	4.08E+00
Pu-239	1.49E-02
Pu-240	5.47E-02
Pu-242	9.51E-04
Ra-226	2.24E-01
Sr-90	1.23E+01
Th-229	4.87E-02
Th-230	4.65E-03
Th-232	2.55E-08
U-233	2.26E-03
U-234	1.46E-03
U-235	5.08E-07
U-236	1.53E-05
U-238	8.01E-08

[a]Based on 1.0E-06 mole/liter (La89).

Table 7.5-18. Parameter ranges and distributions for tuff.

Parameter	Minimum	Maximum	Distribution Type
Solubility	1.0E-09	1.0E-03	Log Uniform
Infiltration rate (mm/yr)	0.1	4.0	Log Uniform
Retardation factors	(a)	(b)	Log Uniform

[a]The sensitivity and uncertainty analyses used retardation factors of one, as well as the "low" values from Table 7.5-16.
[b]See Table 7.5-16.

probability of zero dose during the 10,000-year assessment period. However, these results, which include no retardation, represent bounding conditions for repository performance. They are included to represent the importance of retardation in EPA's model of generic repository performance.

Figures 7.5-44 and 7.5-45 show the effects of varying retardation on concentrations of Ra-226 and total alpha-emitting radionuclides in the ground water. As expected, reducing retardation results in earlier and greater concentrations. Reducing retardation also results in earlier and greater doses from beta- and gamma-emitting radionuclides (Ac-227 and Pb-210) in the ground water (Figure 7.5-46).

7.5.6 Comparison of Media Results

The results of the deterministic calculations for the four repository media are compared in this section. Radionuclide concentrations and dose rates were calculated for a point 2,000 meters down gradient from the repository. Individual dose rates were based on the consumption of two liters per day of ground water. Using the base-case parameter values, the undisturbed ground-water-flow scenario shows no individual doses or ground water contamination at any of the sites during the first 10,000 years after waste disposal. In addition, the bedded salt and tuff sites show no doses or ground-water contamination during the first 100,000 years.

The dose rates at the basalt and granite sites are zero until about year 50,000. As shown in Figure 7.5-47 the individual dose rates at the basalt and granite sites are similar. Figure 7.5-48 shows the Ra-226 concentration in the ground water at the basalt and granite sites. The Ra-226 concentration rises more slowly at the granite site, but the concentration levels are comparable. Figure 7.5-49 shows the total concentrations of alpha-emitting radionuclides. Like the individual dose rates, the concentrations at the basalt and granite sites are very similar. The results for the beta and gamma-emitting radionuclides are shown in Figure 7.5-50. The doses first appear at about year 50,000 and are nearly the same.

7.6 UNCERTAINTY IN THE RISK ASSESSMENT

The generic assessment presented here encompasses many uncertainties which are due to a number of factors such as the following:

- The long time frame over which predictions are needed;
- The simplified nature of the models in comparison with the real physical situation; and
- The generic nature of the modeling.

The purpose of generic risk assessments is to make rough approximations of the capabilities of geologic disposal media to contain radioactive waste. Therefore, despite these uncertainties, the Agency believes that the estimates generated herein provide an adequate technical basis for the associated regulations.

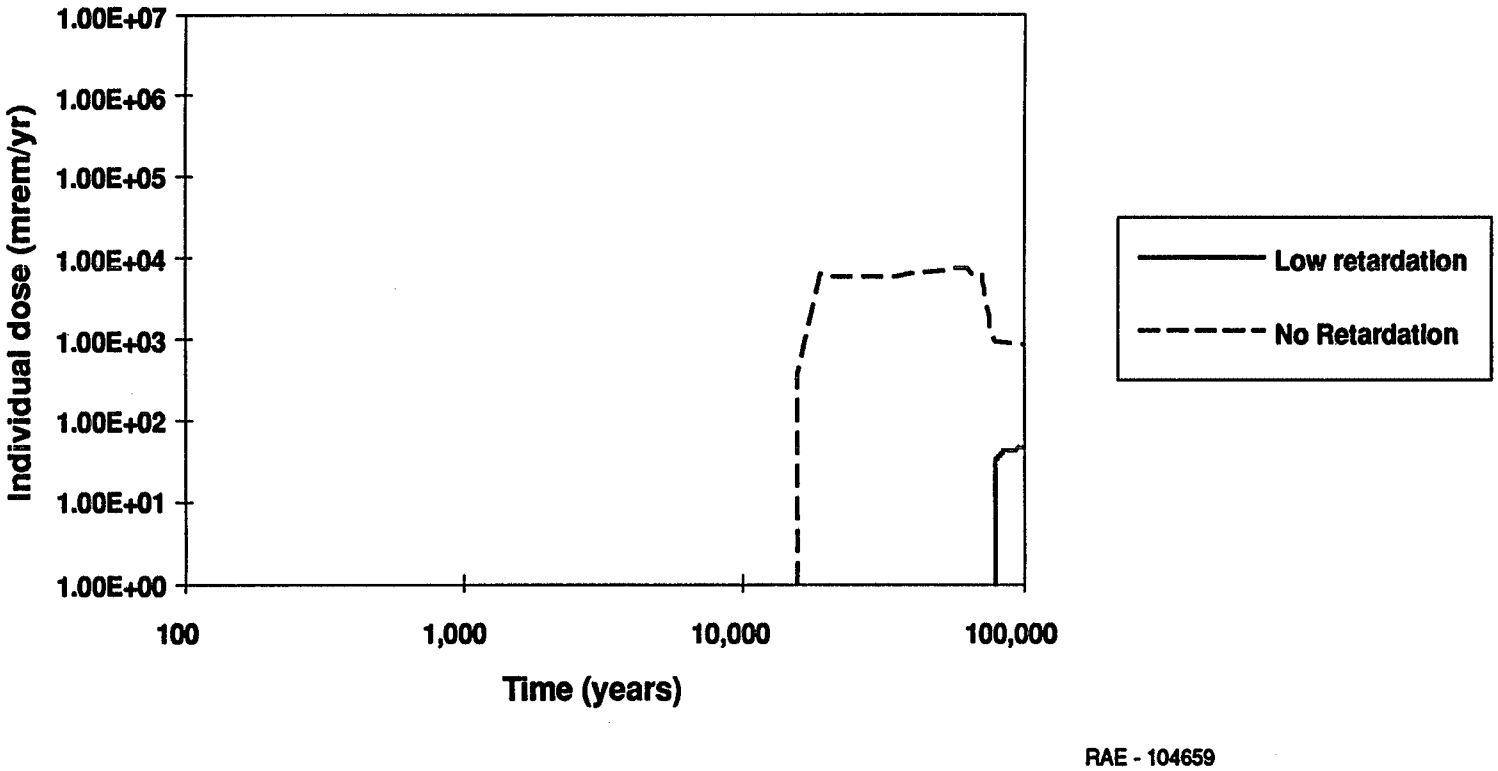

Figure7.5-41. Sensitivity of dose to retardation - tuff.

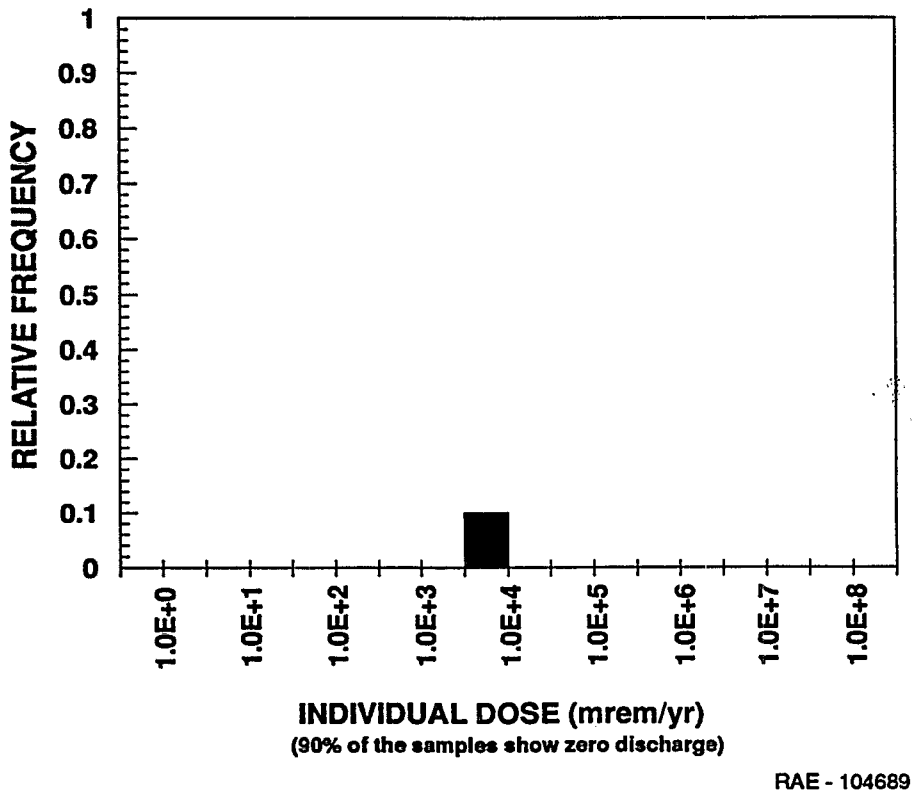

Figure 7.5-42. Distribution of individual dose due to parameter uncertainty (low minimum retardation) - tuff.

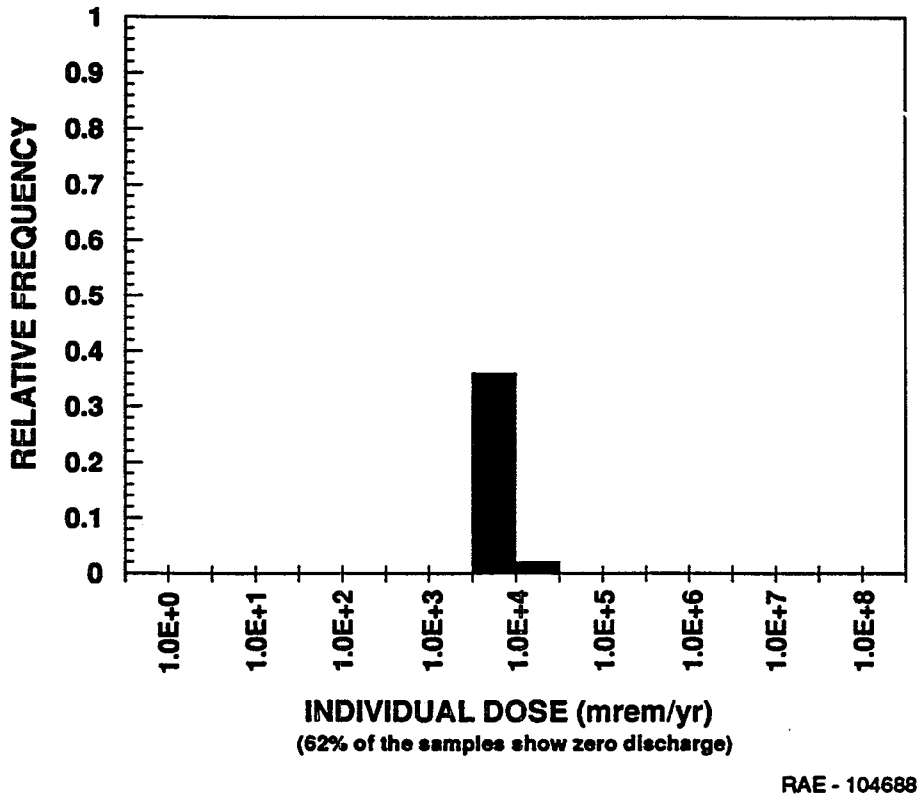

Figure 7.5-43. Distribution of individual dose due to parameter value uncertainty (minimum retardation value of one) - tuff.

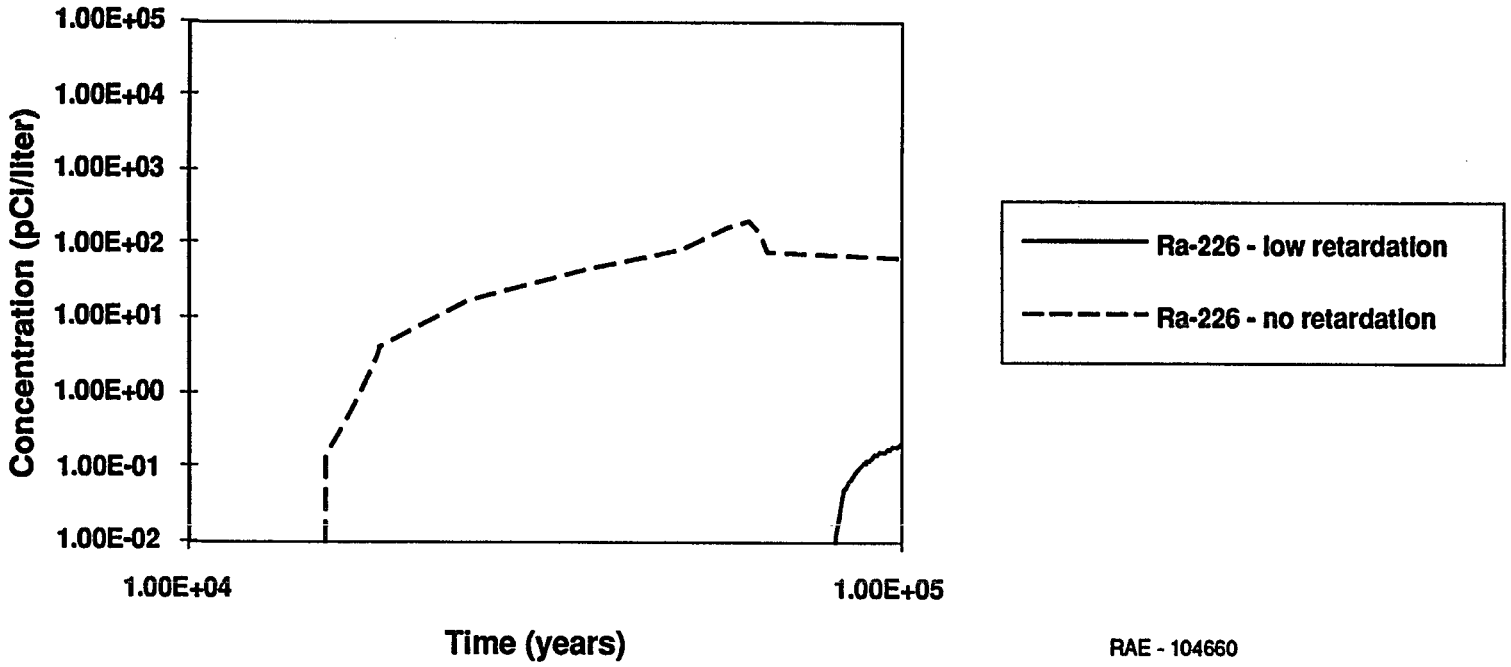

Figure 7.5-44. Sensitivity of Ra-226 groundwater concentrations to retardation - tuff.

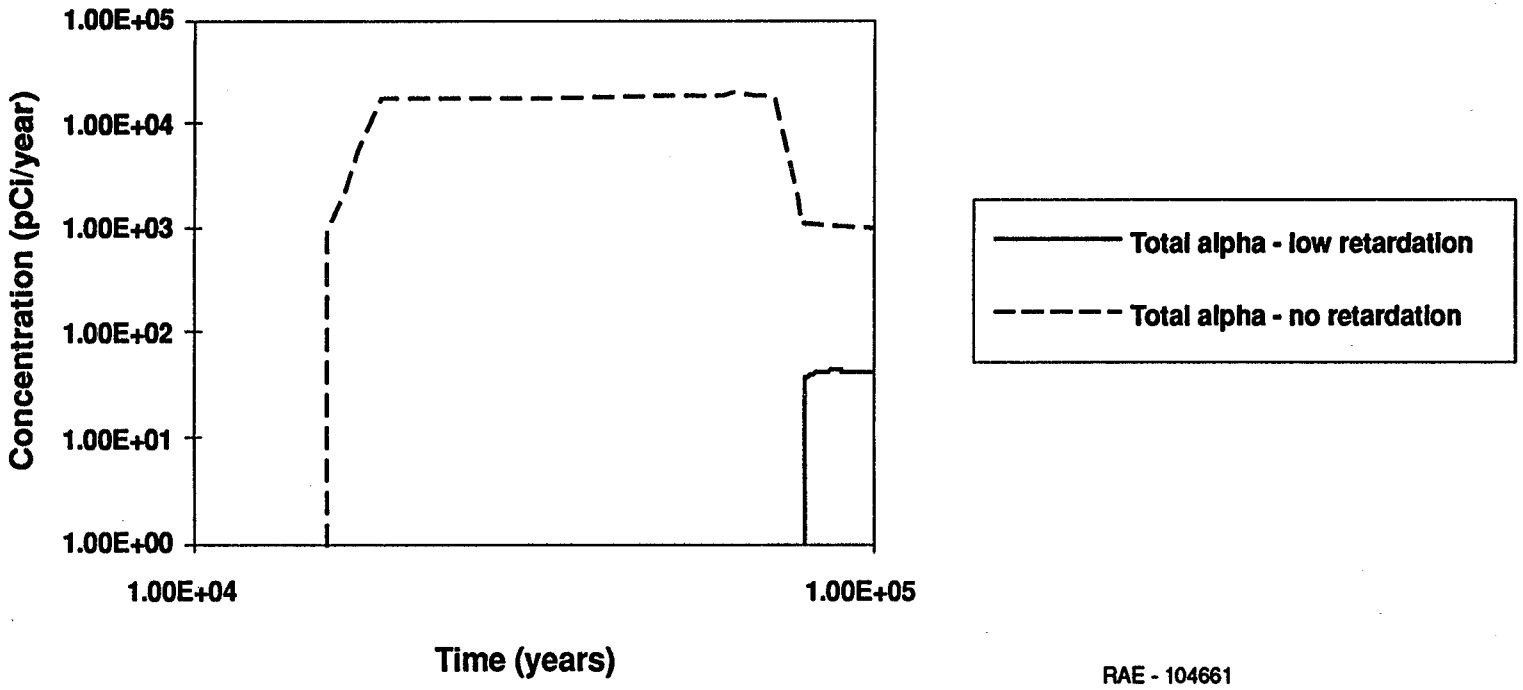

Figure7.5-45. Sensitivity of total alpha groundwater concentrations to retardation - tuff.

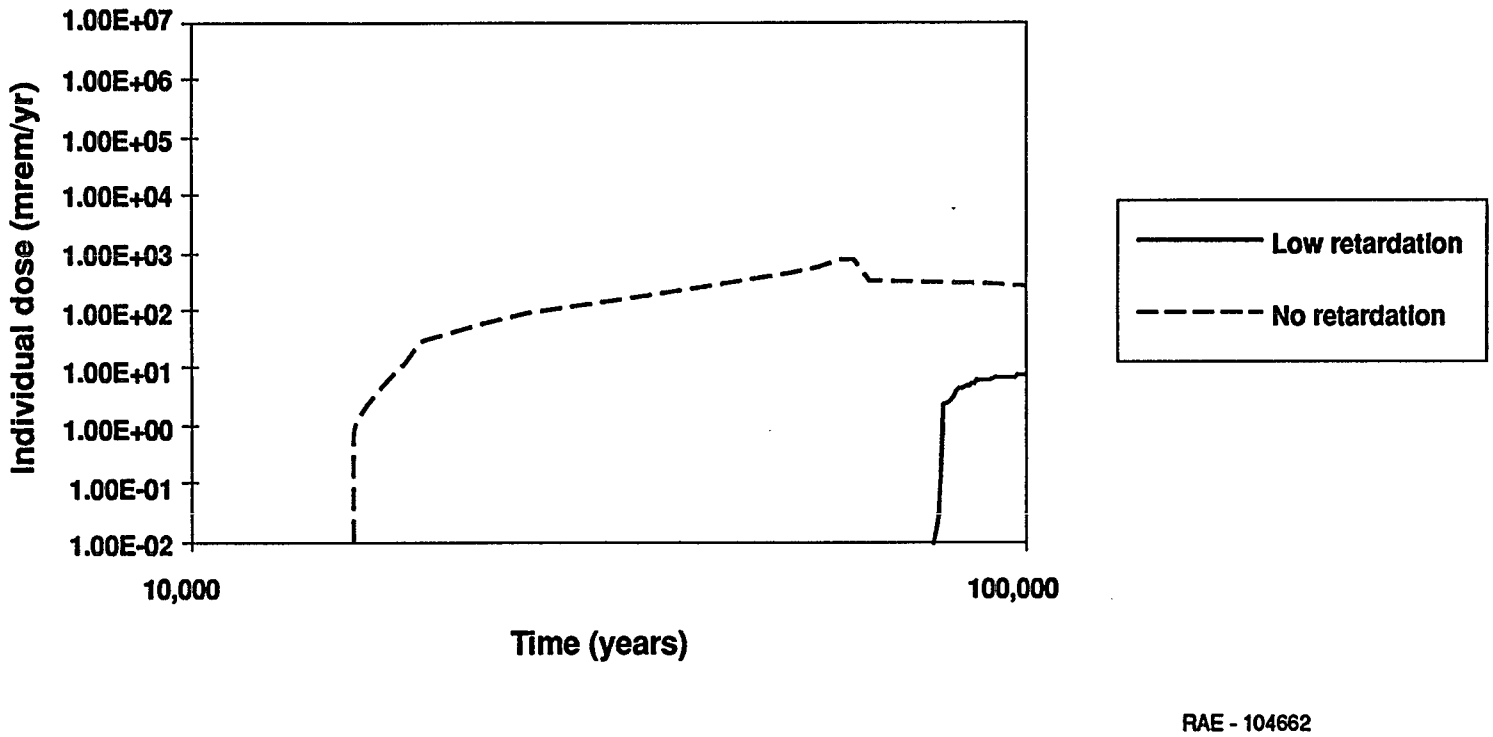

Figure7.5-46. Sensitivity of total beta groundwater dose to retardation - tuff.

In order to lend perspective to the uncertainties in the generic calculations, the Agency has proceeded as follows. First, in estimating parameter values or in choosing models to represent various processes, an attempt has been made to conservatively predict factors that contribute to risks from the repository. This is the same philosophy that was adopted in risk assessments for the proposed rule, although the degree of over-estimation has been reduced in response to recommendations by the Agency's Science Advisory Board. Again, a conservative approach was taken in the selection of many parameter values but sufficient site-specific work has been done by previous studies to provide a high degree of certainty for some parameter values.

Second, use has been made of sensitivity analyses in order to understand how much the results of the assessment change with variations in certain model components or parameters.

Third, in cases where it has been difficult to model the characteristics of a site or a process on a generic basis, several choices of parameter values have been made to understand the range of potential risk results. In the generic assessments, alternative cases were used to model ground-water flow.

Two parameters identified as having a high degree of uncertainty are solubility and retardation. Previously used values for these parameters vary over several orders of magnitude. An area of relatively high uncertainty is the effect of retardation on radionuclide migration. Two alternate cases were used to examine uncertainties in retardation. Both cases use the values from the WISP report (NAS83) for retardation factors. In addition to the recommended nominal values, the high and low sets of values from NAS93 and a retardation factor of one were used in the assessment.

The radionuclide solubility is included as a sensitivity parameter in the TRU waste assessments presented. Neither alternate case had an effect on releases into ground water because the TRU nuclides did not have low enough retardation to travel the flow path in 10,000 years.

It is important to distinguish between the type of uncertainty included in the generic analysis reported here and the uncertainties that would remain with real sites when they are characterized and modeled in connection with the decision on where to put a repository. Many of the uncertainties associated with generic assessments and included here might better be characterized as variabilities. Among actual specific sites there might be a wide variation in the value of the parameter (property) in question. The attempt was made in these generic risk assessments to incorporate such variations, which correspond to an uncertainty in the final results, to determine how well they characterize the performance of the repository.

An assessment of an actual site would include additional uncertainties associated with data collection, site complexity, and difference of opinion about a specific site's characteristics. Section 7.6.1 discusses uncertainties associated with site-specific risk assessments and methods for evaluating these uncertainties. Section 7.6.2 discusses the use of expert judgement in addressing uncertainties in performance assessment calculations.

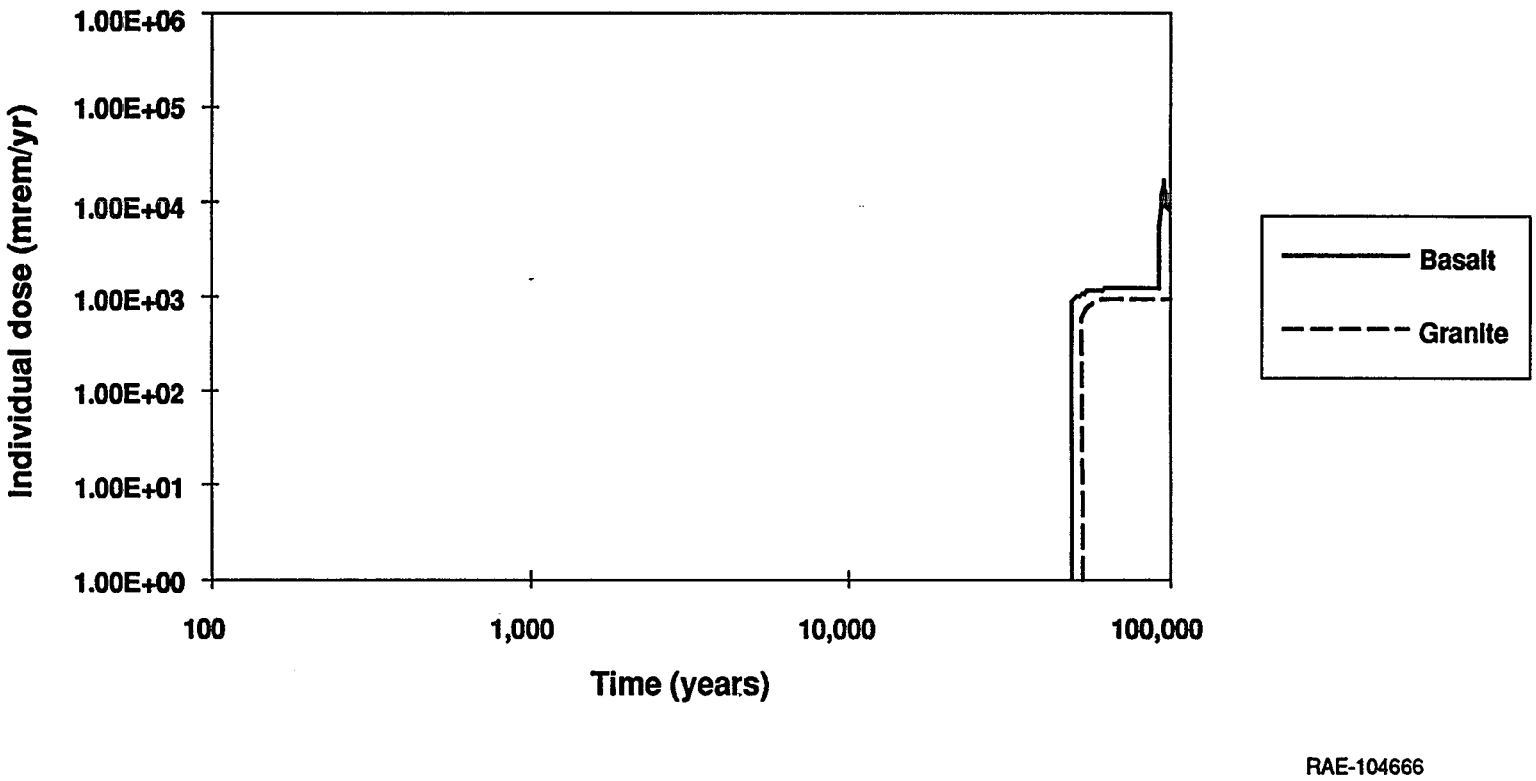

Figure 7.5-47. Comparison of media - individual dose.

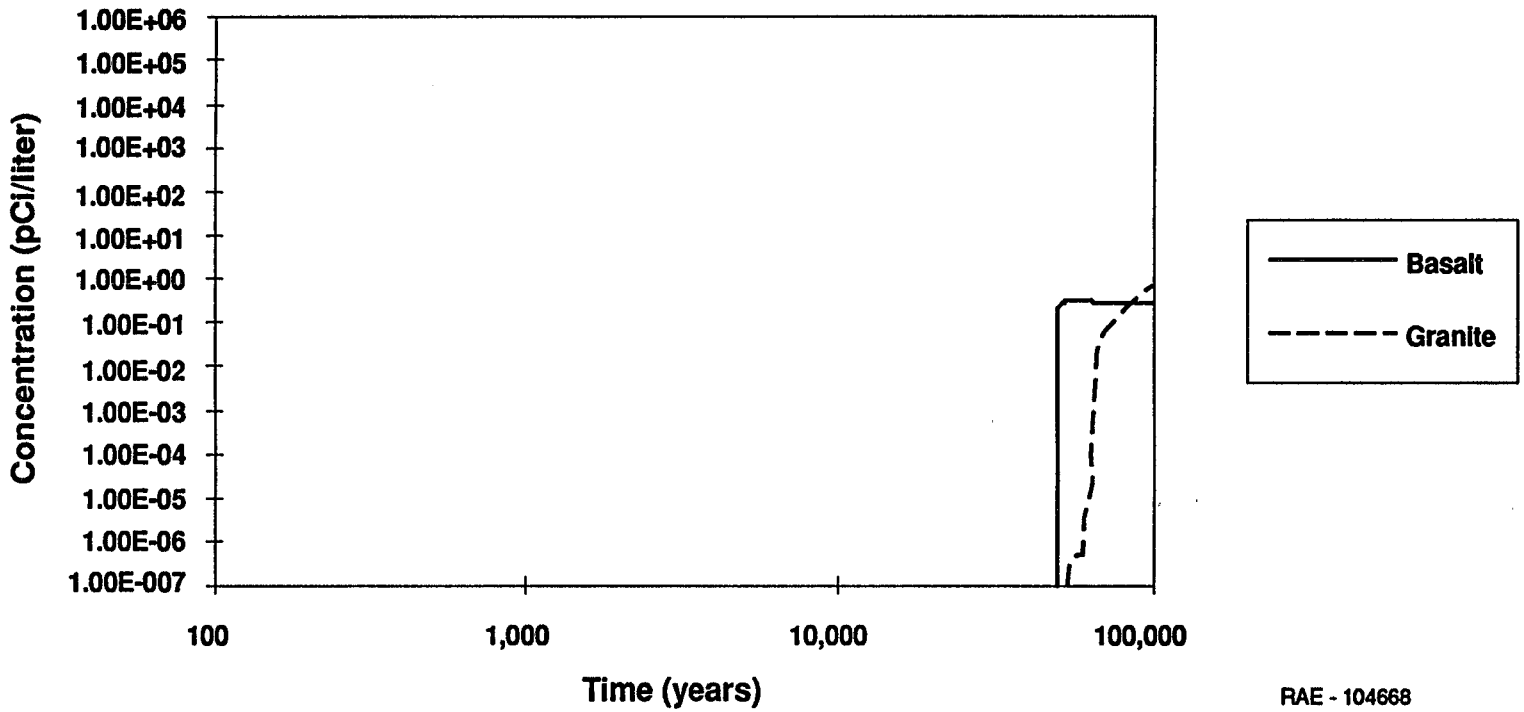

Figure 7.5-48. Comparison of media - Ra-226 concentrations.

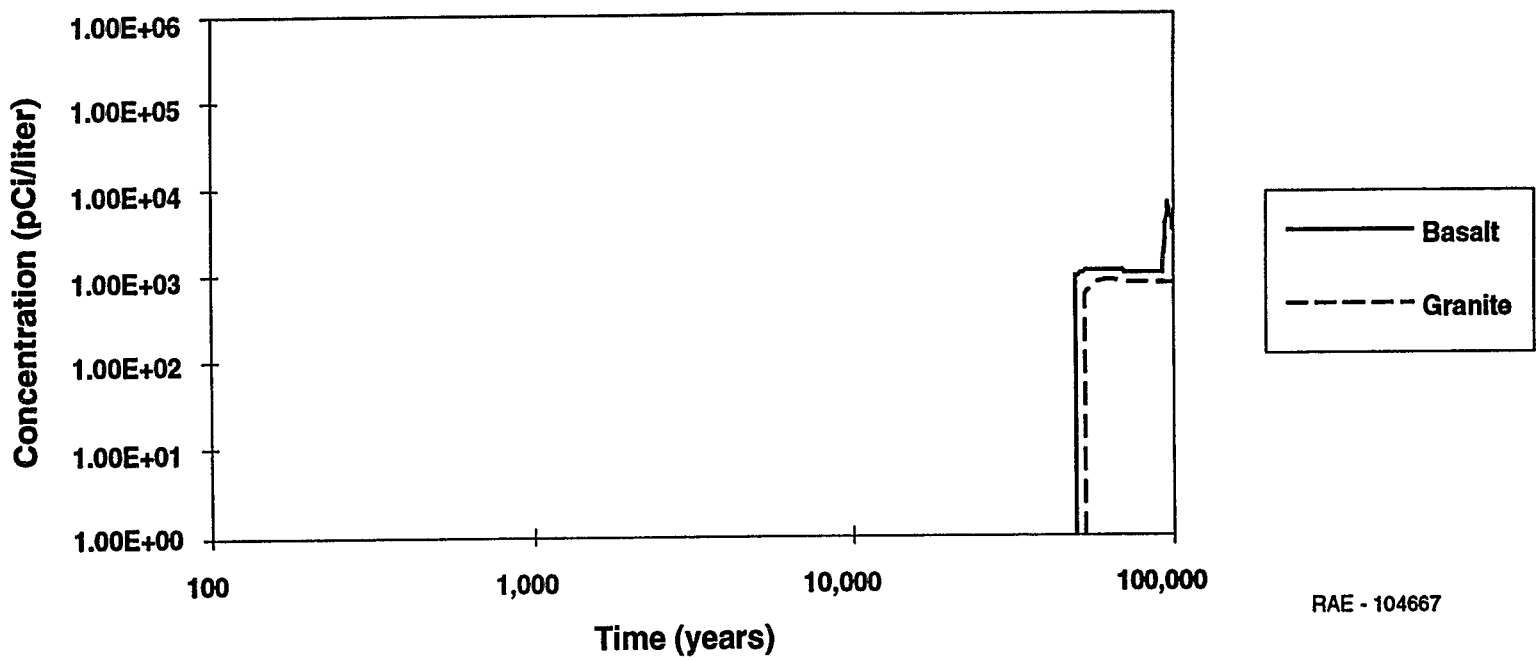

Figure 7.5-49. Comparison of media - concentration of alpha-emitting radionuclides.

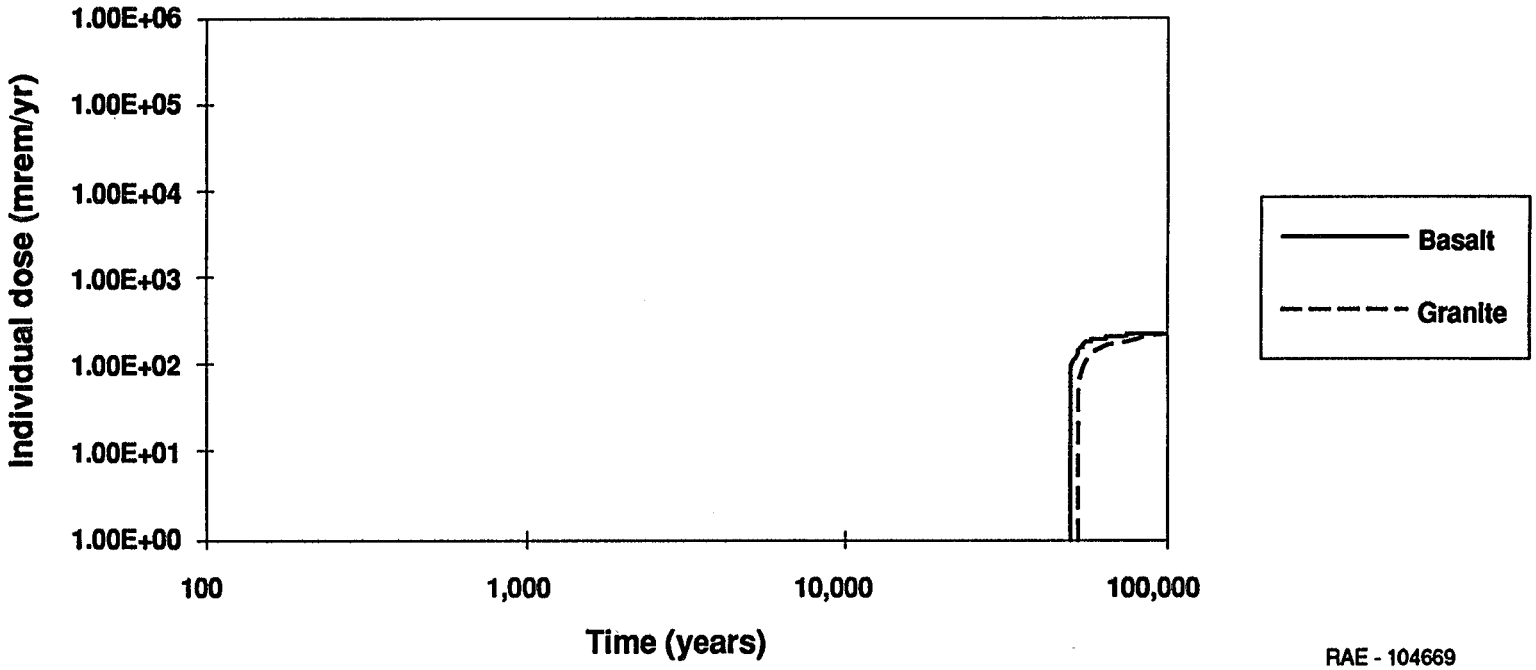

Figure 7.5-50. Comparison of media - dose from beta-emitting radionuclides.

7.6.1 Evaluation of Uncertainties in Site-Specific Risk Assessments

Sources of Uncertainty - Many investigators have grouped uncertainties associated with the assessment of geologic disposal into three categories: scenario uncertainty, model uncertainty, and parameter value or data uncertainty, as summarized in Figure 7.6-1 (Cranwell and Helton, 1981; Hunter et al., 1986; Davis et al., 1990). The following summarizes the description of these categories of uncertainties.

Scenario uncertainty arises from the subjectivity inherent in predicting future conditions. Davis, et al., (Da90) define a scenario as a "combination of anticipated or unanticipated events and processes, either natural or human induced...[that]...could result in the release of radionuclides from the underground facility, their migration through the geosphere and biosphere, and their eventual exposure to humans." Cranwell, et al., (Cr90) define a procedure for scenario development and selection consisting of the following six steps:

1. Identify all possible events and processes relevant to the long-term performance of a geologic waste disposal facility;

2. Group similar events and processes to create a smaller, more manageable set;

3. Screen the set of events and processes based on established criteria, such as a very low probability of occurrence or negligible impact on waste isolation;

4. Systematically combine the remaining events and processes into scenarios, the combined set of scenarios being mutually exclusive and collectively exhaustive;

5. Screen the set of scenarios based on established criteria, such as a very low probability of occurrence or negligible impact on waste isolation; and,

6. Select the set of scenarios that will be used in evaluating repository performance.

Uncertainty related to scenario development has been categorized in several different ways. It is widely recognized that uncertainties will arise with respect to the completeness of the set of scenarios, since it is unrealistic to expect that every possible event or process could be identified in the first step of the scenario development process. Uncertainties can also arise in the subjective screening of the set of events and processes and the set of scenarios, Steps 3 and 5 in the scenario development procedure. Potentially important events and processes or scenarios could be screened out if initial estimates of probabilities of occurrence or consequences are not conservative, i.e., the credibility or consequences of scenarios are underestimated.

Uncertainties also stem from model development and implementation. Models of the waste isolation system must be developed in order to rigorously evaluate each scenario in terms of its probability of occurrence and the resulting consequence (its impact

SOURCES OF UNCERTAINTIES

Scenario Development/Screening

- Completeness
- Unconservative Initial Screening

Modeling (Conceptual, Mathematical, Computational)

- Lack of Process Knowledge
- Oversimplification of System Processes and Dynamics
- Computational Limitations
- Coding Error

Parameter Value/Data

- Quality of Existing Data
- Representativeness of Existing Data
- Selection of Representative Values

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Figure 7.6-1. Sources of Uncertainties.

on waste isolation). Four types of models must be developed: conceptual, mathematical, numerical, and computer.

A conceptual model describes the system in terms of the processes taking place, the variables related to the identified processes, and the temporal and spatial variations in the processes (Davis et al., 1990). The inherent complexity of natural and engineered systems requires that assumptions be made in developing conceptual models. It is likely that processes operating within system components will not be completely understood. Processes may vary within a system component (spatially) at any given time and may vary over time (temporally). Also, the interactions among system components and the processes operating within and among the components are complicated and must be simplified. Such simplifying assumptions result in uncertainties related to the conceptual models.

In order to quantitatively evaluate natural and engineered systems, mathematical models must be developed from the conceptual models. Also, numerical models must be developed as real solutions to the mathematical models. Uncertainties associated with the development of mathematical and numerical models include those associated with insufficient knowledge of the processes operating within the system, insufficient knowledge with respect to temporal and spatial dependencies of processes operating within the system, and the limitations inherent in attempting to represent complex and interdependent system processes by mathematical expressions.

Mathematical models are implemented through the development of computer models. Sources of uncertainty associated with computer codes include coding errors, computational limitations, and user error. Computational errors can be caused, for example, by truncation errors, discretization error, inappropriate convergence and stability error. Another potential source of computational error is the use of numerical algorithms with data beyond the required range for a particular algorithm.

Uncertainties associated with parameter-value and data sets can result from measurement error and from the misinterpretation of the collected data. Also, insufficient knowledge of the system can lead to data uncertainty. The raw data must be reduced to a form suitable for model input. Lack of representativeness, due to unknown spatial variations, and invalid assumptions about the system can lead to the improper selection of the input data sets.

The preceding paragraphs discuss various sources of uncertainties that may arise in conducting performance assessments of a geologic waste disposal system. It is useful to categorize uncertainties according to similarities as well as by source. There are two general categories of uncertainty: random uncertainty and knowledge uncertainty (Wu91). Random, or stochastic, uncertainty results from stochastic variability of some random variable. Knowledge uncertainty results from imperfect knowledge about some fixed value. Wu, et al., state that the essential difference between random uncertainty and knowledge uncertainty is that knowledge uncertainty may be reduced by increased data sampling or experimentation whereas random uncertainty will not be reduced. Random, or stochastic, uncertainty is often referred to as Type 1 uncertainty, and

knowledge uncertainty is often referred to as Type 2 uncertainty (Hofer and Hoffman, 1987).

Uncertainty Evaluation Methods - Each source of uncertainty (scenario uncertainty, model uncertainty, and parameter-value and data uncertainty) should be addressed in an appropriate manner. The intent is to minimize or eliminate uncertainties where possible and to define or quantify where they are unavoidable. The following discussion outlines techniques proposed and employed for evaluating uncertainties in repository performance assessments.

Uncertainties associated with scenario development and model development are generally subjective. Scenario uncertainties are best addressed through the implementation of defensible scenario development methodologies that are well structured and documented. Formalized expert judgement and peer review processes also aid in the minimization and quantification of scenario uncertainties. Even though uncertainties are unavoidable in predicting future conditions, quantification allows the impacts of scenario uncertainties to be evaluated.

Uncertainties associated with model development are addressed in various ways. Uncertainties associated with conceptual models, developed subjectively through the interpretation of data and the hypothesis of system processes, can be addressed similarly to scenario uncertainties. Expert judgement and peer review are primary methods for minimizing and quantifying conceptual model uncertainties. Formal expert judgement processes are discussed in detail in Section 7.6.2. Expert judgement would also play an important role in developing the corresponding mathematical models. In addition, mathematical model uncertainty can be evaluated and minimized through validation exercises. Computational model uncertainty can be addressed through verification and benchmarking exercises. Both mathematical and computational models would be evaluated through peer review. If more than one plausible alternative model has been identified, the sensitivity of results to alternative models should be evaluated.

Parameter-value and data uncertainties are generally evaluated in a more rigorous and quantitative manner than either scenario uncertainties or model uncertainties, since parameter-value and data uncertainties are more readily quantifiable, although expert judgement plays an important role in developing model parameter value input. Evaluating data and parameter-value uncertainty consists of two fundamental steps. First, the uncertainties associated with each parameter are defined quantitatively. Data and parameter-value uncertainties consist of both stochastic (Type 1) and knowledge (Type 2) uncertainties. Deterministic assessments can be made by using single parameter values as input, such as mean or bounding parameter values. But the results of single deterministic calculations are limited in that the uncertainties in the input parameters will not be reflected in the output. Therefore, value ranges are developed for parameters for use in calculations instead of single values. The value ranges are often presented quantitatively in the form of probability distribution functions. Probability distribution functions, developed through data analysis and expert judgement, allow the uncertainties in parameter values to be incorporated into analyses. Typical forms of probability distribution functions are shown in Figure 7.6-2.

Next, the propagation of parameter value uncertainties through system models is evaluated. The following methods may be employed to evaluate the propagation of data and parameter-value uncertainties.

One well-known and often-used method for evaluating parameter uncertainties is the Monte Carlo sampling technique. Commonly, mathematical models and associated models are deterministic, allowing only single, discrete values to be used for input parameters. Deterministic models usually produce single, discrete values for each output parameter value, with no indication of the level of confidence in the output given the uncertainties about the input.

The Monte Carlo technique allows uncertainty in parameter values to be directly taken into account and reflected in the calculated model output. For each input parameter, a sample set is formed consisting of the possible values for the variable. Most likely, the sample set will be in the form of a probability distribution function where the probability of the parameter having any one value is given. If the value of a parameter were known with certainty, the resulting probability distribution function would be a discrete single-value function. More likely, a particular parameter value is random or unknown with the possible values spread over some range. The uncertainty in the parameter value is reflected in the probability distribution function.

The Monte Carlo technique is based on the iterative recalculation of a deterministic model. Instead of a single deterministic calculation of model output, the model is run a large number of times, or iterations. The number of iterations is generally dependent upon the complexity of the model. Prior to an iteration, an input data set, or input vector, is created. The input vector has N dimensions, where N is equal to the number of model input variables. To create an input vector, a value for each input parameter is selected based on the probability distribution function for that input parameter. Using the Monte Carlo technique for each input parameter, values with a higher relative probability of occurring will be selected and used as input to the model more often than values having a lower relative probability of occurring. The use of higher probability parameter values more often as input to the deterministic model will be reflected in the output sample set. The value of the model output parameter is then calculated based on the input vector. Thus, if there are N iterations, there will be N input vectors and N calculated output values in the output parameter set. The sample statistics for the output parameter, such as the mean and the various percentiles, are then calculated. The relationship between the Monte Carlo sampling of input variables and the distribution of output values is illustrated in Figure 7.6-3.

A critical aspect of the Monte Carlo technique is the input parameter sampling scheme employed. The goal is to have the set of N selected values for a particular input parameter reflect closely, in a statistical sense, the uncertainty in that particular input parameter. Several sampling schemes have been developed. Random sampling, as the name implies, involves the selection of values for a particular parameter at random within the predefined probability function for the parameter. A large number of iterations is necessary to ensure that the selected values adequately represent the parameter. Sample representativeness is enhanced by using a more structured sampling scheme. For example, stratified sampling involves the systematic partitioning of the range of values for

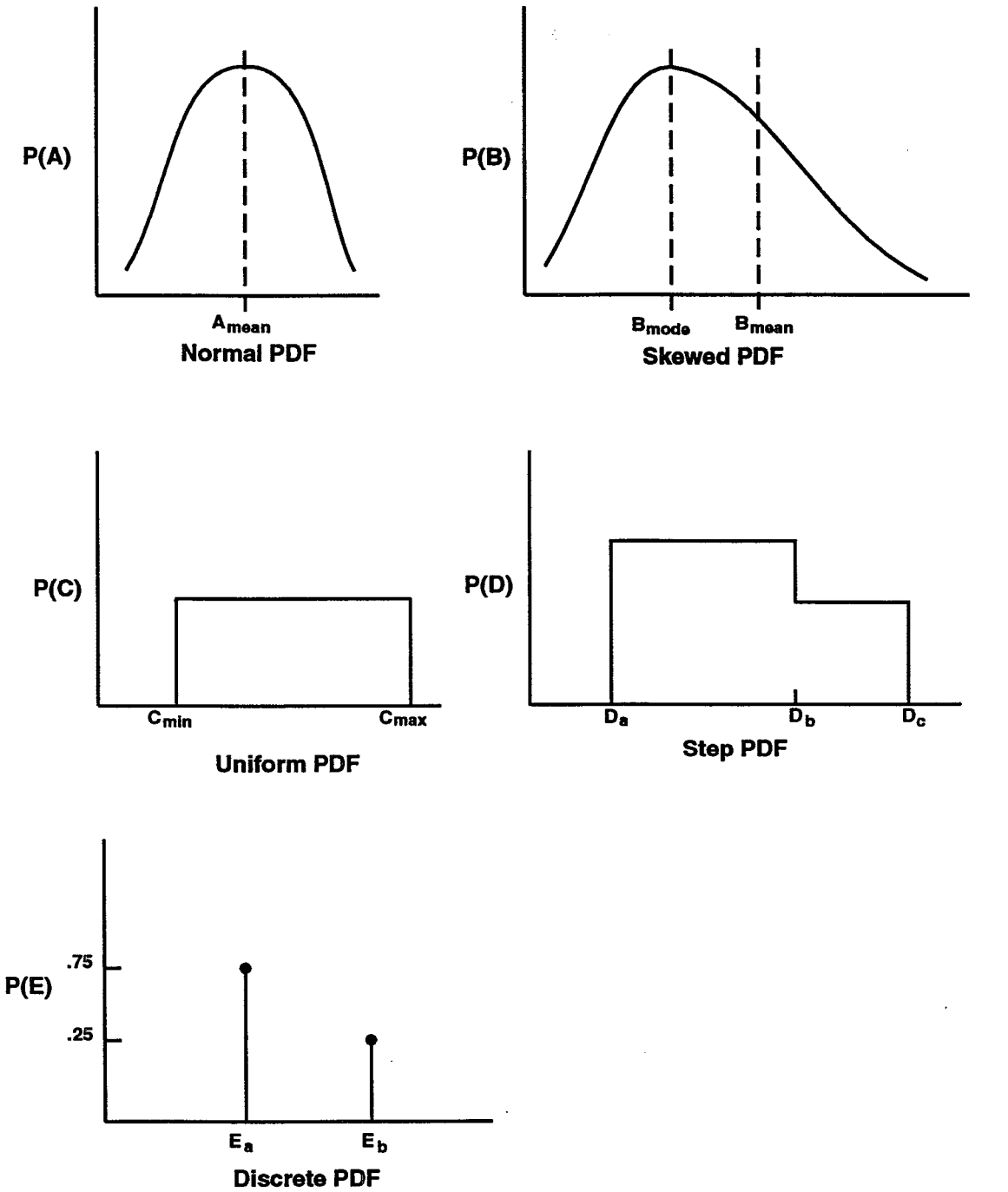


Figure 7.6-2. Typical hypothetical parameter probability density functions (PDFs).

a particular parameter into some number of strata. Samples are then drawn from each stratum, ensuring that the entire range of values is represented. Latin hypercube sampling is a special case of stratified sampling where the range of values for a particular parameter is partitioned into a number of cells equal to the number of iterations, each cell having equal probability. One sample is then drawn from each cell.

The response surface methodology is another technique for evaluating the propagation of uncertainties in risk assessments. The response surface methodology involves the evaluation of uncertainties through the simplification of the deterministic model. The aim is to replace the complex mathematical system model by a relatively simple linear or nonlinear analytical function that is dependent on only a subset of the original input variables.

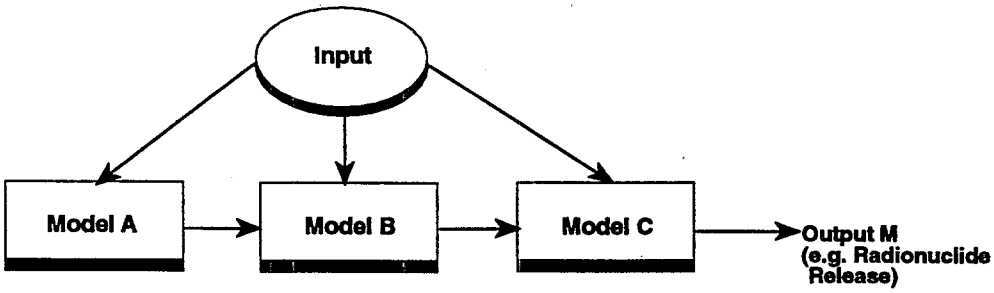
Some number of input parameter sets, or input vectors, are defined and the original complex model is then evaluated deterministically for each of the input vectors. The simplified analytical function is derived through a regression analysis technique such as the least squares method. The simplified function is then used to evaluate the correlation of uncertainties in the input parameters with uncertainty in the output parameter, i.e., sensitivity analyses.

A third technique employed in risk assessments for evaluating the propagation of uncertainties is the differential analysis approach. The differential analysis approach involves replacing the complex deterministic system model with a first order or second order Taylor series expansion. The mean and the variance of the output variable can be approximated by evaluating the series terms, using a mean value for each variable in the input vector. This method is similar to the response surface methodology in that it is very useful in conducting sensitivity analyses. This method is limited, however, in that it allows only a localized evaluation of variance in the output parameter.

As an example of how uncertainties are treated in current site assessments, the performance assessment program for the WIPP, currently under development, is described in detail in the report entitled "Preliminary Comparison with 40 CFR Part 191, Subpart B for the Waste Isolation Pilot Plant," (SAND91). This report, updated periodically, describes the WIPP methodology for evaluating compliance with the disposal requirements of 40 CFR Part 191. The approach to addressing uncertainties in the assessment of performance is described in the first volume of the study, "Methodology and Results."

The WIPP program generally employs the widely accepted approach to performance assessment. Scenarios which are credible and which may impact performance of the facility during the next 10,000 years are systematically identified. The likelihood and consequences of each scenario are evaluated and quantified and the results are combined into a complementary cumulative distribution function (CCDF) to demonstrate compliance with the performance standards.

The Monte Carlo technique was adopted in the WIPP program to evaluate the impact of parameter value uncertainties on the calculated consequences. Five reasons are given in the WIPP compliance report to support this decision. First, it is felt that the



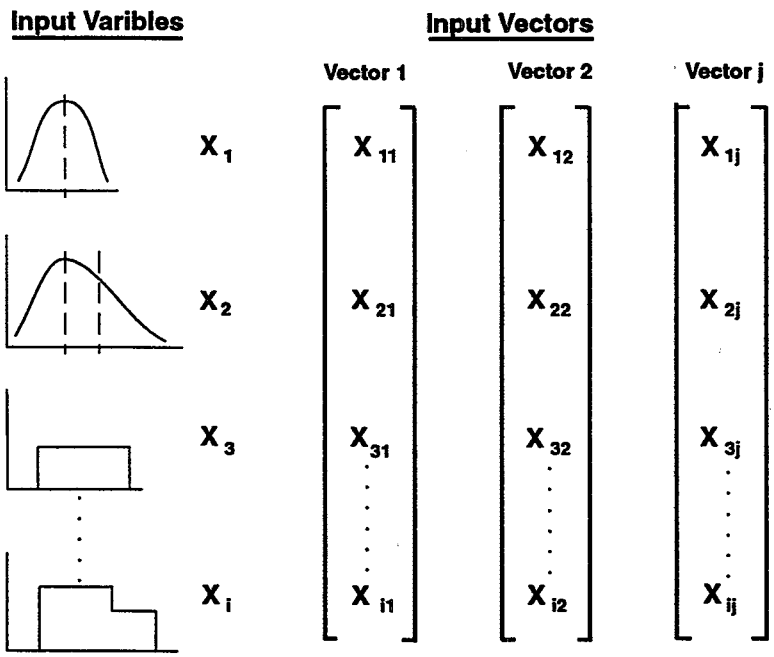


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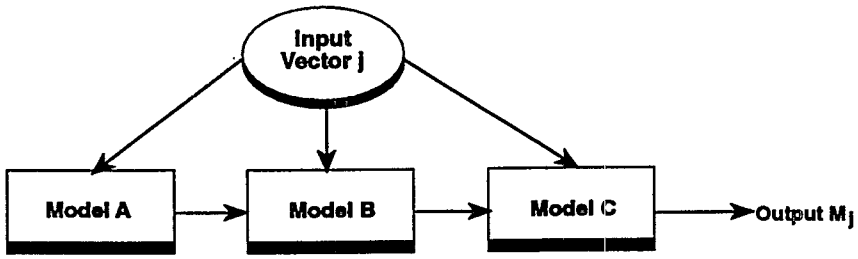
Figure 7.6-3. Monte Carlo Technique (adapted from Hunter et al., 1986).



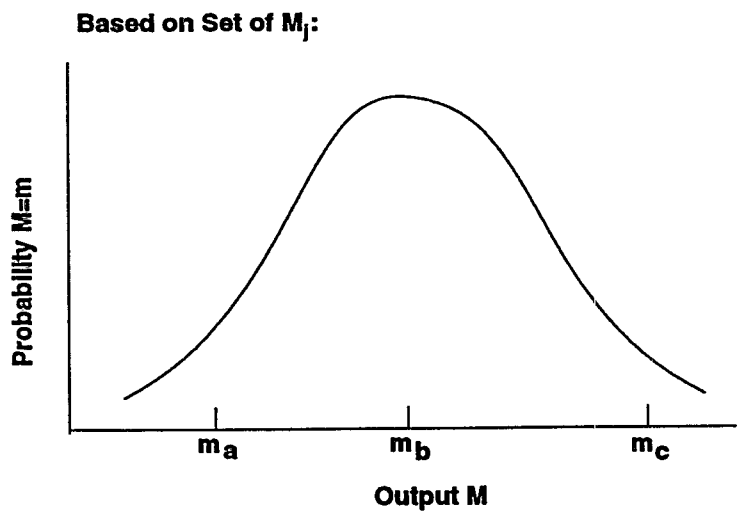


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Figure 7.6-3. Monte Carlo Technique (adapted from Hunter et al., 1986) continued.

Monte Carlo approach best accommodates large uncertainties in input parameters and complex interdependencies among the parameters. Second, the direct result of the Monte Carlo approach is the generation of a distribution function for the output parameter. Third, the Monte Carlo method is straightforward in that complex ancillary calculations are not necessary, just the repetitive recalculation of the deterministic models with a predetermined set of input variable vectors. Fourth, the Monte Carlo technique is amenable to evaluating the propagation of uncertainties through systems of interdependent models. Finally, the Monte Carlo technique allows a direct evaluation of the impacts of input parameter uncertainties on the certainty in the output parameter, i.e., sensitivity studies.

The WIPP program uses the Monte Carlo technique to develop a family of CCDFs. Each CCDF is the result of evaluating the deterministic models with a single input variable vector. Probability distributions are developed for input parameters, based on the existing data base and expert judgement (Section 7.6.2). Input vectors are defined by sampling from the input-parameter-value distributions. Each input vector is then used to calculate scenario probabilities and consequences which are assembled into a CCDF. One hundred iterations (recalculations), for example, will require one hundred input vectors and will generate one hundred CCDFs for the output parameter. From this set of CCDFs, the mean CCDF, or any other statistically significant CCDF (median, percentiles) can be derived and evaluated against the containment standard. It is recognized that this is a simplistic overview of a sophisticated program. The WIPP performance assessment and compliance program is discussed in detail in the WIPP compliance document (SAND91). This overview, however, indicates the rigor with which uncertainties can be quantitatively evaluated in site-specific assessments.

7.6.2 Expert Judgement

It is generally accepted that the use of expert judgement is required in the process of evaluating the long-term containment potential of a geologic waste disposal facility. It is expected that the use of expert judgement will likely be an integral component of the demonstrations of compliance with the Environmental Standards for Disposal (40 CFR Part 191, Subpart B). It is important to discuss the manner in which expert judgement should be used. Expert judgement refers to opinions based on the knowledge of the expert whose experience is relevant to the issue at hand.

The use of expert judgement in the area of nuclear reactor safety is well-documented in the NRC's study, "Severe Accident Risks: An Assessment for Five U.S. Nuclear Plants" (NUREG-1150). Severe accident risks were evaluated for five commercial nuclear power plants. Risks were estimated based on the types and frequencies of accidents that could lead to severe core damage and melting, the response of the containment structure to severe accident loading, the radioactive release that could result from containment failure, and the offsite effects of the potential releases (NRC90). The study was undertaken to provide a risk perspective for the radioactive release resulting from a core meltdown. The long-term objective of the assessment was to provide probabilistic risk assessment models for generic use in research and security risk estimates. The use of expert judgement in the NUREG-1150 study is discussed in Ortiz

et al. (1989), where it is referred to as "part of the largest elicitation task to date" and "an advance over those processes developed in previous probabilistic risk assessments."

Thirty-two issues in the following categories were evaluated:

1. System analysis issues.

2. In-vessel accident progression issues.

3. Containment load issues.

4. Molten core containment issues.

5. Structural issues.

6. Source-term issues.

A panel of twelve experts evaluated the first category and a total of 38 experts evaluated the other five categories. Each expert provided information regarding parameter-value distributions. Consensus distributions were constructed by aggregating the experts' distributions through simple averaging or sampling from the experts' distributions through using the Monte Carlo technique.

Expert judgement is also being employed in the WIPP performance assessment program. The WIPP program is employing a formal elicitation process to evaluate parameters considered significant in assessing performance but with which there is considerable associated uncertainty (SAND91). A formal expert judgement elicitation process is employed in the WIPP program when "data are lacking, either because of the complexity of processes or the time and resources it would take to collect data and/or when data have a major impact on the performance assessment" (SAND91).

An example of this is discussed in Trauth et al. (1991). Radionuclide concentrations in the brine located in the rooms and drifts of the WIPP repository are considered to be a critical parameter in assessing the performance of the facility. However, there is significant uncertainty in these radionuclide concentrations. A formal elicitation process was used to develop concentration distributions. The distributions are used in the performance assessments instead of point values, allowing the impacts of uncertainties in the values of the parameter and associated parameters to be evaluated.

In addition to the evaluation of probability distributions of significant system-parameter values, a formal expert judgement process is being employed in the WIPP program for the identification and evaluation of future human-intrusion scenarios, an inherently qualitative task. The formal elicitation process allows the defensible quantification of the issue as is necessary for inclusion in the quantitative performance assessments. Sixteen experts, external to the WIPP program and representing a diversity of physical and social sciences, were systematically identified and organized into four-member teams. Each team was charged with identifying reasonable, foreseeable "futures" for human society and quantifying the likelihoods of occurrence of these futures.

Also, the teams were asked to evaluate how the futures could result in the intrusion of the WIPP repository and to quantify the likelihoods of such intrusions.

To date, the teams have evaluated future states of human society and identified reasonable "futures." The likelihoods of these futures were identified through a formal elicitation process. The teams have also identified possible modes of intrusion associated with the "futures" and developed quantitative probabilistic estimates of the frequencies of these intrusions. The evaluation of human-intrusion scenarios through a structured expert-judgement process is documented in Hora, et al. (1991).

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APPENDIX A

GLOSSARY AND ACRONYMS

GLOSSARY

actinide: The series of elements beginning with actinium, Element No. 89, and continuing through lawrencium, Element No. 103.

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 radiation.

beta particle: An elementary particle emitted from a nucleus during radioactive decay, with a single electrical charge and a mass equal to 1/1837 that of a proton. A negatively-charged beta particle is identical to an electron. A positively-charged beta particle is called a positron.

contact handled TRU-wastes: TRU wastes that can be handled with just the shielding that is provided by the waste package itself.

critical organ: Specific organ being most susceptible to the effects of a specific type of radiation.

Ci: Curie - the unit rate of radioactive decay; the quantity of any radionuclide which undergoes 3.7×10^{10} disintegrations/second. Several fractions of the curie are in common usage.

Nanocurie (nCi) - one billionth of a curie; 3.7×10^{1} disintegrations/second

Microcurie (uCi) - one-millionth of a curie; 3.7×10^{4} disintegrations/second

Millicurie (mCi) - one-thousandth of a curie; 3.7×10^{7} disintegrations/second

Picocurie (pCi) - one-millionth of a microcurie; $3.7\ 10^{-2}$ disintegrations/second or 2.22 disintegrations/minute

daughter: Synonym for decay product.

decay product: A nuclide resulting from the radioactive disintegration of a radionuclide, 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.

dose: The amount of energy absorbed per unit mass of absorbing tissue as a result of the exposure.

dose equivalent: A term used to express the amount of effective radiation 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.

dosimetry: Quantification of energy absorbed by the population from decaying radionuclides.

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 or radioactive decay.

fissile: Any material fissionable by thermal (slow) neutrons, including but not limited to uranium 235 and 238 and plutonium 239.

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 and generally one or more neutrons. Fission can occur spontaneously, but usually is caused by nuclear absorption of gamma rays, neutrons, or other particles.

fission products: The nuclei formed by the fission of heavy elements, plus the nuclides formed by the fission fragments' radioactive decay.

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. Gamma radiation frequently accompanies alpha and beta emissions and always accompanies fission. Gamma rays are very penetrating, and are best stopped by dense materials.

general environment: The total terrestrial, atmospheric, and aquatic environments outside sites within which any activity, operations, or process associated with the management and storage of spent nuclear fuel, high-level, or transuranic radioactive wastes is conducted.

geometric mean: The mean of a set of numbers, calculated by taking the Nth root of the product of N numbers or by finding the arithmetic means of the logarithms of the individual numbers.

geometric standard deviation: The standard deviation of a set of numbers obtained when calculating the arithmetic mean of the logarithm of the individual numbers. Also see geometric mean.

geosphere: The solid portion of the earth, synonymous with the lithosphere.

GW: Gigawatts = one billion (10^9) watts.

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.

high-temperature gas-cooled reactor: Nuclear reactor using uranium and thorium as a fuel whose core is designed for high fuel utilization efficiency. The heat removal system is based upon helium as a coolant.

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, constitution 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.

kg: Kilogram - the SI unit of mass, approximately equal to 2.2 pounds.

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.

m^3: Cubic meter - the SI unit of volume, approximately equal to 35.3 cubic feet.

management and storage: Any activity, operation, or process, except for transportation, conducted to prepare spent nuclear fuel, high-level or transuranic radioactive wastes for storage or disposal, the storage of any of these materials, or activities associated with disposal of these wastes.

member of the public: Any individual who is not engaged in operations involving the management, storage, and disposal of materials covered by these standards. A worker so engaged is a member of the public except when on duty at a site.

metric ton (t): The SI unit of weight equal to 1000 kilograms or 2205 pounds.

neutron capture: The process in which an atomic nucleus absorbs or captures a neutron. The probability that a given material will capture neutrons is dependent on the energy of the neutron and on the nature of the material.

neutron: An unchanged 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.

noble gas: Any of a group of rare gases that include helium, neon, argon, krypton, xenon, and sometimes radon and exhibit great stability and extremely low reaction rates.

nonstochastic effect: Those health effects that increase in severity with increasing dose and usually have a threshold.

rad (radiation absorbed dose): A measure of the energy imparted to matter by radiation; defined as 100 ergs per gram.

Millirad (mrad)- one-thousandth of a rad.

radioactive decay: A process whereby the nucleus of an atom emits excess energy. The emission of this energy is referred to as radioactivity.

radioactivity: The property of certain nuclides of spontaneously emitting particles or gamma radiation or of emitting X-radiation following orbital electron capture or of undergoing spontaneous fission.

radionuclide: A type of atom which spontaneously undergoes radioactive decay.

RBE: The ratio of the dose (rad) of low-LET radiation to the dose of high-LET radiation producing the same endpoint. It is a measure of the effectiveness of high-LET compared to low-LET radiation in causing the same specific endpoint.

rem (roentgen equivalent man): A measure of equivalence for the relative biological effect of radiations of different types and energies on man.

remotely-handled TRU waste: Those types of TRU wastes that must be handled by robotics.

risk projection: Absolute - risk projection 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 projection based on the assumption that the annual rate of radiation- induced excess cancer deaths is proportional to the ambient rate of occurrence of fatal cancer.

roentgen: R is the symbol for roentgen, a unit of measurement of X-radiation, equivalent to an absorbed dose in tissue of approximately 0.9 rad.

Milliroentgen (mR)- one-thousandth of a roentgen.

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.

standards: The "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.

stochastic effect: Those health effects for which the probability of occurrence is a function of the dose received.

storage: Placement of radioactive wastes with planned capability to readily retrieve such materials.

target: Material subjected to particle bombardment or irradiation in order to induce a nuclear reaction.

target theory (Hit theory): A theory explaining some biological effects of radiation on the basis that ionization occurring in a discreet volume (the target) within the cell, directly causes a lesion which subsequently results in a physiological response to the damage at that location. One, two, or more "hits" (ionizing events within the target) may be necessary to elicit the response.

teratogenesis: Congenital abnormalities or defects.

transuranic waste: Waste containing more than 100 nanocuries of alpha-emitting transuranic isotopes, with half-lives greater than 20 years, per gram of waste.

X-ray: Penetrating electromagnetic radiation whose wave lengths are shorter than those of visible light. They are usually produced by bombarding a metallic target with fast electrons in a high vacuum. 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. These rays are sometimes called roentgen rays.

zircaloy: A zirconium alloy used as fuel cladding in some types of nuclear reactors.

ACRONYMS

AEC	U.S. Atomic Energy Commission
ALAP	As low as practicable
ALARA	As low as reasonably achievable
AMAD	Activity median aerodynamic diameter
ANL	Argonne National Laboratory
BEAR	Biological Effects of Atomic Radiation
BEIR	Biological Effects of Ionizing Radiation
BID	Background Information Document
CFR	Code of Federal Regulations
CH	Contact-handled
DEIS	Draft Environmental Impact statement
DOD	U.S. Department of Defense
DOE	U.S. Department of Energy
DOT	U.S. Department of Transportation
DREF	Dose rate effectiveness factor
DWPF	Defense Waste Processing Facility
ERC	President's Federal Energy Resources Council
ERDA	Energy Research and Development Administration
EPA	U.S. Environmental Protection Agency
FFTF	Fast Flux Test Facility
FRC	Federal Radiation Council

GI	Gastrointestinal
GW(e)	Gigawatts of electric power
HANF	Hanford, Washington
HEW	U.S. Department of Health, Education, and Welfare
HLW	High-level radioactive Waste
HTGR	High-temperature gas-cooled reactor
ICPP	Idaho Chemical Processing Plant
ICRP	International Commission on Radiological Protection
ICRPTG	International Commission on Radiological Protection Task Group
INEL	Idaho National Engineering Laboratory
IRG	Interagency Review Group
LANL	Los Alamos National Laboratory
LET	Linear energy transfer
LLI	Lower large intestine
LMFBR	Liquid metal fast breeder reactor
LQ	Linear quadratic
LWR	Light-water reactor
MFRP	Midwest Fuel Recovery Plant
MIRD	Medical internal radiation dose
MRS	Monitored retrievable storage
MTHM	Metric tons of heavy metal
NAS	National Academy of Sciences

NCHS	National Center for Health Statistics
NCRP	National Council on Radiation Protection and Measurements
NFS	Nuclear Fuel Services
N-P	Nasopharyngeal
NRC	U.S. Nuclear Regulatory Committion
NRPB	National Radiological Protection Board
NWPA	Nuclear Waste Policy Act of 1982
NWTS	National Waste Terminal Storage
OECD	Organization for Economic Coorporation and Development
OMB	Office of Management and Budget
ORNL	Oak Ridge National Laboratory
P	Pulmonary
R	Roentgen
RBE	Relative Biological Effectiveness
RFP	Rocky Flats Plant
RH	Remote-Handled
RIA	Regulatory Impact Analysis
S	Stomach
SAB	Science Advisory Board
SI	Small Intestine
SRP	Savannah River Plant
T-B	Tracheobronchial

TRU	Transuranic
ULI	Upper Large Intestine
UNSCEAR	United Nations Scientific Committee on the Effects of Atomic Radiation
WIPP	Waste Isolation Pilot Plant
WIPP LWA	Waste Isolation Pilot Plant Land Withdrawal Act
WLM	Working Level Month