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Civilian Radioactive Waste Management System
Management & Operating Contractor

CALCULATIONS SUPPORTING EVALUATION OF POTENTIAL ENVIRONMENTAL STANDARDS FOR YUCCA MOUNTAIN

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Preliminary draft results of this study have been presented to the National Academy of Sciences Committee on Technical Bases for Yucca Mountain Standards in December, 1993. During the discussions that followed, committee members made several worthwhile comments that were taken into consideration in completion of this study. We believe that these comments greatly improved the usefulness of the study.

This report was reviewed by Abe Van Luik (M&O/INTERA Inc.), Steven Nesbit (M&O/Duke Engineering & Services, Inc.), Jeremy Boak (DOE), and Chris Pflum and Keith Kersch (Science Applications International Corp.). Their comments greatly improved the final product.

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EXECUTIVE SUMMARY

The Energy Policy Act of 1992, Section 801 (U.S. Congress, 1992) provides for the U. S. Environmental Protection Agency (EPA) to contract the National Academy of Sciences (NAS) to conduct a study and provide findings and recommendations on reasonable standards for the disposal of high-level wastes at the Yucca Mountain site. The NAS study is to provide findings and recommendations which include, among other things, whether a health-based standard based on dose to individual members of the public from releases to the accessible environment will provide a reasonable standard for the protection of the health and safety of the public. The EPA, based upon and consistent with the findings and recommendations of the NAS, is required to promulgate standards for protection of the public from releases from radioactive materials stored or disposed of in a repository at the Yucca Mountain site.

This document presents a number of different "simple" analyses of undisturbed repository performance that are intended to provide input to those responsible for setting appropriate environmental standards for a potential repository at the Yucca Mountain site in Nevada. Each of the processes included in the analyses has been simplified to capture the primary significance of that process in containing or isolating the waste from the biosphere. In these simplified analyses, the complex waste package interactions were approximated by a simple waste package "failure" distribution which is defined by the initiation and rate of waste package "failures". Similarly, releases from the waste package and the engineered barrier system are controlled by the very near field environment and the presence and rate of advective and diffusive release processes. Release was approximated by either a simple alteration-controlled release for the high solubility radionuclides and either a diffusive or advective-controlled release for the solubility-limited radionuclides. In general, the term "simple" performance assessment implies that the complexities associated with process coupling and process interactions have been neglected. This allows the analyses to be more transparent so that the relative importance of a particular component of the system (whether the waste package, or engineered barrier, or geosphere) is highlighted.

The models used for the calculation of releases and doses from a geologic repository are based on assuming that conditions that are now believed to exist will continue long into the future. Uncertainties over these periods stem from uncertainties in the conceptual models themselves, the boundary conditions that are applied, and the ranges of parameters that are used. Necessarily, there is a large amount of uncertainty in these calculations because of the discrepancy between the length of the knowledge base used for the model as compared to the geologic time over which the consequences occur. For this reason, the results of this study should not be viewed as predictions, but as calculations done in an attempt to gain some understanding of potential future repository behavior and its effects on humans. As such, these calculations may be useful in the societal process of setting environmental standards for a geologic repository at Yucca Mountain, Nevada.

In conducting the sensitivity analyses presented in this report, three models were selected each with different attributes that were useful to conducting the analyses. The model UCBNE-41 is a transport model capable of handling radionuclide chains that produces results that are readily related to input parameters. The model UCBNE-41 was selected for use in this report because

it is the basis for previous NAS calculations. A second model, the Repository Integration Program (RIP), was selected because it has had wide application to the total system performance assessment (TSPA) of Yucca Mountain by the Civilian Radioactive Waste Management System, Management and Operations Contractor in TSPA-1993 (Andrews, et al., 1994). RIP is very versatile and is operated in a probabilistic mode so that parameter uncertainty can be incorporated into the analyses. A third model, NEFTRAN-S was selected because it is widely used by the waste management community including the Nuclear Regulatory Commission, and was used by the EPA in support of the recent re-promulgation of 40 CFR 191 (EPA, 1993b). The application of UCBNE-41, RIP, and NEFTRAN-S to the baseline case produced analogous results in terms of the magnitude and time of arrival of the peak dose.

The following conclusions can be made based on the results of this study:

- Doses to an individual at the accessible environment occur well beyond 10,000 years and do not decline significantly for hundreds-of-thousands of years;
- Peak doses and releases at the accessible environment over the 1,000,000 year period are generally unaffected by waste package lifetimes of up to 100,000 years;
- Dose and release are significantly affected by unsaturated-zone pore velocity;
- Dispersion in the unsaturated zone is more important than that in the saturated zone;
- The wide range of ^{237}Np solubility has a significant effect on the magnitude of the peak dose at times approaching several hundred thousand years; and
- The effects of a diffusion path length formed by a capillary barrier has a significant effect on dose and release even at long times (of the order of hundreds-of thousands of years).

Numerous differences exist between the "simple" analyses reported in this document and the more complete analyses conducted as part of TSPA-1993 (see Andrews et al., 1994). A primary difference is that the TSPA-1993 analyses attempt to present representative analyses based on the best site-specific and design-specific information available at the time, accounting for the uncertainty in the parameters. In addition, the TSPA-1993 analyses attempt to incorporate as much realism in the individual processes potentially affecting the containment and isolation of radioactive wastes in a potential repository at Yucca Mountain, including the thermo-hydrologic regime in the vicinity of the repository as a function of repository loading, time, and location and the effects of the thermo-hydrologic regime on the initiation and rate of waste package degradation, waste form alteration, and release from the engineered barrier system. The TSPA-1993 analyses were performed with a range of possible thermal loads, a range of possible waste package designs, two possible criteria for the initiation of aqueous corrosion, and two conceptual representations of the aqueous corrosion rates. As a result of these differences, making direct comparisons of the results presented herein to the TSPA-1993 analyses is not possible. However, some useful insights as described below can be gained.

- The 10,000 year integrated release (the 10,000 cumulative release normalized to the values in Table 1 of 40 CFR 191) to the accessible environment determined in TSPA-1993 is totally determined by the release of gaseous ^{14}C. As a result, these results are not at all comparable to the results in this document where it was assumed the ^{14}C was transported in the aqueous phase.

- In TSPA-1993, for the baseline thermal load and waste package design, the normalized integrated aqueous ^{99}Tc release over 10,000 years (with the normalization to Table 1 of 40 CFR 191) at the accessible environment varied from 10^{-16} to 10^{-4} for the range of possible percolation fluxes from 5×10^{-4} to 2×10^{-3} m/yr. This release is analogous to the results presented here. In addition the 100,000 year releases are very similar between TSPA-1993 and the results presented in this report using "simple" analyses.

- The very long term peak doses from TSPA-1993 are analogous to the results presented here for comparable waste package failure distributions.

The analyses of the potential populations that may be sustainable on the available ground water and analyses of repository equivalent uranium ore bodies provide information that is relevant to the setting of new environmental standards for Yucca Mountain. The sensitivity analyses of possible post-closure consequences (release or dose) associated with a potential repository at Yucca Mountain also provides insight into repository behavior that should be considered in setting standards.

- There is little ground water available to support large populations in the vicinity of Yucca Mountain, and there is little water for dilution of releases from a potential repository. Calculations show that for a farming scenario in which all water for food production, drinking, and household use are derived from ground water, the available water could support a population that ranges from 14 to 150 persons. For a scenario where the available ground water is used for drinking and household use it could potentially support a population that ranges from 1,200 to 13,000 persons. Water for larger populations would dilute the concentration in water that could be potentially contaminated by the repository.

- The analysis of two representative repository equivalent uranium ore bodies, one in a chemically reducing environment the other in a oxidizing environment, indicates that doses to an individual from drinking water range from 30 to 400 mrem/yr, respectively. The health effects integrated over 10,000 years range from 2,000 to 17,000. These analyses suggest that if a uranium ore body were used as an analog to the high-level waste repository that the dose to an individual in a new standard could be 100 mrem/yr or higher, and that the integrated health effects could be considerably higher than the 1,000 used as the basis for 40 CFR 191.

- The sensitivity analyses show that the relative response of the repository system is virtually the same for integrated release to the accessible environment and peak dose to an individual at the accessible environment except at very long times.

- Releases to the accessible environment from a repository at Yucca Mountain may begin after a few tens of thousand years and extend to times over one million years;

- Short-term releases (less than 10,000 years) could be influenced by long-lived waste packages (i.e., lifetimes of several thousand years);

- Mid-term releases (10,000 to 100,000 years) are influenced by diffusion out of the waste package or very long-lived packages; and

- Long-term doses (beyond 500,000 years) from ^{237}Np can be affected by an engineered capillary barrier that creates a diffusion-controlled release from the engineered barrier system or a highly sorptive low solubility near field geochemical environment.

Based on the results of this study, the following conclusions can be drawn related to characterization activities that would aid performance assessment. In addition, the results have implications for repository design.

- Additional study of the solubility of ^{237}Np under anticipated near field thermo-hydrologic-geochemical conditions would reduce the uncertainty of long-term performance assessment calculations, if the range of measured solubilities can be narrowed;

- The pore velocity in the unsaturated zone is an important parameter to performance assessment (as was already known), and uncertainty in performance assessment calculations could be reduced if the range of percolation flux could be reduced;

- Dispersivity in the unsaturated zone is far more important than dispersivity in the saturated zone, with the dispersivity being very important if the period being considered is on the order of about half of the retarded radionuclide travel time (for example, if the time considered is 10,000 years and the retarded travel time is about 20,000 years, then the dispersivity will control the arrival of the radionuclide);

- Additional research on the coefficient of diffusion in capillary barriers could provide important information that would significantly lower the calculated doses at the accessible environment, even at very long times (beyond 1,000,000 years); and

- The waste package lifetime has little effect on dose and release over very long periods (on the order of a million years).

- Waste form alteration/dissolution rate information used in the analyses that were conducted have been based on the value used in the WISP study (NAS, 1983). This value has been substantiated by recent flow-through tests conducted by DOE contractors (see Andrews et al., 1994). This value may be very conservative given the possible limitation on dissolution caused by the lack of available chemical reactants at the waste form-water contact. More realistic (quasi-static) tests may yield substantially lower alteration/dissolution rates.

TABLE OF CONTENTS

ACKNOWLEDGEMENTS iii

EXECUTIVE SUMMARY iv

TABLE OF CONTENTS viii

LIST OF APPENDICES xi

LIST OF TABLES xii

LIST OF FIGURES xiii

1. INTRODUCTION 1-1

1.1 BACKGROUND 1-2

1.2 SIMPLE PERFORMANCE ASSESSMENTS 1-3

1.3 PURPOSE AND OBJECTIVES 1-5

1.4 APPROACH 1-6

2. ANALYSES USING THE UCBNE-41 MODEL 2-1

2.1 DESCRIPTION OF UCBNE-41 2-1

2.2 BASELINE CASE USING YUCCA MOUNTAIN PARAMETERS 2-3

2.3 SENSITIVITY TO GROUND-WATER TRAVEL TIME 2-7

2.4 SENSITIVITY TO WASTE PACKAGE LIFETIME 2-8

2.5 SENSITIVITY TO SOLUBILITY 2-9

2.6 THE NUCLEAR REGULATORY COMMISSION (NRC) RELEASE LIMIT 2-10

2.7 SUMMARY 2-10

TABLE OF CONTENTS
(Continued)

3. ANALYSES USING THE REPOSITORY INTEGRATION PROGRAM (RIP) 3-1

3-1 GENERAL DESCRIPTION OF RIP 3-1

3.2 COMPARISON WITH THE BASELINE CASE 3-3

3.3 CUMULATIVE RELEASE PERFORMANCE MEASURE 3-6

3.4 PEAK DOSE PERFORMANCE MEASURE 3-7

3.5 EFFECT OF WASTE PACKAGE FAILURE ON PEAK DOSE . . . 3-8

3.6 EFFECT OF DIFFUSIVE RELEASE FROM THE ENGINEERED BARRIER SYSTEM 3-9

3.7 SUMMARY 3-11

4. ANALYSES USING THE NEFTRAN-S MODEL 4-1

4.1 GENERAL DESCRIPTION OF NEFTRAN-S 4-1

4.2 COMPARISON WITH THE BASELINE CASE 4-1

4.3 EFFECT OF DISPERSIVITY ON DOSE 4-3

4.4 EFFECT OF WASTE PACKAGE LIFETIME ON RELEASE AND DOSE 4-4

4.5 EFFECT OF UNSATURATED ZONE VELOCITY ON RELEASE AND DOSE 4-5

4.6 EFFECT OF DIFFUSIVE RELEASE FROM THE ENGINEERED BARRIER SYSTEM ON RELEASE AND DOSE 4-5

4.7 SUMMARY 4-7

5. SUMMARY AND CONCLUSIONS 5-1

TABLE OF CONTENTS
(Continued)

5.1 INTRODUCTION 5-1

5.2 COMPARISON OF RESULTS FROM UCBNE-41, RIP, AND NEFTRAN-S 5-2

5.3 SUMMARY OF SENSITIVITY ANALYSES 5-4

5.4 COMPARISON OF RESULTS TO TSPA-1993 5-6

5.5 SIGNIFICANCE OF RESULTS TO STANDARD SETTING 5-7

5.6 SIGNIFICANCE OF RESULTS TO SITE CHARACTERIZATION AND DESIGN 5-10

6. REFERENCES 6-1

LIST OF APPENDICES

APPENDIX A	POTENTIAL POPULATION IN THE VICINITY OF YUCCA MOUNTAIN
APPENDIX B	POTENTIAL EFFECTS OF URANIUM ORE BODIES
APPENDIX C	COMPARISON WITH THE RESULTS OF THE WASTE ISOLATION SYSTEM PANEL

LIST OF TABLES

Table 1-1 Characteristics and Features of the Models UCBNE-41, RIP, and NEFTRAN-S 1-8

Table 1-2 Sensitivity Analyses Conducted Using UBCNE-41, RIP, and NEFTRAN-S 1-9

Table 2-1 Source Term Information Used for the Yucca Mountain Baseline Case 2-12

Table 2-2 Parameters Used for the Yucca Mountain Baseline Case 2-14

Table 2-3 Retardation Factors and Dose Conversion Factors Used for the Yucca Mountain Baseline Case 2-15

Table 2-4 Leach Times and Initial Concentrations for the Baseline Case Modified by Assuming Advective-Diffusive Releases from the Waste Package 2-17

Table 2-5 Summary of Peak Dose to an Individual from ^{99}Tc, ^{129}I, and ^{237}Np and Time of Occurrence for the Peaks for each Simulation 2-18

Table 3-1 Radionuclide Inventories and Solubilities Used in RIP Analyses . . 3-12

Table 3-2 Specific Activities, Retardation Factors, EPA Limits, and Dose Conversion Factors Used in RIP Analyses 3-14

Table 3-3 Waste Package Release Parameters Used in RIP Analyses 3-16

Table 3-4 Unsaturated-Zone Transport Properties Used in RIP Analyses 3-16

Table 3-5 Saturated-Zone Transport Properties Used in RIP Analyses 3-16

Table 3-6 Parameters Used in the RIP Analyses 3-17

Table 4-1 Description of NEFTRAN Output Options 4-9

Table 4-2 Description of Source Term Options for NEFTRAN 4-10

Table 4-3 Simplified Inventory Used for Analyses Using the Model NEFTRAN-S 4-11

LIST OF FIGURES

Figure 2-1 Schematic Diagram of a Geologic Repository at Yucca Mountain 2-20

Figure 2-2 Dose to an Individual at the Accessible Environment for the Baseline Case 2-21

Figure 2-3 Dose to an Individual at the Accessible Environment Modifying the Baseline Case by Assuming Advective-Diffusive Releases from the Waste Package 2-21

Figure 2-4 Dose to an Individual at the Accessible Environment for a Ground-Water Travel Time of 10,000 Years (4.0 mm/yr percolation flux) 2-22

Figure 2-5 Dose to an Individual at the Accessible Environment for a Ground-Water Travel Time of 100,000 Years (0.21 mm/yr percolation flux) 2-22

Figure 2-6 Dose to an Individual at the Accessible Environment for a Ground-Water Travel Time of 100,000 Years (0.21 mm/yr percolation flux), and a Dispersion Coefficient of 12.5 m^2/yr 2-23

Figure 2-7 Dose to an Individual at the Accessible Environment for a 25,000 Year Ground-Water Travel Time and a 10,000 Year Waste Package 2-24

Figure 2-8 Dose to an Individual at the Accessible Environment for a 25,000 Year Ground-Water Travel Time and a 100,000 Year Waste Package 2-24

Figure 2-9 Dose to an Individual at the Accessible Environment with Solubilities of Np, Cm, Am, Pu, and U Increased by a Factor of 100 2-25

Figure 2-10 Dose to an Individual at the Accessible Environment with Solubilities of Cm, Am, Pu, and U Increased by a Factor of 100 and that of Np Increased by a Factor of 1,000 2-25

Figure 2-11 Dose to an Individual at the Accessible Environment with Solubilities of Cm, Am, Pu, and U Increased by a Factor of 100 and that of Np Increased by a Factor of 100,000 2-26

Figure 2-12 Dose to an Individual at the Accessible Environment with a Reduced Retardation Factor for Neptunium 2-26

LIST OF FIGURES
(Continued)

Figure 2-13 Release of all Radionuclides at a Rate of 10^{-5} yr^{-1} of the 1,000 Year Inventory 2-27

Figure 2-14 Sensitivity to Ground-Water Travel Time of Dose to an Individual at the Accessible Environment from ^{99}Tc, ^{129}I, and ^{237}Np 2-28

Figure 2-15 Sensitivity to Waste Package Lifetime of Dose to an Individual at the Accessible Environment from ^{99}Tc, ^{129}I, and ^{237}Np 2-28

Figure 2-16 Sensitivity to Solubility and Retardation Factor of Dose to an Individual at the Accessible Environment from ^{237}Np 2-29

Figure 2-17 Peak Dose to an Individual at the Accessible Environment from ^{99}Tc, ^{129}I, and ^{237}Np for all Analyses 2-29

Figure 3-1 Dose to an Individual at the Accessible Environment for the Baseline Case 3-19

Figure 3-2 Dose to an Individual at the Accessible Environment for the Baseline Case for the First 100,000 Years 3-20

Figure 3-3 Dose to an Individual at the Accessible Environment with Diffusive Release from the Waste Package 3-21

Figure 3-4 Dose to an Individual at the Accessible Environment for the First 100,000 Years with Diffusive Release from the Waste Package 3-22

Figure 3-5 Effects of Increased ^{237}Np Solubility on Dose to an Individual at the Accessible Environment 3-23

Figure 3-6 Scatter Plot of Normalized Cumulative Release at 10,000 Years as a Function of Unsaturated-Zone Velocity 3-24

Figure 3-7 Scatter Plot of Normalized Cumulative Release at 100,000 Years as a Function of Unsaturated-Zone Velocity 3-25

Figure 3-8 Scatter Plot of Normalized Cumulative Release at 1,000,000 Years as a Function of Diffusion Coefficient in the Waste Package 3-26

Figure 3-9 Scatter Plot of Normalized Cumulative Release at 1,000,000 Years as a Function of Unsaturated-Zone Velocity 3-27

LIST OF FIGURES
(Continued)

Figure 3-10 Scatter Plot of Normalized Cumulative Release at 1,000,000 Years as a Function of ^{237}Np Solubility 3-28

Figure 3-11 Scatter Plot of Normalized Cumulative Release at 1,000,000 Years as a Function of Diffusion Coefficient in the Waste Package with ^{237}Np Solubility Fixed at 1.0 g/m^3 3-29

Figure 3-12 Scatter Plot of Normalized Cumulative Release at 1,000,000 Years as a Function of Unsaturated-Zone Velocity with ^{237}Np Solubility Fixed at 1.0 g/m^3 3-30

Figure 3-13 Scatter Plot of 10,000 Year Peak Dose to an Individual as a Function of Unsaturated-Zone Velocity 3-31

Figure 3-14 Scatter Plot of 100,000 Year Peak Dose to an Individual as a Function of Unsaturated-Zone Velocity 3-32

Figure 3-15 Scatter Plot of 1,000,000 Year Peak Dose to an Individual as a Function of Diffusion Coefficient in the Waste Package 3-33

Figure 3-16 Scatter Plot of 1,000,000 Year Peak Dose to an Individual as a Function of Unsaturated-Zone Velocity 3-34

Figure 3-17 Scatter Plot of 1,000,000 Year Peak Dose to an Individual as a Function of ^{237}Np Solubility 3-35

Figure 3-18 Scatter Plot of 1,000,000 Year Peak Dose to an Individual as a Function of Diffusion Coefficient in the Waste Package with ^{237}Np Solubility Fixed at 1.0 g/m^3 3-36

Figure 3-19 Scatter Plot of 1,000,000 Year Peak Dose to an Individual as a Function of Unsaturated-Zone Velocity with ^{237}Np Solubility Fixed at 1.0 g/m^3 3-37

Figure 3-20 Dose to an Individual at the Accessible Environment with a Linear Waste Package Failure Rate (1.0 % per 1,000 years) Between Zero and 100,000 Years 3-38

Figure 3-21 Dose to an Individual at the Accessible Environment with a Waste Package Failure Rate (1.0 % per 1,000 years) Between 10,000 and 110,000 Years 3-39

LIST OF FIGURES
(Continued)

Figure 3-22 Dose to an Individual at the Accessible Environment with a Waste Package Failure Rate (1.0 % per 1,000 years) Between 100,000 and 200,000 Years 3-40

Figure 3-23 Dose to an Individual at the Accessible Environment with a Waste Package Failure Rate (1.0 % per 10,000 years) Between Zero and 1,000,000 Years 3-41

Figure 3-24 Dose to an Individual at the Accessible Environment with a Diffusion Coefficient of 1.0 x 10^{-4} m^2/yr through a 1.0 m Engineered Barrier 3-42

Figure 3-25 Dose to an Individual at the Accessible Environment with a Diffusion Coefficient of 1.0 x 10^{-5} m^2/yr through a 1.0 m Engineered Barrier 3-43

Figure 3-26 Dose to an Individual at the Accessible Environment with a Diffusion Coefficient of 1.0 x 10^{-6} m^2/yr through a 1.0 m Engineered Barrier 3-44

Figure 3-27 Scatter Plot of Normalized Cumulative Release to the Accessible Environment Over 1,000,000 Years for a Range of Diffusion Coefficients through the Engineered Barrier as a Function of Unsaturated-Zone Velocity 3-45

Figure 3-28 Scatter Plot of Normalized Cumulative Release to the Accessible Environment Over 1,000,000 Years for a Range of Unsaturated-Zone Velocities as a Function of Diffusion Coefficient through the Engineered Barrier 3-46

Figure 3-29 Scatter Plot of Peak Dose to an Individual at the Accessible Environment Over 1,000,000 Years as a Function of Unsaturated-Zone Velocity for a Range of Diffusion Coefficients through the Engineered Barrier 3-47

Figure 3-30 Scatter Plot of Peak Dose to an Individual at the Accessible Environment Over 1,000,000 Years as a Function of Diffusion Coefficient through the Engineered Barrier for a Range of Unsaturated-Zone Velocities 3-48

LIST OF FIGURES
(Continued)

Figure 4-1 Dose to an Individual at the Accessible Environment for the Baseline Case Using NEFTRAN-S 4-12

Figure 4-2 Dose to an Individual at the Accessible Environment from the ^{237}Np and ^{235}U Chains for the Baseline Case 4-13

Figure 4-3 Dose to an Individual at the Accessible Environment from the ^{238}U and ^{232}Th Chains for the Baseline Case 4-14

Figure 4-4 The Effects of a Range of Dispersivities (alpha) on ^{99}Tc Dose to an Individual at the Accessible Environment 4-15

Figure 4-5 Comparison of Dose to an Individual at the Accessible Environment from ^{99}Tc and ^{129}I Assuming a Single Leg and a Double Leg Transport Path 4-16

Figure 4-6 Effect of Variation of Unsaturated Zone Dispersivity on ^{99}Tc Dose to an Individual at the Accessible Environment 4-17

Figure 4-7 Effect of Variation of Saturated Zone Dispersivity on ^{99}Tc Dose to an Individual at the Accessible Environment 4-18

Figure 4-8 Normalized Release to the Accessible Environment as a Function of Source-Term Release Over 10,000 Years 4-19

Figure 4-9 Normalized Release to the Accessible Environment as a Function of Source-Term Release Over 100,000 Years 4-20

Figure 4-10 Peak Dose to an Individual Over 10,000 Years as a Function of Source-Release Time 4-21

Figure 4-11 Peak Dose to an Individual Over 100,000 Years as a Function of Source-Release Time 4-22

Figure 4-12 Normalized Release to the Accessible Environment Over 10,000 Years as a Function of Unsaturated Zone Pore Velocity 4-23

Figure 4-13 Normalized Release to the Accessible Environment Over 100,000 Years as a Function of Unsaturated Zone Pore Velocity 4-24

LIST OF FIGURES
(Continued)

Figure 4-14 Peak Dose to an Individual at the Accessible Environment Over 10,000 Years as a Function of Pore Velocity in the Unsaturated Zone 4-25

Figure 4-15 Peak Dose to an Individual at the Accessible Environment Over 100,000 Years as a Function of Pore Velocity in the Unsaturated Zone 4-26

Figure 4-16 Normalized Release to the Accessible Environment Over 10,000 Years as a Function of Diffusive Release from the Engineered Barrier System 4-27

Figure 4-17 Normalized Release to the Accessible Environment Over 100,000 Years as a Function of Diffusive Release from the Engineered Barrier System 4-28

Figure 4-18 Dose to an Individual at the Accessible Environment from ^{129}I as a Function of Time for a Range of Values of Diffusion Through the Engineered Barrier System 4-29

Figure 4-19 Peak Dose to an Individual at the Accessible Environment Over 10,000 Years as a Function of Diffusive Release from the Engineered Barrier System 4-30

Figure 4-20 Peak Dose to an Individual at the Accessible Environment Over 100,000 Years as a Function of Diffusive Release from the Engineered Barrier System 4-31

Figure 5-1 Correlation Between Peak Dose and Normalized Cumulative Release Over 10,000 Years 5-11

Figure 5-2 Correlation Between Peak Dose and Normalized Cumulative Release Over 100,000 Years 5-12

Figure 5-3 Correlation Between Peak Dose and Normalized Cumulative Release Over 1,000,000 Years 5-13

1. INTRODUCTION

Standards and regulations are set taking into account in part predictions based on available technical knowledge about related systems. Setting these standards can be relatively simple if the period of regulation is similar to the scale of available observations, because widely accepted models commonly exist. In the case of a geologic repository for high-level radioactive wastes, the period in which consequences of disposal may occur is more than a few thousand years, and may extend to millions of years. Knowledge of ground-water flow, contaminant transport, and effects of radioactivity in the biosphere is based on human observations that extend over no more than a few tens of years. Thus, the setting of standards for such systems becomes extremely difficult.

The geologic record provides important constraints on potential long term behavior, through study of natural analogs, and the stability of the geosphere is a primary reason for favoring geologic disposal. However, large uncertainties remain because of the short observational base and the sometimes imprecise or qualitative constraints of the geologic record. The models constructed to predict the long-term consequences of radioactive waste disposal are limited in their predictive capability by these uncertainties. For this reason, the results of this study should not be viewed as accurate predictions of consequence. They should be viewed as providing conceptual understanding of a repository system based on realistic but conservative descriptions of the Yucca Mountain site. As such, these calculations may be useful technical and scientific input to the societal process of setting environmental standards for a geologic repository at Yucca Mountain.

Environmental regulations may be based on several fundamental premises of protection. One may desire to protect the population as a whole or some potentially exposed individual of a critical portion of the population. One may desire to protect members of all future societies which may potentially be exposed to the hazard during the period when the hazard poses potential health consequences to members of society, or to limit the period of interest to some period when the environmental risks are greatest. A regulation may be based on an assessment of the risk associated with the potential exposure of individuals or to population groups to the long-term hazard. Risk is generally defined as the probability that a particular consequence occurs times the actual consequences of the activity or hazard, with the consequences either defined with respect to the number of latent health effects (i.e., excess cancers associated with the activity or hazard), or doses, or some other measure of consequence. The probability of a particular consequence may include the probability that alternate futures occur, with these alternate futures including both future natural and anthropogenic environments; as well as the probability that the existing conditions are known precisely (i.e., incorporating the effects of uncertain conceptual representations and the associated uncertain parameters that describe the fate of the hazard in the natural setting and biosphere).

Addressing all potential measures of safety (whether health-based, dose-based, concentration-based, or mass release-based), all potential exposed groups and individuals, and all potential probabilities that may affect the release, transport, uptake, dose, and health effects associated with a particular environmental hazard such as radioactive wastes is a task well beyond the scope of this report.

1.1 BACKGROUND

The Energy Policy Act of 1992, Section 801 (U.S. Congress, 1992) provides the following direction to the U. S. Environmental Protection Agency (EPA), the National Academy of Sciences (NAS), and the U. S. Nuclear Regulatory Commission (NRC):

- The EPA is to contract the NAS to conduct a study and provide findings and recommendations on reasonable standards for the disposal of high-level wastes at the Yucca Mountain site;

- The NAS study is to provide findings and recommendations which include;

  - Whether a health-based standard based on dose to individual members of the public from releases to the accessible environment will provide a reasonable standard for protection of the health and safety of the public

  - Whether it is reasonable to assume that a system for post-closure oversight of the repository can be developed, based upon active institutional controls, that prevent an unreasonable risk of breaching the repository's engineered or geologic barriers or increasing the exposure of individual members of the public to radiation beyond allowable limits

  - Whether it is possible to make scientifically supportable predictions of the probability that the repository's engineered or geologic barriers will be breached as a result of human intrusion over a period of 10,000 years

- The EPA, based upon and consistent with the findings and recommendations of the NAS, is required to promulgate standards for protection of the public from releases from radioactive materials stored or disposed of in a repository at the Yucca Mountain site; and

- The NRC is directed to modify 10 CFR 60 (NRC, 1991) to be consistent with the new EPA standards.

Based on Section 801, the NAS formed the Committee on Technical Bases for Yucca Mountain Standards, which is currently conducting periodic meetings to arrive at their findings and recommendations. This Committee will make recommendations, under the Energy Policy Act of 1992, to the EPA, on Yucca Mountain Standards.

After passage of the Energy Policy Act of 1992, the Department of Energy (DOE) recognized the need for a uniform position on issues related to standards for Yucca Mountain. In order to develop positions, a thorough understanding of the generalized behavior of repositories at arid sites is required. The analyses that follow provide technical input to the DOE in formulating positions on issues regarding new standards, and will accompany presentations of those positions to the NAS committee.

1.2 SIMPLE PERFORMANCE ASSESSMENTS

Calculations of the expected performance of any potential waste repository may be made with varying levels of detail/"reality" with the emphasis placed on various possible components controlling the containment and isolation of the waste from the biosphere. In addition, these assessments may consider a range of possible measures of performance or "safety" and a range of possible times over which the performance is to be compared. Several assessments of the post-closure performance of the potential repository at Yucca Mountain have been published over the past decade. These range from simple assessments conducted in the mid 1980's (NAS, 1983; Sinnock et al., 1984; EPA, 1985b; and DOE, 1986) to more complete and complex assessments published in the last five years (Doctor et al., 1992; NRC, 1990; McGuire et al., 1990; McGuire et al., 1992; Miller et al., 1992; Barnard et al., 1992; Eslinger et al., 1993; INTERA, 1993; Wilson et al., 1994; and Andrews et al., 1994). The more complete analyses attempt to incorporate the significant processes affecting the containment and isolation of the radionuclide wastes. With the exception of the most recent analyses reported as part of the total system performance assessment (TSPA) exercise called TSPA-1993 (Wilson et al., 1994 and Andrews et al., 1994) which investigated the performance over a range of time up to 1,000,000 years and addressed both release and individual dose performance measures, most of the previous analyses have focused on the total system performance requirements specified in 40 CFR 191, namely the integrated release of radionuclide activity for 10,000 years normalized to the values specified in Table 1 of 40 CFR 191.

There is uncertainty in the regulatory requirements following the passage of Section 801 of the Energy Policy Act of 1992, and the evaluation of the importance of various components of the multi-barrier disposal system is difficult in the more complete assessments of total system performance referenced above. Because of these two factors it is worthwhile to conduct some "simple" evaluations of the long-term undisturbed performance of a potential repository in an unsaturated medium in an arid environment such as exists at Yucca Mountain. The aim of these assessments is to extend the analyses conducted by the Waste Isolation System Panel (WISP) members (NAS, 1983) by incorporating conditions more applicable to unsaturated media and potential waste repositories placed above the water table.

The "simple" assessments described in the following chapters differ from the more complete assessments in several ways. These differences include the following:

- The waste package lifetime in the "simple" assessments is considered to be a parameter with a starting time that is assumed to be uncertain and a duration of "failures" that is assumed to be either rapid (i.e., all packages "fail" at once) or spread out over some uncertain duration. The waste package lifetime in the more complete analyses is considered to be a function of time, the waste package design, the near field thermo-hydrologic environmental conditions, and the conceptual representation of "failure" mechanisms.

- The waste form alteration in the "simple" assessments is considered to be a fixed parameter with no uncertainty. The waste form alteration in the more complete analyses is assumed to be a function of the near field thermo-chemical environment around the waste form once the waste package has "failed".

- The releases from the waste package and engineered barrier system in the "simple" assessments is considered to be either advective or diffusive-controlled and may be alteration-rate- or solubility-limited depending on the value of uncertain radionuclide solubilities, diffusion coefficients, and advective fluxes. The releases in the more complete total system performance assessments are also functions of the above parameters, but these parameters are dependent on the spatially and temporally varying hydrologic, thermal, and geochemical environment in which the waste packages reside.

- The transport of radionuclides in the geosphere (both the unsaturated and saturated zones) in the "simple" assessments consider the flux to be constant in space and time and that the conceptual model for flow can be represented as an equivalent porous medium with a constant equivalent porosity and radionuclide retardation (i.e., distribution) coefficient. The "simple" assessments have assumed the ^{14}C is transported in the aqueous phase to be consistent with the WISP analyses (NAS, 1983). In more complete analyses ^{14}C is transported primarily in the gaseous phase and for this reason the ^{14}C values presented in this study are not realistic. The aqueous flow and radionuclide transport in the more complete total system performance assessments have considered spatially varying transport properties (i.e., effective porosity, distribution coefficients, bulk density, degree of saturation) depending on the rock type of the different hydrostratigraphic units.

- The biosphere models used in both the "simple" and more complete assessments of total system performance have been based on a constant dilution in the saturated zone and the same dose conversion factors to convert from concentration at the accessible environment (which is the form of the result calculated by the release and transport model) to dose to a potential maximally exposed individual who uses the tuff aquifer for his or her domestic use, including irrigation of a small garden.

Another significant difference between the "simple" total system performance assessments presented in this document and the more complete analyses described, for example, in the TSPA-1993 reports (Andrews et al., 1994 and Wilson et al., 1994) is the incorporation of a wide range of parameter values to determine the sensitivity of the system response to some of the uncertain characteristics of the system components. In the "simple" analyses, no presumption is made regarding the "best estimate" or "expected value" or mean of the uncertain parameter distributions used in the analyses. In the more complete analyses, the range of values used in the analyses is meant to capture the uncertainty and/or variability in the site and design-related parameters and processes as well as being substantially conservative.

The key components of the system which we have treated as being uncertain in these "simple" assessments of post-closure performance include:

- The "lifetime" or time to "failure" of the waste package containment (start and duration),

- The form of the release from the "failed" waste packages (i.e., whether controlled by advective or diffusive processes, or both),

- The solubility of the actinides (in particular neptunium),

- The advective flux (or effective velocity) through the unsaturated zone,
- The effect of dispersion in the unsaturated zone, and
- The effect of a diffusive barrier caused by a low water saturation/low capillary pressure crushed rock backfill around or underneath the waste packages.

This is not to imply that other parameters are not potentially important to the total system performance (whether defined as a release or dose or some other measure of "safety" such as health effects or risk). However, it is the above attributes of the system which have been identified in the numerous performance assessments conducted to date as well as in the Site Characterization Plan (DOE, 1988b) as being the most significant.

It also warrants emphasizing that the "simple" assessments presented herein have made several conservative assumptions due generally to a lack of any available information with which to conclude otherwise. For example, once the waste package has "failed" it is assumed that the entire waste matrix is exposed, i.e., no credit is taken for the cladding or for the fact that only a small portion of the package may have been breached with the remaining portions still providing a very limited area for advective or diffusive release. In addition, once the package is breached, it is assumed that water is available to come into contact with the waste matrix in order for alteration of the fuel to commence. Geochemical changes in the near-field environment which could alter the solubility and transport characteristics of the radionuclides are not considered. Finally, the potential near field and far field perturbations on the hydrology that would be caused by the thermal regime following the emplacement of the waste are not incorporated in the analyses. Thermal effects were included in TSPA-1993 (Andrews et al., 1994) and a comparison with those results is prsented in Chapter 5. All of these assumptions make the releases and doses calculated in Chapters 2 to 4 conservative, meaning that they are likely to be overestimated.

To reiterate, the overall aim of the "simple" analyses is to evaluate a range of possible performance measures and a range of possible performance times given a range of uncertain properties which affect the long-term post-closure performance of a potential repository at Yucca Mountain. The sensitivity analyses conducted as part of this study aid in evaluating the robustness of the system response to the major uncertainties. The results are purposely presented in terms of either dose histories for specific parameter values or scatter plots illustrating the cumulative normalized release or peak dose for selected times versus a range of possible parameter values. The results are not presented in any probabilistic fashion [whether probability density functions (PDFs) or complementary cumulative distribution functions (CCDFs)] as any inference about the probability of any particular consequence would be inappropriate. What is presented is a relatively detailed sensitivity analysis to illustrate the potential impact of uncertain parameters on varying measures of "safety". In that sense, the results should be evaluated in a relative mode rather than in any absolute sense of comparison to a particular limit or standard.

1.3 PURPOSE AND OBJECTIVES

The purpose of this report is to document the assumptions and describe the performance assessment analyses that were carried out on the behalf of the DOE, in support of formulating

DOE positions on issues related to new environmental standards for the Yucca Mountain site. This report was also written in order to provide the necessary documentation for the sensitivity and performance assessment analyses that will be presented by the DOE as information that may be useful to the NAS Committee on Technical Bases for Yucca Mountain Standards.

1.4 APPROACH

The approach to determining which calculations would be useful in support of environmental standards for Yucca Mountain stems from several sources: the site is in an arid region which limits the population that can be potentially exposed (Appendix A); the long period of release from a repository which can be controlled in part by engineered barriers (Chapters 2 through 4); the desire not to pass undue risk to future generations makes studies of natural analogs desirable (Appendix B); and the belief that the NAS would rely heavily on their previous work on high-level waste repositories (Appendix C and Chapter 2).

The primary source of release from a geologic repository is by the ground-water pathway. Water that reaches the waste containers may corrode the waste package to the point of failure, dissolve the radionuclides from the waste form, and transport radionuclides to the accessible environment where it may be used by humans. In an arid region there are no streams, rivers, or surface water bodies to dilute the radionuclide concentration in the ground water prior to its use by humans. Also, in an arid region the population is largely controlled by the amount of water available. This reasoning leads to the calculation of the potential population in the vicinity of Yucca Mountain based on the amount of available ground water which is presented in Appendix A.

The premise that the risk from a repository should be no greater than that from the unmined uranium ore from which the spent fuel was derived satisfies the desire of not passing risk to future generations. Uranium ore bodies also provide a reasonable analog for a spent fuel repository in that both are subject to dissolution of long-lived radionuclides that are transported by ground water that can be used by humans. This reasoning and the fact that early in the history of the development of the EPA Standard the analog of uranium ore bodies was used (Klett, 1991), lead to the calculations that are presented in Appendix B.

The NAS report (NAS, 1990) entitled "Rethinking High-Level Radioactive Waste Disposal" provides a general philosophy for the approach that the Committee on Technical Bases for Yucca Mountain Standards may follow. However, the detailed calculations of high-level waste repositories that were conducted by the NAS (NAS, 1983) Waste Isolation System Panel (WISP) will likely be a starting point for technical input to the Committee. Based on these assumptions, the sensitivity analyses presented in Chapter 2 were preceded by a comparison to the WISP results (Appendix C) using UCBNE-41(the model used by the NAS for the WISP calculations). The calculations are then updated to Yucca Mountain parameters for a baseline case and parameter sensitivity analyses are conducted around this baseline.

The model UCBNE-41 was selected for use in this report because it is the basis for previous NAS calculations, it has been used by the NRC in some of their analyses of geologic repositories, and is widely accepted by the technical community. A second model, the Repository Integration Program (RIP), was selected because it has had wide application to total system performance assessment (TSPA) of Yucca Mountain by Golder Associates (Golder Associates Inc., 1993) and

by the Civilian Radioactive Waste Management System, Management and Operations Contractor in TSPA-1993 (Andrews, et al., 1994). A third model, NEFTRAN-S was selected because it is widely used by the waste management community including the Nuclear Regulatory Commission, and was used by the EPA in support of the recent re-promulgation of 40 CFR 191 (EPA, 1993b). In support of their justification for extending the ground-water protection and individual protection standards from 1,000 to 10,000 years the EPA, in reference to NEFTRAN-S states:

> "The Agency believes improvements in modeling capability since 1985 have facilitated demonstrating compliance with individual dose limits over periods longer than 1,000 years."

An even stronger reason for selecting a group of models to use in this study stems from the fact that the models mentioned above have different features and characteristics that allow them to be applied more effectively to different parts of the sensitivity analyses. For example, UCBNE-41 is a deterministic ground-water transport model in which a concentration at the waste package is input and a concentration at a down-gradient point is output. As such, it cannot incorporate a distribution of parameters in a probabilistic mode as can RIP and NEFTRAN-S. The characteristics and features of each of the models are presented in Table 1-1. Based on these characteristics and features, the types of analyses conducted with each model were determined. The types of analyses conducted using each of these models are presented in Table 1-2, and the results using UCBNE-41, RIP, and NEFTRAN-S are presented in Chapter 2, 3, and 4, respectively. The results from each model are summarized at the end of each Chapter, and are compared in the Summary and Conclusions, Chapter 5. A baseline case of dose to an individual at the accessible environment was calculated as a comparison among the three models, and as a reference case for comparison with other calculations using the same model. The dose to an individual at the accessible environment in the three models uses the same dose conversion factors which represent an individual drinking, bathing, and deriving all subsistence from the use of ground water. Chapter 5 also includes a comparison with the results of TSPA-1993, and the significance of the sensitivity analyses discussed in this report to standards setting, and to site characterization and design.

Table 1-1. Characteristics and Features of the Models UCBNE-41, RIP, and NEFTRAN-S

FEATURE	UCBNE-41	RIP	NEFTRAN-S
Solution Mode	Deterministic	Probabilistic	Probabilistic
Source Term	Concentration from spreadsheet	Waste package with advective or diffusive release	Simple waste package with advective release
Radionuclide Decay	One chain at a time	All chains at once	All chains at once
Radionuclide Solubility	Proportioned dissolution	Competing ions	Competing ions
Waste Package Failure Time	Discrete	Distribution	Distribution
Waste Package Failure Mode	All at once	All at once or sequentially	All at once
Transport Path	One-D analytical solution/single leg	One-D analytical solution/multiple legs	Multi-D analytical or numerical solution/multiple legs
Chain Decay	Chains	Chains	Chains
Biosphere	Post processed using spreadsheet	Internal	Internal

Table 1-2. Sensitivity Analyses Conducted Using UBCNE-41, RIP, and NEFTRAN-S

SENSITIVITY ANALYSES	UCBNE-41	RIP	NEFTRAN-S
Baseline Case	X	X	X
Percolation Flux	Discrete values	Distribution	Distribution
Waste Package Failure Times	Discrete values	Distribution	Distribution
Waste Package Failure Modes	All at once	All at once or sequentially	All at once
Np Solubility	Discrete values	Distribution	Distribution
Advective Release From Waste Package	X	X	X
Diffusive Release From Waste Package	X	X	
Diffusive Release From EBS		X	X

2. ANALYSES USING THE UCBNE-41 MODEL

The analyses in this chapter were conducted using the model UCBNE-41(Lung et al., 1987). This model was selected because it was the basis for the calculations conducted by the Waste Isolation System Panel (WISP) in 1983 (NAS, 1983). Parameter ranges used in these analyses were selected to be representative of Yucca Mountain, and then sensitivity analyses were conducted by varying these parameters over the expected range. Prior to beginning the sensitivity a comparison was made with the WISP results. The comparison with the WISP results provides a tie between this study and that of the NAS (Appendix C).

2.1 DESCRIPTION OF UCBNE-41

The Model UCBNE-41 is based on an analytical solution of the transport equation in a one-dimensional space, where dispersion, retardation, and chain decay of radionuclides are considered. The transport equation is written in terms of concentration; it begins with an initial concentration near the waste and provides the concentration at a given distance from the waste over a selected period of time. The evaluation requires the following initial conditions and parameters:

- The initial concentration of each radionuclide at the source (waste);
- The radionuclides considered, their half-lives, and the decay chains;
- The retardation coefficient for each radionuclide;
- The ground-water pore velocity (or the flux and the effective porosity);
- The distance to the point where final concentrations are desired;
- The time increments at which final concentrations are desired, and the total time period considered; and
- The coefficient of dispersion along the transport pathway.

In order for a transport model to be used effectively, two additional models are required; one for calculating the initial concentration of each radionuclide at the source (a source term model), and the other model for calculating the radiation dose to persons at the point being considered (a dose model). These two models are usually in the form of computer spread sheets, although more elaborate source term and dose models are available.

The source term model used here is based on the following:

- The time of failure of the waste packages;
- The inventory of each radionuclide in the waste package at the time of failure;
- The release mechanism for each radionuclide (alteration or solubility controlled);

- The alteration rate of the fuel;
- The solubility of each solubility-controlled radionuclide;
- The ground-water flux past the waste package and the effective porosity;
- The dimensions of the package;
- The number of waste packages;
- The coefficient of diffusion out of the waste package; and
- The geometry of the repository.

Given these parameters, the initial concentration of each radionuclide can be calculated for input into the transport model. The output from the transport model is then used as input to the dose assessment model. The dose assessment model requires the following parameters:

- The concentration of each radionuclide at the point of water use over time;
- The calculated dilution factor based on mixing with the saturated zone flux prior to its use;
- The water use scenario (drinking, household, farming, etc.);
- Dose conversion factors that convert radionuclide concentration in the water to dose for the given water use scenario; and
- The number of persons using the water (individual or population dose).

Because there are a large number of radionuclides in the initial inventory, a composite inventory is developed that contains only those radionuclides that are expected to be released or that produce daughters that are expected to be released in quantities sufficient to produce significant doses over the period of repository release are used in the analyses. This inventory is composed of those radionuclides that make up more than 99.9 percent of the dose from the waste in the repository. Thus, the inventory consists of those radionuclides that could produce significant dose or yield daughter radionuclides that could produce significant dose, and that can be transported over large distances before they decay.

For advective release, the fractional release for each radionuclide is determined by the following equation (NAS, 1983):

$$f_{j,A} = \frac{8 N_j D_j^{0.5} n_e U^{0.5} \left(1 + \frac{R}{L}\right)}{(\pi R)^{1.5} n_j} \quad \text{, when } \frac{UR}{D_j} > 4 \qquad \text{(Eq. 2-1)}$$

where:

$f_{j,A}$ is the fractional release rate of radionuclide j (yr^{-1});
N_j is the radionuclide solubility (g/m^3);
D_j is the diffusion coefficient (m^2/yr);
n_e is the effective porosity;
U is the pore velocity (m/yr);
R is the package radius (m);
L is the package length (m); and
n_j is the bulk density of the radionuclide in the waste (g/m^3).

When U R / D_j is less than 4, the release is no longer advection controlled, but is controlled by diffusion. The fractional release is then given by the following equation (NAS, 1983):

$$f_{j,D} = \frac{\beta n_e D_j N_j}{n_j} \qquad \text{(Eq. 2-2)}$$

where:

β is 3 / R_s^2; and
R_s is the radius of a sphere that has a surface area equal to that of the package.

The leach time is then calculated as:

Leach time = 1 / f_j (either advection or diffusion controlled; $f_j = f_{j,A}$ or $f_{j,D}$)

For radionuclides that are released through alteration of the waste (alteration limited) the leach time is simply the inverse of the alteration rate.

2.2 BASELINE CASE USING YUCCA MOUNTAIN PARAMETERS

The Yucca Mountain baseline case is different from the case that was considered in the WISP report (Appendix C). Since 1983, a decision has been made by the DOE that the repository at Yucca Mountain would be in the unsaturated zone, and the EPA regulation 40 CFR 191 (EPA, 1985b) was promulgated which reduced the distance to the accessible environment by a factor of two (5,000 m instead of 10,000 m). In addition, the actual system performance measure adopted by the EPA was integrated release of radionuclide activity over 10,000 years at the accessible environment, with the release normalized by radionuclide specific values to assure no more than 1,000 health effects over 10,000 years (0.1 per year) per 100,000 metric ton heavy metal (MTHM). The potential repository is assumed to be situated 200 m above the water table as shown in the schematic diagram of the repository in Figure 2-1. (Although recent designs

place the potential repository some 250 to 300 m above the water table, the 200 m value has been retained for the analyses presented in this report). The geometry of the repository was taken from the Site Characterization Plan (DOE, 1988b) and from the 1991 total system performance assessment by Barnard et al. (1992). The waste packages were assumed to be placed in the drifts with their long axes perpendicular to the unsaturated ground-water flux. The repository was assumed to contain 63,000 MTHM of spent fuel, and the inventory was assumed to be 30 years old at the time of repository closure. Although the repository is expected to also contain 7,000 MTHM of high-level defense waste in a glass waste form, for these calculations, this defense waste inventory was neglected. Had the defense waste been included the peak doses for ^{99}Tc and ^{237}NP would be increased by, at most, 10%.

A simplifying assumption was made that the release of radionuclides was advection controlled, and the unsaturated zone flux continued through the horizontal waste packages which are 4.76 m long and have a diameter of 0.66 m. The flux dissolves the solubility-controlled radionuclides and transports those that are alteration controlled (with an assumed alteration rate of 1.0×10^{-4} yr^{-1}). The radionuclides that are alteration controlled are ^{14}C, ^{79}Se, ^{94}Nb, ^{99}Tc, ^{129}I, and ^{135}Cs, and the remaining radionuclides are assumed to have solubility-controlled release. The radionuclides are assumed to be diluted by the flux of water through the area of the repository less the area of waste packages, and by the volume of the ground-water flow in the saturated zone [(saturated zone flux) x (thickness of the saturated zone) x (repository width)]. Here the thickness of the saturated zone is assumed to be 2400 m, the value used for the EPA calculations (EPA, 1985a and 1993a). These later EPA calculations were done in support of the new 40 CFR 191 (EPA, 1993b) which no longer applies to Yucca Mountain.

The inventory was taken to be a mix of 9.52×10^4 pressurized water reactor (PWR) assemblies which contain 0.428 metric ton uranium (MTU) each with a burn up of 42.3 GWd and 1.25×10^5 boiling water reactor (BWR) assemblies which contain 0.178 MTU each with a burn up of 32.25 GWd. This inventory was taken from the DOE Characteristics of Potential Repository Wastes "Data Base", which is a computerized version of the DOE report (DOE, 1992). The age of the waste was assumed to be 30 years at the time of closure. Table 2-1 presents a listing of the radionuclides considered, their inventory, half-lives, specific activities, and assumed solubilities.

For the baseline calculation the rate of recharge from infiltration was assumed to be 1.0 mm/yr, the pore velocity in the saturated zone was assumed to be 1 m/yr, and the effective porosity in both the saturated and unsaturated zones was assumed to be 0.1. These values yield a ground-water travel time of 25,000 years (20,000 years in the unsaturated zone and 5,000 years in the saturated zone), which is within the range of travel times expected at Yucca Mountain. These and other parameters required for the calculation of the Yucca Mountain baseline case are presented in Table 2-2 for reference by the reader. These parameters were used to calculate leach times and initial concentrations for each of the 40 radionuclides shown in Table 2-1. The solubilities shown in Table 2-1 were taken from the 1991 TSPA (Barnard et. al., 1992) and from the WISP study (NAS, 1983), and are proportioned on the basis of the mass of each isotope of a particular element that is present in the inventory at 30 years.

The retardation factors that were assumed were taken largely from the WISP report (NAS, 1983), and were modified somewhat based on the parameters from the 1991 TSPA (Barnard et. al.,

1992). This resulted in lowering the retardation factors for selenium and neptunium, and raising those for tin, plutonium, and americium. Dose conversion factors were taken largely from the values used by Pacific Northwest Laboratories (PNL) in their 1991 TSPA (Eslinger et al., 1993). The retardation and dose conversion factors used in the Yucca Mountain baseline case are presented in Table 2-3. The initial concentrations and leach times from Table 2-1, the parameters from Table 2-2, and the retardation and dose conversion factors from Table 2-3 were used along with the model UCBNE-41 and a spread sheet dose model to determine the dose to an individual at the accessible environment (5,000 m down gradient from the repository). These doses are plotted as a function of time in Figure 2-2. The dose to an individual peaks beyond 10,000 yrs, with the first arrival being composed of ^{14}C and ^{129}I. The early arrival (i.e., ahead of the ground-water travel time of 25,000 years) for ^{14}C and ^{129}I is caused by dispersion (which accounts for some transport pathways being faster). The estimated dose from ^{14}C is too high because a significant portion of it would have been transported in the gaseous phase, which cannot be analyzed using UCBNE-41. The dose to an individual peaks at about 1×10^{-3} Sv/yr (100 mrem/yr) and drops below 1×10^{-4} Sv/yr (10 mrem/yr) at about 4×10^{6} yrs (with the exception of two short periods of time on either side of the ^{99}Tc peak (Figure 2-2). This case is referred to as the baseline case because it will serve as the basis for all of the sensitivity analyses that follow.

The parameters used for the baseline case are within the range of parameters expected for the Yucca Mountain site. However one could argue that the percolation flux is too large, the depth of mixing in the saturated zone is too large, and the solubility of the actinides is too small. The ranges of the parameters percolation flux and solubility are addressed in the sensitivity analyses that follow. The depth of mixing of the radionuclides (2,400 m) over the entire saturated thickness is uncertain. The effect of mixing to a depth of 1,000 meters would increase the peak doses by a factor of 2.4, and the effect of mixing only to a depth of 100 meters in the saturated zone would increase doses by a factor of 25. The variation of the factor of 25 from the expected value of 24 is caused by the fact that the concentration is also diluted by the flux through the footprint of the repository which does not directly pass through the waste packages.

There are two ways to calculate the initial concentration for input to the transport model. The first is that of the WISP report, in which either Equation 2-1 or Equation 2-2 is used, and the second is to assume advection directly through the package as though it were a porous medium. In order to make a comparison between these two approaches, it must first be determined which equation for fractional release governs (Equation 2-1 or 2-2). This may be done as follows:

The R_s^2 of the β term of Equation 2-2 is determined by equating the surface area of a sphere to the surface area of a cylinder;

$$4 \pi R_s^2 = 2 \pi R L + 2 \pi R^2 \quad \text{or}$$

$$R_s^2 = 1/2 \; R \; (L+R)$$

To compare the fractional release from advection with the fractional release from diffusion [to answer the question of whether advective fractional release is greater than diffusive fractional release (?/>)] , the expression for R_s^2 is substituted into Equation 2-2.

$$f_{j,A} \stackrel{?}{>} f_{j,D}$$

$$\frac{8\, N_j\, D_j^{0.5}\, n_e\, U^{0.5}\left(1+\frac{R}{L}\right)}{(\pi R)^{1.5}\, n_j} \stackrel{?}{>} \frac{3\, n_e\, D_j\, N_j}{0.5\, R\, (L+R)\, n_j}$$

Immediately the terms N_j , n_e, and n_j drop out. Then multiplying both sides by $R/D_j^{0.5}$ yields

$$\frac{8\, U^{0.5}\left(1+\frac{R}{L}\right)}{\pi^{1.5}\, R^{0.5}} \stackrel{?}{>} \frac{6\, D_j^{0.5}}{L+R} \qquad \text{(Eq. 2-3)}$$

Note that;

- The only dependencies are on the waste package dimensions (R + L), the pore velocity (U), and the diffusion coefficient (D_j),
- For a given package, whether advection or diffusion dominates will be determined by the relative values of U and D_j; and
- R = 0.33 m and L = 4.76 m

Then:

$$\frac{8\, U^{0.5}\left(1+\frac{R}{L}\right)}{\pi^{1.5}\, R^{0.5}} = 2.67\, U^{0.5} \quad \text{and} \quad \frac{6\, D_j^{0.5}}{L+R} = 1.18\, D_j^{0.5}$$

$$2.67\, U^{0.5} \stackrel{?}{>} 1.18 D_j^{0.5}$$

Using the parameters from Table 2-2 in Equation 2-3 above and a coefficient of diffusion of 10^{-3} m^2/yr (which is appropriate for partially saturated conditions), yields;

$$0.267\ yr^{-1} > 0.0373\ yr^{-1}$$

or

$$f_{j,A} > f_{j,D}$$

Thus, advection will always dominate in the fractional release assuming an advective flux of 10^{-3} m/yr and an effective porosity of 0.1. Because of the power of 0.5 dependance, either U would have to decrease by a factor of 100, or D_j would have to increase by a factor of 100, before the mechanism of diffusion would dominate the fractional release from the waste package. For recharge rates at Yucca Mountain in the range from 0.1 mm/yr to about 1.0 mm/yr, advective release will dominate (unless an engineered barrier is designed to limit advective flux and make the near field environment diffusion dominated).

Table 2-4 shows the leach times and the initial concentration of radionuclides based on Equation 2-1. Because the radionuclides ^{14}C, ^{79}Se, ^{94}Nb, ^{99}Tc, ^{129}I, and ^{135}Cs are assumed to be alteration controlled, the initial concentrations of these radionuclides remain the same as in the baseline case shown in Table 2-1. The coefficient of diffusion is assumed to be 1.0×10^{-3} m^2/yr, and all other parameters remain the same as in the baseline case. The value for the diffusion coefficient was selected because it was the value used by the WISP. A range of values for the diffusion coefficient (1.0×10^{-3} to 1.0×10^{-5} m^2/yr) were used to obtain the doses to an individual at the accessible environment over the period of repository release that are presented in Figure 2-3 for solubility-limited radionuclides. The dose from ^{231}Pa is nearly the same as for ^{227}Ac here and in the analyses presented in the Sections that follow. The doses from the alteration-limited radionuclides are not affected because they are released at the alteration rate. The doses from the solubility-controlled radionuclides are similar to those of the baseline case (Figure 2-2).

For the remaining analyses in this chapter the simplifying assumption of advective release by flux through the waste package and dilution with recharge through the repository area was used.

2.3 SENSITIVITY TO GROUND-WATER TRAVEL TIME

The recharge to the unsaturated zone at Yucca mountain is uncertain. The translation of this recharge to the repository level percolation flux is also uncertain. Recharge may be as low as 0.1 mm/yr (or lower) and may range to as high as a few millimeters per year. To investigate the effects of recharge (infiltration, below the root zone), on dose to an individual, the percolation rates of 4.0 mm/yr and 0.21 mm/yr were selected for analysis. This range of percolation results in a ground-water travel time through the 200 m unsaturated zone of 5,000 and 95,000 years for 4.0 and 0.21 mm/yr percolation rate, respectively, assuming an effective porosity of 0.1. The ground-water travel time to the accessible environment in the saturated zone remains at 5,000 years, and all other parameters remain the same as in the baseline case. The dose to an individual at the accessible environment for 10,000 and 100,000 year total ground-water travel time are presented in Figures 2-4 and 2-5, respectively. For the 10,000 year travel time the peak doses occur earlier than in the baseline case (Figure 2-2) which has a travel time of 25,000 years. The dose due to solubility-controlled radionuclides increases over the baseline case because there is more flux to dissolve more element mass. The alteration-controlled radionuclides increase somewhat because there is less time for decay and dispersion. The lower dispersion causes the "peaks" to be narrower. For these two cases the dispersion coefficient was assumed to be 50 m^2/yr which effectively implies increasing the dispersivity value for the lower pore velocity (high water travel time) case.

In the 100,000 yr travel time case (Figure 2-5) the peak doses are considerably lower because of radionuclide decay for the shorter half-life radionuclides. In this case, however, the

assumption of no change of dispersion coefficient has a considerable effect. This is responsible for the early arrival of radionuclides especially ^{14}C and ^{129}I. Here it is interesting to note that in the case of ^{14}C, release occurred prior to significant decay because of the high dispersion even though the mean water travel time was 100,000 years. In order to investigate this, another simulation of 100,000 year travel time was conducted in which the dispersion was decreased to 12.5 m^2/yr. This simulation has the same dispersivity as the baseline case and the dispersion is modified to be in agreement with the pore velocity. The coefficient of dispersion is defined by:

$$D = \alpha V \qquad \text{(Eq. 2-4)}$$

where D is the coefficient of dispersion (m^2/yr), α is the dispersivity (m), and V is the pore velocity (m/yr). The result of simulating the 100,000 yr travel time and the correct dispersion are shown in Figure 2-6. Note that the time to the first arrival of all radionuclides is increased, and the peak doses of the shorter half-life radionuclides is decreased significantly because there is more time for radioactive decay. Decay is even apparent for ^{135}Cs, which has over a two million year half-life, because its arrival at the accessible environment is shifted a few million years out in time due the correct dispersion and to its high sorption.

2.4 SENSITIVITY TO WASTE PACKAGE LIFETIME

The waste package and the engineered barrier system of a geologic repository are designed to contain radionuclides until the shorter half-life radionuclides have decayed. The requirement of the Nuclear Regulatory Commission(NRC), 10 CFR 60 (NRC, 1991), of a 300 year waste package would allow ten half-lives of decay of radionuclides such as ^{90}Sr and ^{137}Cs (which have approximately 30 yr half-lives). In the Yucca Mountain Project waste packages that could endure for thousands to tens of thousands of years are under consideration.

The baseline case considered a zero waste package lifetime, and additional simulations were conducted to examine the effect of package lifetime on dose to an individual at the accessible environment. Three additional simulations were conducted for package lifetimes of 1,000, 10,000, and 100,000 years. All of the parameters used in the baseline case remained the same in these simulations except the source term, which was decayed to simulate the package lifetime. The results for the 10,000 and 100,000 year waste packages are shown in Figure 2-7 and Figure 2-8, respectively. The results of the 1,000 year waste package are not shown because they appear identical to the baseline case (i.e., there is not enough change from the baseline case to appear because of the log scale of the plot).

It is interesting to note that the baseline case (Figure 2-2) and the 10,000 year waste package (Figure 2-7) are nearly identical except the peaks are moved out in time by 10,000 years. In addition, the peak ^{14}C dose is reduced by about a factor of four, corresponding to the two half-lives that have passed prior to the container being breached. For a 100,000 year package, the effect of time is more dramatic for ^{14}C (dose is reduced by more than four orders of magnitude). The small amount of ^{227}Ac is an artifact of the modeling. The leach time of ^{227}Ac (about 10^3 yrs) was assigned to the entire chain. Had the chain been assigned the longer leach time of ^{239}Pu (about 10^8 yrs) the results for ^{227}Ac would have been similar to those in Figure 2-7, but the dose would have been somewhat lower. With the exception of these two radionuclides, the primary effect of a 100,000 year waste package is only to shift the doses farther out in time. The

apparent narrowing of the peaks is due to the log plot. The integral under the curve for the long-lived radionuclides remains essentially unchanged. Thus the effects of a long package lifetime

on dose to an individual at the accessible environment from the long-lived radionuclides such as ^{129}I and ^{237}Np is small. It only occurs at a later time.

2.5 SENSITIVITY TO SOLUBILITY

In Section 2.3 the sensitivity of individual dose to flux past the waste package (ground-water travel time) was shown. This sensitivity indicates that individual dose is sensitive to the behavior of the source term. In addition, the solubilities of Np, Cm, Am, Pu, and U are low in comparison to the values used in TSPA-1993 (Andrews et al., 1994). The solubilities for Cm, Am, Pu, and U are low by a factor of 100, and the solubility for Np is low by a factor of as much as 100,000. To investigate the effects of these increased solubilities on the baseline case, three additional simulations were conducted. The first simulation increased the solubility values of all of these radionuclides (Np, Cm, Am, Pu, and U) by a factor of 100. The results of this simulation are presented in Figure 2-9. This figure indicates that ^{227}Ac and ^{237}Np doses increased by a factor of 100 over the baseline case (Figure 2-2), which would be expected. The width of the ^{237}Np peak decreases by about an order of magnitude from the baseline case (compare Figure 2-2 and 2-9).

The second simulation is the same as the first except the solubility of Np was increased by a factor of 1,000 (to 1.0 g/m^3) over the baseline case (e.g., Cm, Am, Pu, and U are increased by a factor of 100; and Np is increased by a factor of 1,000). The results of this simulation are shown in Figure 2-10, and as expected the dose from ^{237}Np increased by a factor of 1,000 over the baseline case. The third simulation increased the solubility of Np by a factor of 100,000 (to 100 g/m^3) over the baseline case (e.g., Cm, Am, Pu, and U are increased by a factor of 100; and Np is increased by a factor of 100,000). The results of this simulation are shown in Figure 2-11, and are somewhat unexpected. The dose to an individual did not increase by a factor of 100,000 over the baseline case, but remained similar to the case when the neptunium solubility was increased by 1,000 (to 1.0 g/m^3, Figure 2-10). This behavior is caused by the release of Np over a relatively short period of time (in approximately 4000 years as compared to 10^8 years, Table 2-1). Another way of viewing this result is that there is not enough Np to increase the Np dose by a factor of 100,000 over that of the baseline case. Doses from Cm, Am, Pu, and U are not significant as compared to Np (see Section 3.2 and Section 4.2).

A retardation factor of 16 was used for ^{237}Np in the baseline case and in the solubility sensitivity analyses. Because the long-term dose to an individual is dependent on ^{237}Np, it was selected for a sensitivity analysis to retardation factor. For this simulation the retardation factor for Np was reduced from 16 to 2 while all other parameters remained the same as in the baseline case. The results of this simulation are presented in Figure 2-12. This reduction in retardation causes Np to arrive earlier and the peak to be broader than in the baseline case. In addition, the peak is slightly higher than in the baseline case. This is because the peak occurs earlier, in about 10^5 yrs (Figure 2-12) instead of 10^6 years (Figure 2-2), which means that there is less time for decay.

2.6 THE NUCLEAR REGULATORY COMMISSION (NRC) RELEASE LIMIT

The Nuclear Regulatory Commission in 10 CFR 60 requires that the annual rate of release of any radionuclide not exceed one part in 100,000 of the inventory of that radionuclide calculated to be present at 1,000 years after closure (NRC, 1991). To investigate how exactly meeting this requirement would affect the individual dose at the accessible environment, an alteration rate for all radionuclides of 10^{-5} yr^{-1} was assumed and the inventory at 1,000 years was used. All other parameters of the baseline case remained the same. The results of this simulation are presented in Figure 2-13. Here the alteration-controlled radionuclides of the baseline case (^{14}C, ^{99}Tc, ^{129}I, ^{79}Se, and ^{135}Cs) decreased in individual dose somewhat because their annual release rate was reduced from 10^{-4} to 10^{-5}. Individual dose from the radionuclides ^{237}Np and ^{227}Ac increased substantially over the baseline case (Figure 2-2), and their peaks became narrower (because they are behaving like alteration-controlled instead of solubility-limited radionuclides). It is also interesting to note that the peak dose from ^{237}Np in Figure 2-13 is somewhat higher than the peak dose when the ^{237}Np solubility was increased by a factor of 100,000 (Figure 2-11). In the discussion in Section 2.5 the case was made that the dose from ^{237}Np could not increase because all of it had been released rapidly over about 4,000 years. In this case (Figure 2-11) the parent of Np was released at a rate lower than 10^{-5}. Thus, in the case of the annual release rate of 10^{-5} (Figure 2-13) the higher release rate of the parents of neptunium (^{241}Am and ^{237}U) more than makes up for the lower release of ^{237}Np to produce a higher individual dose rate. In considering Figure 2-13, it should be remembered that the doses calculated are unrealistic because Np, Ac, and their parents are generally solubility limited and would not be released as rapidly as the NRC limit.

2.7 SUMMARY

The results obtained by the NAS Waste Isolation System Panel for a saturated tuff site were first duplicated using the model UCBNE-41(See Appendix C). The model was then used to simulate a baseline case for a repository in the unsaturated zone at Yucca Mountain, and then to conduct sensitivity analyses around this baseline case. The sensitivity of dose to an individual at the accessible environment was examined for the following: mode of release from the package (advection through the package compared to diffusion out of the package); dilution; ground-water travel time (percolation flux); waste package lifetime; and the solubility of neptunium and other actinides. The effect of exactly complying with the NRC annual release limit of 10^{-5} of the inventory at 1,000 years was investigated. In addition, the effects of neptunium retardation and of dispersion were investigated to a limited extent.

The primary radionuclides responsible for dose to an individual at the accessible environment are ^{14}C, ^{99}Tc, ^{129}I, and ^{237}Np. For all of the simulations ^{14}C was assumed to be released by way of the ground-water pathway when in fact it will largely be released by way of the gaseous pathway at Yucca Mountain. This means that the ^{14}C releases shown are higher than would actually occur by way of the ground-water pathway, and for this reason they are not summarized here.

The peak doses to an individual from ^{99}Tc, ^{129}I, and ^{237}Np, and their time of occurrence, are presented in Table 2-5 for selected simulations. Simulations such as those with extremely high dispersion coefficients and those based on the NRC release limit are not included because the parameters and mechanisms of release, respectively, are not realistic.

The effect of dilution is also not shown in Table 2-5, but can be easily calculated for each of the simulations . The simulations presented in Table 2-5 are based on dilution by mixing the release to a depth of 2400 meters in the saturated zone. The effect of mixing to a depth of 1,000 meters

would increase the peak doses by a factor of 2.4, and the effect of mixing only to a depth of 100 meters in the saturated zone would increase doses by a factor of 25.

The effects of ground-water travel time on dose to an individual at the accessible environment are summarized in Figure 2-14. Doses for shorter travel times are higher because of increased flux through the waste package and less radioactive decay. Over the range of ground-water travel times of 10,000 to 100,000 years, dose to an individual from neptunium decreases by a factor of 2.5, iodine reduces by a factor of 15, and technetium reduces by a factor of 220.

The sensitivity to waste package lifetime of dose to an individual from ^{99}Tc, ^{129}I, and ^{237}Np is presented in Figure 2-15. Here the effect of the waste package is only to allow for decay of radionuclides before they are transported to the accessible environment. The doses from ^{99}Tc, ^{129}I, and ^{237}Np remain of the same order of magnitude for the range of waste package lifetime of zero to 100,000 years. This is because all of these radionuclides have half-lives greater than 100,000 years (Table 2-1).

Figure 2-16 shows the effects of solubility and retardation factor on dose to an individual at the accessible environment from ^{237}Np. The solubility of neptunium for the baseline case is 0.001 g/m^3 and this value is increased by a factor of 100,000 (to 100 g/m^3). Here it should be noted that the last factor of 100 increase in solubility does not produce a factor of 100 increase in dose. This is because all of the neptunium is released over a short period of time and thereafter there is not enough neptunium to cause a 100 fold increase in dose. Also, the solubility of the parents of neptunium were only increased by a factor of 100 over the baseline case, while neptunium was increased by a factor of 100,000.

A summary of the peak doses from ^{99}Tc, ^{129}I, and ^{237}Np for all of the simulations listed in Table 2-5 is presented in Figure 2-17. This figure indicates that for realistic values of parameters, doses from ^{99}Tc, ^{129}I, and ^{237}Np occur from 10,000 to 1,000,000 years after repository closure and range from millirems to tens of rems per year. Higher doses could be calculated by considering less dilution, as discussed above.

Table 2-1. Source Term Information Used for the Yucca Mountain Baseline Case

RADIO-NUCLIDE	HALF-LIFE [1] yrs	INVEN-TORY Ci/MTHM	SPECIFIC-ACTIVITY Bq/g	SOLU-BILITY g/m^3	LEACH TIME yrs	INIT. CONC. g/m^3
^{14}C	5.73×10^{3}	1.48×10^{0}	1.65×10^{11}		1.00×10^{4}	2.22×10^{-2}
^{79}Se	6.50×10^{4}	4.80×10^{-1}	2.58×10^{9}		1.00×10^{4}	4.60×10^{-1}
^{94}Nb	2.03×10^{4}	8.91×10^{-1}	6.94×10^{9}		1.00×10^{4}	3.18×10^{-1}
^{99}Tc	2.13×10^{5}	1.51×10^{1}	6.28×10^{8}		1.00×10^{4}	5.95×10^{1}
^{126}Sn	1.00×10^{5}	9.25×10^{-1}	1.05×10^{9}	1.00×10^{-3}	2.18×10^{7}	1.00×10^{-3}
^{129}I	1.57×10^{7}	3.72×10^{-2}	6.54×10^{6}		1.00×10^{4}	1.41×10^{1}
^{135}Cs	2.30×10^{6}	5.67×10^{-1}	4.26×10^{7}		1.00×10^{4}	3.29×10^{1}
^{243}Am	7.38×10^{3}	2.82×10^{1}	7.38×10^{9}	1.10×10^{-5}	8.59×10^{9}	1.10×10^{-5}
^{243}Cm	2.85×10^{1}	1.70×10^{1}	1.91×10^{12}	1.64×10^{-5}	1.34×10^{7}	1.64×10^{-5}
^{239}Np	6.45×10^{-3}	2.82×10^{1}	8.58×10^{15}	1.76×10^{-10}	4.61×10^{8}	1.76×10^{-10}
^{239}Pu	2.41×10^{4}	3.75×10^{2}	2.30×10^{9}	6.25×10^{-4}	6.45×10^{9}	6.25×10^{-4}
^{235}U	7.04×10^{8}	1.68×10^{-2}	8.00×10^{4}	8.21×10^{-6}	6.32×10^{11}	8.21×10^{-6}
^{231}Th	2.91×10^{-3}	1.68×10^{-2}	1.97×10^{16}	1.37×10^{-9}	1.54×10^{4}	1.37×10^{-9}
^{231}Pa	3.28×10^{4}	3.59×10^{-5}	1.75×10^{9}	9.70×10^{-4}	5.23×10^{2}	9.70×10^{-4}
^{227}Ac	2.18×10^{1}	1.97×10^{-5}	2.68×10^{12}	1.00×10^{-4}	1.82×10^{0}	1.00×10^{-4}
^{244}Cm	1.76×10^{1}	1.41×10^{3}	3.08×10^{12}	8.44×10^{-4}	1.34×10^{7}	8.44×10^{-4}
^{240}Pu	6.54×10^{3}	5.73×10^{2}	8.44×10^{9}	2.60×10^{-4}	6.45×10^{9}	2.60×10^{-4}
^{236}U	2.34×10^{7}	2.93×10^{-1}	2.40×10^{6}	4.78×10^{-6}	6.32×10^{11}	4.78×10^{-6}
^{232}Th	1.41×10^{10}	4.71×10^{-10}	4.06×10^{3}	1.86×10^{-4}	1.54×10^{4}	1.86×10^{-4}
^{245}Cm	8.50×10^{3}	4.27×10^{-1}	6.36×10^{9}	1.24×10^{-4}	1.34×10^{7}	1.24×10^{-4}
^{241}Pu	1.44×10^{1}	3.56×10^{4}	3.81×10^{12}	3.58×10^{-5}	6.45×10^{9}	3.58×10^{-5}
^{241}Am	4.32×10^{2}	3.92×10^{3}	1.27×10^{11}	8.88×10^{-5}	8.59×10^{9}	8.88×10^{-5}
^{237}U	1.85×10^{-2}	8.73×10^{-1}	3.02×10^{15}	1.13×10^{-14}	6.32×10^{11}	1.13×10^{-14}

[1] Conversion factors: 3.7×10^{10} Bq/Ci, flux past waste is 94.3 m^3/yr, and 63,000 MTHM.

Table 2-1. Source Term Information Used for the Yucca Mountain Baseline Case (Continued)

RADIO-NUCLIDE	HALF-LIFE [2] yrs	INVEN-TORY Ci/MTHM	SPECIFIC-ACTIVITY Bq/g	SOLU-BILITY g/m^3	LEACH TIME yrs	INIT. CONC. g/m^3
^{237}Np	2.14×10^{6}	4.87×10^{-1}	2.61×10^{7}	1.00×10^{-3}	4.61×10^{8}	1.00×10^{-3}
^{233}Pa	7.40×10^{-2}	4.87×10^{-1}	7.68×10^{14}	3.00×10^{-5}	5.23×10^{2}	3.00×10^{-5}
^{233}U	1.59×10^{5}	7.82×10^{-5}	3.58×10^{8}	8.53×10^{-12}	6.32×10^{11}	8.53×10^{-12}
^{229}Th	7.34×10^{3}	4.32×10^{-7}	7.88×10^{9}	8.80×10^{-8}	1.54×10^{4}	8.80×10^{-8}
^{225}Ra	4.05×10^{-2}	4.33×10^{-7}	1.45×10^{15}	6.96×10^{-9}	1.06×10^{0}	6.96×10^{-9}
^{246}Cm	4.73×10^{3}	9.38×10^{-2}	1.14×10^{10}	1.52×10^{-5}	1.34×10^{7}	1.52×10^{-5}
^{242m}Am	1.52×10^{2}	2.34×10^{1}	3.60×10^{11}	1.87×10^{-7}	8.59×10^{9}	1.87×10^{-7}
^{242}Am	1.83×10^{-3}	2.33×10^{1}	2.99×10^{16}	2.24×10^{-12}	8.59×10^{9}	2.24×10^{-12}
^{242}Cm	4.46×10^{-1}	1.93×10^{1}	1.23×10^{14}	2.90×10^{-7}	1.34×10^{7}	2.90×10^{-7}
^{242}Pu	3.76×10^{5}	2.18×10^{0}	1.45×10^{8}	5.75×10^{-5}	6.45×10^{9}	5.75×10^{-5}
^{238}Pu	8.77×10^{1}	3.57×10^{3}	6.34×10^{11}	2.16×10^{-5}	6.45×10^{9}	2.16×10^{-5}
^{238}U	4.47×10^{9}	3.14×10^{-1}	1.24×10^{4}	9.87×10^{-4}	6.32×10^{11}	9.87×10^{-4}
^{234}Th	6.60×10^{-2}	3.15×10^{-1}	8.57×10^{14}	5.90×10^{-7}	1.54×10^{4}	5.90×10^{-7}
^{234}U	2.45×10^{5}	1.43×10^{0}	2.31×10^{8}	2.42×10^{-7}	6.32×10^{11}	2.42×10^{-7}
^{230}Th	7.70×10^{4}	3.79×10^{-4}	7.47×10^{8}	8.13×10^{-4}	1.54×10^{4}	8.13×10^{-4}
^{226}Ra	1.60×10^{3}	2.64×10^{-6}	3.66×10^{10}	1.00×10^{-2}	1.78×10^{-1}	1.00×10^{-2}
^{210}Pb	2.23×10^{1}	7.15×10^{-7}	2.83×10^{12}	1.00×10^{-1}	6.25×10^{-5}	1.00×10^{-1}

[2] See EPA (1985a)

Table 2-2. Parameters Used for the Yucca Mountain Baseline Case

PARAMETER	VALUE
Recharge Rate (Percolation flux)	1.0 mm/yr
Pore velocity saturated zone	1.0 m/yr
Distance to accessible environment	5,000 m
Distance to saturated zone	200 m
Thickness of saturated zone	2,400 m
Effective porosity	0.10
Repository area	5.75×10^{6} m^2
Repository width	3,400 m
Number of waste packages	30,000
Repository capacity	63,000 MTHM
Age of spent fuel	30 yrs
Waste package length	4.76 m
Waste package diameter	0.66 m
Total flux past waste packages	9.43×10^{1} m^3/yr
Dilution factor	1.15×10^{-4}
Dispersion coefficient	50 m^2/yr
Alteration rate of release	1.0×10^{-4} yr^{-1}

Table 2-3. Retardation Factors and Dose Conversion Factors Used for the Yucca Mountain Baseline Case

RADIONUCLIDE	RETARDATION FACTOR	DOSE CONVERSION FACTOR ($Sv\text{-}m^3$ / Bq-yr)
^{14}C	1	1.04×10^{-7}
^{79}Se	25	4.25×10^{-7}
^{94}Nb	50	3.49×10^{-7}
^{99}Tc	5	1.23×10^{-8}
^{126}Sn	1500	5.79×10^{-7}
^{129}I	1	3.13×10^{-7}
^{135}Cs	150	2.89×10^{-7}
^{243}Am	1500	2.20×10^{-6}
^{243}Cm	500	1.53×10^{-6}
^{239}Np	16	3.69×10^{-8}
^{239}Pu	1820	5.40×10^{-8}
^{235}U	40	1.85×10^{-8}
^{231}Th	5000	1.98×10^{-10}
^{231}Pa	1500	6.43×10^{-6}
^{227}Ac	1500	8.54×10^{-6}
^{244}Cm	500	1.22×10^{-6}
^{240}Pu	1820	9.80×10^{-9}
^{236}U	40	1.87×10^{-8}
^{232}Th	5000	4.00×10^{-7}
^{245}Cm	500	2.27×10^{-6}
^{241}Pu	1820	1.04×10^{-9}
^{241}Am	1500	2.21×10^{-6}
^{237}U	40	2.18×10^{-10}

Table 2-3. Retardation Factors and Dose Conversion Factors Used for the Yucca Mountain Baseline Case (Continued)

RADIONUCLIDE	RETARDATION FACTOR	DOSE CONVERSION FACTOR ($Sv\text{-}m^3$ / Bq-yr)
^{237}Np	16	5.02×10^{-5}
^{233}Pa	1500	2.20×10^{-9}
^{233}U	40	3.80×10^{-8}
^{229}Th	5000	5.56×10^{-7}
^{225}Ra	500	1.40×10^{-6}
^{246}Cm	500	2.25×10^{-6}
^{242m}Am	1500	2.13×10^{-6}
^{242}Am	1500	8.56×10^{-10}
^{242}Cm	500	6.97×10^{-8}
^{242}Pu	1820	9.50×10^{-9}
^{238}Pu	1820	4.89×10^{-8}
^{238}U	40	2.91×10^{-8}
^{234}Th	5000	2.00×10^{-9}
^{234}U	40	1.97×10^{-8}
^{230}Th	5000	8.03×10^{-8}
^{226}Ra	500	2.40×10^{-6}
^{210}Pb	50	3.07×10^{-5}

Table 2-4. Leach Times and Initial Concentrations for the Baseline Case Modified by Assuming Advective-Diffusive Releases from the Waste Package

RADIO-NUCLIDE	LEACH TIME (yrs)	INITIAL CONC. (g/m^3)	RADIO-NUCLIDE	LEACH TIME (yrs)	INITIAL CONC. (g/m^3)
^{14}C	1.00×10^{4}	2.22×10^{-2}	^{241}Pu	1.21×10^{9}	1.90×10^{-4}
^{79}Se	1.00×10^{4}	4.60×10^{-1}	^{241}Am	1.62×10^{9}	4.72×10^{-4}
^{94}Nb	1.00×10^{4}	3.18×10^{-1}	^{237}U	1.19×10^{11}	6.01×10^{-14}
^{99}Tc	1.00×10^{4}	5.95×10^{1}	^{237}Np	8.68×10^{7}	5.31×10^{-3}
^{126}Sn	1.00×10^{4}	1.00×10^{-3}	^{233}Pa	9.84×10^{1}	1.59×10^{-4}
^{129}I	1.00×10^{4}	1.41×10^{1}	^{233}U	1.19×10^{11}	4.53×10^{-11}
^{135}Cs	1.00×10^{4}	3.29×10^{1}	^{229}Th	2.90×10^{3}	4.68×10^{-7}
^{243}Am	1.62×10^{9}	5.95×10^{-5}	^{225}Ra	2.00×10^{-1}	3.70×10^{-8}
^{243}Cm	2.52×10^{6}	8.72×10^{-5}	^{246}Cm	2.52×10^{6}	8.08×10^{-5}
^{239}Np	8.68×10^{7}	9.35×10^{-10}	^{242m}Am	1.62×10^{9}	1.19×10^{-11}
^{239}Pu	1.21×10^{9}	3.32×10^{-3}	^{242}Am	1.62×10^{9}	9.95×10^{-7}
^{235}U	1.19×10^{11}	4.36×10^{-5}	^{242}Cm	2.52×10^{6}	1.54×10^{-6}
^{231}Th	2.90×10^{3}	7.28×10^{-9}	^{242}Pu	1.21×10^{9}	3.06×10^{-4}
^{231}Pa	9.84×10^{1}	5.15×10^{-3}	^{238}Pu	1.21×10^{9}	1.15×10^{-4}
^{227}Ac	3.42×10^{1}	5.31×10^{-4}	^{238}U	1.19×10^{11}	5.24×10^{-3}
^{244}Cm	2.52×10^{6}	4.49×10^{-3}	^{234}Th	2.90×10^{3}	3.13×10^{-6}
^{240}Pu	1.21×10^{9}	1.38×10^{-3}	^{234}U	1.19×10^{11}	1.28×10^{-6}
^{236}U	1.19×10^{11}	2.54×10^{-5}	^{230}Th	2.90×10^{3}	4.32×10^{-3}
^{232}Th	2.90×10^{3}	9.68×10^{-4}	^{226}Ra	3.36×10^{-2}	5.31×10^{-2}
^{245}Cm	2.52×10^{6}	6.59×10^{-4}	^{210}Pb	1.18×10^{-5}	5.31×10^{-1}

Table 2-5. Summary of Peak Dose to an Individual from ^{99}Tc, ^{129}I, and ^{237}Np and Time of Occurrence for the Peaks for Each Simulation

Simulation	^{99}Tc		^{129}I		^{237}Np	
	Peak Dose (Sv/yr)	**Time (yrs)**	**Peak Dose (Sv/yr)**	**Time (yrs)**	**Peak Dose (Sv/yr)**	**Time (yrs)**
Baseline Fig. 2-2	4.0×10^{-3}	1.0×10^{5}	1.7×10^{-3}	2.5×10^{4}	1.5×10^{-4}	6.3×10^{5}
Waste Package Diffusive Release, $D=10^{-3}$ m^2/yr Fig. 2-3	4.0×10^{-3} (not on Fig.)	1.0×10^{5} (not on Fig.)	1.7×10^{-3} (not on Fig.)	2.5×10^{4} (not on Fig.)	7.8×10^{-4}	6.3×10^{5}
Waste Package Diffusive Release, $D=10^{-4}$ m^2/yr Fig. 2-3	4.0×10^{-3} (not on Fig.)	1.0×10^{5} (not on Fig.)	1.7×10^{-3} (not on Fig.)	2.5×10^{4} (not on Fig.)	2.5×10^{-4}	6.3×10^{5}
Waste Package Diffusive Release, $D=10^{-5}$ m^2/yr Fig. 2-3	4.0×10^{-3} (not on Fig.)	1.0×10^{5} (not on Fig.)	1.7×10^{-3} (not on Fig.)	2.5×10^{4} (not on Fig.)	7.8×10^{-5}	6.3×10^{5}
Ground-water travel time=10^4k years Fig. 2-4	1.8×10^{-2}	5.0×10^{4}	3.2×10^{-3}	1.6×10^{4}	6.8×10^{-4}	2.5×10^{5}
Ground-water travel time=10^5k years Fig. 2-5	4.0×10^{-4}	4.0×10^{5}	4.7×10^{-4}	1.0×10^{5}	1.7×10^{-5}	2.5×10^{6}
Waste Package lifetime=10^4k years Fig. 2-7	3.9×10^{-3}	1.1×10^{5}	1.7×10^{-3}	3.5×10^{4}	2.3×10^{-4}	6.4×10^{5}

Table 2-5. Summary of Peak Dose to an Individual from ^{99}Tc, ^{129}I, and ^{237}Np and Time of Occurrence for the Peaks for Each Simulation (Continued)

Simulation	^{99}Tc		^{129}I		^{237}Np	
	Peak Dose (Sv/yr)	**Time (yrs)**	**Peak Dose (Sv/yr)**	**Time (yrs)**	**Peak Dose (Sv/yr)**	**Time (yrs)**
Waste Package lifetime=10^5k years Fig. 2-8	2.9×10^{-3}	2.0×10^5	1.7×10^{-3}	1.3×10^5	1.2×10^{-4}	7.3×10^5
Np Sol=0.1 g/m^3 Fig. 2-9					1.2×10^{-2}	6.3×10^5
Np Sol=1.0 g/m^3 Fig. 2-10					1.2×10^{-1}	6.3×10^5
Np Sol=100 g/m^3 Fig. 2-11					2.2×10^{-1}	3.2×10^5
Np R_d=2 Fig. 2-12					1.8×10^{-4}	1.0×10^5

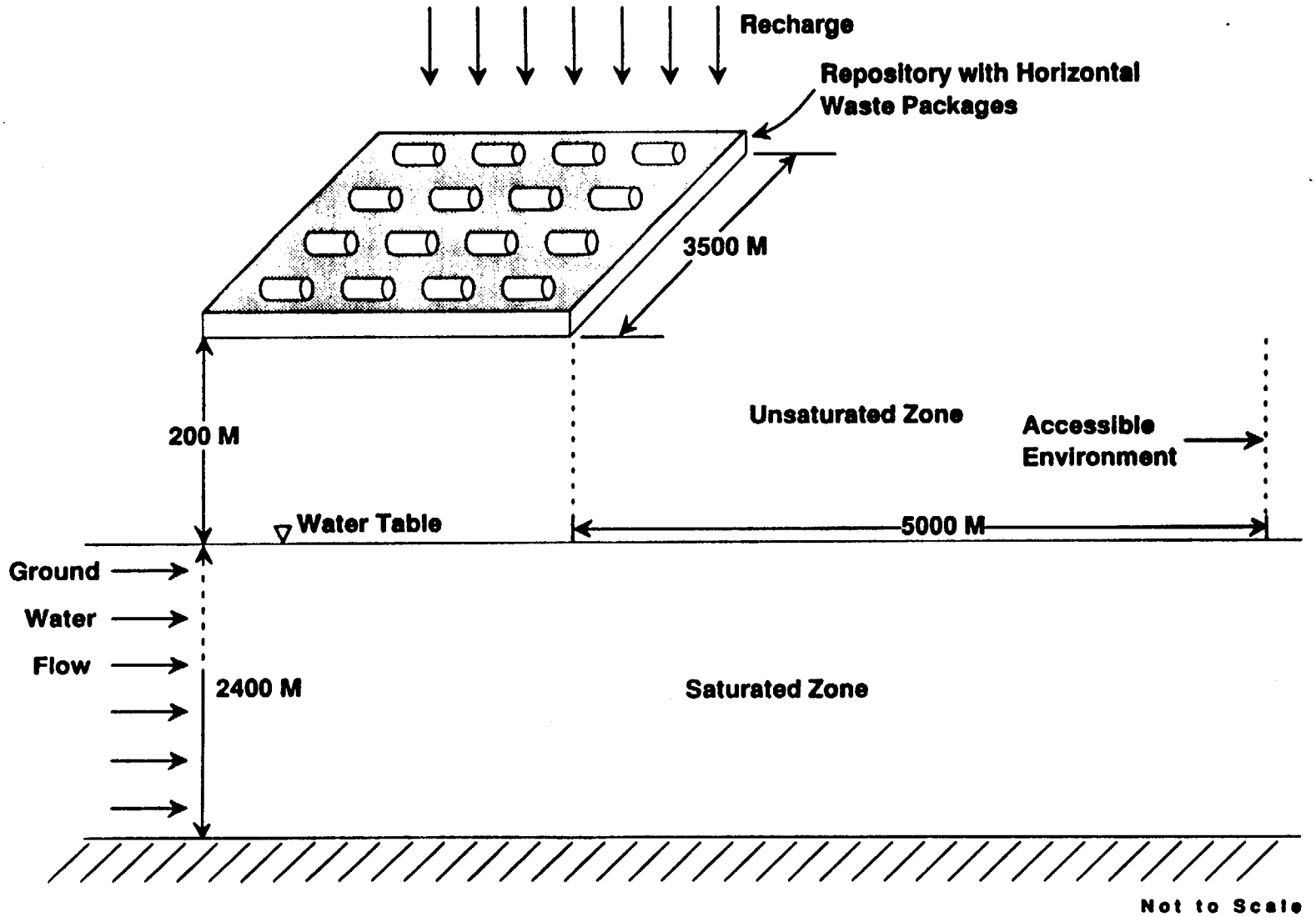


Figure 2-1. Schematic Diagram of a Geologic Repository at Yucca Mountain.

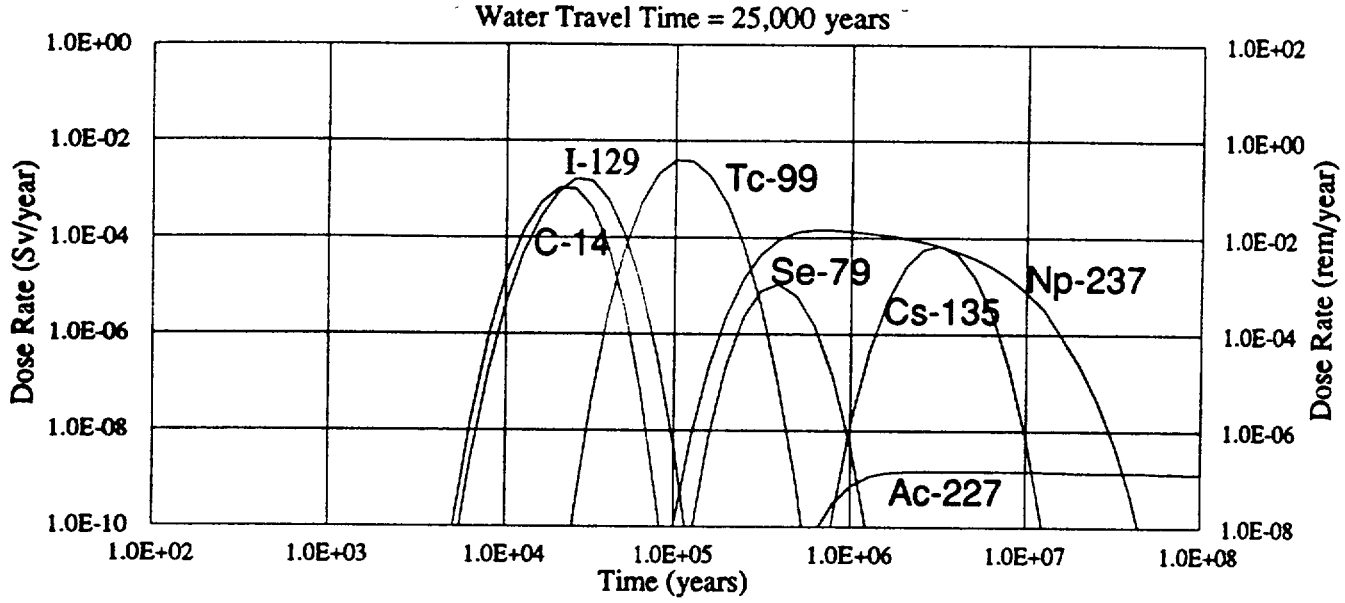


Figure 2-2. Dose to an Individual at the Accessible Environment for the Baseline Case.

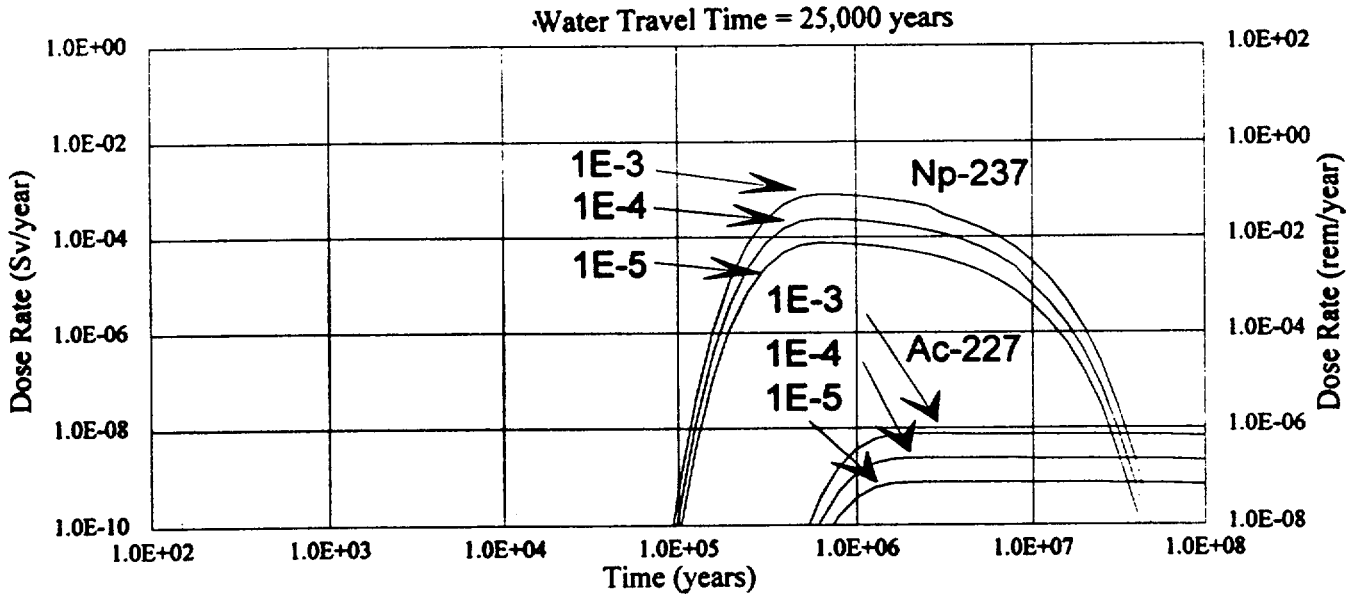


Figure 2-3. Dose to an Individual at the Accessible Environment Modifying the Baseline Case by Assuming Advective-Diffusive Releases from the Waste Package.

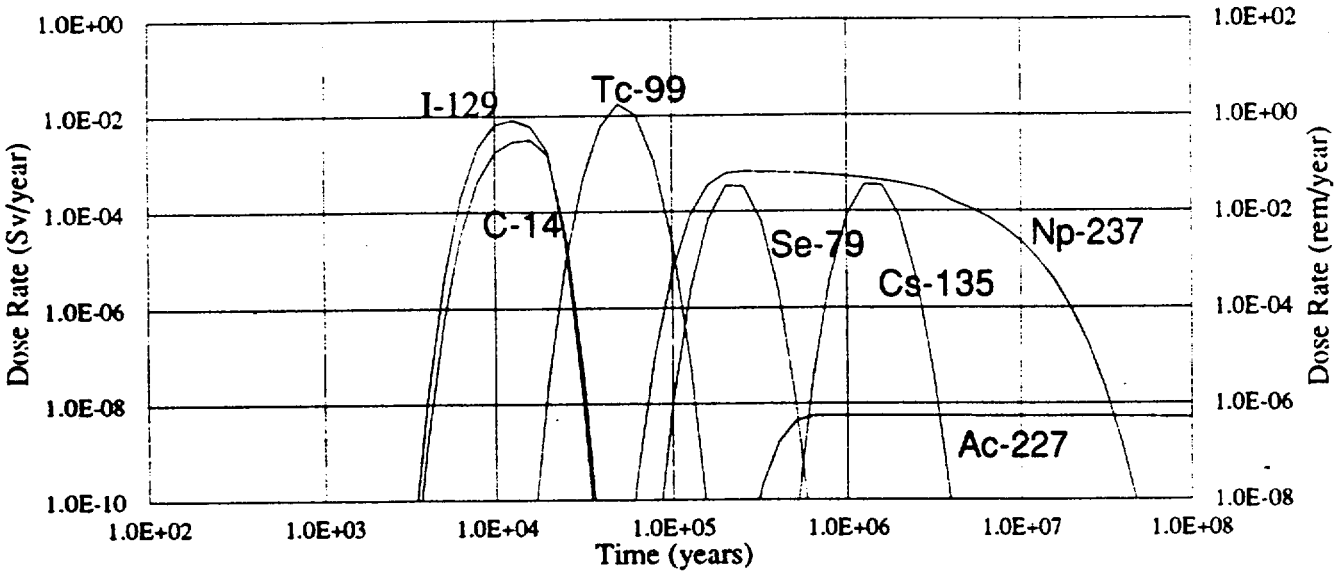


Figure 2-4. Dose to an Individual at the Accessible Environment for a Ground-Water Travel Time of 10,000 Years (4.0 mm/yr percolation flux).

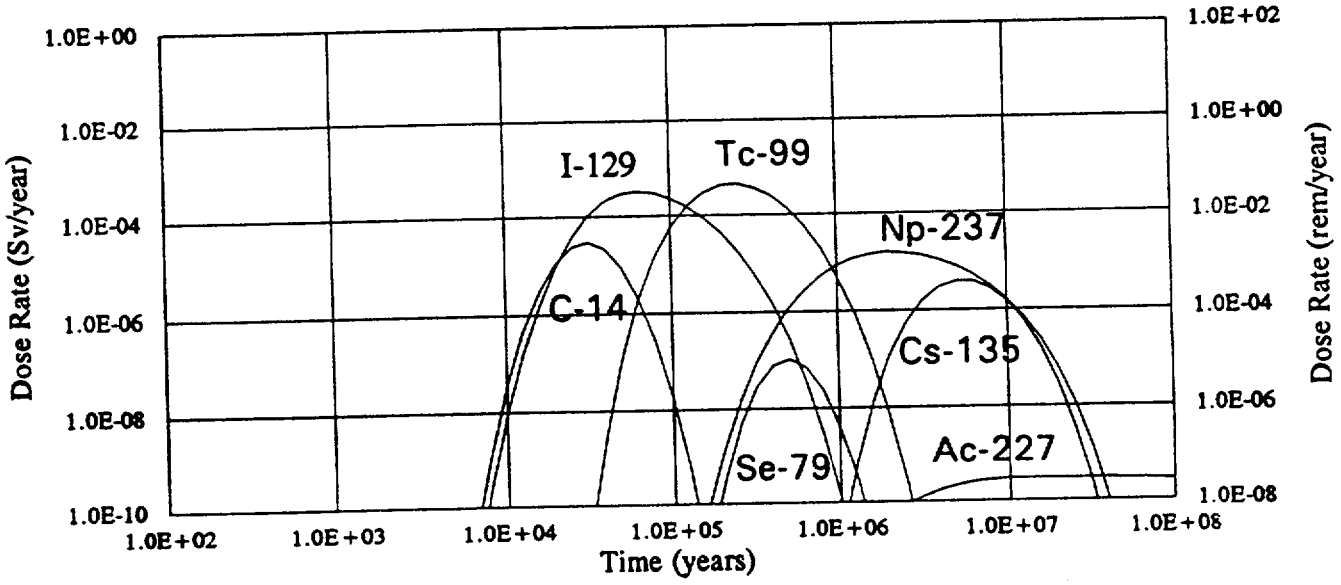


Figure 2-5. Dose to an Individual at the Accessible Environment for a Ground-Water Travel Time of 100,000 Years (0.21 mm/yr percolation flux).

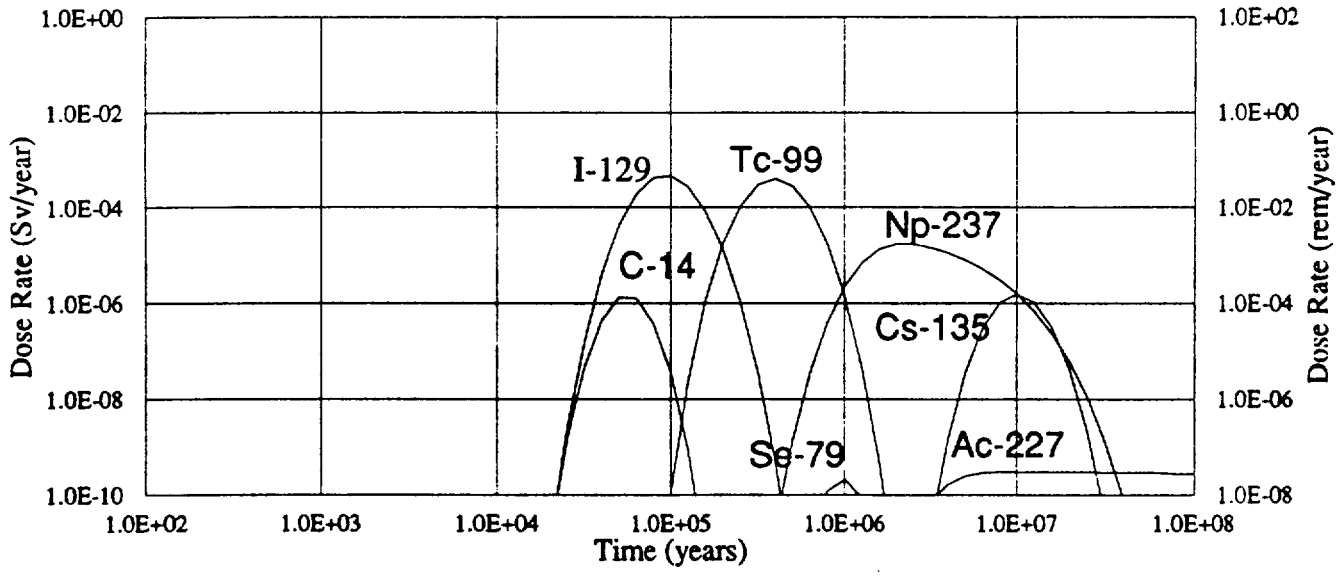


Figure 2-6. Dose to an Individual at the Accessible Environment for a Ground-Water Travel Time of 100,000 Years (0.21 mm/yr percolation flux), and a Dispersion Coefficient of 12.5 m^2/yr.

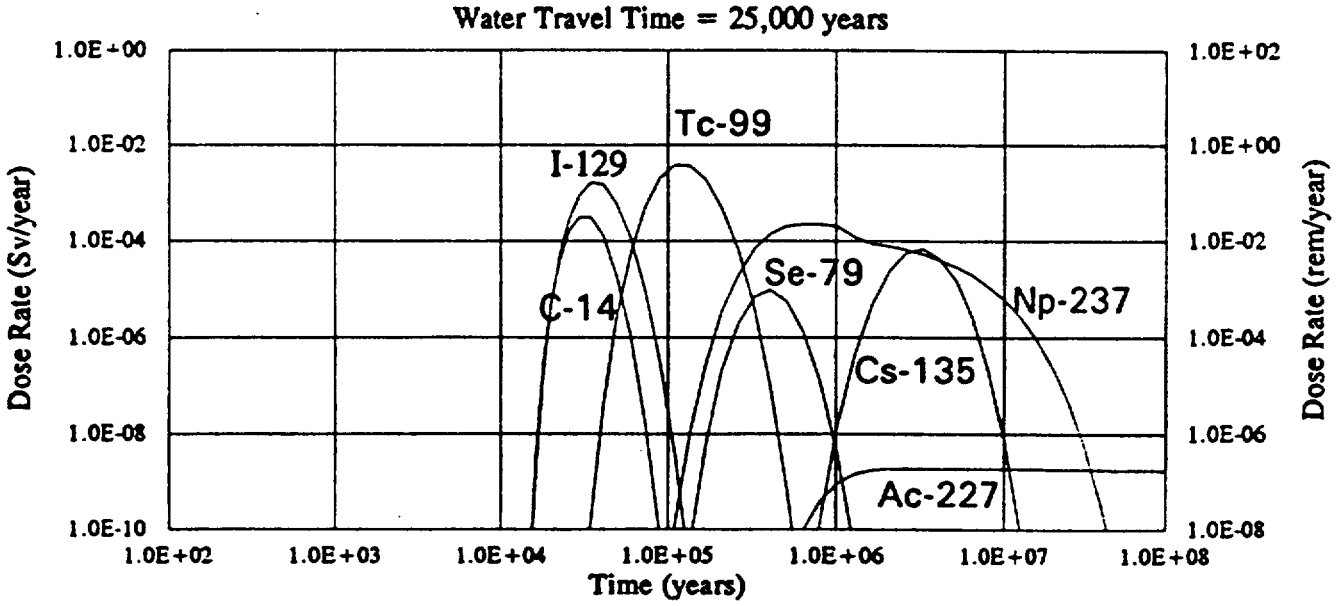


Figure 2-7. Dose to an Individual at the Accessible Environment for a 25,000 Year Ground-Water Travel Time and a 10,000 Year Waste Package.

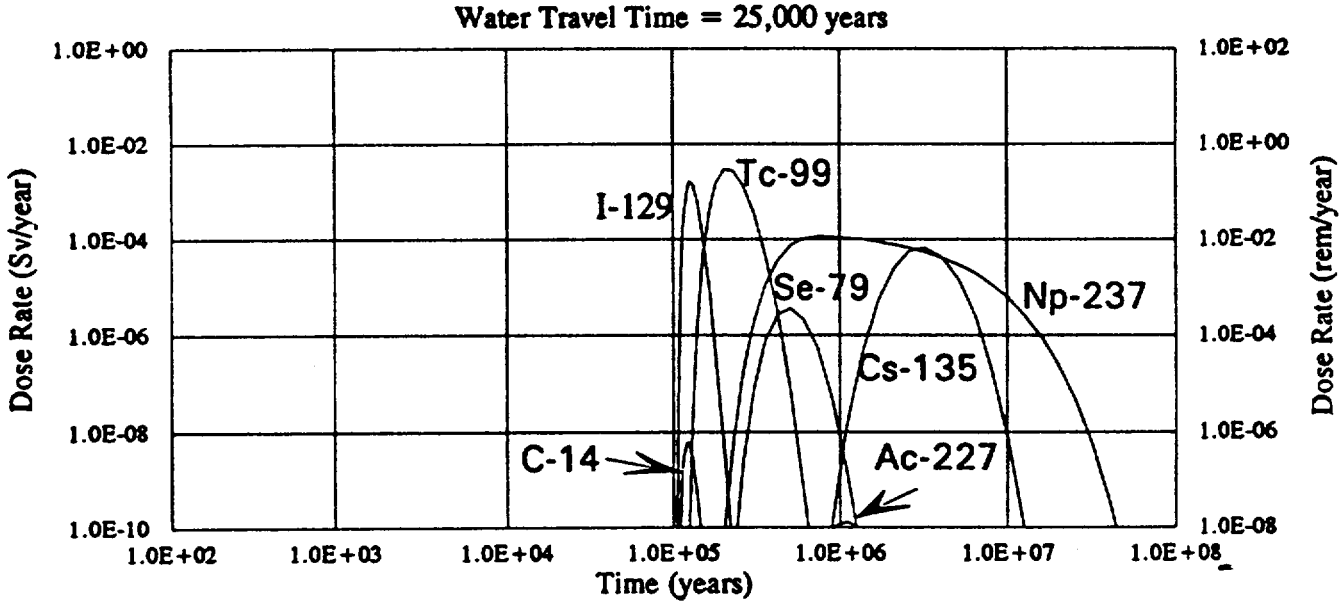


Figure 2-8. Dose to an Individual at the Accessible Environment for a 25,000 Year Ground-Water Travel Time and a 100,000 Year Waste Package.

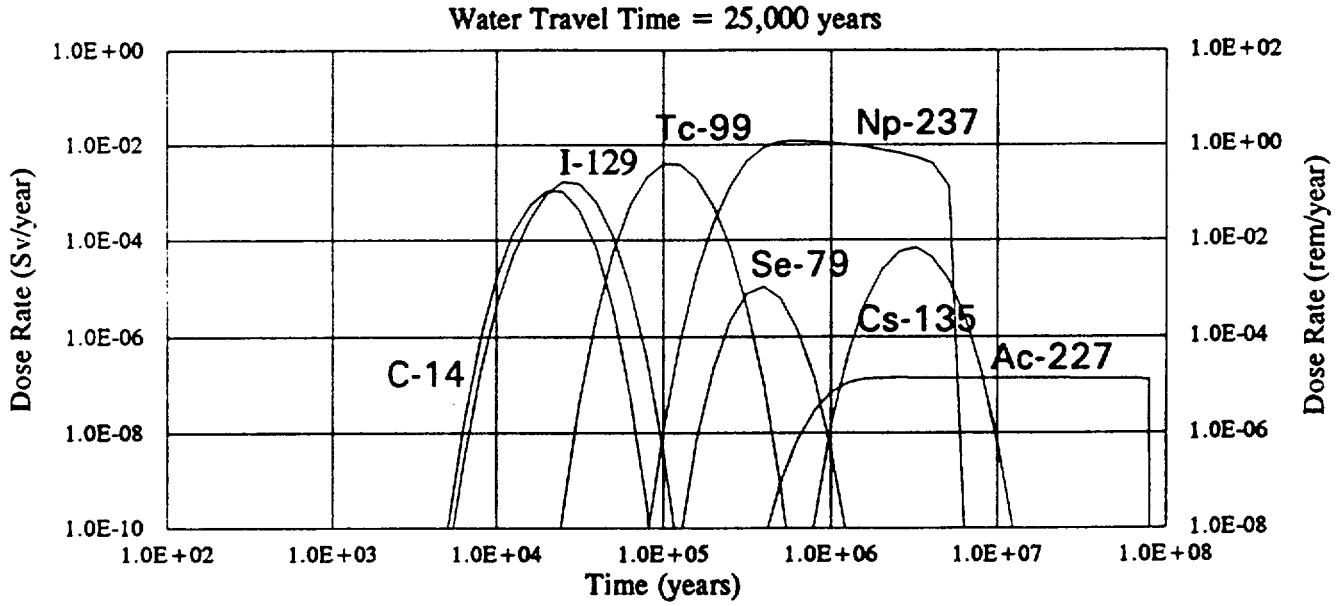


Figure 2-9. Dose to an Individual at the Accessible Environment with Solubilities of Np, Cm, Am, Pu, and U Increased by Factor of 100.

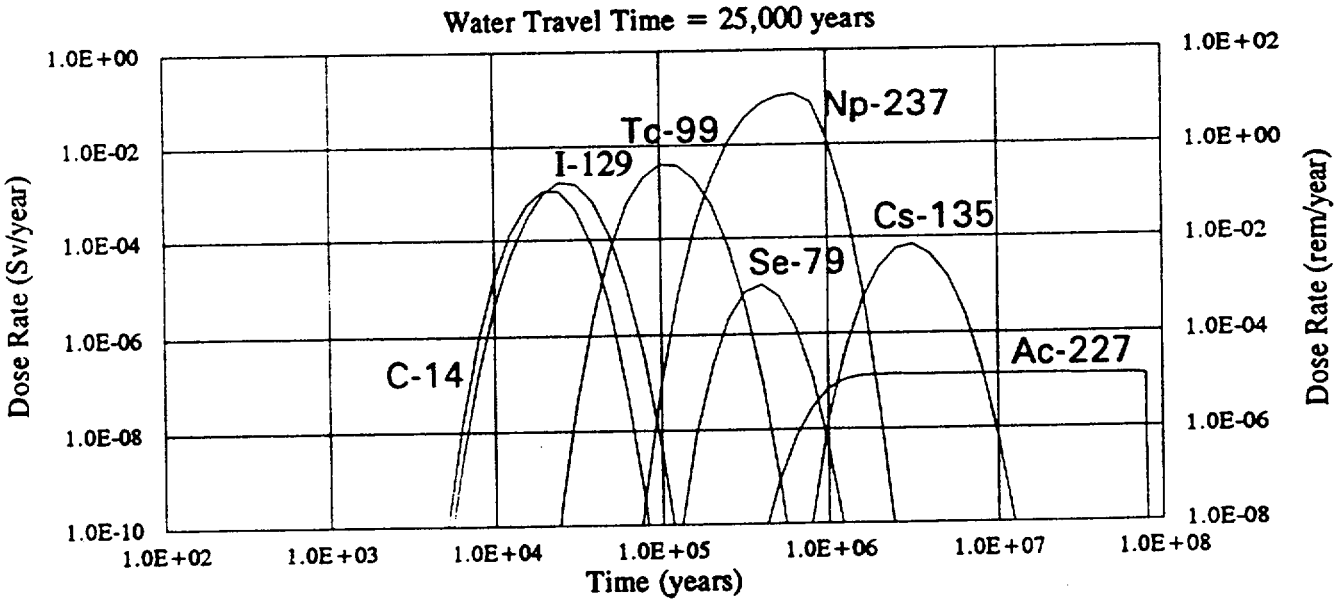


Figure 2-10. Dose to an Individual at the Accessible Environment with Solubilities of Cm, Am, Pu, and U Increased by a Factor of 100 and that of Np Increased by a Factor of 1,000.

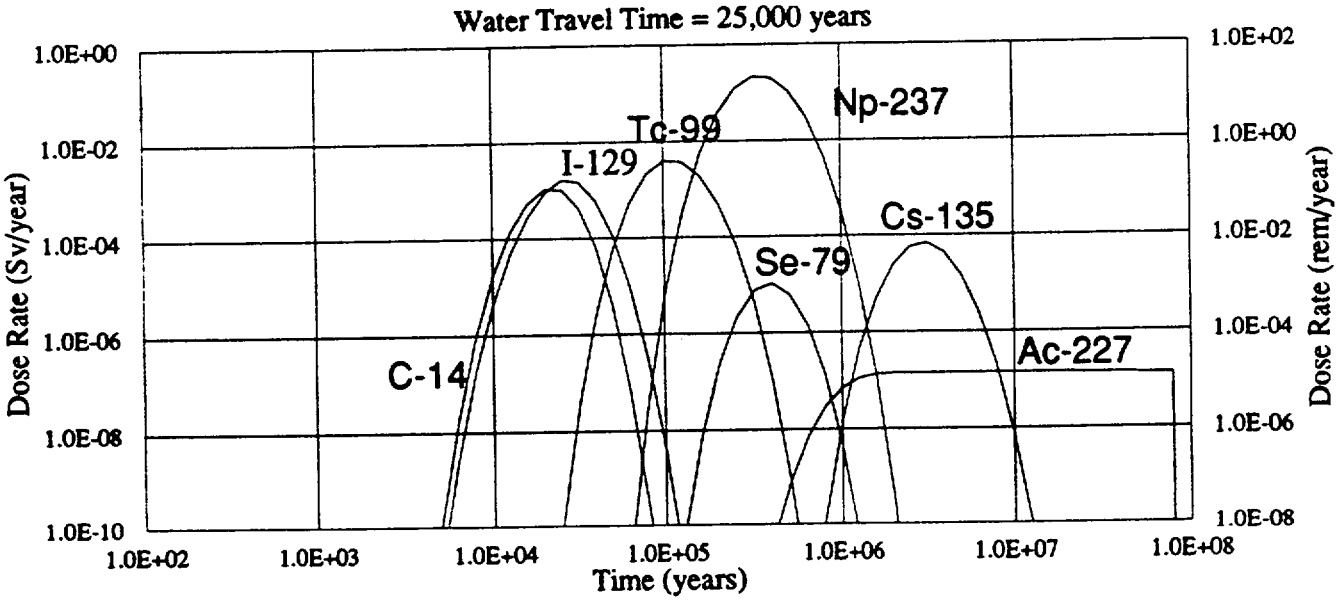


Figure 2-11. Dose to an Individual at the Accessible Environment with Solubilities of Cm, Am, Pu, and U Increased by a Factor of 100 and that of Np Increased by a Factor of 100,000.

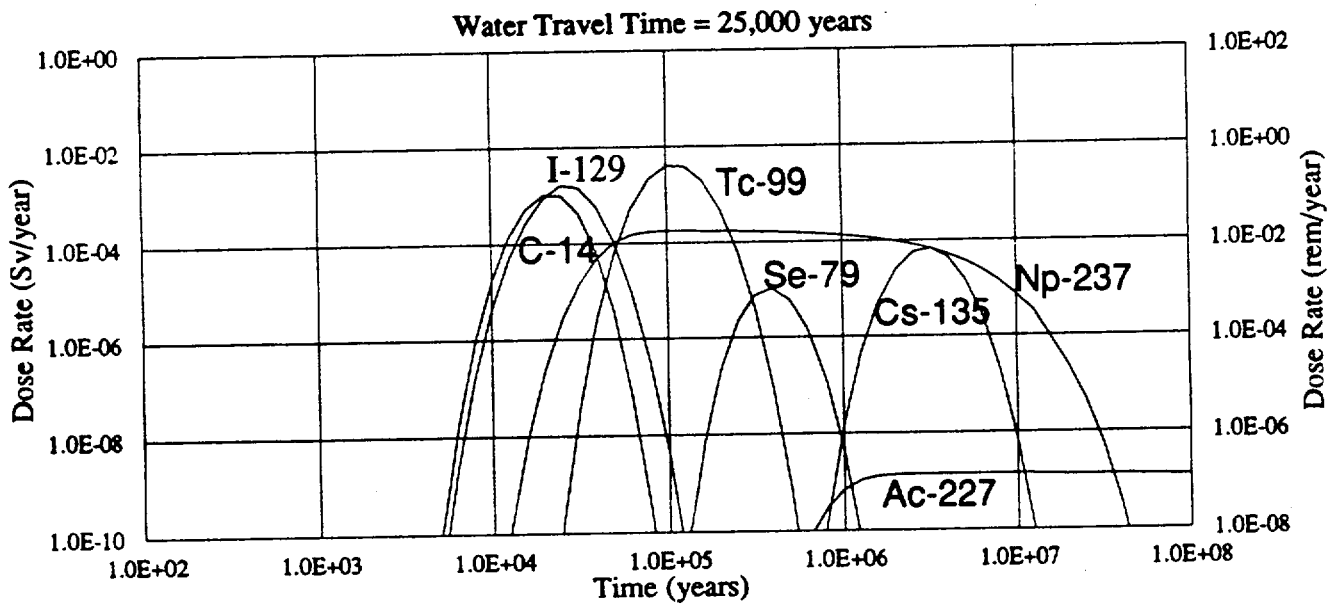


Figure 2-12. Dose to an Individual at the Accessible Environment with a Reduced Retardation Factor for Neptunium.

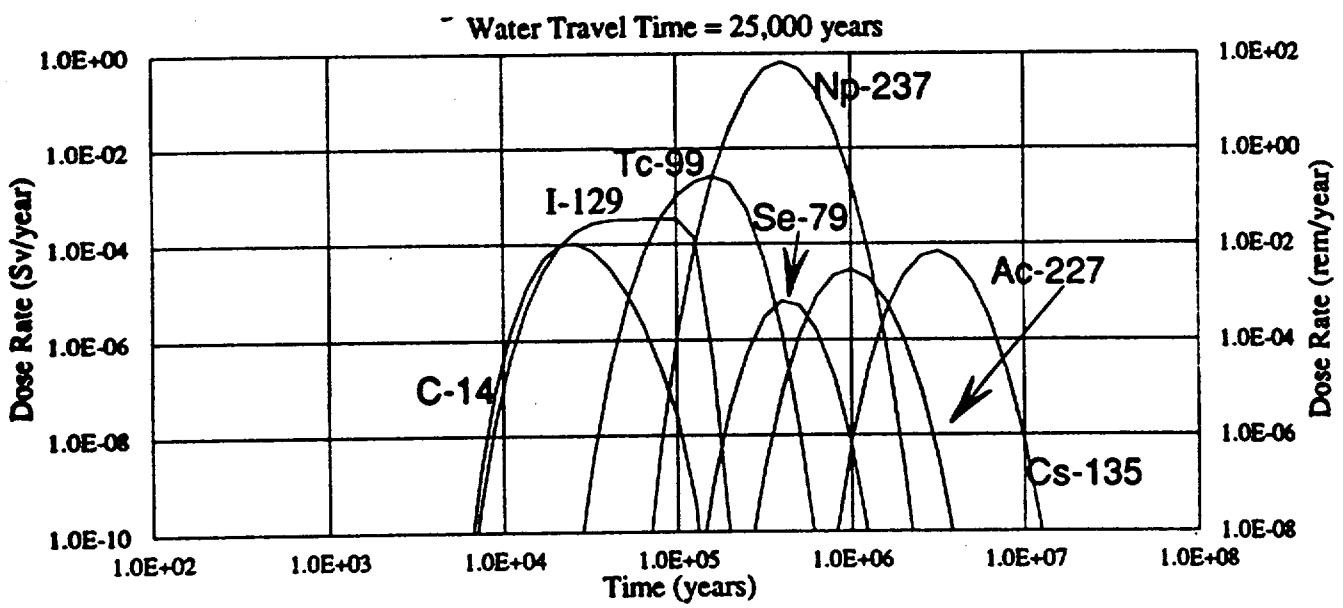


Figure 2-13. Release of all Radionuclides at a Rate of 10^{-5} yr^{-1} of the 1,000 Year Inventory

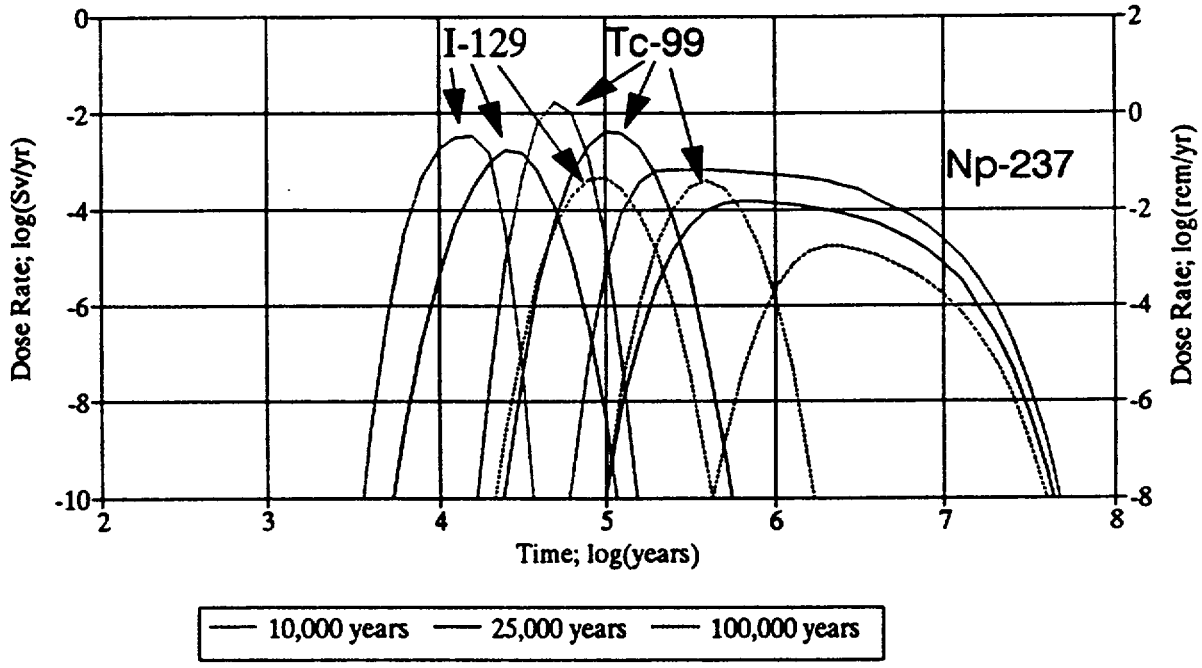


Figure 2-14. Sensitivity to Ground-Water Travel Time of Dose to an Individual at the Accessible Environment from ^{99}Tc, ^{129}I, and ^{237}Np.

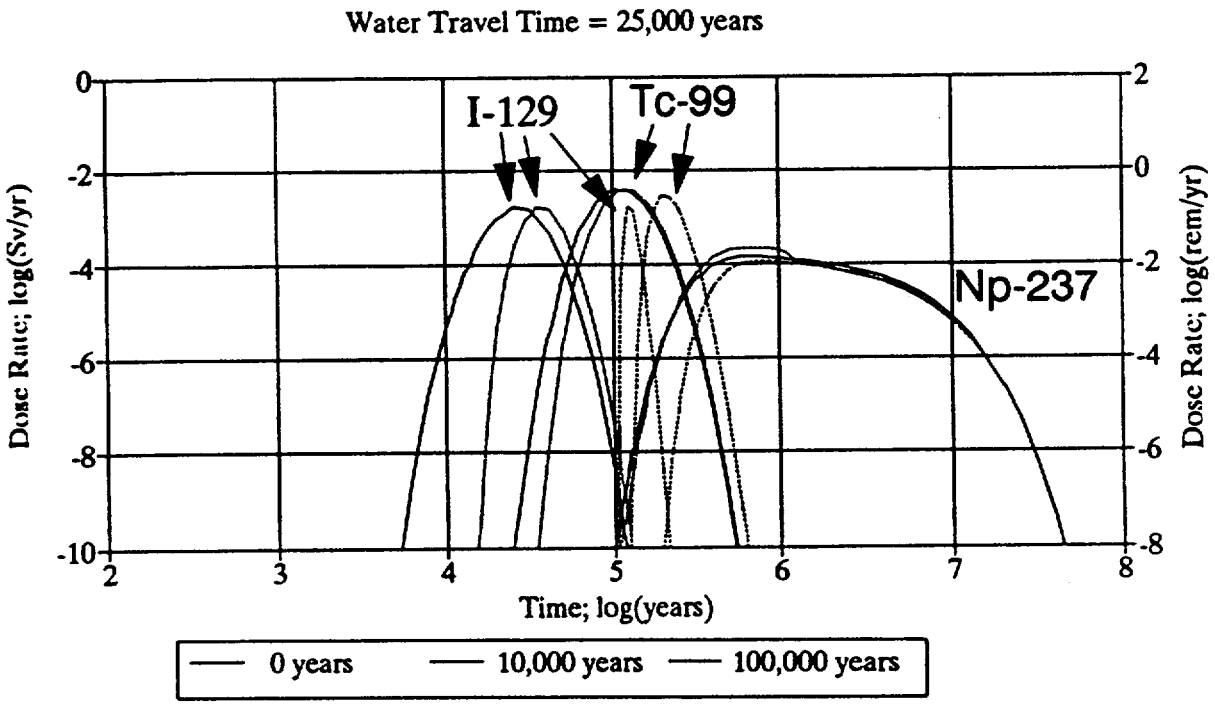


Figure 2-15. Sensitivity to Waste Package Lifetime of Dose to an Individual at the Accessible Environment from ^{99}Tc, ^{129}I, and ^{237}Np.

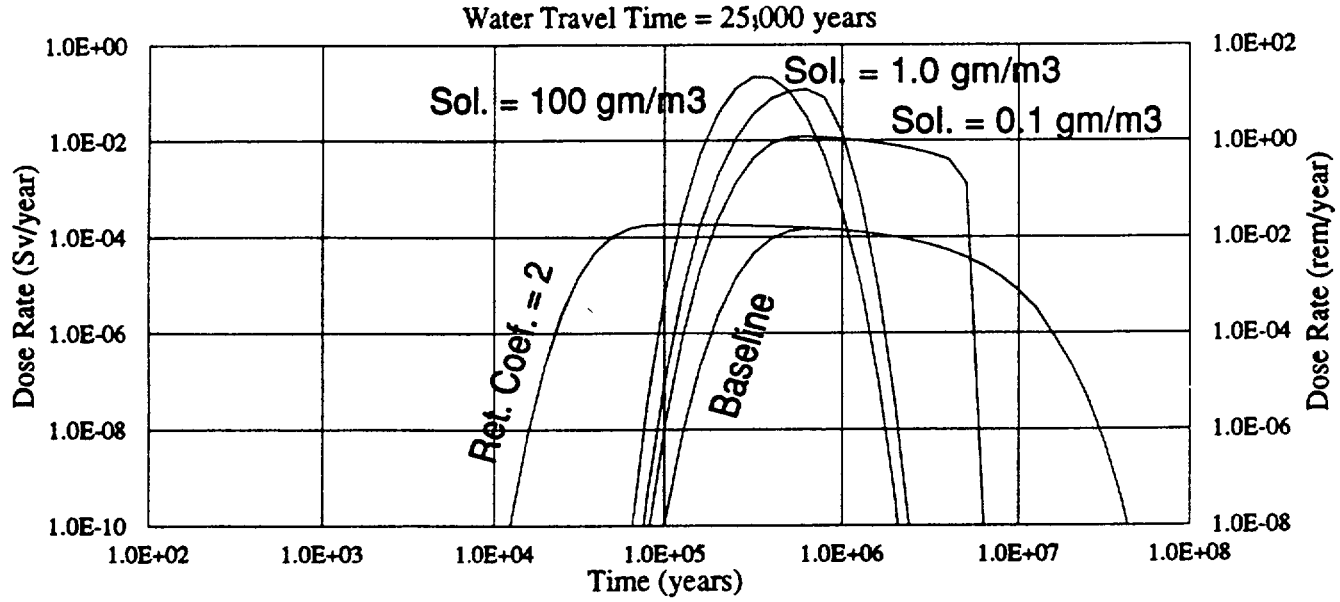


Figure 2-16. Sensitivity to Solubility and Retardation of Dose to an Individual at the Accessible Environment from ^{237}Np.

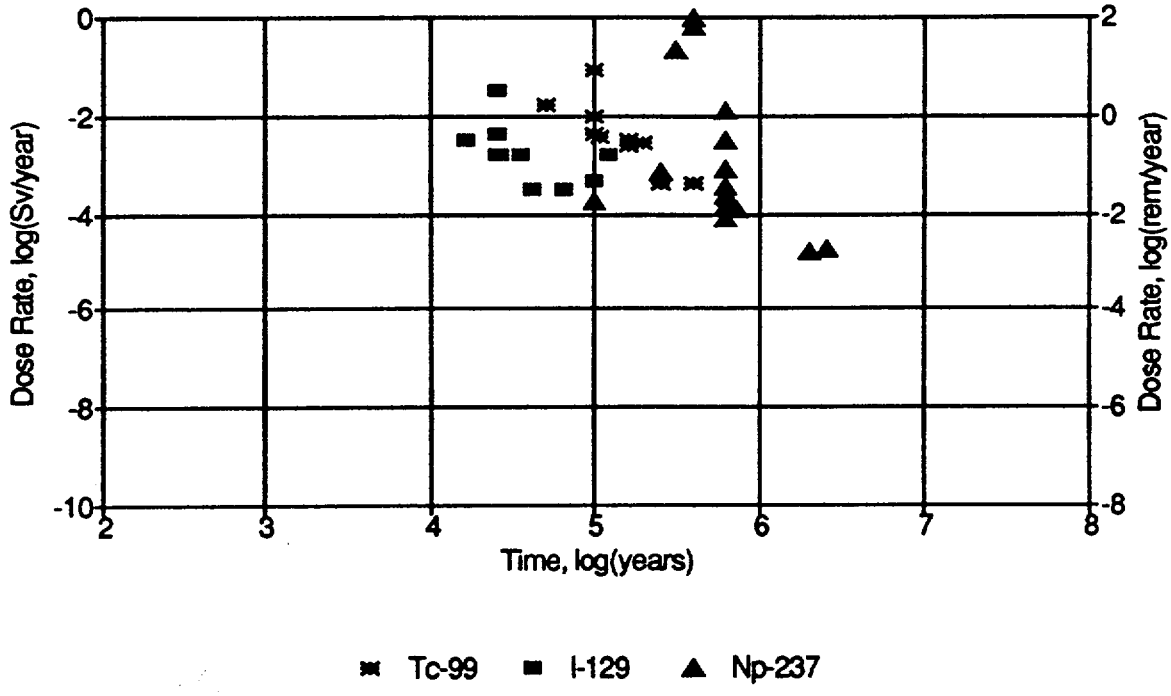


Figure 2-17. Peak Dose to an Individual at the Accessible Environment from ^{99}Tc, ^{129}I, and ^{237}Np for all Analyses.

3. ANALYSES USING THE REPOSITORY INTEGRATION PROGRAM (RIP)

3.1 GENERAL DESCRIPTION OF RIP

The Repository Integration Program (RIP) is a model for conducting probabilistic total system performance assessments of a geologic repository. It is not a model in the normal sense in that it does not explain the behavior of the repository system or its components but it attempts to describe the behavior. The description of the system is left entirely for the user to define by the use of simple algebraic functional relationships. In a sense, RIP is similar to a spreadsheet. The current version of RIP contains a large amount of built-in logic and calculational capabilities, but the problem solved must be entirely defined by the user. The theory and capability of RIP are described by Miller et al. (1993) and a user's guide is also available (Kossik and Hachey, 1993). RIP has been applied to evaluate a potential repository at Yucca Mountain by INTERA Inc. (1993), Miller et al. (1992), and Andrews et al (1994).

RIP consists of a front-end, a back-end, and a post-processor. The front-end is where the parameter values and functional relationships are defined and sampled using a Monte Carlo-type sampling algorithm. The back-end is where the actual computation occurs. The back-end is run for each sampled realization created in the front end. The post-processor is used to display the results [whether time histories or complementary cumulative distribution functions (CCDFs)] and to conduct simple sensitivity analyses (one- and two-dimensional scatter plots and simple parameter correlations). The CCDFs can be plotted in terms of either dose or release. The computational part of RIP presented in this report is divided into two domains (models): the waste package/engineered barrier system radionuclide release model and the near and far field (geosphere) radionuclide transport model. RIP also has the capability of disruptive events simulations, which are beyond the scope of this report.

The waste package/engineered barrier component of RIP is used to describe several processes which, if they occur, lead to radionuclide releases to the geosphere. These processes include container failure, exposure of radionuclides, and the mass transfer of radionuclides from the waste package to the host rock. Each process can be dependent on environmental parameters in the near field; in particular the temperature, geochemistry, and hydrology. The primary and secondary (cladding or pour canister) containers may fail either sequentially or simultaneously.

Once the container fails, the radionuclide inventory is exposed. There are three inventories within RIP, the free inventory which is released instantaneously once the primary container fails, a gap inventory which is released instantaneously once the secondary container fails, and a matrix (bound) inventory which is released as the matrix is altered and dissolved. The exposure of the bound radionuclides in the fuel or glass matrix is a function of the dissolution rate, the surface area exposed, and the percent of the surface area that is in contact with water.

Once radionuclides have been dissolved, they may be transferred to the host rock by advective transport, diffusive transport, or a combination of advective and diffusive transport. Aqueous-phase advective releases are a function of the radionuclide concentration in contact with the waste (which may or may not be solubility limited) and the flux past each waste package. Diffusive releases are a function of the radionuclide concentration, the effective diffusion coefficient, and a geometric factor that embodies the cross-sectional area of the container surface through which

diffusive releases occur and the length of the diffusive path. Only steady-state diffusive releases can be modeled within RIP. The release from the waste package is not dependent on the ability of the geosphere to transport the radionuclides.

The geosphere may be discretized into multiple pathways that are combined either in parallel or in series. These pathways represent different flow regimes (aqueous or gaseous), different flow domains (saturated or unsaturated zone), or different cross-sections of the repository, dependent on user specifications. In addition, different flow modes may be applied to each pathway. A single-flow mode may be used to represent an equivalent porous medium, while a multiple mode pathway may be used to describe fracture-matrix coupling in a dual porosity-dual permeability medium. For single mode pathways, RIP uses an analytical solution to the one-dimensional advection-dispersion equation. For multiple mode pathways, it uses a modified Markovian process to predict the transition between the two modes (Miller et al., 1993).

The source term model within RIP calculates the amount and the rate of waste available for transfer between the waste package and the geosphere. The waste inventory input within RIP requires the number of different types of waste streams, the number of waste canisters per waste stream, the waste-stream burnup, the amount of MTHM per container, and the inventory, in Ci/container, for each specific radionuclide present in the waste containers. The location and fraction of the waste within the containers can be specified at three different locations. First is the fraction of the waste located between the inner and outer container and instantaneously exposed once the outer container has failed. The second is the fraction of waste located within the inner container that is instantaneously exposed once the inner container has failed. These two fractions are important when considering the gaseous transport of radionuclides away from the waste package. In this analysis, no gaseous transport was considered. The remaining waste is bound within the waste matrix, with the waste matrix available for alteration once the inner container has failed. The waste container failure rates must be specified within RIP and can be included as either instantaneous, constant over a specified period, or as a distribution versus time. These container failure rates can be specified for both the outer container and inner container.

Once the waste matrix is exposed, the source term model determines the mass of each radionuclide available for release due to alteration of the waste matrix. The amount of waste available for release is limited by the solubility of the individual radionuclides. The alteration of the waste matrix is calculated using (Miller et al., 1993)

$$k_{wat} = R_{dis} \; S \; f_w \qquad \text{(Eq. 3-1)}$$

where k_{wat} is the aqueous alteration rate (yr^{-1}), R_{dis} is the matrix dissolution rate ($g/m^2/yr$), S is the effective surface area of the waste matrix per unit mass (m^2/g) and f_w is the fraction of the effective surface area that is wet.

Once the rate of exposed mass is determined, the amount of this mass that can be transferred away from the matrix and into solution, under steady state conditions, is determined based on both advective and diffusive mechanisms. For aqueous advective mass transfer, the transfer rate is calculated using (Miller et al., 1993)

$$k_a = F \; A \; C_s \qquad \text{(Eq. 3-2)}$$

where k_a is the advective mass transfer rate (g/yr), F is the ground-water flow rate through the repository (m/yr), A is the waste-package cross-sectional catchment area (m^2), and C_s is the saturation concentration of the radionuclide in the water at the waste package (g/m^3). This saturation concentration is the maximum amount of mass that can be transferred into solution based on either the amount of waste that has been exposed (alteration limited) or the amount of exposed waste that can be transferred into solution based on the radionuclide solubility (solubility limited).

The diffusive mass transfer away from the exposed waste matrix is determined using (Miller et al., 1993)

$$k_d = D_{eff} \; \omega \; C_s \qquad \text{(Eq. 3-3)}$$

where k_d is the diffusive mass transfer rate (g/yr), D_{eff} is the effective diffusion coefficient (m^2/yr), and ω is the geometric factor for diffusion (m). Assuming a spherical waste form, ω is calculated using (Chambre et al., 1985)

$$\omega = 4 \; \pi \; r \; \eta_e \qquad \text{(Eq. 3-4)}$$

where r is the radius (m) and η_e is the effective porosity through which the diffusion will occur. These equations are equivalent to the diffusive controlled fractional release rate used in NAS (1983) and presented in Eq. 2-2.

The mass transfer mechanisms within RIP move the mass from the waste package and into the first geosphere transport pathway. This first pathway, and all subsequent pathways, is completely defined by the user and can be specified to represent transport mechanisms through an unsaturated zone, a saturated zone, or a gravel backfilled zone around the waste package (Miller et al., 1993). At the end of the system, the mass is discharged to the accessible environment. The mass discharge into the accessible environment can be determined as either a discharge rate or as a cumulative discharge over a specified period using isotope specific activities. The mass released to the accessible environment can also be converted to a dose using a user specified, simple linear dose conversion factor.

3.2 COMPARISON WITH THE BASELINE CASE

The baseline case presented in Section 2.2 using UCBNE-41 was also simulated using RIP. Here the baseline case can be more closely approximated because RIP is capable of treating both the unsaturated and saturated flow as sequential portions of the flow path. As discussed earlier, the

model UCBNE-41 had to assume an average pore velocity over the 5,200 m flow path. Using RIP, the individual flow regimes through the 200 m unsaturated zone and the 5,000 m saturated zone can be simulated. In addition, RIP can handle competing isotopes of the same element in the dissolution process which could not be handled using UCBNE-41 except by perportioning on a mass basis.

The waste inventory for the baseline simulation consisted of 63,000 MTHM of 30 year old waste in 30,000 containers. The 30,000 containers were composed of approximately 35 percent boiling water reactor (BWR) waste and 65 percent pressure water reactor (PWR) waste. The waste package containers were assumed to be placed horizontally within the underground drifts with the direction of ground-water flow occurring perpendicular to the long axis of the containers. This orientation resulted in a cross-sectional area perpendicular to flow of 3.14 m^2 per waste container assuming a 4.76-m container with a diameter of 0.66 m . Tables 3-1 and 3-2 present a listing of the waste package inventories along with the element solubility, isotopic specific activities, half-life, EPA Table 1 limits (EPA,1985), and daughter products. These tables differ from Tables 2-1 and 2-2 only in the units that are used as input to the RIP model. Table 3-3 presents the parameters that are necessary to define the release of radionuclides from the waste packages. The calculated waste alteration rate (k_{wat}) of 1 x 10^{-4} yr^{-1} (Table 3-3) is maintained throughout the analyses presented in Chapter 3. The mass release from the waste package occurs at the alteration limit or the solubility limit, depending on which is limiting. The waste package lifetime for the baseline simulation was specified such that the containers failed instantaneously at time zero.

For the baseline simulation, the mass released from the waste package is transferred into the unsaturated zone domain simulated within RIP. The 200-m unsaturated zone is discretized into four 50-m pathways connected in series. Table 3-4 presents the specified parameters that describe the transport of mass through the unsaturated zone. The average linear velocity was assumed to be 1 x 10^{-2} m/yr with an effective porosity of ten percent. The area perpendicular to flow was specified to be 5.75 x 10^6 m^2, which is in effect setting the domain boundary equal to the areal footprint of the repository (DOE, 1988b).

The simulated mass release from the base of the unsaturated zone was discharged into the start of the saturated-zone domain simulated within RIP. The 5000-m path length in the saturated zone is discretized into five 1000-m pathways, also connected in series. The accessible environment was specified to be located at the end of the saturated zone, 5000 m down gradient from the repository. Table 3-5 presents the specified parameters that describe the transport of mass through the saturated zone. The average linear velocity was assumed to be 1.0 m/yr with an effective porosity specified at ten percent. The area perpendicular to flow was specified to be 8.16 x 10^6 m^2, assuming a width of 3400 m (DOE, 1988b) and a mixing depth of 2400 m (EPA, 1985a and 1993a). The release of mass to the accessible environment is converted into an activity release rate (Ci/yr) based on the specific activity values listed in Table 3-2.

The release of mass has been converted into a dose based on specified dose conversion factors. To accurately track mass for dose calculations, an additional pathway or biocell was attached to the end of the saturated zone. This pathway has no spatial dimensions, but is used only as an accumulation and mixing stage, given the mixing volumetric flow rate of the saturated zone (area x flux). The amount of mass that is discharged into the biocell is mixed with the corresponding

volumetric flow in each time step, and then converted to a dose during the same time step. Table 3-2 presents the dose conversion factor for each isotope. The dose received by the receptor is considered to be a dose for that of a maximally exposed individual in a farming scenario where all crops are produced by irrigation with contaminated ground water and ground water is used for livestock, drinking water, and household use (the dose conversion factors used in UCBNE-41, RIP, and NEFTRAN-S were the same).

Figure 3-1 presents the baseline time-history plot showing the dose exposure (Sv/yr) to an individual over a 1,000,000 year period. The baseline simulation parameters are summarized in Table 3-6. The plot consists of all isotopes that yield dose results higher that 10^{-10} Sv/yr. Figure 3-2 presents the same results as Figure 3-1, but extending the analyses to only 100,000 years to better illustrate the early arrival portion of the unretarded radionuclides. [The 100,000 year simulations are conducted with a 100 year time step to better capture these early arrivals, while the 1,000,000 year simulations are conducted with a 1,000 year time step. This difference can cause slight differences in the first arrivals as well as the peak height because there is more numerical dispersion when the larger time steps are used.] The diffusion coefficient was lowered to 1.0 x 10^{-7} m^2/yr so that the release from the waste package would be advection controlled. These figures can be compared to Figure 2-2 which shows the baseline case using UCBNE-41. As indicated by the comparison of these two figures, RIP and UCBNE-41, give analogous results.

The above baseline simulation assumed that the releases from the waste package were controlled by advective transport. The sensitivity of the dose to only diffusive waste package releases was investigated by specifying the ground-water flux through the repository as zero, while maintaining ground-water flow through the unsaturated zone. This would be analogous to the expected case where the aqueous flux preferentially stays in the more negative capillary pressure rock as opposed to the drift and any associated backfill materials. The effective diffusion coefficient out of the waste package was also modified to a value of 1.0 x 10^{-3} m^2/yr from the value of 1.0 x 10^{-7} m^2/yr which was used for Figures 3-1 and 3-2 (to assure that only advective releases from the package occurred in order to allow a comparison with the UCBNE-41 assumptions). The dose versus time plots generated from this run are presented in Figure 3-3 and 3-4 for 1,000,000 and 100,000 years, respectively. The alteration-limited radionuclides such as ^{14}C, ^{129}I, ^{99}Tc, and ^{135}Cs show no noticeable change in the time-history curves. However, the solubility-limited radionuclides show a slightly longer time to breakthrough and a lower peak dose. For example, the peak ^{237}Np dose decreased from approximately 1 x 10^{-4} Sv/yr to approximately 5 x 10^{-5} Sv/yr. This decrease is due to the slower transport of radionuclides away from the waste package surface.

The simulations presented above incorporated the neptunium solubility of 1.0 x 10^{-3} g/m^3 used in the WISP analysis (NAS, 1983) and many of the analyses in Chapter 2. The neptunium solubility was increased to 1.0 g/m^3 and the simulation was repeated (Figure 3-5). The dose versus time results presented in Figure 3-5 are equivalent to Figure 3-1 except for the peak dose of ^{237}Np and related daughter products. As specified in Tables 3-1 and 3-2, ^{233}U and ^{229}Th are the daughter and grand-daughter products, respectively, of ^{237}Np. Figure 3-5 shows that an increase in the neptunium solubility by three orders of magnitude resulted in an increase in the peak dose of ^{237}Np, ^{233}U, and ^{229}Th by an equivalent three orders of magnitude. For example, the ^{237}Np peak dose exposure increased from approximately 5 x 10^{-5} Sv/yr to approximately 5 x 10^{-2} Sv/yr.

3.3 CUMULATIVE RELEASE PERFORMANCE MEASURE

The three simulations in Section 3.1 presented dose time-history plots generated from deterministic simulations (i.e., based on a single realization). To obtain a better understanding of the sensitivity of mass release and dose to certain parameters, stochastic simulations were generated that incorporated multiple realizations. The mass release and dose results generated from multiple realization simulations could be presented in the form of complimentary cumulative density functions (CCDFs), with the CCDFs representing the probability of a certain magnitude, or greater, of cumulative mass release or peak dose. Because such a probability plot would imply expected values rather than simply the sensitivity of the results to the uncertain parameters, no CCDFs will be presented in the analyses below. When considering cumulative mass release, the values are normalized to the EPA Table 1 limits (EPA, 1985) for release to the accessible environment at 10^4 years. Even though the Table 1 release limits are not applicable to larger times, the normalization values are maintained at the 10^4 year limit for simulations that consider releases over 10^5 and 10^6 years. Scatter plots have been generated from multiple realizations. These plot normalized cumulative release or peak dose and the sampled value of an uncertain parameter. The plots are useful in showing the sensitivity of the normalized cumulative release or peak dose to parameter value ranges.

The simulation parameters were modified to incorporate the uncertainty of key parameters, with log-uniform distributions being applied to the effective diffusion coefficient out of the waste package, the solubility of neptunium, and the unsaturated-zone average linear velocity. Figure 3-6 presents a scatter plot of unsaturated-zone average linear velocity versus normalized cumulative release for each realization. This figure shows a strong correlation between the value of the average linear velocity and the normalized cumulative release. The normalized releases over the 10,000 year period are primarily composed of the dispersive fronts of ^{14}C and ^{129}I. The scatter plot in Figure 3-6 shows that as the average linear velocity is increased from 1×10^{-3} m/yr to 1×10^{-2} m/yr, the normalized cumulative release is increased approximately 11 orders of magnitude. The large dependence on the average linear velocity is due to the different degree to which the dispersive fronts of ^{14}C and ^{129}I have reached the accessible environment by 10^4 years. The effect of neptunium solubility is not seen in Figure 3-6 because it arrives at the accessible environment beyond 100,000 years. The relatively smooth curve is caused by the limited effect of the range of the diffusion coefficient. Also, at such low normalized cumulative release values, some numerical dispersion within RIP should be expected.

The simulation time was extended to 10^5 years and the normalized cumulative release to the accessible environment calculated. Figure 3-7 presents a scatter plot of the unsaturated-zone average linear velocity and normalized cumulative release for each realization. This figure shows a strong correlation between the value of the average linear velocity versus the normalized cumulative release. The normalized releases over 100,000 years are primarily composed of ^{14}C, ^{129}I, and ^{99}Tc. The sensitivity of the normalized cumulative release to the unsaturated-zone average linear velocity is similar to the 10^4 year simulation, but the magnitude of change in the release is not as large. Still, the normalized cumulative release is very sensitive to the value of the average linear velocity in the unsaturated zone. The normalized release over 100,000 years is not sensitive to the effective diffusion coefficient.

The simulation time was also extended from 10^4 years to 10^6 years and the normalized cumulative release to the accessible environment calculated. As the simulation time increased, some of the radionuclides that were solubility limited and not highly retarded were discharged prior to 10^6 years, thereby increasing the normalized cumulative release to approximately 0.2 for all 100 realizations. Also, eight realizations generated normalized releases greater than 10. Figures 3-8, 3-9, and 3-10 present scatter plots of effective diffusion coefficient, unsaturated-zone average linear velocity, and the solubility of neptunium, respectively, versus normalized cumulative release for each realization. The normalized cumulative release shows a strong sensitivity to the average linear velocity in the unsaturated zone (Figure 3-9). However, the variation in the magnitude of the normalized cumulative release is not as great as in the 10^4 year or 10^5 year simulations. This is due to the relatively constant release, after an initial buildup, of the solubility-limited radionuclides (especially ^{237}Np) after 10^5 years. This general behavior was seen in the time-history dose plots presented in Section 3.2. The normalized cumulative release at 10^6 years shows slight sensitivity to the effective diffusion coefficient (Figure 3-8) and the solubility of neptunium (Figure 3-10), at the higher values, but the variance in the magnitude of the release is dominated by the average linear velocity in the unsaturated zone. It is interesting to note that the high normalized release realizations (those lying above the generally linear response illustrated in Figure 3-9) require both a high neptunium solubility and a high diffusion coefficient. If either the neptunium solubility is less than 1 g/m^3 or the diffusion coefficient is less than 10^{-4} m^2/yr, then the response is totally controlled by the unsaturated zone velocity.

To further investigate the effect of the neptunium solubility on the 10^6 year normalized cumulative releases, the neptunium solubility was maintained at a constant value of 1.0 g/m^3 and the simulation conducted again. With all other parameters remaining the same, the variation in the two analyses should be due solely to the sampled distribution of the neptunium solubility. At a simulation time of 10^6 years, the normalized cumulative release is not very sensitive to the uncertainty in the neptunium solubility because the release is dominated by the release of technetium and iodine. Figures 3-11 and 3-12 present scatter plots showing the sensitivity of the normalized cumulative release to the effective diffusion coefficient and the unsaturated-zone average linear velocity. With the removal of the uncertainty in the neptunium solubility, the scatter plots show that the normalized cumulative release at 10^6 years is sensitive to both the effective diffusion coefficient and the unsaturated-zone average linear velocity.

3.4 PEAK DOSE PERFORMANCE MEASURE

Section 3.3 presented the normalized cumulative release to the accessible environment at simulation times of 10^4, 10^5, and 10^6 years. The maximally exposed, whole body dose to an individual located at the accessible environment was determined for the four simulations discussed in Section 3.3. The peak dose (Sv/yr) was determined for each realization, resulting in 100 independent peak dose values. The analyses and comparisons presented in Section 3.3 will be repeated with the peak dose exposure used as the performance measure.

The scatter plot of the 10,000 year peak dose versus the sampled value of the unsaturated-zone average linear velocity is presented in Figure 3-13. The strong sensitivity of the peak dose exposure to the value of the unsaturated-zone average linear velocity is similar to the sensitivity of the normalized cumulative release (Figure 3-6). The dose exposure within 10^4 years is

dominated by ^{14}C and ^{129}I, which are both alteration-limited, non-retarded isotopes. The peak dose exposure within 10^4 years shows virtually no sensitivity to the effective diffusion coefficient.

For a simulation period of 10^5 years, all 100 realizations resulted in peak dose exposures of at least 2×10^{-5} Sv/yr, with two realizations having peak dose exposures of about 0.07 Sv/yr. Figure 3-14 presents a scatter plot of the unsaturated-zone average linear velocity versus the peak dose for 100 realizations of 10^5 years. The peak dose exposure is very sensitive to the unsaturated-zone average linear velocity. Within 10^5 years, the dose exposure to ^{14}C and ^{129}I has already peaked and the dose exposure to ^{99}Tc is continuing to increase (see Figures 3-1 and 3-2, for example). The bimodal shape of the data plotted in Figure 3-14 is attributed to the ^{129}I peak dose at lower unsaturated-zone average linear velocities, and to the ^{99}Tc maximum dose exposure, within 10^5 years, at higher velocities. As the average linear velocity is increased, the maximum dose exposure of ^{99}Tc becomes greater than the peak dose exposure of ^{129}I at an average linear velocity value of approximately 6×10^{-3} m/yr. As in the 10^4 year simulation, the peak dose exposure was not sensitive to the effective diffusion coefficient. The two realizations with peak doses of about 0.07 Sv/yr correspond to very high neptunium solubilities and high unsaturated zone velocities.

For a simulation period of 10^6 years, all 100 realizations resulted in peak dose exposures of at least 8×10^{-5} Sv/yr, with at least 20% of the realizations having peak dose exposures greater than 1×10^{-2} Sv/yr, and 10 % of the realizations having peak dose exposures greater than 1×10^{-1} Sv/yr. Figures 3-15 through 3-17 present scatter plots of the effective diffusion coefficient, the unsaturated-zone average linear velocity, and the neptunium solubility, respectively, versus the 1,000,000 year peak dose for 100 realizations. Of the three scatter plots, the peak dose exposure is most sensitive to the unsaturated-zone average linear velocity over the entire distribution range, and is also slightly sensitive to the larger values of both the effective diffusion coefficient and the neptunium solubility.

The neptunium solubility range of 10^{-3} to 10^2 g/m^3 was modified to maintain a constant value of 1.0 g/m^3. The analyses gave similar results, demonstrating that the peak dose exposure for 1,000,000 years is more sensitive to the unsaturated zone velocity and effective diffusion coefficient than to the neptunium solubility. Scatter plots showing the sensitivity of the peak dose exposure to the unsaturated-zone average linear velocity and the effective diffusion coefficient are presented in Figures 3-18 and 3-19, respectively. The peak dose 1,000,000 years is sensitive to the unsaturated-zone average linear velocity over the entire distribution of values of velocity evaluated (Figure 3-19). The peak dose exposure also shows some sensitivity to the effective diffusion coefficient values above approximately 1×10^{-4} m^2/yr (Figure 3-18).

3.5 EFFECT OF WASTE PACKAGE FAILURE ON PEAK DOSE

The simulation results presented in Sections 3.3 and 3.4 assumed that the failure of the waste package containers occurred instantaneously at the start of the simulation. Therefore, the entire waste matrix was instantly available for alteration. This assumption of instantaneous waste package failure was selected due to its conservative nature. Additional sensitivity simulations have been conducted which demonstrate the predicted dose exposure over 10^6 years for various waste package container failure initiation times and failure rates. The dose exposure results are

presented as time-history plots (dose versus time) generated by conducting one realization per simulation data set.

Figure 3-20 presents the time-history plot for the case when the waste package failure rate is one percent every 10^3 years starting at time zero. This results in a uniform distribution of failures where all the containers have failed by 10^5 years. Figure 3-20 can be compared directly to the dose time-history curve presented in Figure 3-5. The only variation in the two data sets is the waste package container failure distributions. The results illustrated in Figure 3-20 show a slight lowering and broadening of the peak for the alteration-limited radionuclides, such as ^{14}C , ^{129}I, and ^{99}Tc. However, there is no change in the solubility limited release of radionuclides such as neptunium. The slower failure rate of waste packages creates a limiting effect on the rate of mass exposure and thus on the rate of mass transfer out of the waste package.

Figure 3-21 presents the 10^6 year time-history dose exposure plot for the case when the waste-package failure rate was maintained at one percent per 10^3 years, but the failure initiation time was extended from zero to 10^4 years. The results for the two simulations are almost identical except for the decrease in ^{14}C, due to its short half-life, and a shift of the curves approximately 10^4 years on the time axis. Figure 3-22 presents the 10^6 year time-history dose exposure plot for the case when the waste-package failure rate was maintained at one percent per 10^3 years, but the failure initiation time was extended to 10^5 years. This results in a uniform failure of the waste packages between 1 x 10^5 and 2 x 10^5 years. The peak dose curves for this simulation are almost identical to the earlier results except for the lack of ^{14}C due to decay and a shift of the curves by 10^5 years on the time axis.

The waste-package failure rate was modified so that one percent of the waste packages would fail every 10^4 years. Specifying a failure initiation time of zero years results in a uniform failure of all the waste packages over 10^6 years. The resulting simulation results are presented in Figure 3-23 as a dose exposure time-history curve. This waste-package failure rate results in the alteration of a constant fraction of each radionuclide per year over the entire 10^6 year simulation. Once the dispersive front of each radionuclide breaks through to the accessible environment, the dose exposure should remain constant over the entire simulation period of interest until the inventory is depleted or decayed away. It is interesting to note that there is virtually no change in the neptunium dose even for the very slow failure rate of 1% per 10,000 years. This is because the solubility-limited release rate is equivalent to this rate of container failures. Had a larger solubility been used, then a more pronounced effect would have been observed. This holds true as long as the radionuclide half-life is sufficiently large to maintain the constant source strength. This is seen in Figure 3-23 where there is a slight lowering of the peak dose exposure value for most radionuclides, as compared to Figure 3-20, but the release to the receptor for the alteration-limited radionuclides has been extended out to 10^6 years. The one exception is the decrease in ^{14}C because of its short half-life.

3.6 EFFECT OF DIFFUSIVE RELEASE FROM THE ENGINEERED BARRIER SYSTEM

The sensitivity analyses presented above assumed the waste packages were in direct contact with the host rock, where advective or diffusive releases from the waste package were immediately available for advective transport into and through the unsaturated zone. The remaining sensitivity

analyses presented in this section will present the predicted response in the normalized cumulative release and the peak dose exposure due to the inclusion of a 1.0-m capillary barrier around the waste package. The capillary barrier was assumed to be a 1.0-m thick gravel layer between the waste package and the host rock. Radionuclide transfer was assumed to be diffusive out of the waste package with transport across the barrier assumed to be by diffusive transport, with the velocity or travel time through the 1.0-m barrier being a function of the specified effective diffusion coefficient. [See Section 4.6 for a discussion of the relationship between the transport velocity and the effective diffusion coefficient]. A continuous thin water film was assumed to envelop the gravel particles, and no advective radionuclide transport was assumed to occur within this 1.0-m pathway (i.e., the advective flow would be limited to the fine-grained material because of capillary forces). Table 3-6 presents a listing of the simulation parameters for the analyses presented in this section.

Figure 3-24 presents a time-history plot of the dose exposure over 10^6 years for an effective diffusion coefficient of 1.0×10^{-4} m^2/yr. The average linear velocity through the diffusive pathway was specified as 1.0×10^{-3} m/yr, with a resulting travel time of 10^3 years. Figure 3-24 can be compared to Figure 3-5 with the principal difference being a slightly higher effective diffusion coefficient and the 1.0-m diffusive pathway. The addition of the 1.0-m diffusive pathway, with the assumed effective diffusion coefficient, resulted in a slight lowering of the peak-dose exposure for the alteration-limited radionuclides. The addition of the 10^3 year travel time through the diffusive pathway is almost imperceivable within a simulation period of 10^6 years. Figures 3-25 and 3-26 present dose exposure time-history plots with the effective diffusion coefficients lowered to 1.0×10^{-5} m^2/yr and 1.0×10^{-6} m^2/yr, respectively. As the effective diffusion coefficient is lowered by one order of magnitude, the travel time through the 1.0-m diffusive pathway is correspondingly increased by one order of magnitude. This decrease in the effective diffusion coefficient results in a delay in mass breakthrough and a decrease in the peak dose exposures, especially for the solubility-limited radionuclides.

The constant parameter values used in the single realization simulations were modified and specified as distributions (Table 3-6) to show the sensitivity of the normalized cumulative release and the peak-dose exposure for a set of 100 realizations. For the range of diffusion coefficients investigated (10^{-6} to 10^{-2} m^2/yr), there is only a limited sensitivity to the actual diffusion coefficient sampled, except at the lowest values. [Note that a 10^{-6} m^2/yr diffusion coefficient corresponds to a mean travel time of 1,000,000 years through the 1 m diffusive barrier.] Scatter plots showing the sensitivity of the normalized cumulative release value to the unsaturated-zone average linear velocity and the effective diffusion coefficient are presented in Figures 3-27 and 3-28, respectively. The normalized cumulative release shows more sensitivity to the effective diffusion coefficient than in previous analyses. The normalized cumulative release still shows sensitivity to the unsaturated-zone average linear velocity.

Figures 3-29 and 3-30 present scatter plots showing the sensitivity of the peak-dose exposure for 1,000,000 years to the unsaturated-zone average linear velocity and the effective diffusion coefficient, respectively. The peak-dose exposure is more sensitive to the effective diffusion coefficient, especially at high values (Figure 3-29). This is due to greater diffusive releases from the waste package, and lesser retention times in the diffusive pathway. The sensitivity to the unsaturated-zone average linear velocity (Figure 3-30) is slightly less than in the cases without the 1.0-m diffusive pathway.

3.7 SUMMARY

The performance assessment code RIP was used to simulate the baseline case defined in Chapter 2, to show the general agreement with UCBNE-41 calculations. The RIP was then used to demonstrate the sensitivity of the normalized cumulative release and the dose exposure calculated at the accessible environment to the variation in certain waste package/engineered barrier system and geosphere parameters. These included variations in the effective diffusion coefficient, the solubility of neptunium, the unsaturated-zone average linear velocity, the waste package failure time and rate, and the existence of a capillary barrier. The results demonstrating the simulated sensitivity to these parameters were presented as time-history plots, and scatter plots. The generalized results for the dose exposure and the normalized cumulative release are sufficiently similar so as to allow the discussion below to mention only the dose exposure summaries and conclusions.

By comparing the scatter plots for each of the 100 realization runs, it can be seen that for this system, the unsaturated-zone average linear velocity is the dominant factor controlling the magnitude of the dose exposure. This is especially true in cases where the dispersive fronts of some of the significant dose contributors are estimated to reach the accessible environment for simulation times of 10^4 and 10^5 years. The dose exposures are so sensitive to the uncertainty in the unsaturated-zone average linear velocity that it tends to obscure the possible sensitivity of the releases from the waste package to the effective diffusion coefficient and the neptunium solubility.

The time-history plots of peak dose indicate that, as the parameters and engineered barrier system configurations were modified in the sensitivity analysis over the 10,000 to 100,000 year period, the alteration-limited radionuclides and the early breakthrough portions of the solubility-limited radionuclide curves were modified. However, at larger times, the solubility-limited and retarded radionuclides, especially ^{237}Np, were released to the accessible environment at approximately the same concentrations. This is to say that the inclusion of the 1.0-m diffusive pathway and the delay in the waste-package failures had an effect on the early time release of ^{237}Np, but they had no significant effect on the late time releases, especially approaching 10^6 years.

The single realization time-history plots have demonstrated that the peak ^{99}Tc and ^{237}Np dose exposures remained fairly constant between the various sensitivity runs for a 10^6 year simulation period. The only significant decrease in the peak dose exposure of ^{99}Tc and other alteration-limited, low radionuclide retardation, radionuclides occurred by significantly decreasing the waste-package failure rate to one percent per 10^4 years. The most significant decrease in the ^{237}Np peak dose exposure occurred by the inclusion of the 1-m capillary barrier between the waste package and the host rock, and by specifying the effective diffusion coefficient at the lower bound in the log-uniform distribution.

The sensitivity analyses conducted using RIP demonstrated that the peak dose exposures and the normalized cumulative release at the accessible environment over a 10^6 year period were not that sensitive to the change in the engineered barrier system configurations considered. When considering a shorter simulation time of 10^4 or 10^5 years, the sensitivity to the engineered barrier system configurations can be significant. This is especially true for the decrease in the waste-package failure rates and for the system with the 1-m diffusive pathway and low effective diffusion coefficient.

Table 3-1. Radionuclide Inventories and Solubilities Used in RIP Analyses

Radio-nuclide	Total Inventory Ci/MTHM	Total Inventory Ci/container	BWR Inventory Ci/container	PWR Inventory Ci/container	Solubility g/m^3
^{227}Ac	1.97×10^{-5}	4.14×10^{-5}	1.46×10^{-5}	2.68×10^{-5}	1.00×10^{-4}
^{241}Am	3.92×10^{3}	8.23×10^{3}	2.91×10^{3}	5.32×10^{3}	1.00×10^{-4}
^{242}Am	2.33×10^{1}	4.89×10^{1}	1.73×10^{1}	3.17×10^{1}	1.00×10^{-4}
^{242m}Am	2.34×10^{1}	4.91×10^{1}	1.74×10^{1}	3.18×10^{1}	1.00×10^{-4}
^{243}Am	2.82×10^{1}	5.92×10^{1}	2.09×10^{1}	3.83×10^{1}	1.00×10^{-4}
^{14}C	1.48×10^{0}	3.11×10^{0}	1.10×10^{0}	2.01×10^{0}	n/a
^{242}Cm	1.93×10^{1}	4.05×10^{1}	1.43×10^{1}	2.62×10^{1}	1.00×10^{-3}
^{243}Cm	1.70×10^{1}	3.57×10^{1}	1.26×10^{1}	2.31×10^{1}	1.00×10^{-3}
^{244}Cm	1.41×10^{3}	2.96×10^{3}	1.05×10^{3}	1.92×10^{3}	1.00×10^{-3}
^{245}Cm	4.27×10^{-1}	8.97×10^{-1}	3.17×10^{-1}	5.80×10^{-1}	1.00×10^{-3}
^{246}Cm	9.38×10^{-2}	1.97×10^{-1}	6.96×10^{-2}	1.27×10^{-1}	1.00×10^{-3}
^{135}Cs	5.67×10^{-1}	1.91×10^{0}	4.21×10^{-1}	7.70×10^{-1}	n/a
^{129}I	3.72×10^{-2}	7.81×10^{-2}	2.76×10^{-2}	5.05×10^{-2}	n/a
^{94}Nb	8.91×10^{-1}	1.87×10^{0}	6.61×10^{-1}	1.21×10^{0}	n/a
^{237}Np	4.87×10^{-1}	1.02×10^{0}	3.61×10^{-1}	6.62×10^{-1}	1.00×10^{-3}
^{239}Np	2.82×10^{1}	5.92×10^{1}	2.09×10^{1}	3.83×10^{1}	1.00×10^{-3}
^{231}Pa	3.59×10^{-5}	7.54×10^{-5}	2.66×10^{-5}	4.88×10^{-5}	1.00×10^{-3}
^{233}Pa	4.87×10^{-1}	1.02×10^{0}	3.61×10^{-1}	6.62×10^{-1}	1.00×10^{-3}
^{210}Pb	7.15×10^{-7}	1.50×10^{-6}	5.30×10^{-7}	9.71×10^{-7}	1.00×10^{-1}
^{238}Pu	3.57×10^{3}	7.50×10^{3}	2.65×10^{3}	4.85×10^{3}	1.00×10^{-3}

Table 3-1. Radionuclide Inventories and Solubilities Used in RIP Analyses (Continued)

Radio-nuclide	Total Inventory Ci/MTHM	Total Inventory Ci/container	BWR Inventory Ci/container	PWR Inventory Ci/container	Solubility g/m^3
^{239}Pu	3.75×10^{2}	7.88×10^{2}	2.78×10^{2}	5.09×10^{2}	1.00×10^{-3}
^{240}Pu	5.73×10^{2}	1.20×10^{3}	4.25×10^{2}	7.78×10^{2}	1.00×10^{-3}
^{241}Pu	3.56×10^{4}	7.48×10^{4}	2.64×10^{4}	4.84×10^{4}	1.00×10^{-3}
^{242}Pu	2.18×10^{0}	4.58×10^{0}	1.62×10^{0}	2.96×10^{0}	1.00×10^{-3}
^{225}Ra	4.33×10^{-7}	9.09×10^{-7}	3.21×10^{-7}	5.88×10^{-7}	1.00×10^{-2}
^{226}Ra	2.64×10^{-6}	5.54×10^{-6}	1.96×10^{-6}	3.59×10^{-6}	1.00×10^{-2}
^{79}Se	4.80×10^{-1}	1.01×10^{0}	3.56×10^{-1}	6.52×10^{-1}	n/a
^{126}Sn	9.25×10^{-1}	1.94×10^{0}	6.86×10^{-1}	1.26×10^{0}	1.00×10^{-3}
^{99}Tc	1.51×10^{1}	3.17×10^{1}	1.12×10^{1}	2.05×10^{1}	n/a
^{229}Th	4.32×10^{-7}	9.07×10^{-7}	3.20×10^{-7}	5.87×10^{-7}	1.00×10^{-3}
^{230}Th	3.79×10^{-4}	7.96×10^{-4}	2.81×10^{-4}	5.15×10^{-4}	1.00×10^{-3}
^{231}Th	1.68×10^{-2}	3.53×10^{-2}	1.25×10^{-2}	2.28×10^{-2}	1.00×10^{-3}
^{232}Th	4.71×10^{-10}	9.89×10^{-10}	3.49×10^{-10}	6.40×10^{-10}	1.00×10^{-3}
^{234}Th	3.15×10^{-1}	6.62×10^{-1}	2.34×10^{-1}	4.28×10^{-1}	1.00×10^{-3}
^{233}U	7.82×10^{-5}	1.64×10^{-5}	5.80×10^{-5}	1.06×10^{-4}	1.00×10^{-3}
^{234}U	1.43×10^{0}	3.00×10^{0}	1.06×10^{0}	1.94×10^{0}	1.00×10^{-3}
^{235}U	1.68×10^{-2}	3.53×10^{-2}	1.25×10^{-2}	2.28×10^{-2}	1.00×10^{-3}
^{236}U	2.93×10^{-1}	6.15×10^{-1}	2.17×10^{-1}	3.98×10^{-1}	1.00×10^{-3}
^{237}U	8.73×10^{-1}	1.83×10^{0}	6.48×10^{-1}	1.19×10^{0}	1.00×10^{-3}
^{238}U	3.14×10^{-1}	6.59×10^{-1}	2.33×10^{-1}	4.27×10^{-1}	1.00×10^{-3}

Table 3-2. Specific Activities, Retardation Factors, EPA Limits, and Dose Conversion Factors Used in RIP Analyses

Radionuclide	Specific Activity Ci/g	Retardation Factor	EPA Table 1 Limit Ci/ MTHM	Dose Conversion Factor Sv-m^3/g-yr	Principal Dose Producing
^{227}Ac	7.24×10^{1}	1,500	2.00×10^{-1}	2.29×10^{7}	--
^{241}Am	3.43×10^{0}	1,500	1.00×10^{-1}	2.81×10^{5}	^{237}Np
^{242}Am	8.08×10^{5}	1,500	1.00×10^{-1}	2.56×10^{7}	^{238}Pu
^{242m}Am	9.73×10^{0}	1,500	1.00×10^{-1}	7.67×10^{5}	^{238}Pu
^{243}Am	1.99×10^{-1}	1,500	1.00×10^{-1}	1.62×10^{4}	^{239}Pu
^{14}C	4.46×10^{0}	1	1.00×10^{-1}	1.72×10^{4}	--
^{242}Cm	3.32×10^{3}	500	1.00×10^{0}	8.57×10^{6}	^{238}U
^{243}Cm	5.16×10^{1}	500	1.00×10^{0}	2.92×10^{6}	^{239}Pu
^{244}Cm	8.32×10^{1}	500	2.00×10^{-1}	3.76×10^{6}	^{240}Pu
^{245}Cm	1.72×10^{-1}	500	1.00×10^{-1}	1.44×10^{4}	^{241}Am
^{246}Cm	3.08×10^{-1}	500	1.00×10^{-1}	2.57×10^{4}	^{242}Pu
^{135}Cs	1.15×10^{-3}	150	1.00×10^{1}	1.23×10^{1}	--
^{129}I	1.77×10^{-4}	1	1.00×10^{-1}	2.05×10^{0}	--
^{94}Nb	1.88×10^{-1}	50	1.00×10^{0}	2.42×10^{3}	--
^{237}Np	7.05×10^{-4}	16	1.00×10^{-1}	1.31×10^{3}	^{233}U
^{239}Np	2.32×10^{5}	16	1.00×10^{-1}	3.17×10^{8}	^{239}Pu
^{231}Pa	4.73×10^{-2}	1,500	1.00×10^{-1}	1.13×10^{4}	^{227}Ac
^{233}Pa	2.08×10^{4}	1,500	1.00×10^{-1}	1.69×10^{6}	^{233}U
^{210}Pb	7.65×10^{1}	50	1.00×10^{0}	2.17×10^{7}	--
^{238}Pu	1.71×10^{1}	1,820	1.00×10^{-1}	3.10×10^{4}	^{234}U

Table 3-2. Specific Activities, Retardation Factors, EPA Limits, and Dose Conversion Factors Used in RIP Analyses (Continued)

Radionuclide	Specific Activity Ci/g	Retardation Factor	EPA Table 1 Limit Ci/ MTHM	Dose Conversion Factor Sv-m^3/g-yr	Principal Dose Producing
^{239}Pu	6.22×10^{-2}	1,820	1.00×10^{-1}	1.24×10^{2}	^{235}U
^{240}Pu	2.28×10^{-1}	1,820	1.00×10^{-1}	4.56×10^{2}	^{236}U
^{241}Pu	1.03×10^{2}	1,820	2.00×10^{-1}	3.96×10^{3}	^{241}Am
^{242}Pu	3.92×10^{-3}	1,820	1.00×10^{-1}	7.44×10^{0}	^{238}U
^{225}Ra	3.92×10^{4}	500	1.00×10^{-1}	2.03×10^{9}	--
^{226}Ra	9.89×10^{-1}	500	1.00×10^{-1}	8.78×10^{4}	^{210}Pb
^{79}Se	6.97×10^{-2}	25	1.00×10^{0}	1.10×10^{3}	--
^{126}Sn	2.84×10^{-2}	1500	1.00×10^{0}	6.08×10^{2}	--
^{99}Tc	1.70×10^{-2}	5	1.00×10^{1}	7.72×10^{0}	--
^{229}Th	2.13×10^{-1}	5,000	1.00×10^{-1}	4.38×10^{3}	--
^{230}Th	2.02×10^{-2}	5,000	1.00×10^{-2}	6.00×10^{1}	^{226}Ra
^{231}Th	5.32×10^{5}	5,000	1.00×10^{-2}	3.90×10^{6}	^{231}Pa
^{232}Th	1.10×10^{-7}	5,000	1.00×10^{-2}	1.62×10^{-3}	--
^{234}Th	2.32×10^{4}	5,000	1.00×10^{-2}	1.71×10^{6}	^{234}U
^{233}U	9.68×10^{-3}	40	1.00×10^{-1}	1.36×10^{1}	^{229}Th
^{234}U	6.24×10^{-3}	40	1.00×10^{-1}	4.55×10^{0}	^{230}Th
^{235}U	2.16×10^{-6}	40	1.00×10^{-1}	1.48×10^{-3}	^{231}Pa
^{236}U	6.49×10^{-5}	40	1.00×10^{-1}	4.49×10^{-2}	^{232}Th
^{237}U	8.16×10^{4}	40	1.00×10^{-1}	6.58×10^{5}	^{237}Np
^{238}U	3.35×10^{-7}	40	1.00×10^{-1}	3.61×10^{-4}	^{234}U

Table 3-3. Waste Package Release Parameters Used in RIP Analyses

Matrix Dissolution Rate ($g/m^2/yr$)	Matrix Surface Area (m^2/g)	Fraction of Waste Wetted	Package Area (m^2)	Diffusion Coefficient (m^2/yr)	Geometric Factor for Diffusion (m)
1.0×10^{-4}	1.0	1.0	3.14	LU [1] 1.0×10^{-6} to 1.0×10^{-2}	1.0

[1] LU = Log Uniform

Table 3-4. Unsaturated-Zone Transport Properties Used in RIP Analyses

Unsaturated Zone Thickness (m)	Longitudinal Dispersivity (m)	Pore Velocity (m/yr)	Effective Porosity
200	15	LU[1] 1.0×10^{-3} to 1.0×10^{-1}	0.1

[1] LU = Log Uniform

Table 3-5. Saturated-Zone Transport Properties Used in RIP Analyses

Saturated Zone Length (m)	Longitudinal Dispersivity (m)	Pore Velocity (m/yr)	Effective Porosity
5,000	50	1.0	0.1

Table 3-6. Parameters Used in the RIP Analyses

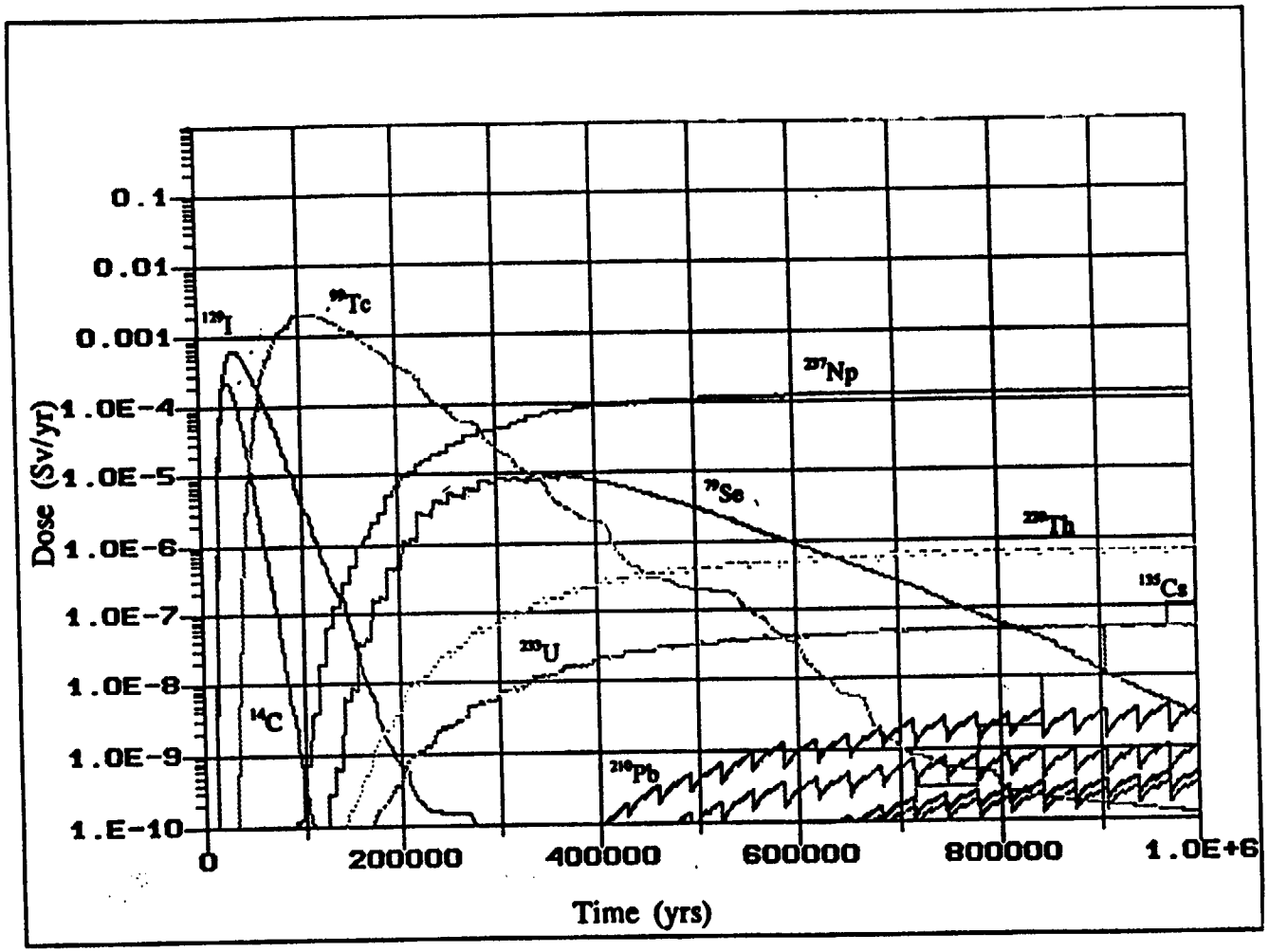
Diffusion Coefficient (m^2/yr)	**Neptunium Solubility (g/m^3)**	**Repository Flux (m/yr)**	**Unsat.-Zone Velocity (m/yr)**	**Failure Time (yrs)**	**Figure**
1.0×10^{-7}	1.0×10^{-3}	1.0×10^{-3}	1.0×10^{-2}	0.0	Fig. 3-1 & 3-2
1.0×10^{-3}	1.0×10^{-3}	0.0	1.0×10^{-2}	0.0	Fig. 3-3 & 3-4
1.0×10^{-3}	1.0	0.0	1.0×10^{-2}	0.0	Fig. 3-5
LU (1) 1.0×10^{-6} to 1.0×10^{-2} (out of waste package)	LU 1.0×10^{-3} to 1.0×10^{2}	0.0	LU 1.0×10^{-3} to 1.0×10^{-1}	0.0	Fig. 3-6, 3-7, 3-8, 3-9, 3-10, 3-13, 3-14, 3-15, 3-16, &3-17
LU 1.0×10^{-6} to 1.0×10^{-2} (out of waste package)	1.0	0.0	LU 1.0×10^{-3} to 1.0×10^{-1}	0.0	Fig. 3-11, 3-12, 3-18, & 3-19

(1) LU = Log Uniform

Table 3-6. Parameters Used in the RIP Analyses (Continued)

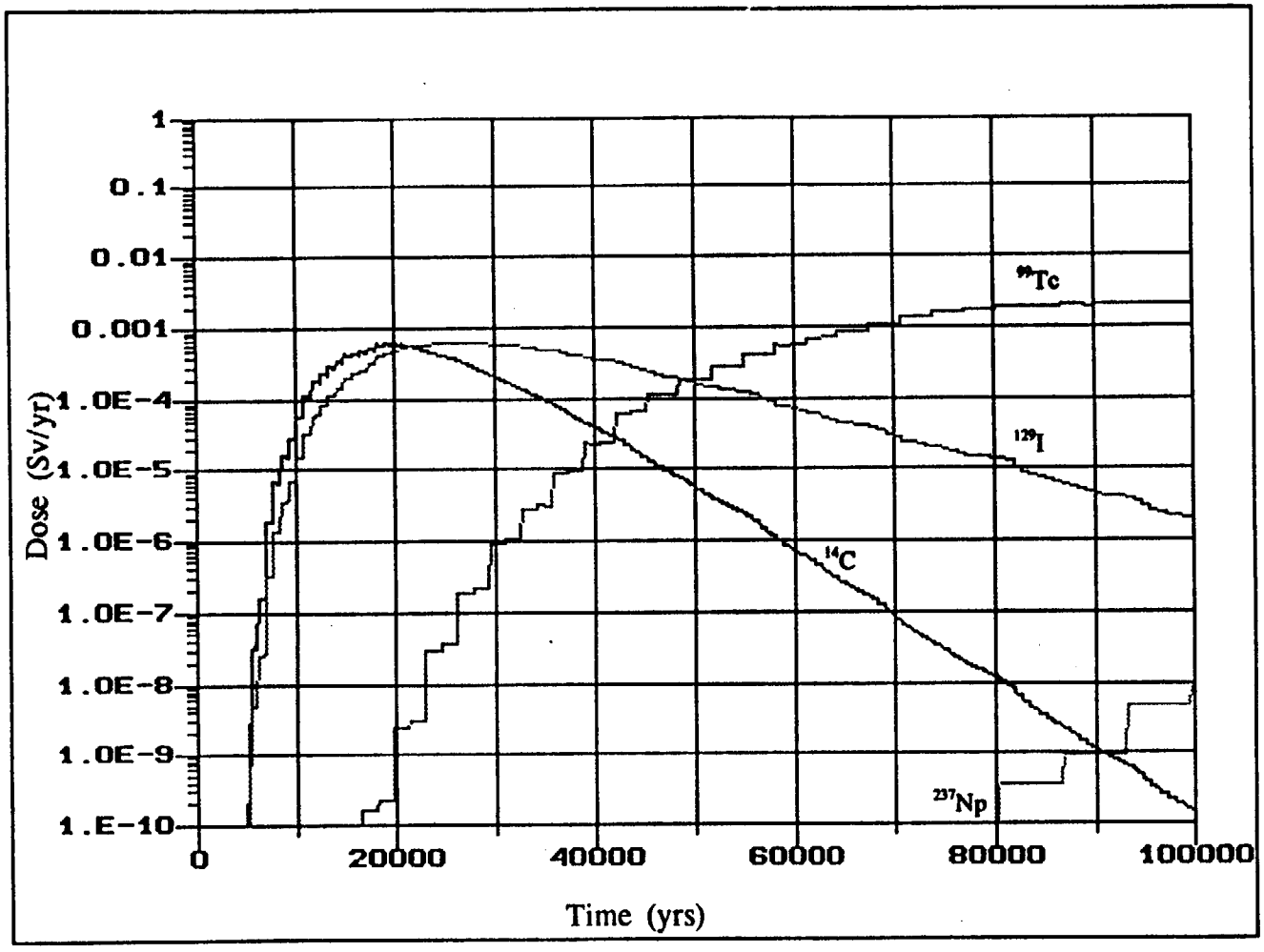
Diffusion Coefficient (m^2/yr)	Neptunium Solubility (g/m^3)	Repository Flux (m/yr)	Unsat.-Zone Velocity (m/yr)	Failure Time (yrs)	Figure
1.0×10^{-3}	1.0	0.0	1.0×10^{-2}	0.0 (3)	Fig. 3-20
1.0×10^{-3}	1.0	0.0	1.0×10^{-2}	1.0×10^{4} (3)	Fig. 3-21
1.0×10^{-3}	1.0	0.0	1.0×10^{-2}	1.0×10^{5} (3)	Fig. 3-22
1.0×10^{-3}	1.0	0.0	1.0×10^{-2}	0.0 (4)	Fig. 3-23
1.0×10^{-4} (2)	1.0	0.0	1.0×10^{-2}	0.0	Fig. 3-24
1.0×10^{-5} (2)	1.0	0.0	1.0×10^{-2}	0.0	Fig. 3-25
1.0×10^{-6} (2)	1.0	0.0	1.0×10^{-2}	0.0	Fig. 3-26
LU(1) 1.0×10^{-6} to 1.0×10^{-2} (2)	1.0	0.0	LU 1.0×10^{-3} to 1.0×10^{-1}	0.0	Fig. 3-27, 3-28, 3-29, & 3-30

(1) LU = Log Uniform
(2) Through engineered barrier
(3) Failure rate 1%/1,000 years
(4) Failure rate 1%/10,000 years



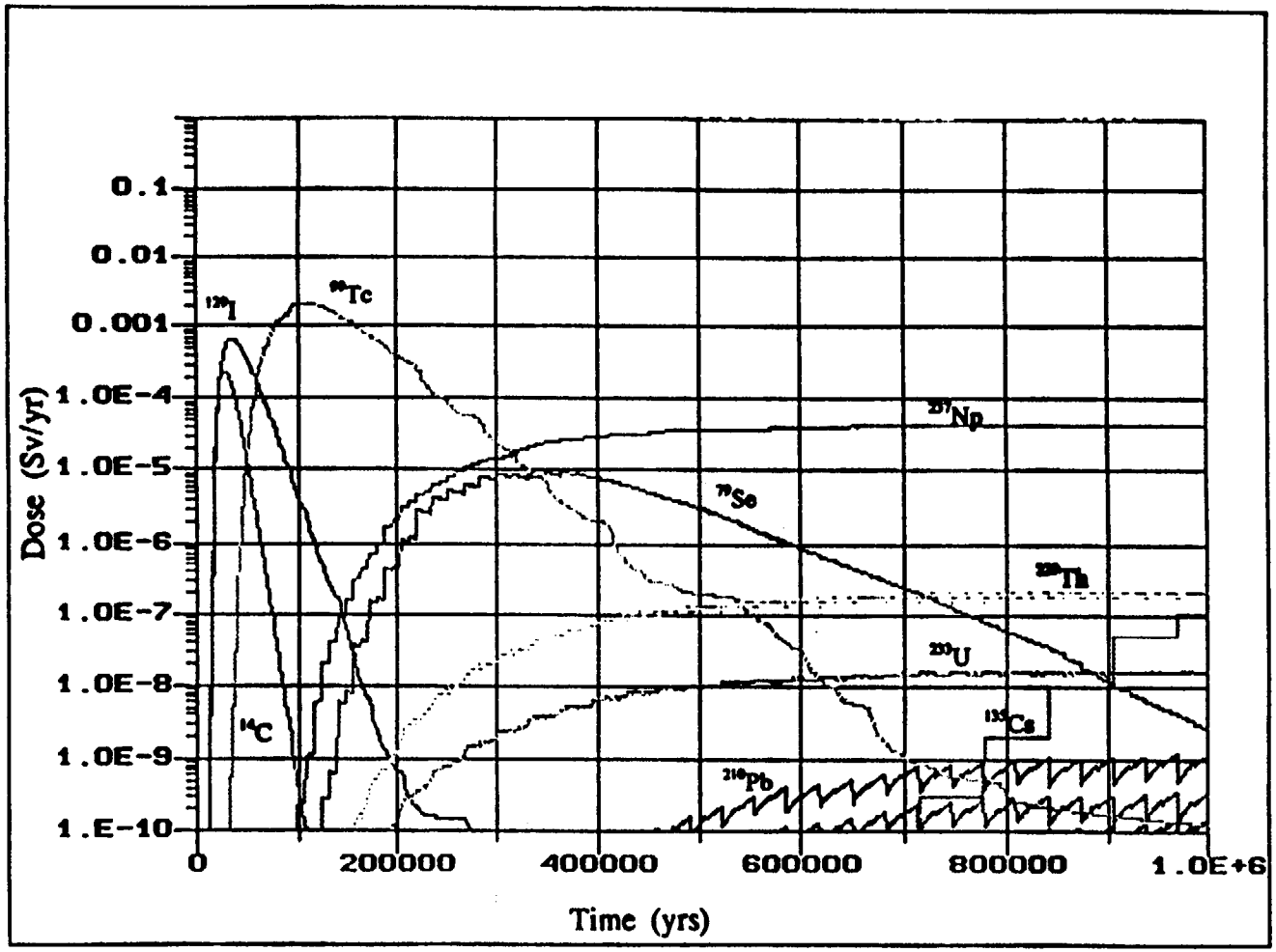
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
1.0×10^{-7}	1.0×10^{-3}	1.0×10^{-3}	1.0×10^{-2}	0.0

Figure 3-1. Dose to an Individual at the Accessible Environment for the Baseline Case



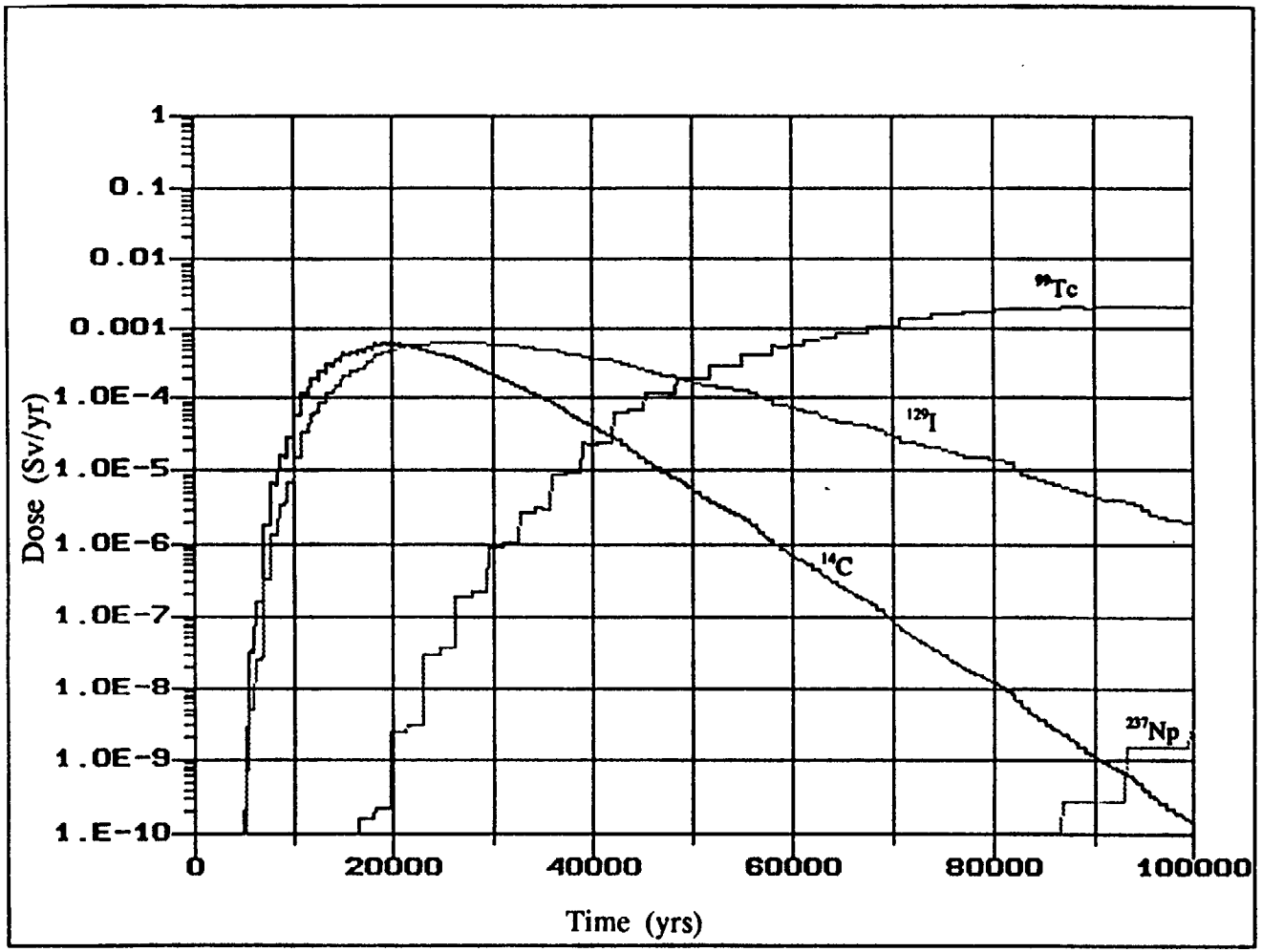
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
1.0×10^{-7}	1.0×10^{-3}	1.0×10^{-3}	1.0×10^{-2}	0.0

Figure 3-2. Dose to an Individual at the Accessible Environment for the Baseline Case for the First 100,000 Years



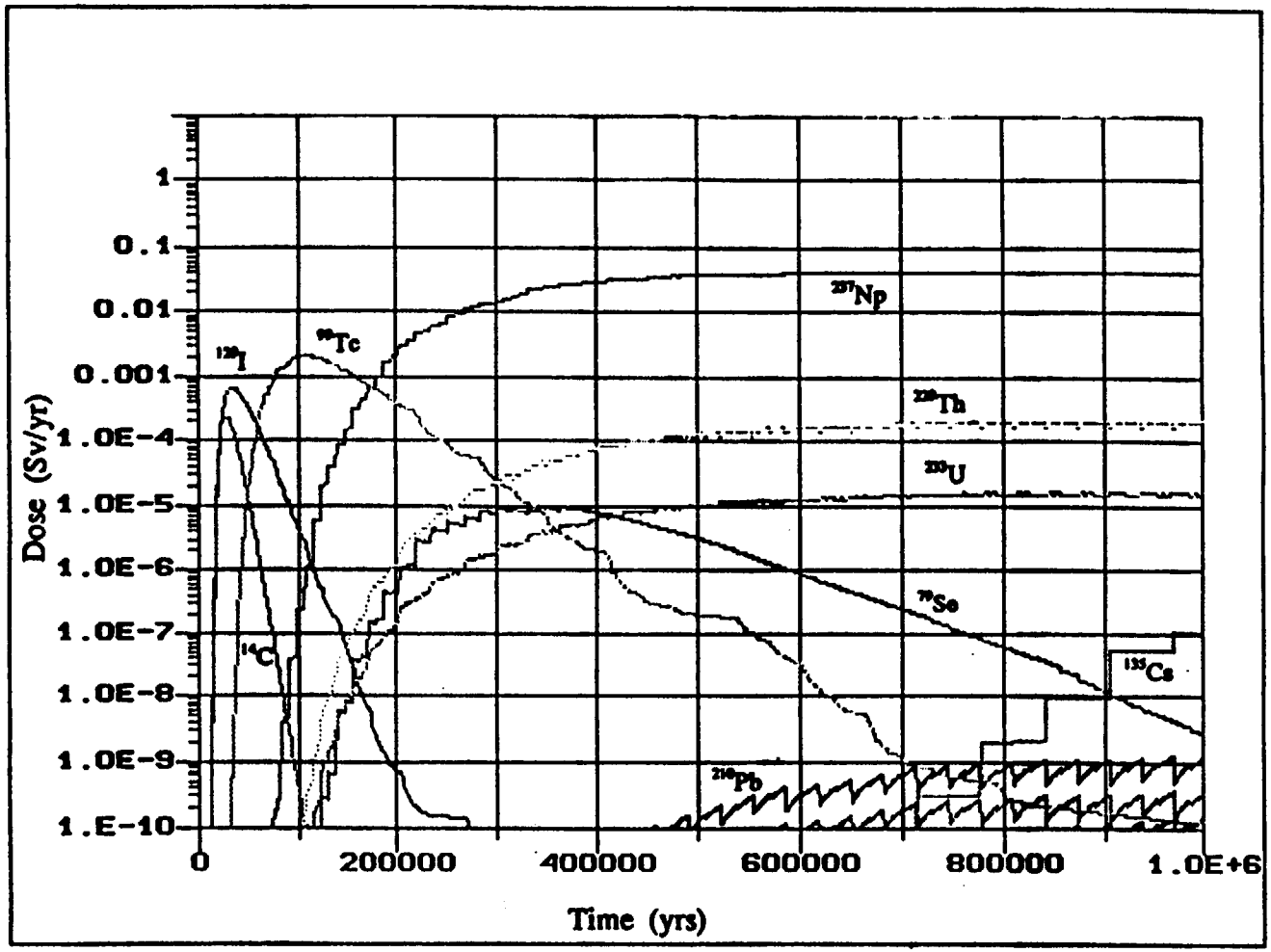
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
1.0×10^{-3}	1.0×10^{-3}	0.0	1.0×10^{-2}	0.0

Figure 3-3. Dose to an Individual at the Accessible Environment with Diffusive Release from the Waste Package



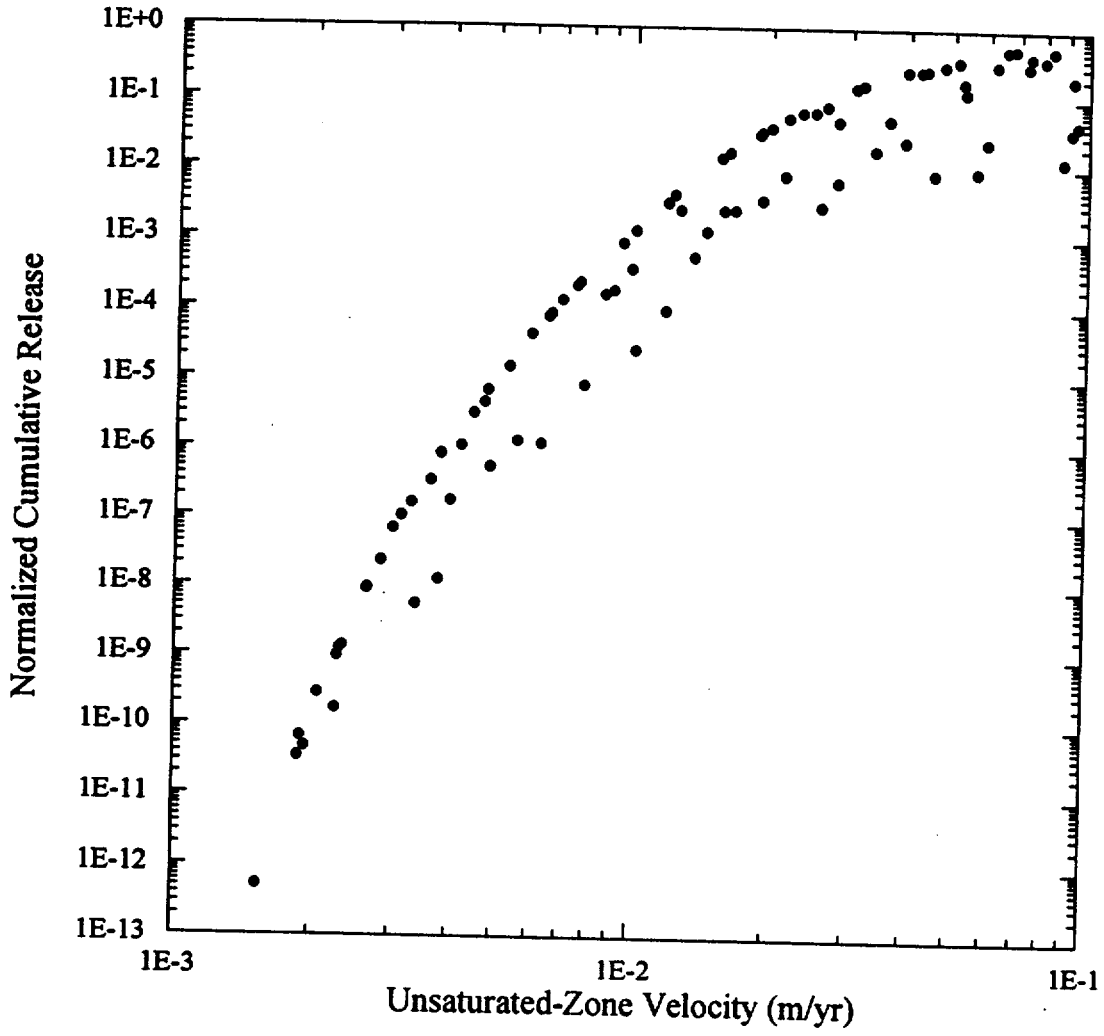
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
1.0×10^{-3}	1.0×10^{-3}	0.0	1.0×10^{-2}	0.0

Figure 3-4. Dose to an Individual at the Accessible Environment for the First 100,000 Years with Diffusive Release from the Waste Package



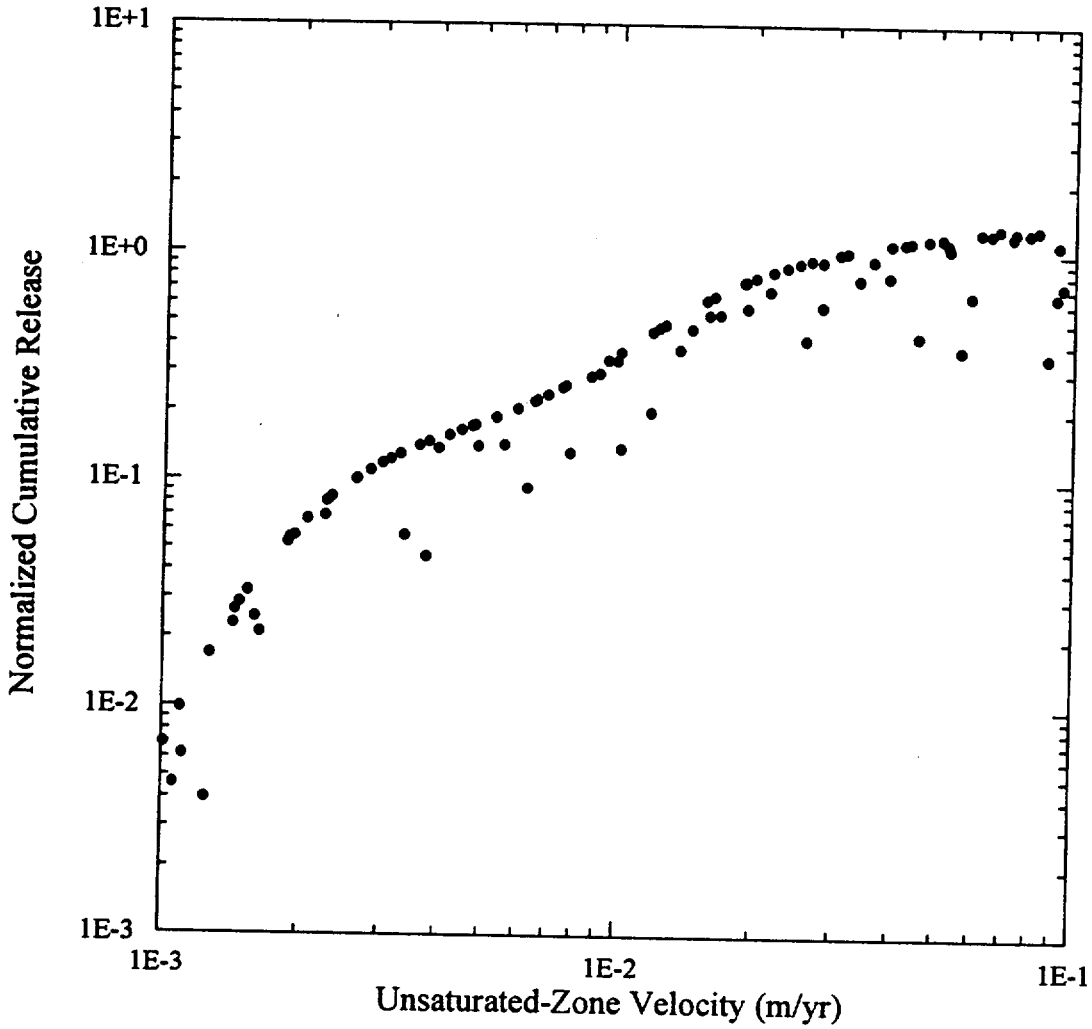
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
1.0×10^{-3}	1.0	0.0	1.0×10^{-2}	0.0

Figure 3-5. Effects of Increased ^{237}Np Solubility on Dose to an Individual at the Accessible Environment



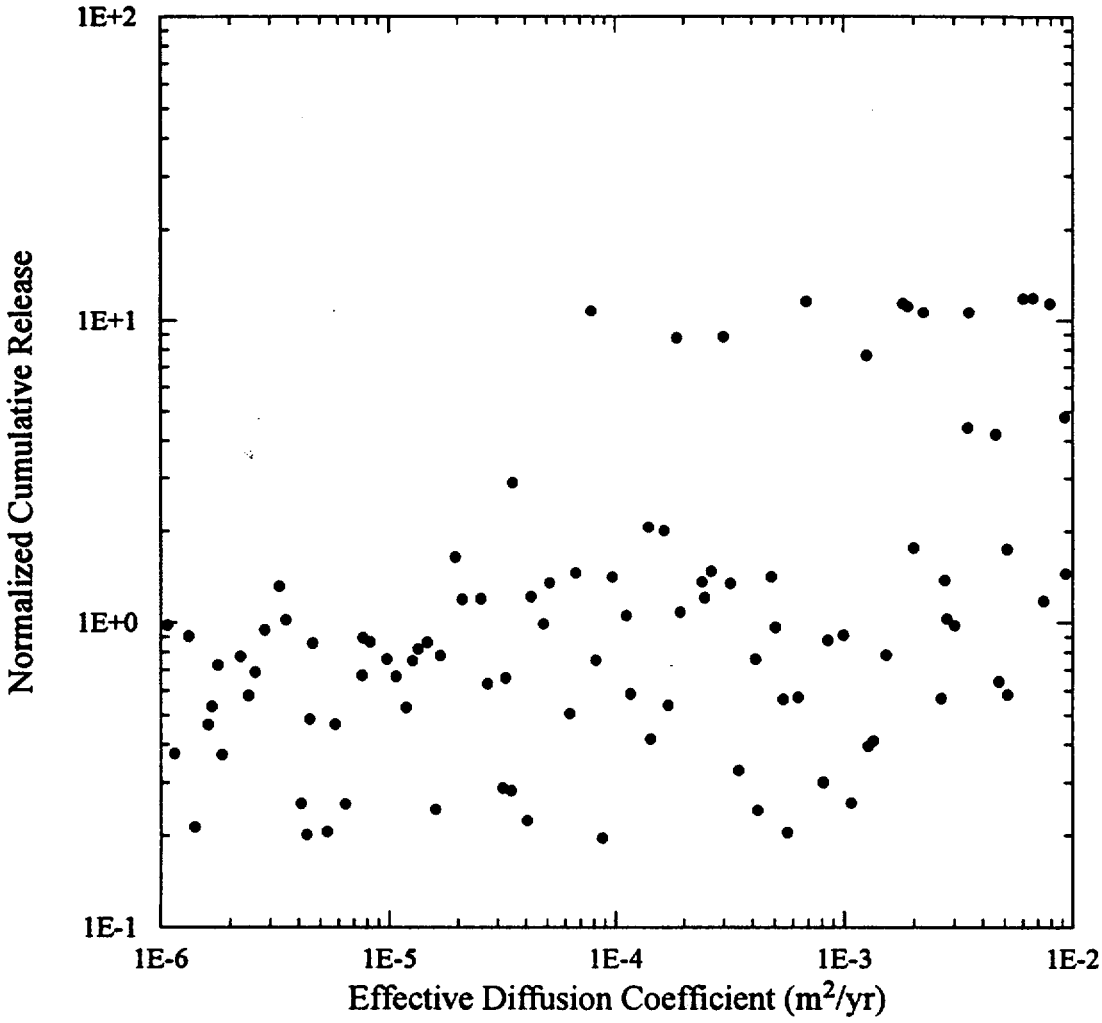
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	10^{-3} to 10^{2}	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-6. Scatter Plot of Normalized Cumulative Release at 10,000 Years as a Function of Unsaturated-Zone Velocity



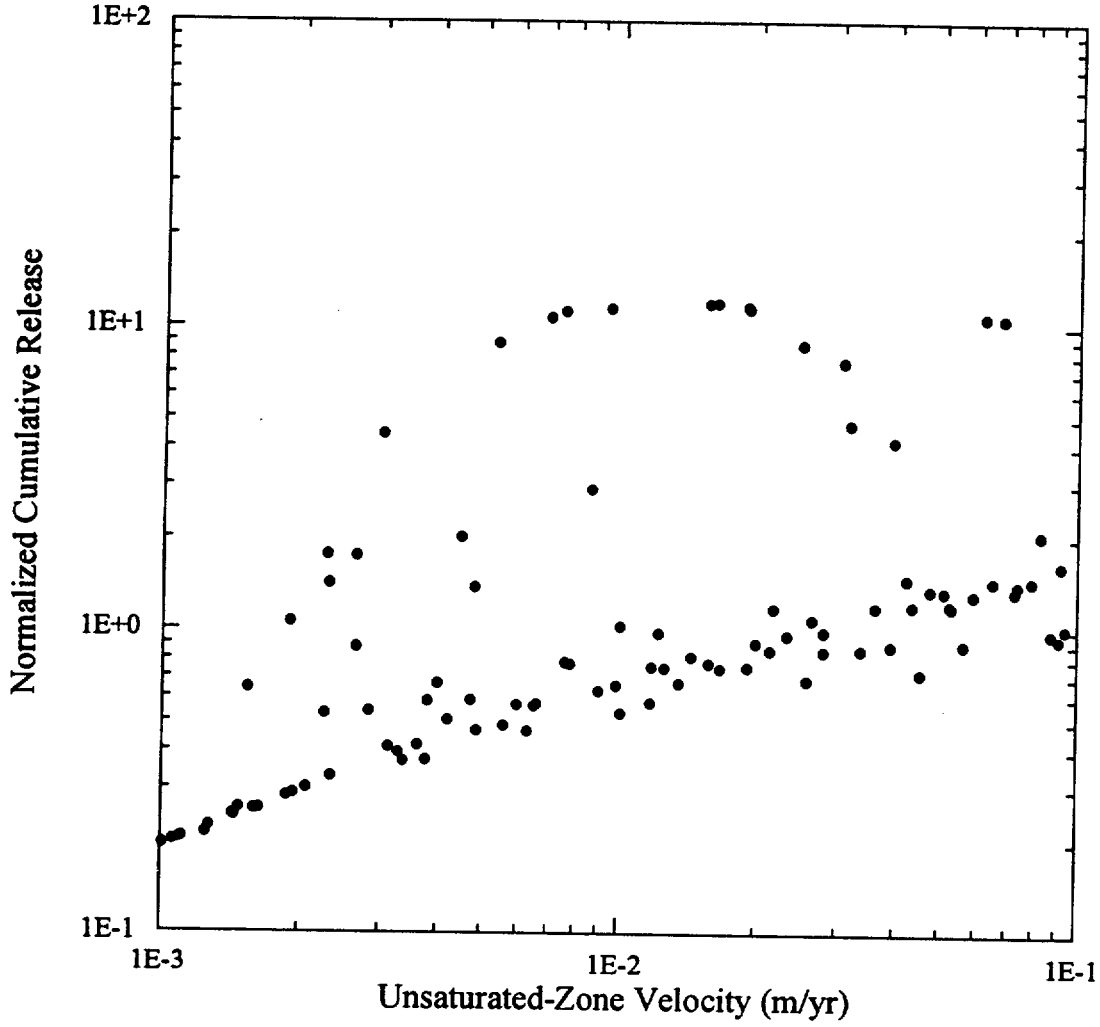
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	10^{-3} to 10^{2}	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-7. Scatter Plot of Normalized Cumulative Release at 100,000 Years as a Function of Unsaturated-Zone Velocity



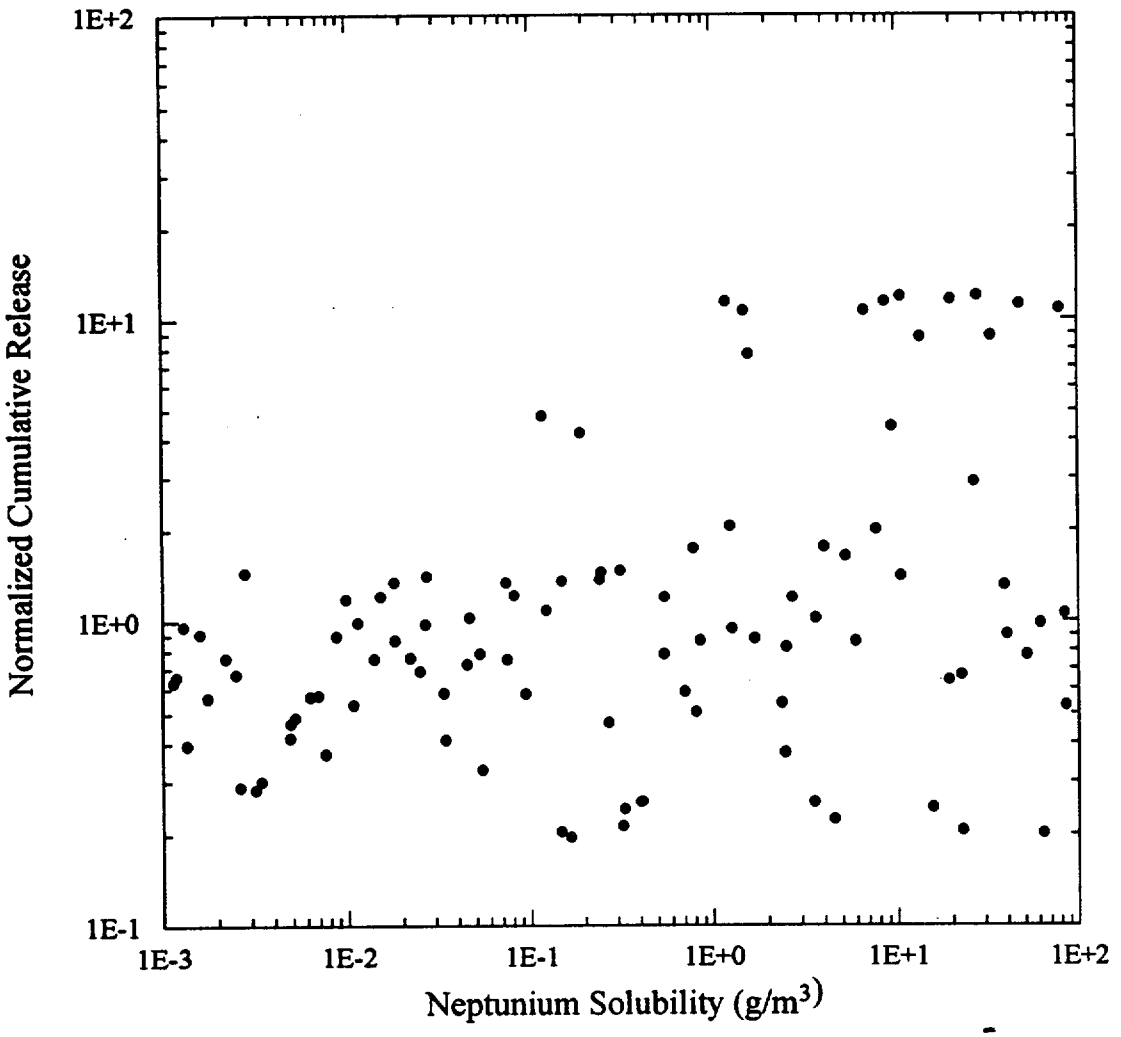
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	10^{-3} to 10^{2}	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-8. Scatter Plot of Normalized Cumulative Release at 1,000,000 Years as a Function of Diffusion Coefficient in the Waste Package



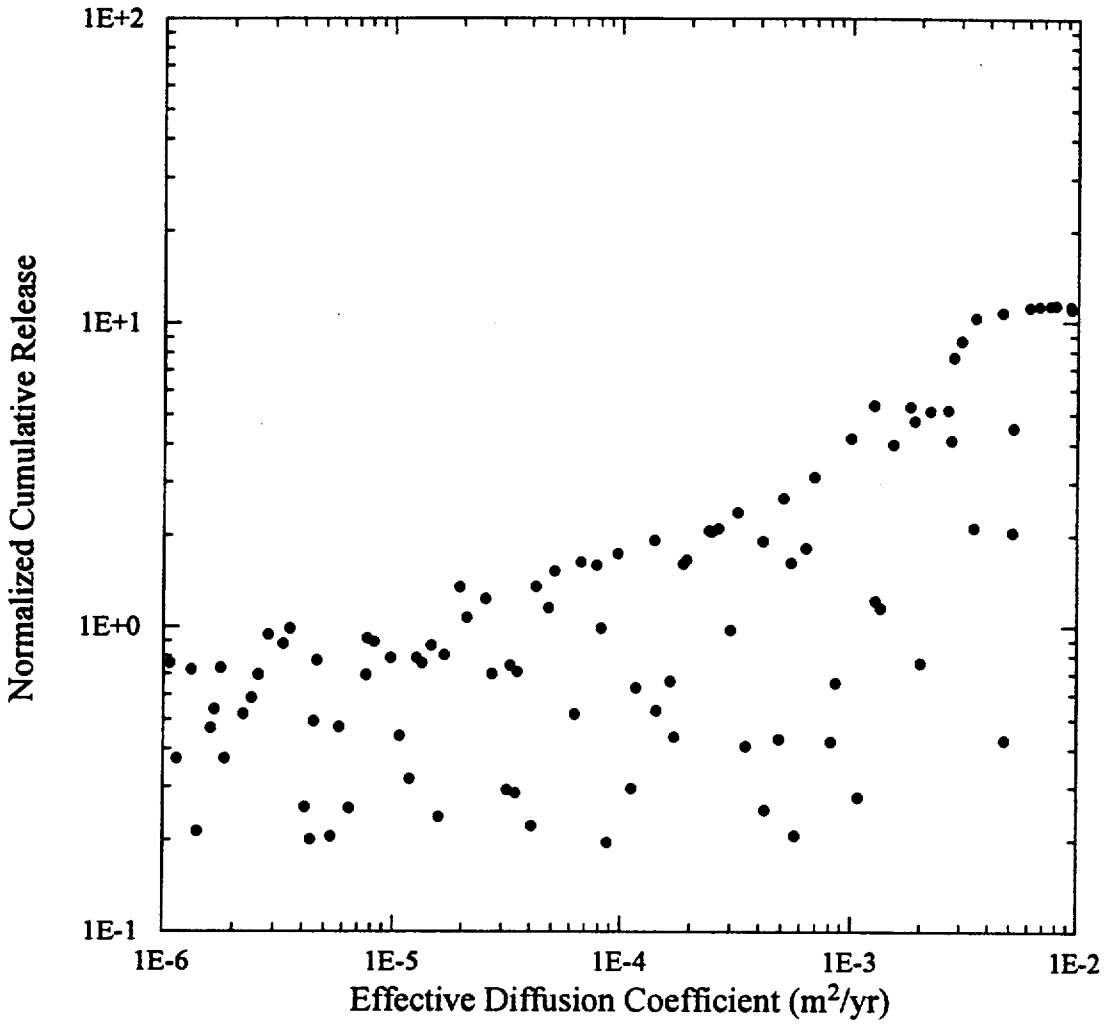
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	10^{-3} to 10^{2}	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-9. Scatter Plot of Normalized Cumulative Release at 1,000,000 Years as a Function of Unsaturated-Zone Velocity



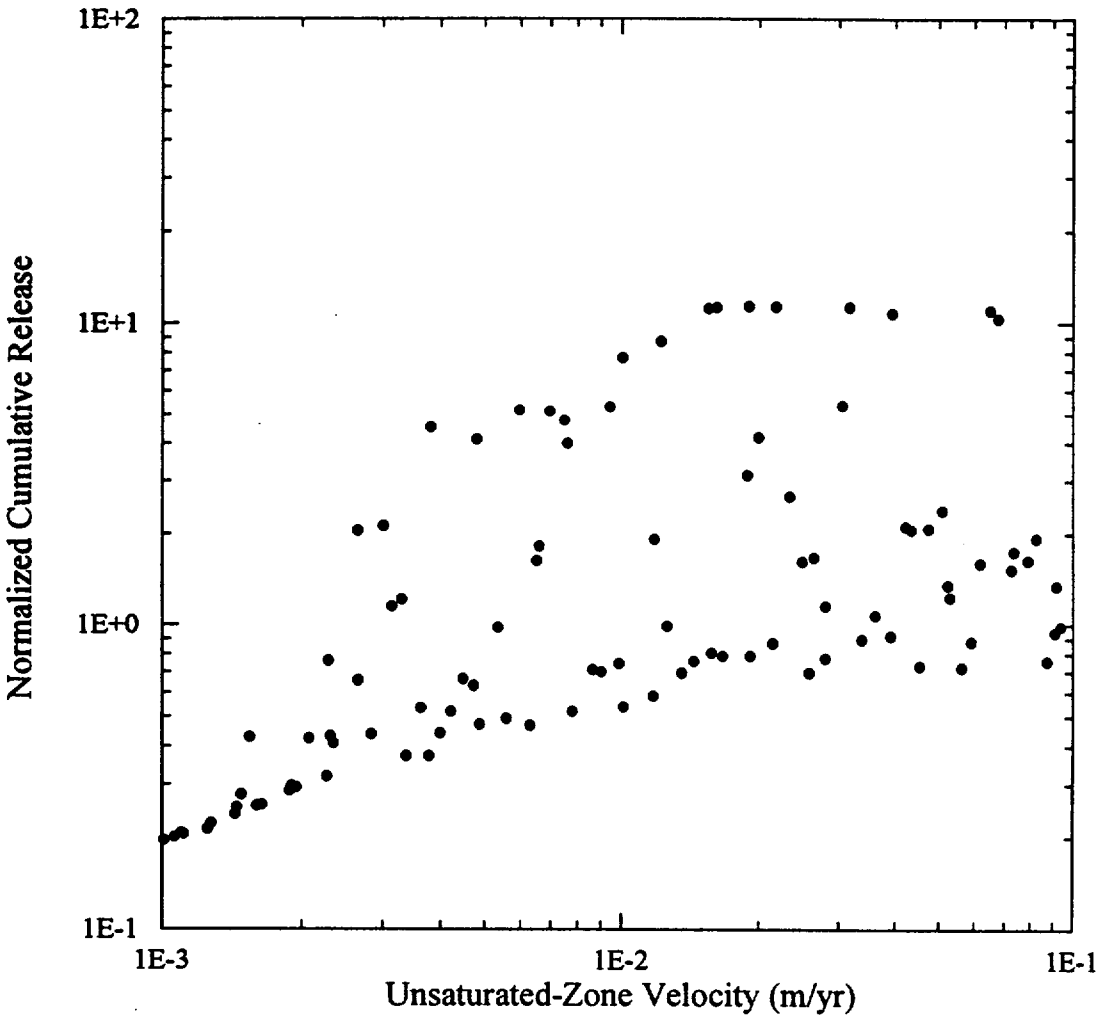
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	10^{-3} to 10^{2}	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-10. Scatter Plot of Normalized Cumulative Release at 1,000,000 Years as a Function of ^{237}Np Solubility.



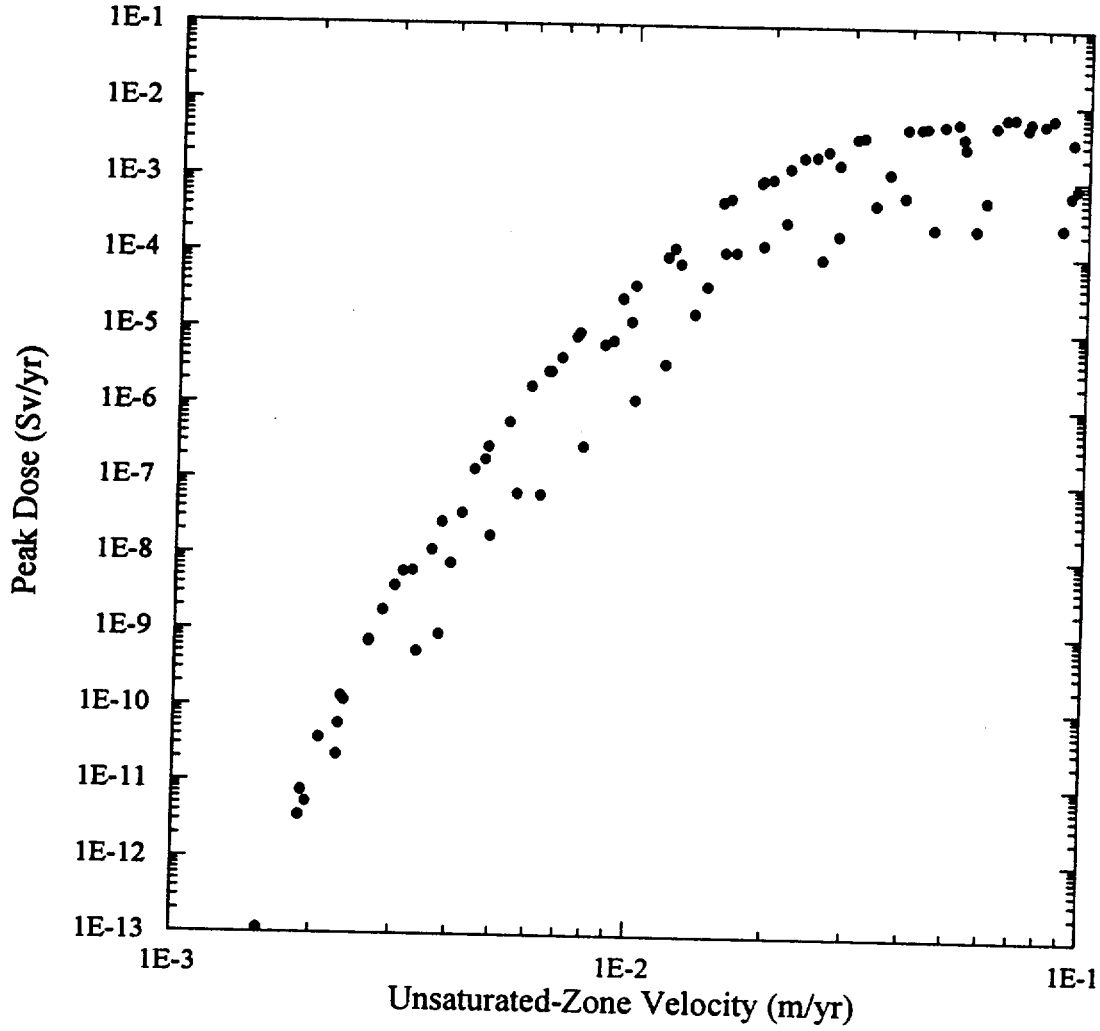
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	1.0	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-11. Scatter Plot of Normalized Cumulative Release at 1,000,000 Years as a Function of Diffusion Coefficient in the Waste Package with ^{237}Np Solubility Fixed at 1.0 g/m^3



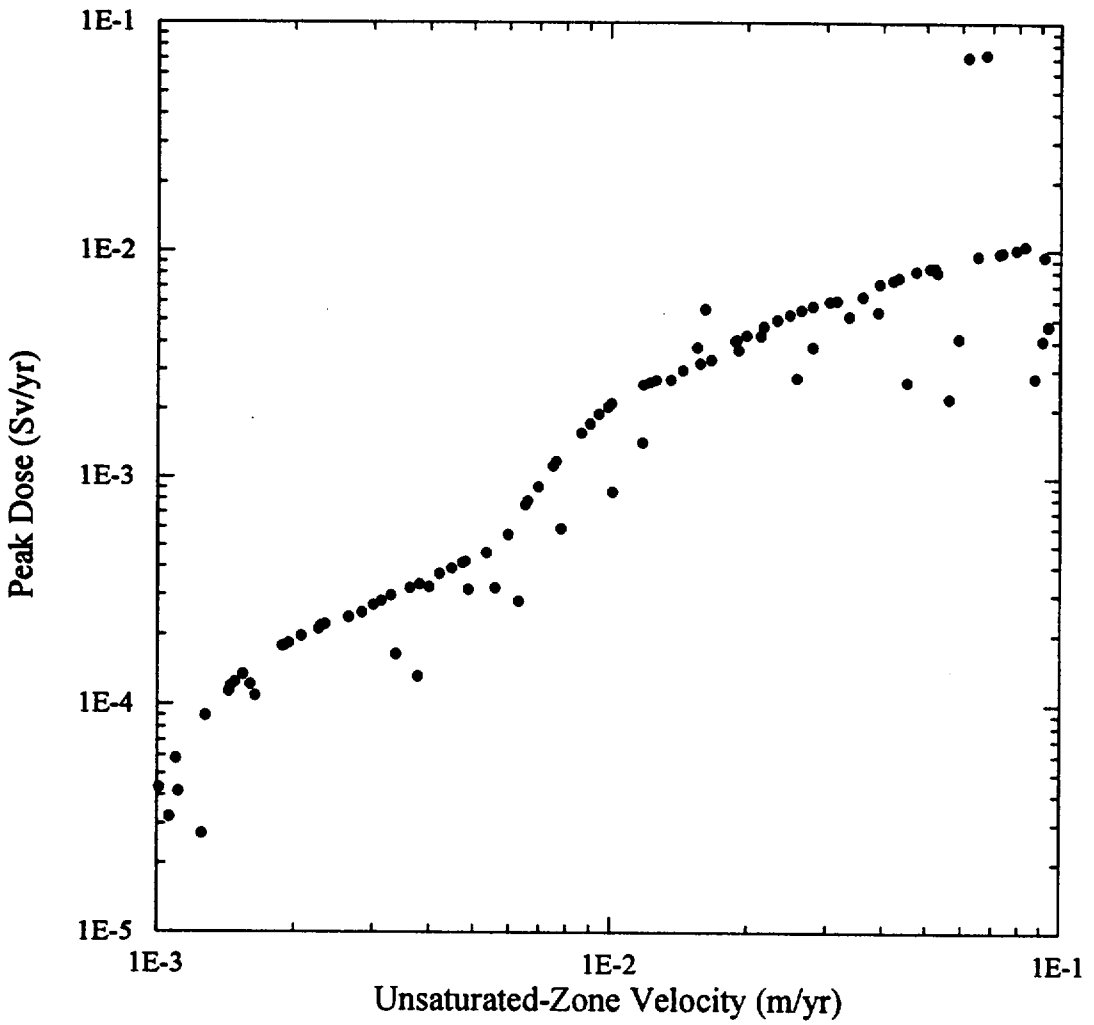
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	1.0	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-12. Scatter Plot of Normalized Cumulative Release at 1,000,000 Years as a Function of Unsaturated-Zone Velocity with ^{237}Np Solubility Fixed at 1.0 g/m^3



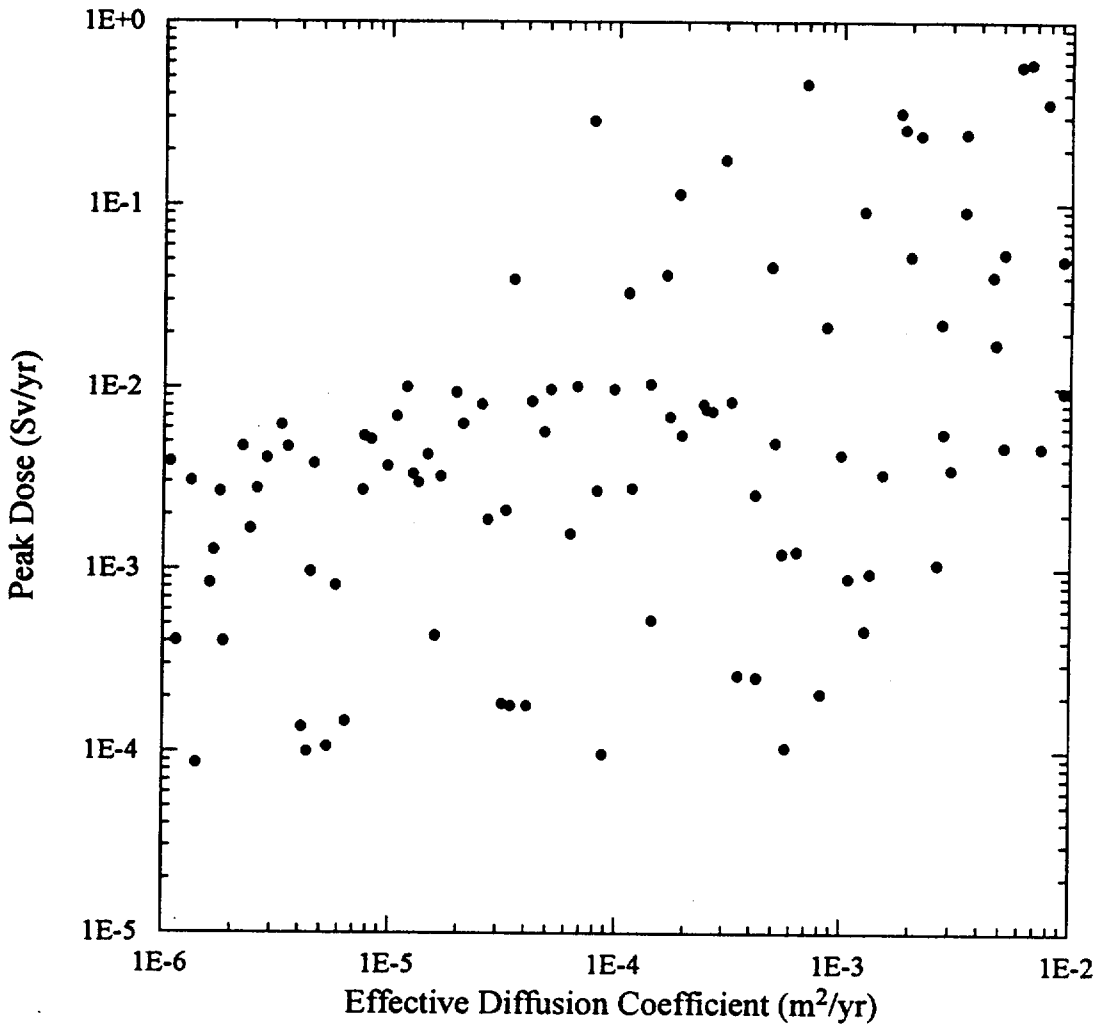
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	10^{-3} to 10^{2}	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-13. Scatter Plot of Peak Dose to an Individual Over 10,000 Years as a Function of Unsaturated-Zone Velocity



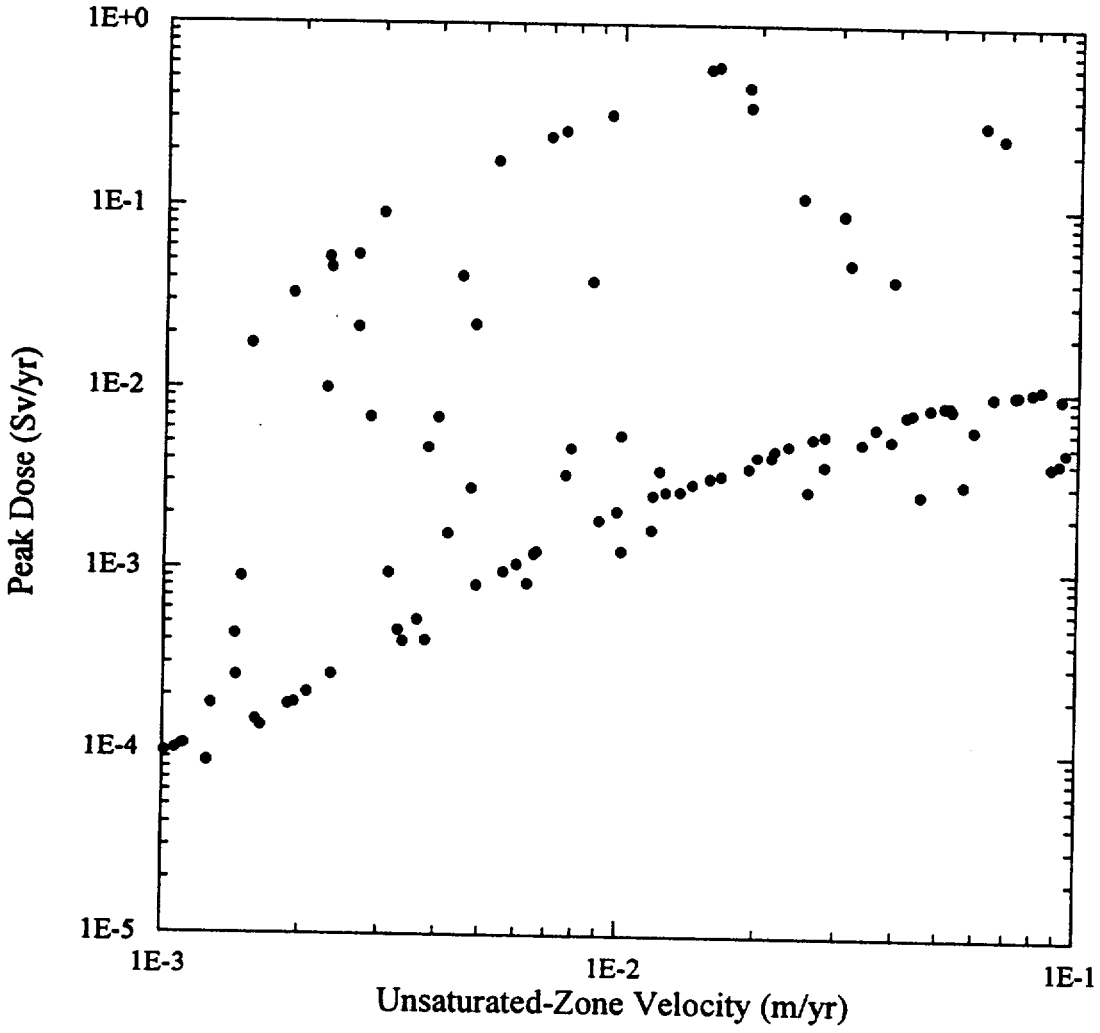
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	10^{-3} to 10^{2}	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-14. Scatter Plot of Peak Dose to an Individual Over 100,000 Years as a Function of Unsaturated-Zone Velocity



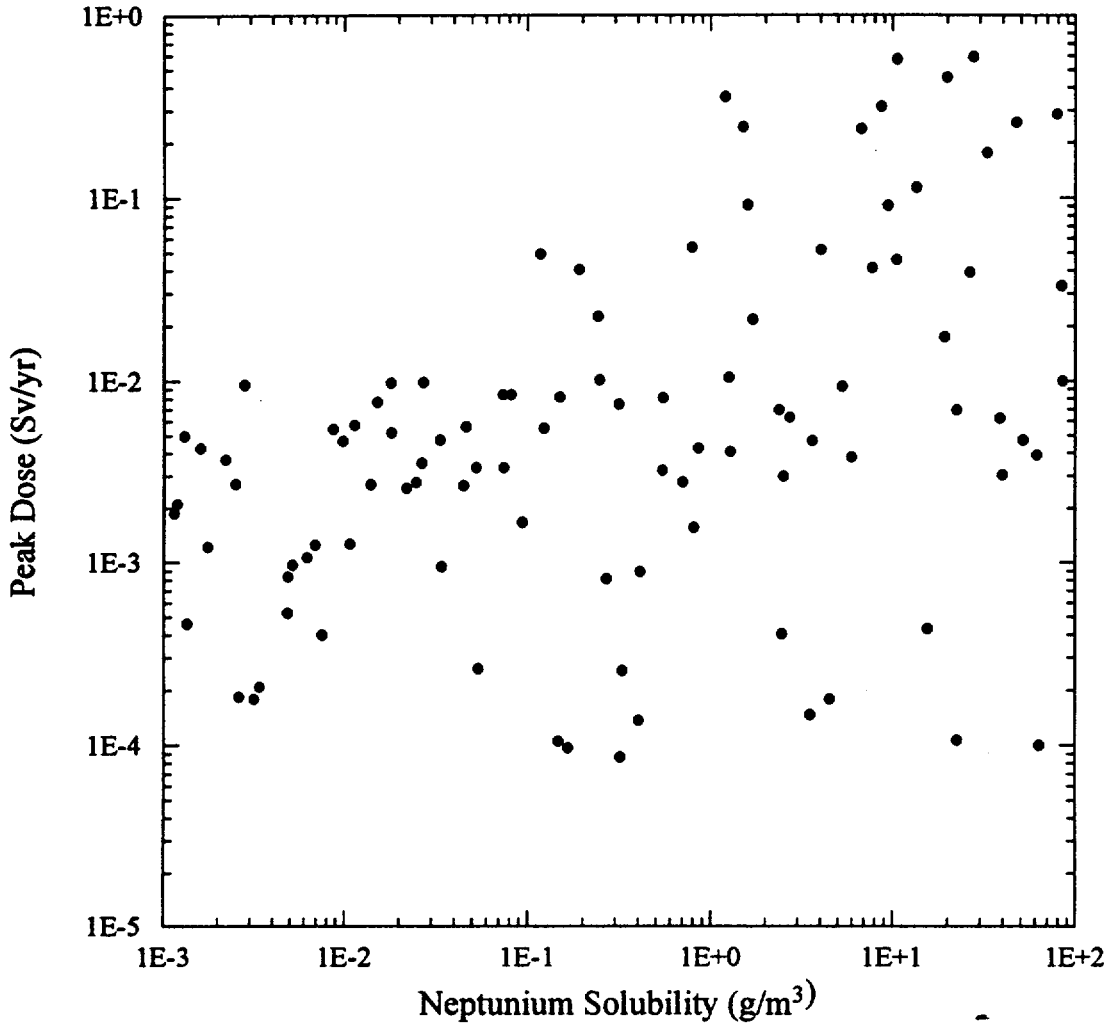
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	10^{-3} to 10^{2}	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-15. Scatter Plot of Peak Dose to an Individual Over 1,000,000 Years as a Function of Diffusion Coefficient in the Waste Package



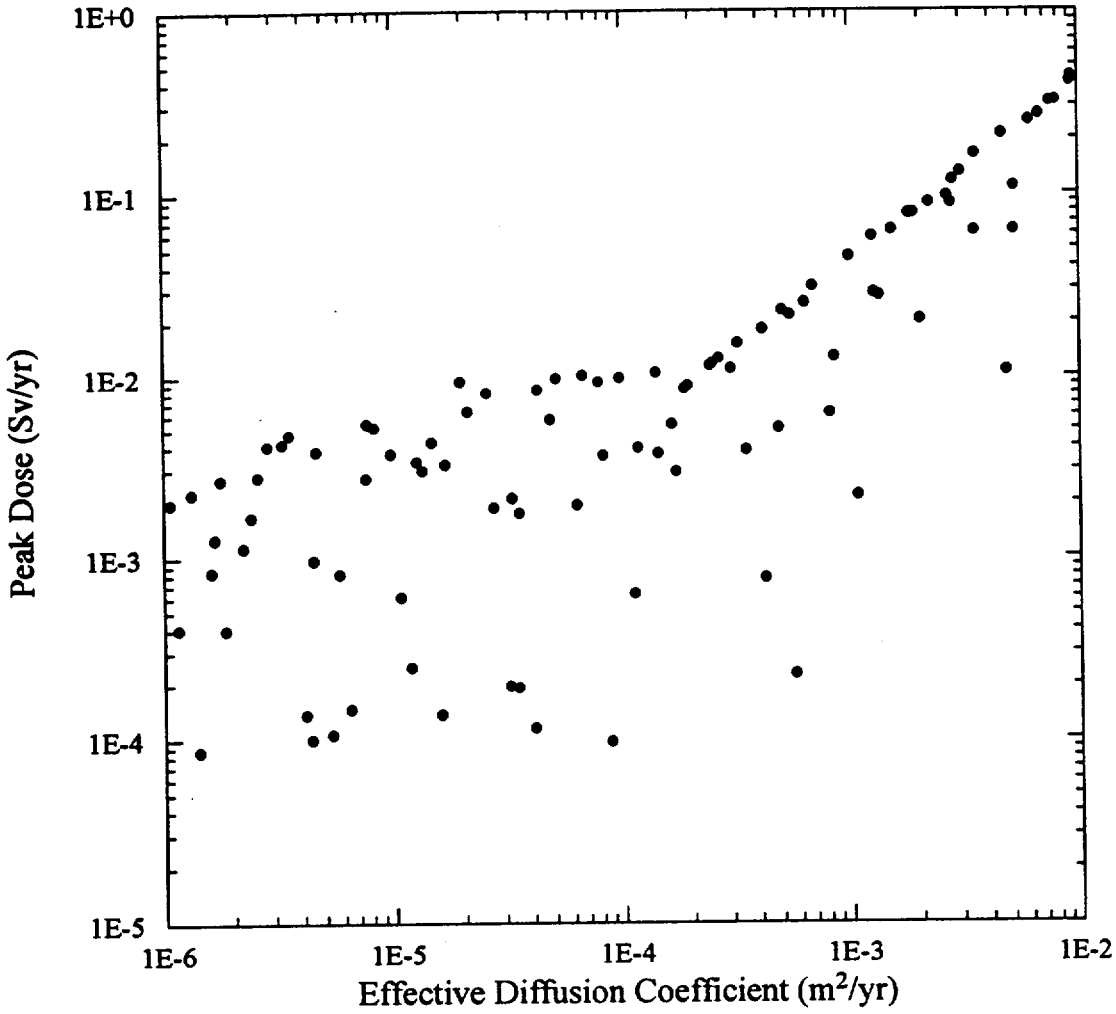
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	10^{-3} to 10^{2}	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-16. Scatter Plot Peak Dose to an Individual Over 1,000,000 Years as a Function of Unsaturated-Zone Velocity



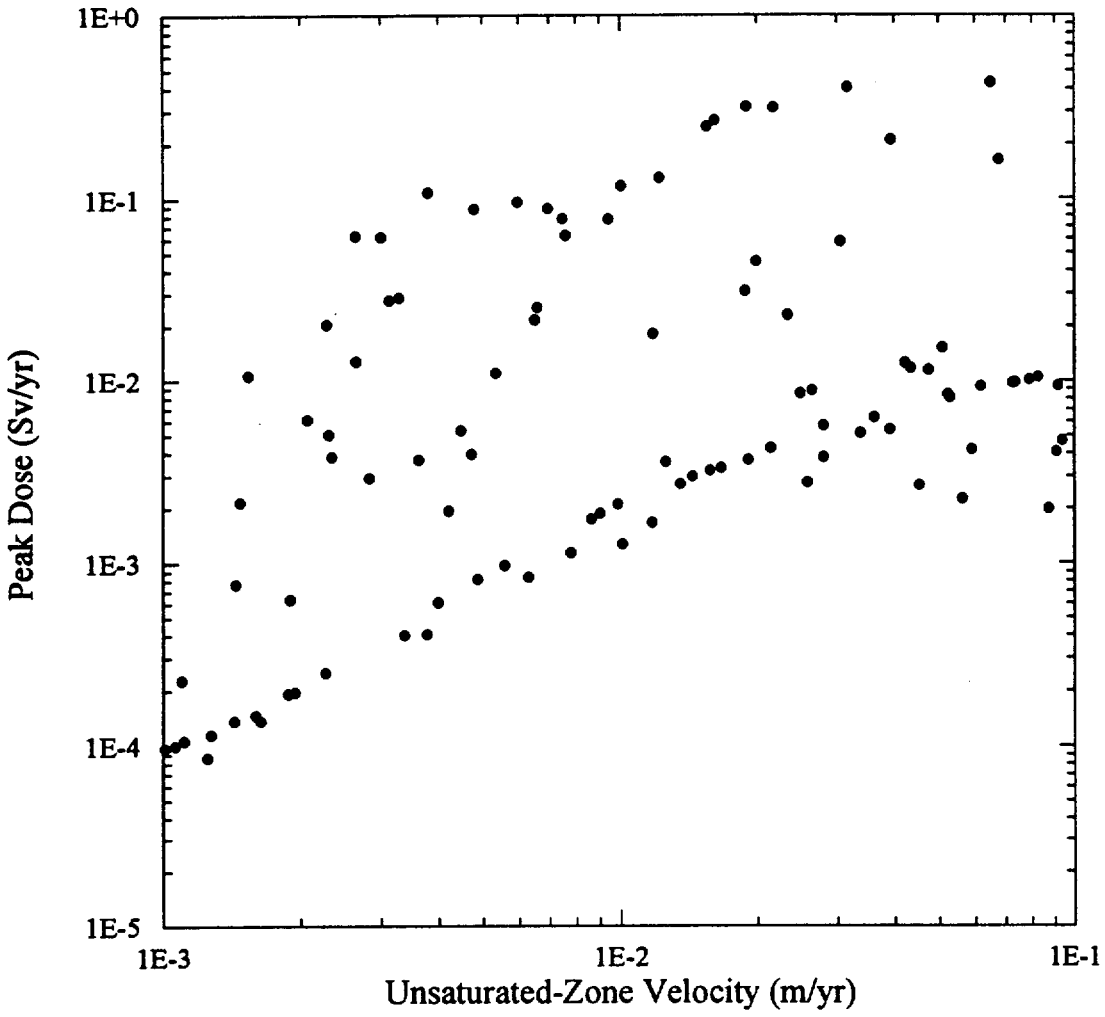
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	10^{-3} to 10^{2}	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-17. Scatter Plot of Peak Dose to an Individual Over 1,000,000 Years as a Function of ^{237}Np Solubility



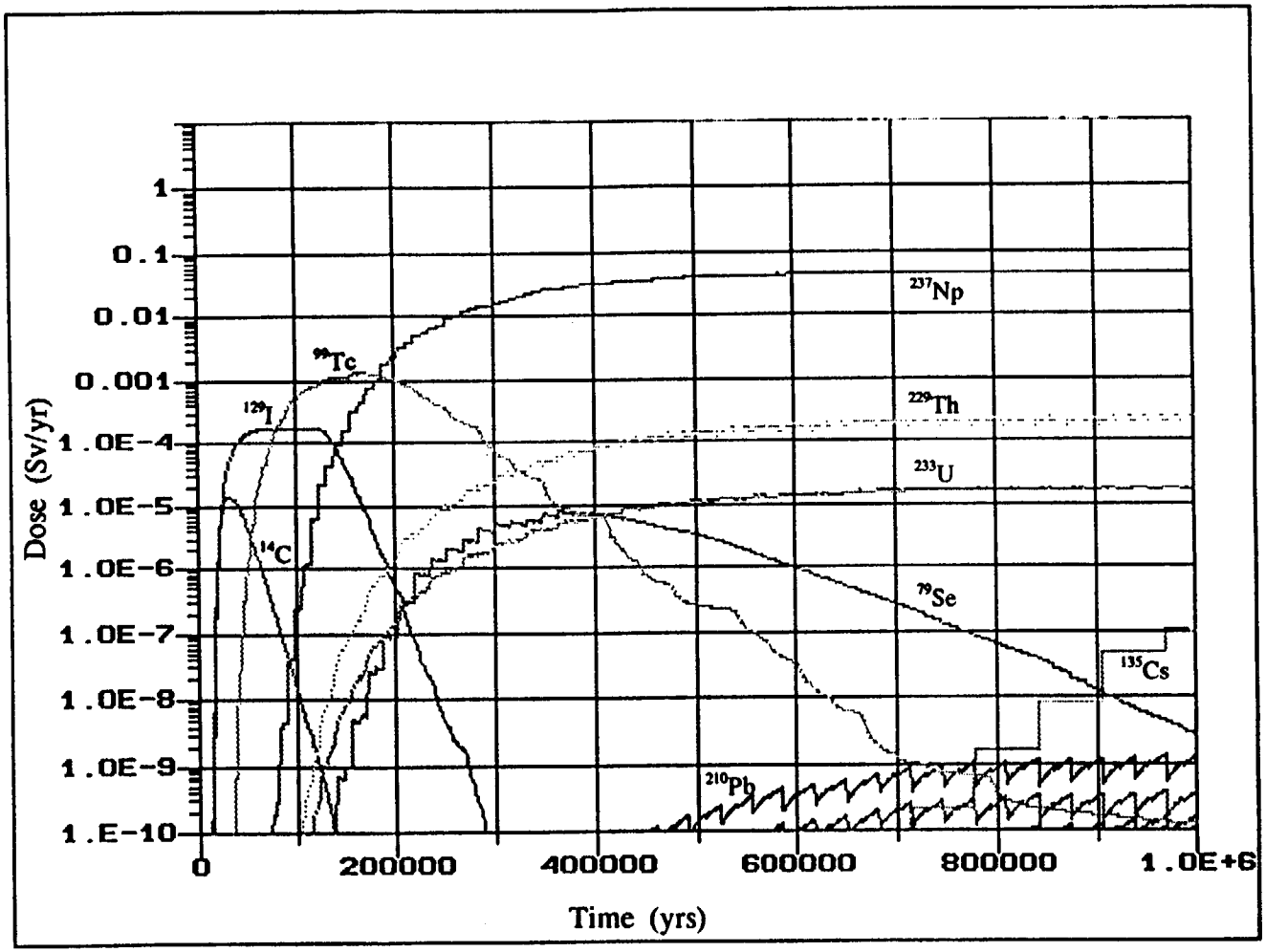
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	1.0	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-18. Scatter Plot of Peak Dose to an Individual Over 1,000,000 Years as a Function of Diffusion Coefficient in the Waste Package with ^{237}Np Solubility Fixed at 1.0 g/m^3



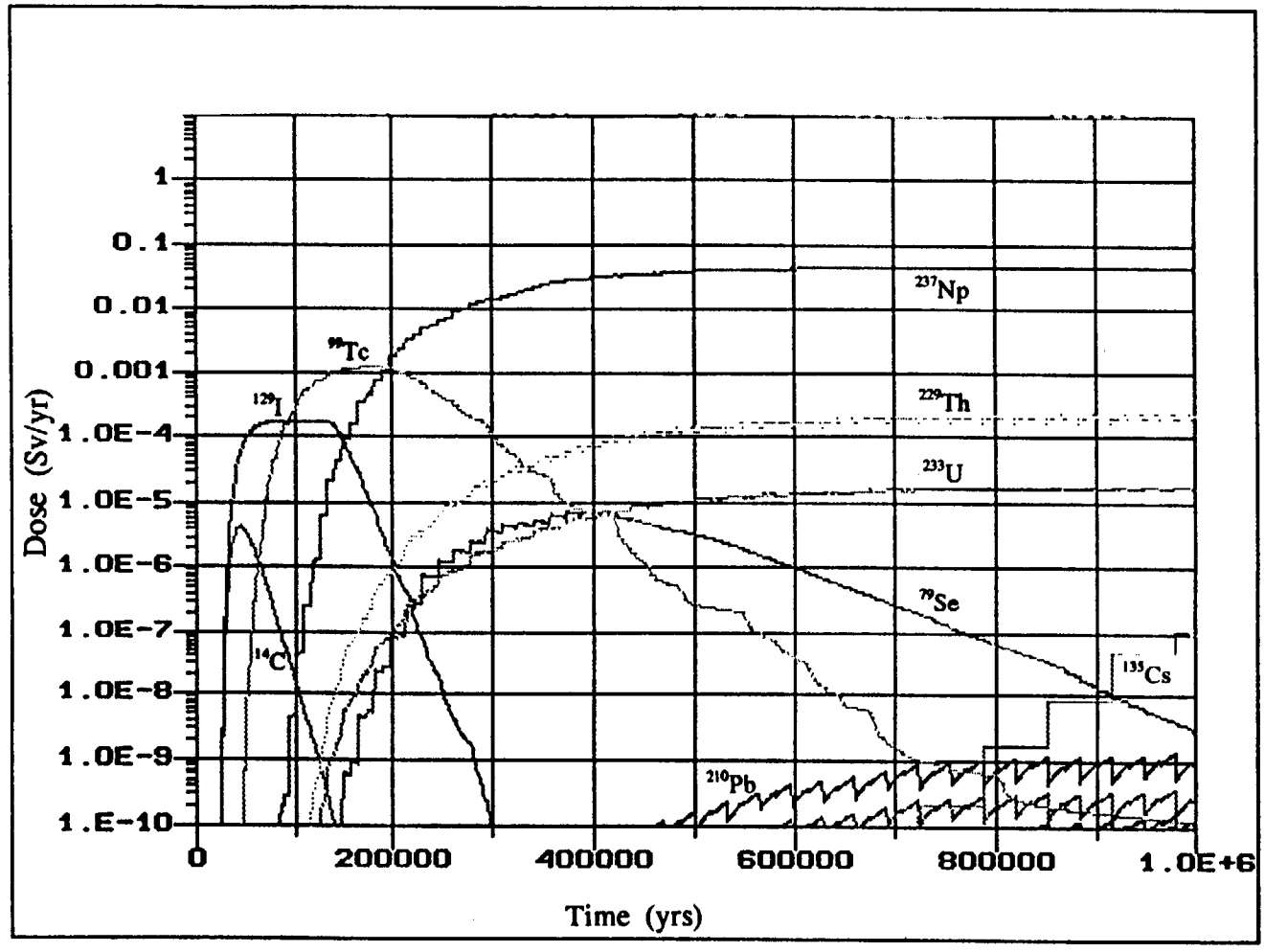
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	1.0	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-19. Scatter Plot of Peak Dose to an Individual Over 1,000,000 Years as a Function of Unsaturated-Zone Velocity with ^{237}Np Solubility fixed at 1.0 g/m^3.



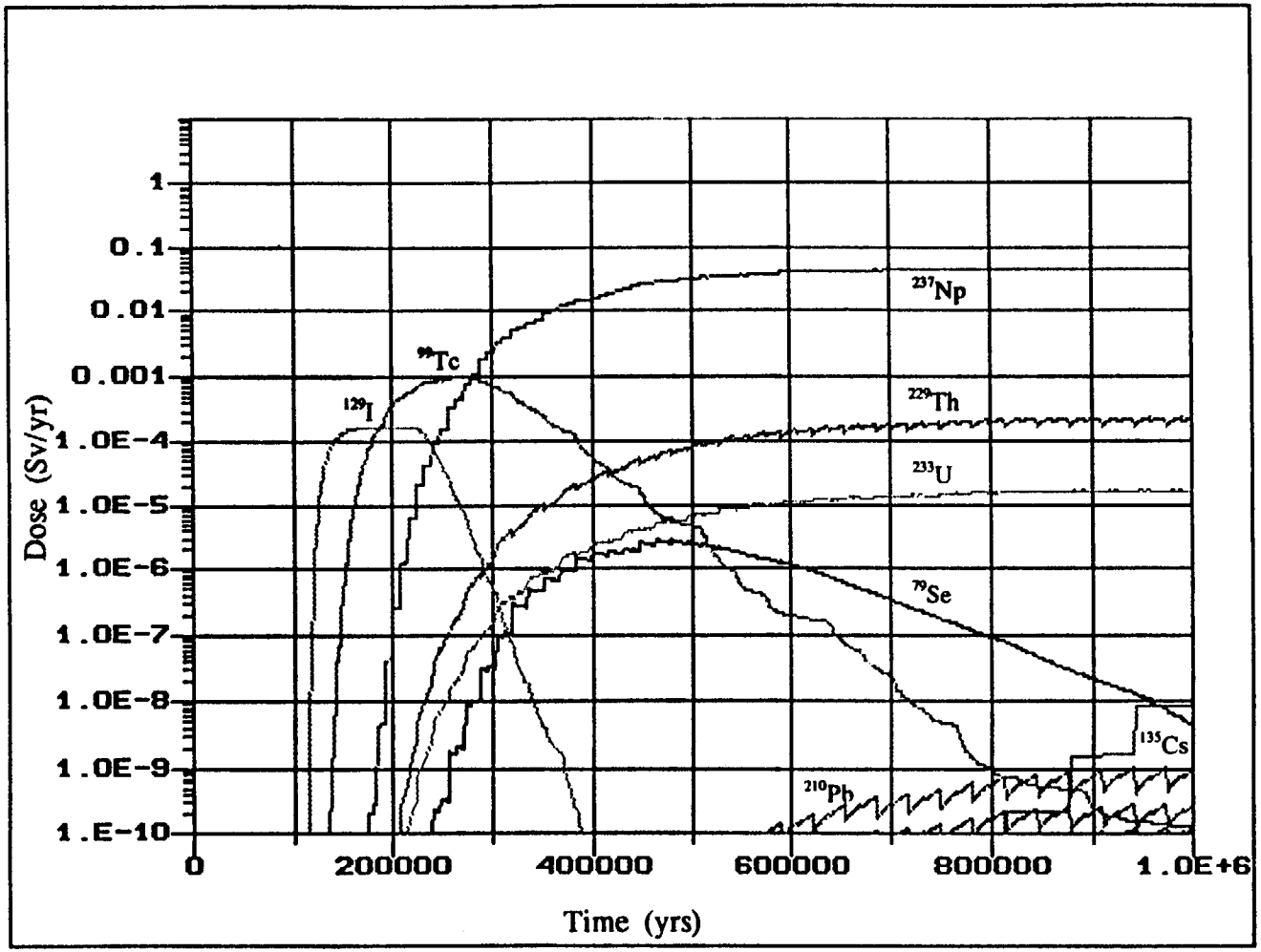
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Rate (yrs)	Fail Time (yrs)
1.0×10^{-3}	1.0	0.0	1.0×10^{-2}	$1\%/10^3$ yrs	0.0

Figure 3-20. Dose to an Individual at the Accessible Environment with a Linear Waste Package Failure Rate (1.0 % per 1,000 years) Between Zero and 100,000 Years



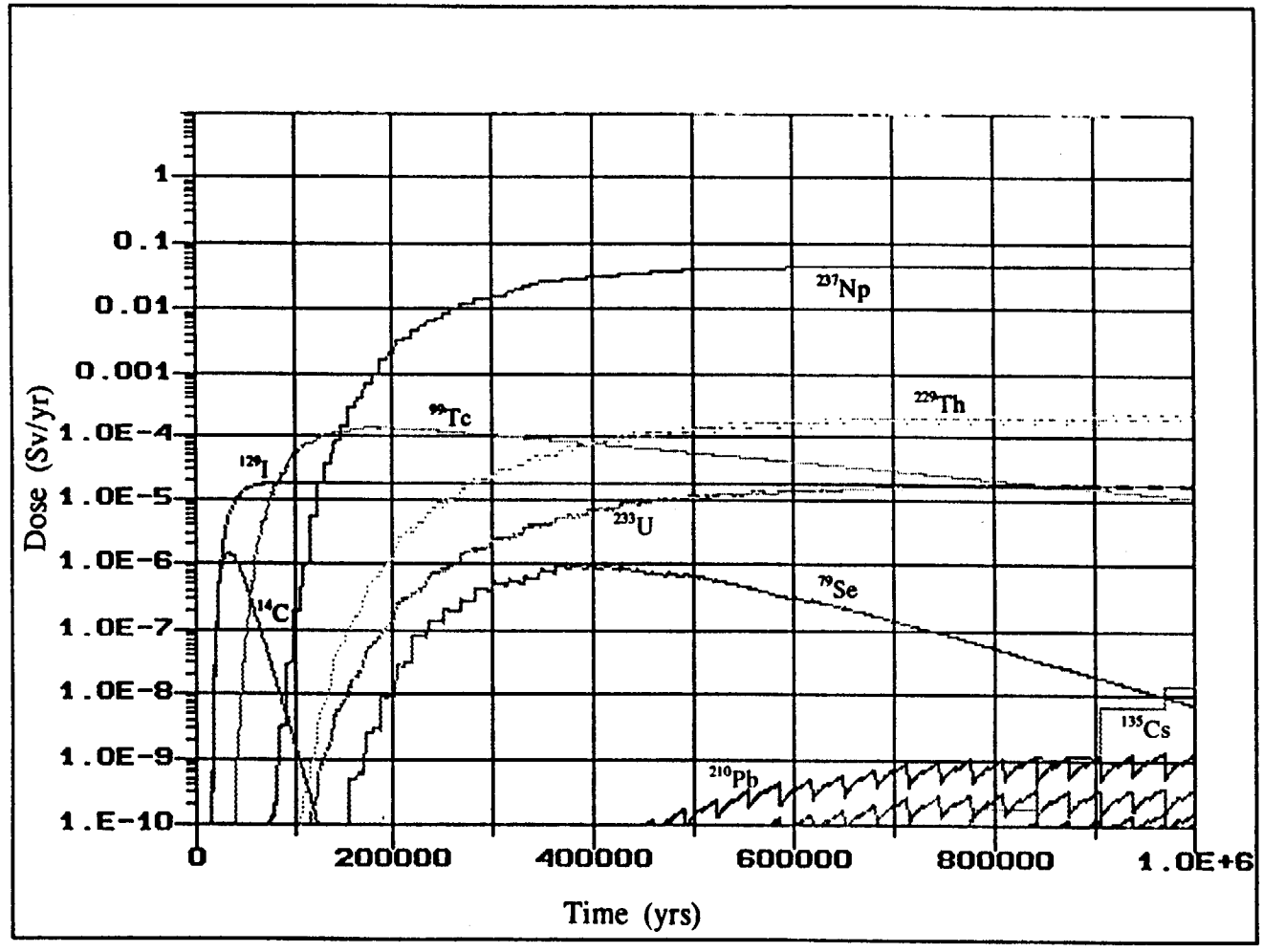
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Rate (yrs)	Fail Time (yrs)
1.0×10^{-3}	1.0	0.0	1.0×10^{-2}	$1\%/10^3$ yrs	1.0×10^4

Figure 3-21. Dose to an Individual at the Accessible Environment with a Waste Package Failure Rate (1.0% per 1,000 years) Between 10,000 and 110,000 Years



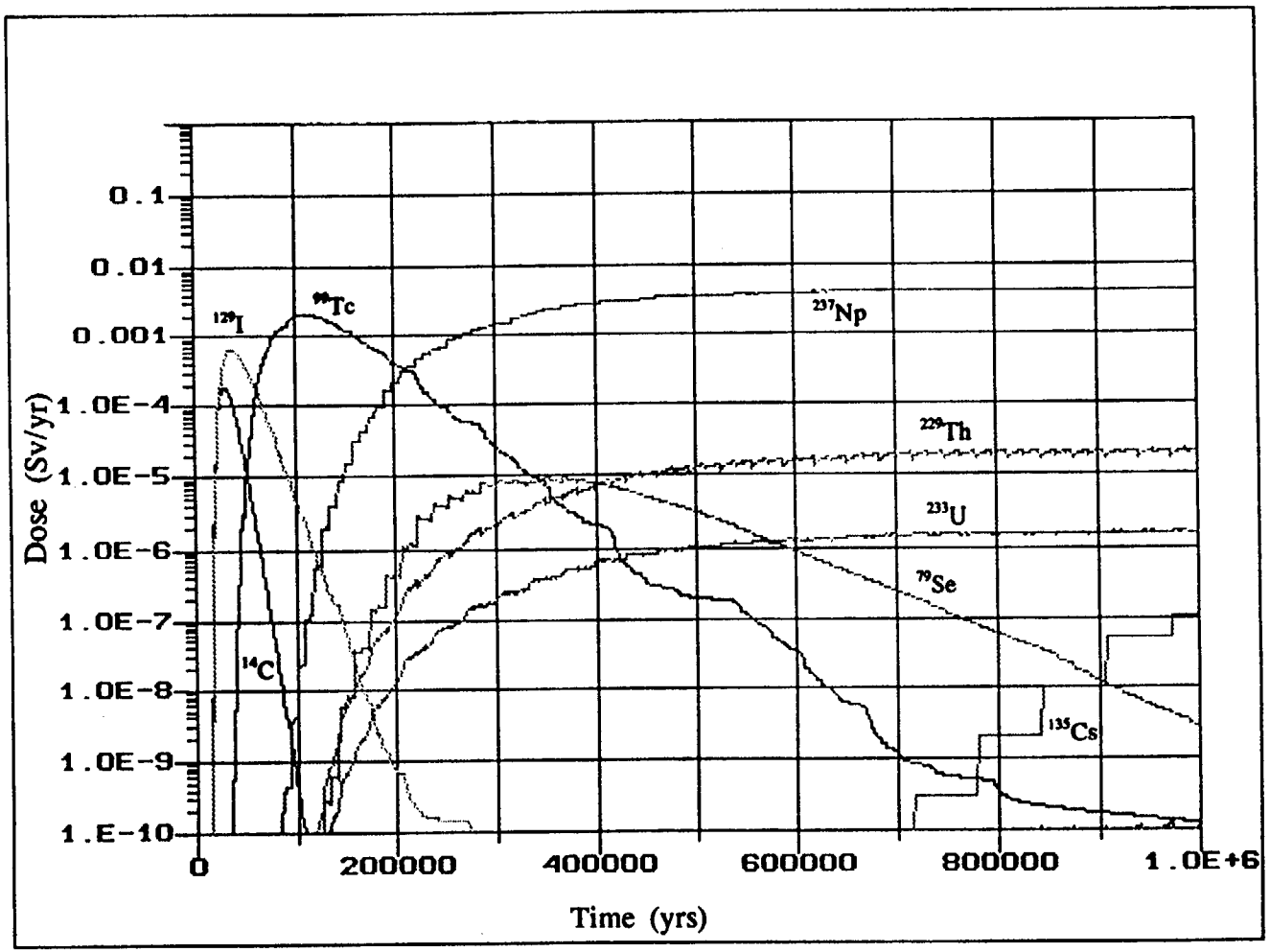
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Rate (yrs)	Fail Time (yrs)
1.0×10^{-3}	1.0	0.0	1.0×10^{-2}	$1\%/10^3$ yrs	1.0×10^5

Figure 3-22. Dose to an Individual at the Accessible Environment with a Waste Package Failure Rate (1.0 % per 1,000 years) Between 100,000 and 200,000 Years



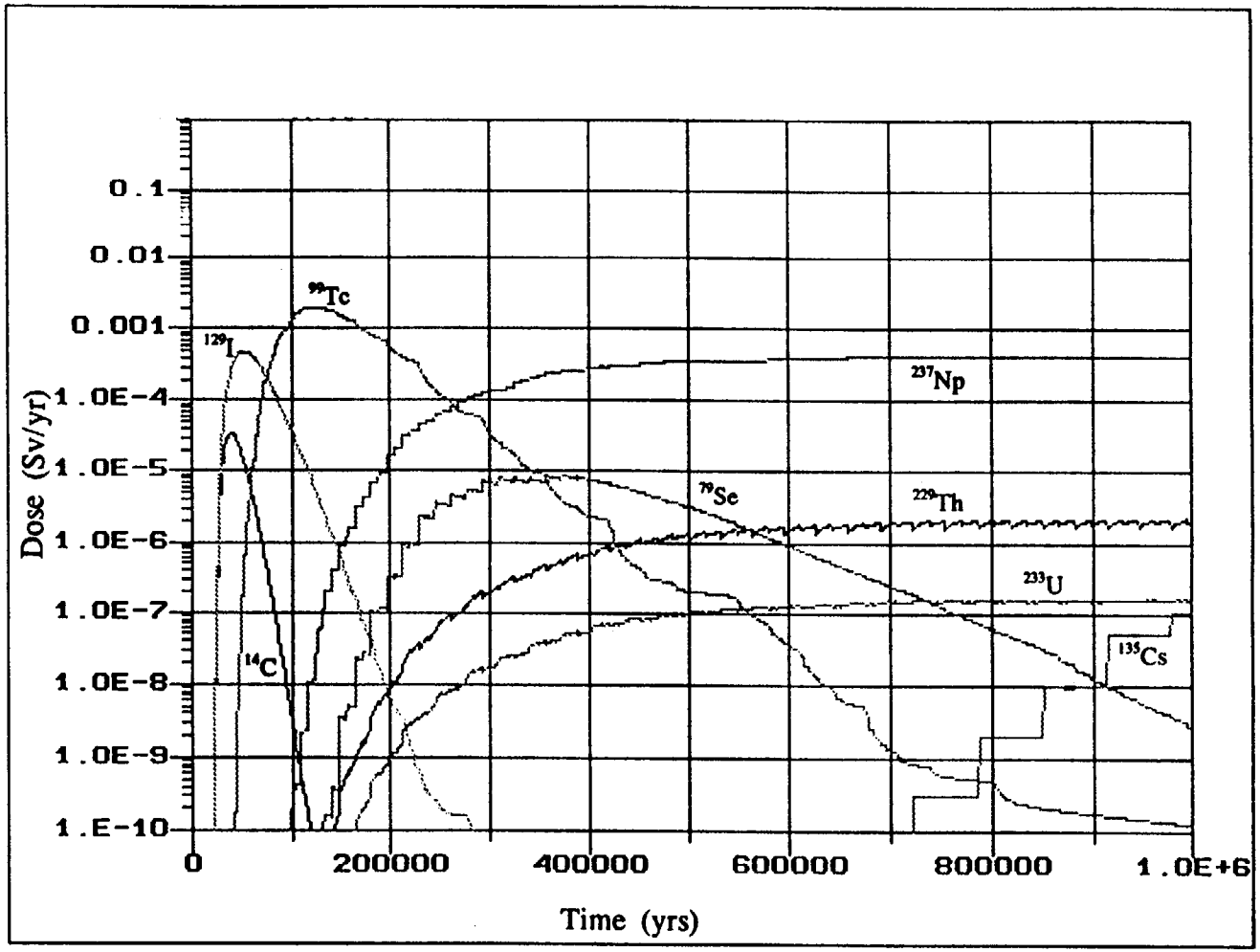
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Rate (yrs)	Fail Time (yrs)
1.0×10^{-3}	1.0	0.0	1.0×10^{-2}	$1\%/10^4$ yrs	0.0

Figure 3-23. Dose to an Individual at the Accessible Environment with a Waste Package Failure Rate (1.0 % per 10,000 years) Between Zero and 1,000,000 Years



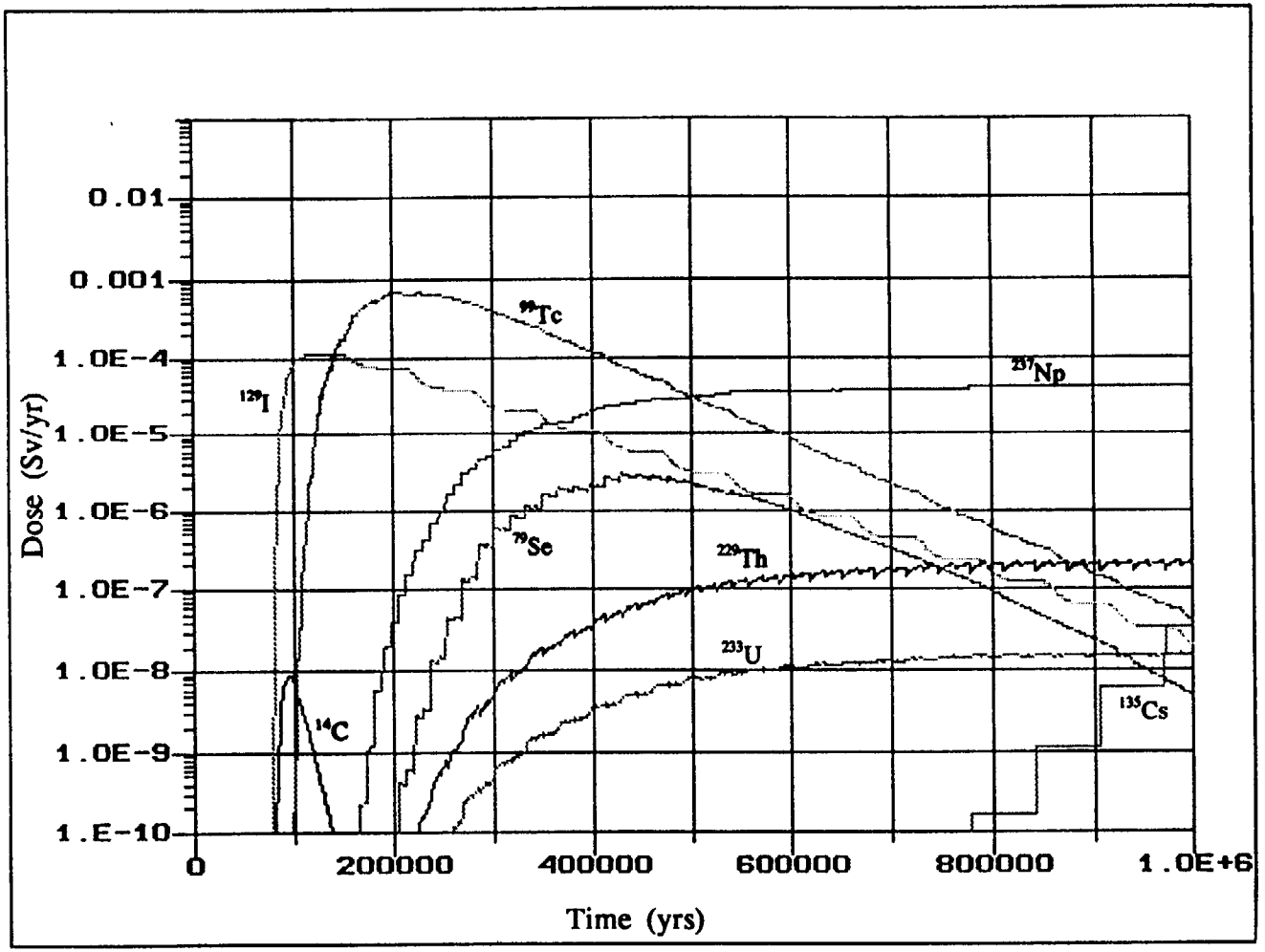
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
1.0×10^{-4}	1.0	0.0	1.0×10^{-2}	0.0

Figure 3-24. Dose to an Individual at the Accessible Environment with a Diffusion Coefficient 1.0×10^{-4} m^2/yr through a 1.0 m Engineered Barrier



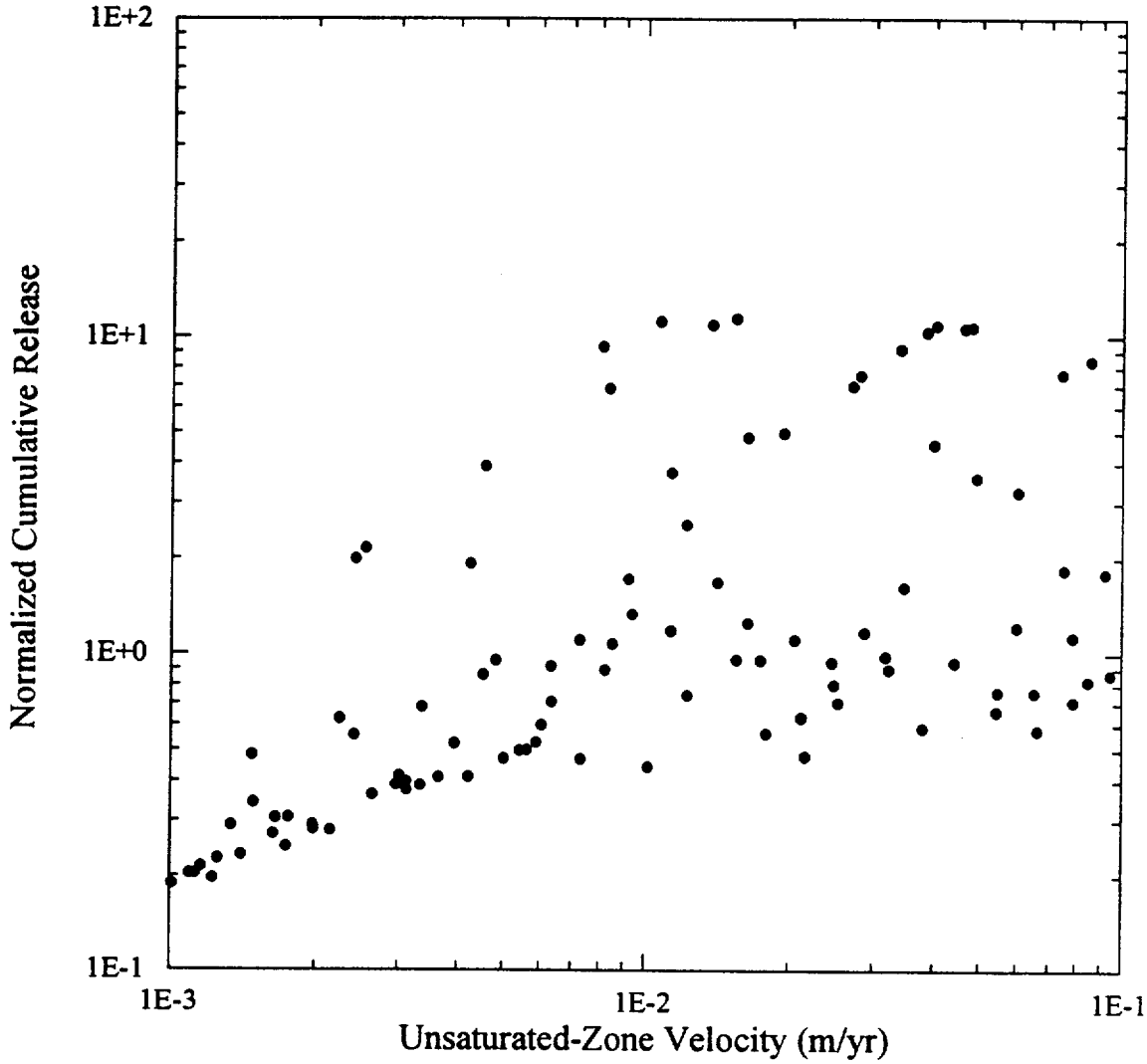
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
1.0×10^{-5}	1.0	0.0	1.0×10^{-2}	0.0

Figure 3-25. Dose to an Individual at the Accessible Environment with a Diffusion Coefficient of 1.0×10^{-5} m^2/yr through a 1.0 m Engineered Barrier



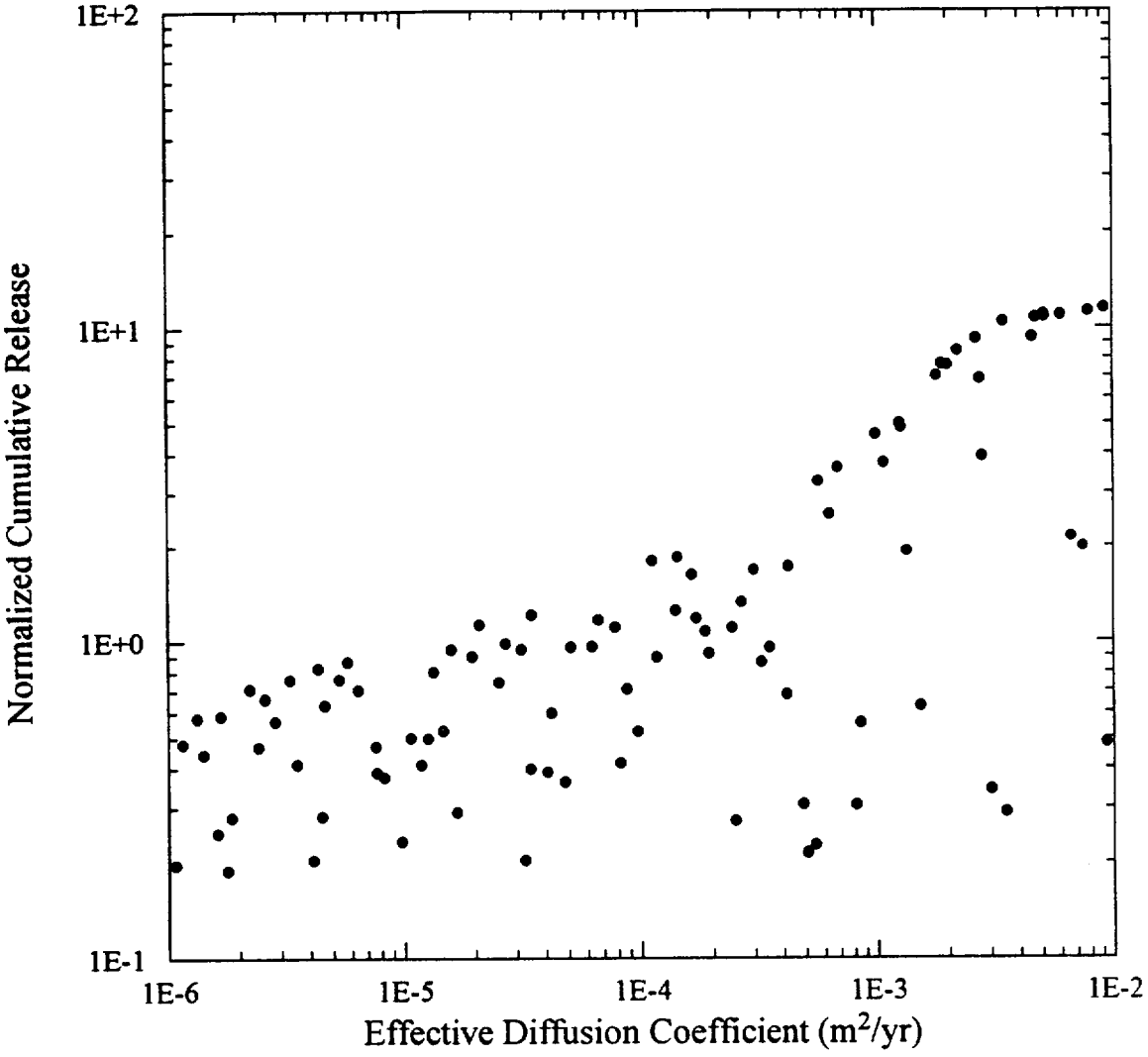
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
1.0×10^{-6}	1.0	0.0	1.0×10^{-2}	0.0

Figure 3-26. Dose to an Individual at the Accessible Environment with a Diffusion Coefficient of 1.0×10^{-6} m^2/yr through a 1.0 m Engineered Barrier



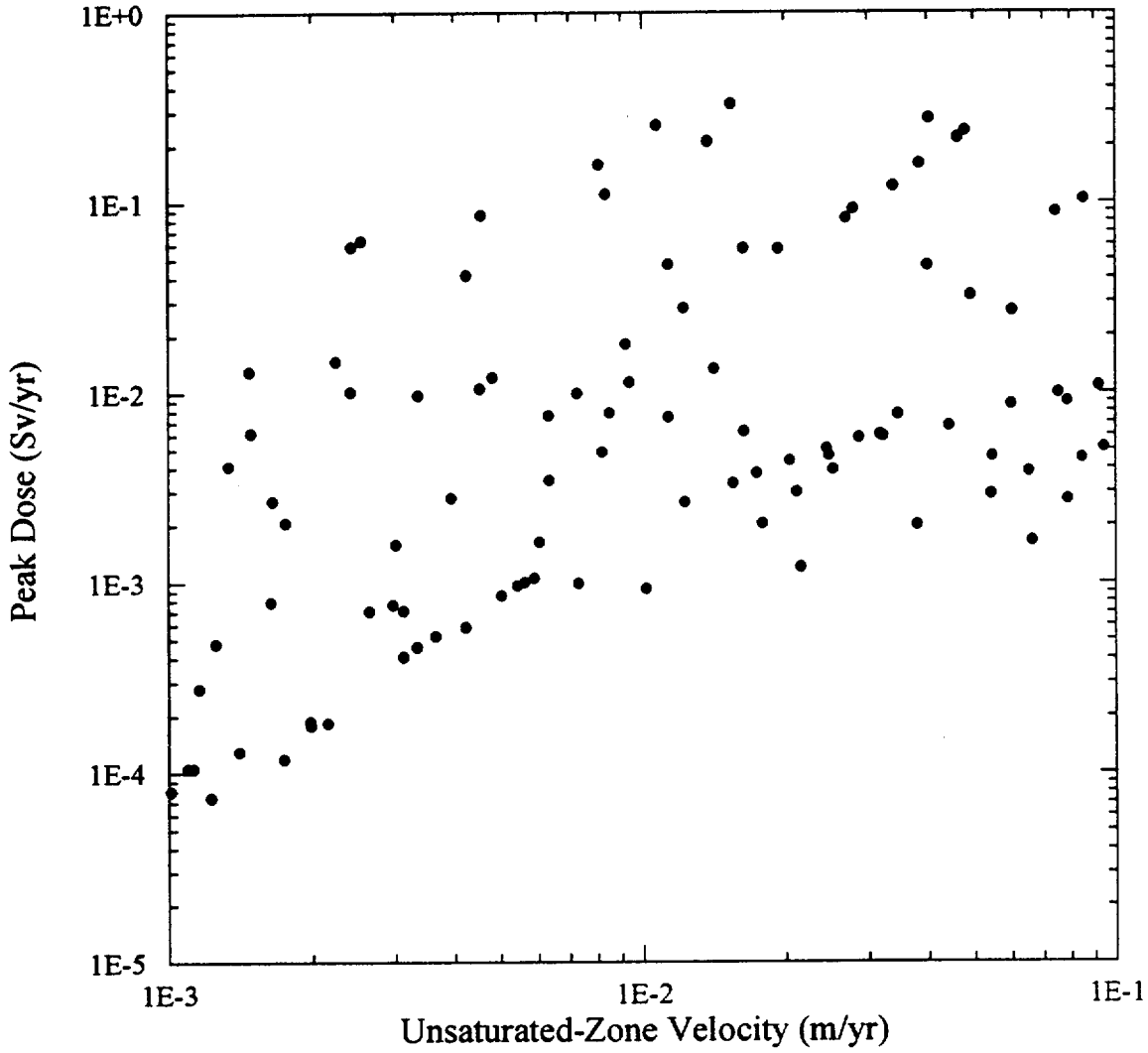
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	1.0	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-27. Scatter Plot of Normalized Cumulative Release to the Accessible Environment Over 1,000,000 Years for a Range of Diffusion Coefficients through the Engineered Barrier as a Function of Unsaturated-Zone Velocity



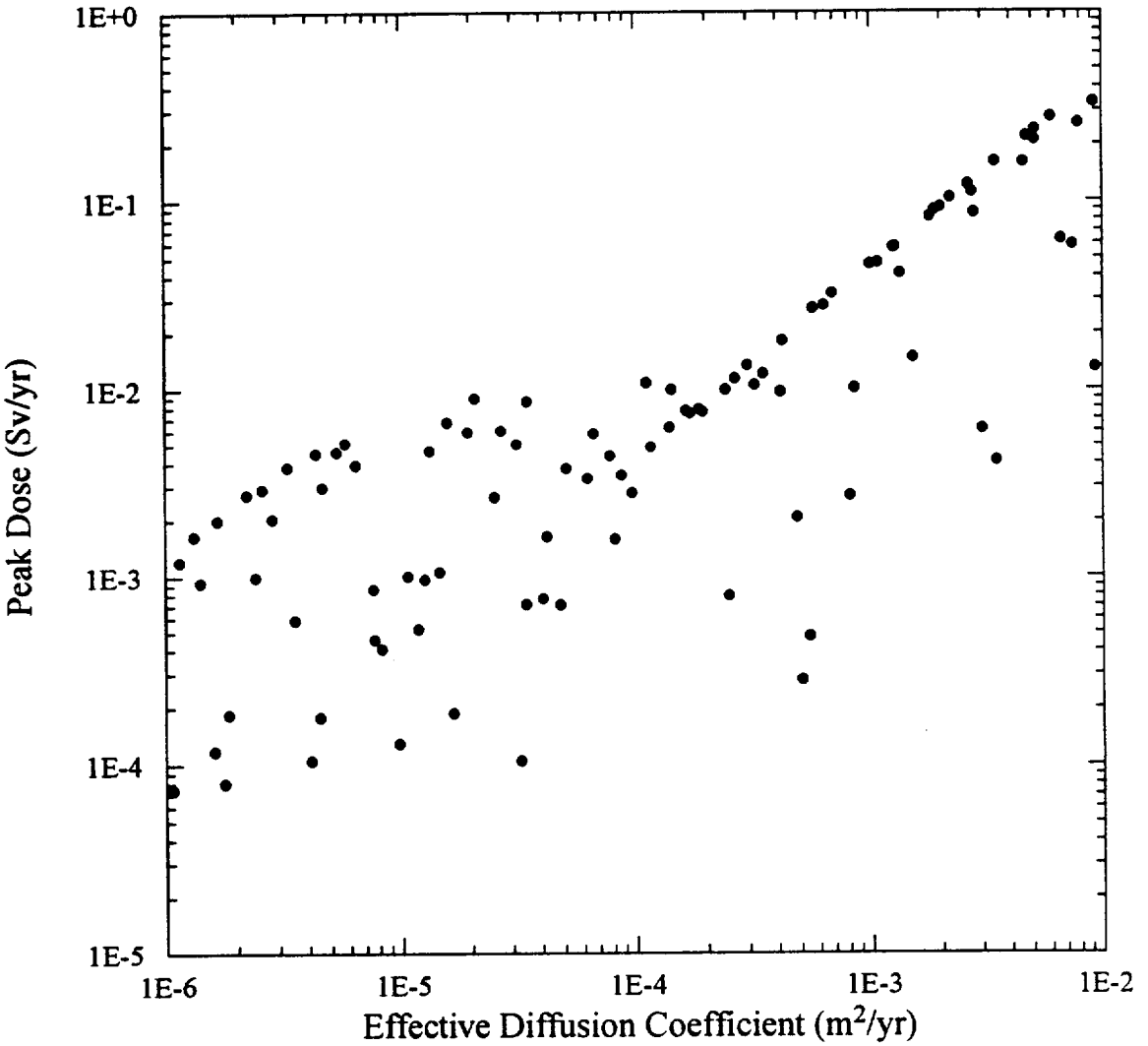
Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	1.0	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-28. Scatter Plot of Normalized Cumulative Release to the Accessible Environment Over 1,000,000 Years for a Range of Unsaturated-Zone Velocity as a Function of Diffusion Coefficient through the Engineered Barrier



Diff. Coef. (m^2/yr)	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	1.0	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-29. Scatter Plot of Peak Dose to an Individual at the Accessible Environment Over 1,000,000 Years as a Function of Unsaturated-Zone Velocity for a Range of Diffusion Coefficients through the Engineered Barrier



Diff. Coef. (m^2/yr	Np Sol. (g/m^3)	Rep. Flux (m/yr)	UZ Vel. (m/yr)	Fail Time (yrs)
10^{-6} to 10^{-2}	1.0	0.0	10^{-3} to 10^{-1}	0.0

Figure 3-30. Scatter Plot of Peak Dose to an Individual at the Accessible Environment Over 1,000,000 Years as a Function of Diffusion Coefficient through the Engineered Barrier for a Range of Unsaturated-Zone Velocities

4. ANALYSES USING THE NEFTRAN-S MODEL

4.1 GENERAL DESCRIPTION OF NEFTRAN-S

The model NEFTRAN-S (Campbell et al., 1991) incorporates the use of a statistical driver (using either Latin-Hypercube or Monte-Carlo sampling) to conduct sensitivity/uncertainty analyses. The NEFTRAN structure assumes that the physical system may be characterized by a network of one-dimensional "legs" (segments). With appropriate boundary conditions prescribed both at the external boundaries and at the leg interfaces, the code simulates both flow and transport processes. Assuming steady state and a single-porosity medium throughout, the code solves the appropriate Darcy conservation equation or the fluid velocity may be specified in each leg. The transport analysis uses a pseudo-steady state dual-porosity formulation. Under transient conditions the code solves the appropriate advection-dispersion equations for either single- or dual-porosity media. These solutions incorporate radionuclide decay chains of arbitrary length.

Because the flow and transport models are embedded within a statistical driver, either deterministic or probabilistic calculations can be performed. In either case, the user must choose the dependent variables to be calculated. Up to 15 different dependent variables may be selected, each of which represents one of the seven types presented in Table 4-1. The first four dependent variables are appropriate for probabilistic calculations, and the last three are appropriate for deterministic calculations (Table 4-1). For deterministic calculations, time is always the independent variable. For probabilistic calculations, time is either fixed or at specified values, and the uncertain parameters become the independent variables. The analysis overrides the fixed parameter values of a deterministic analysis with values sampled from distribution functions. The model places these overrides in one of four categories which are the leg properties, the rock properties, the element properties, and the source term. Leg properties include dispersivity and length; rock properties include hydraulic conductivity, type of medium (porosity), and mass transfer rate; and element properties include radionuclide solubilities and retardation factors. The source term is characterized by seven source variables that are described in Table 4-2.

When the user defines the dependent variables, "output blocks" are also defined. For deterministic calculations, the output blocks contain values of the independent variable, time, and values of the selected dependent variables. Output blocks for probabilistic calculations contain values of variable parameters which have been sampled from the input distribution and calculated results based on those input parameters. The output blocks may be rewritten to allow post-processing of the results. The results can be plotted as x-y plots for deterministic analyses or as CDFs, CCDFs, histograms, or scatter plots for probabilistic analyses. Typically x-y plots would present concentration, dose, or release as a function of time, and CDFs and CCDFs are plotted in terms of dose or release.

4.2 COMPARISON WITH THE BASELINE CASE

The baseline problem is defined in the schematic diagram of a geologic repository at Yucca Mountain (Figure 2-1). Beginning at the repository, the flow path to the accessible environment is 200 m through the unsaturated zone and 5,000 m in the saturated zone. The percolation flux is assumed to be 1 mm/yr and the pore velocity in the saturated zone is assumed to be 1.0 m/yr. The effective porosity was assumed to be 0.1. This yields a ground-water travel time of 25,000

years (20,000 yrs in the unsaturated zone and 5,000 yrs in the saturated zone). The modeler has a choice of running this problem as a single leg problem with an average velocity and uniform dispersivity (240 m) or as a two leg problem with paired velocity and dispersivity of 0.0001 m/yr and 15 m and 1.0 m/yr and 50 m for the unsaturated and saturated legs, respectively. The model UCBNE-41 can only be used to simulate the baseline case as a single leg problem, but with RIP and NEFTRAN-S the user has a choice of assuming one or multiple legs.

Prior to simulation of the baseline case the inventory of radionuclides shown in Table 2-1 was simplified. This was done because for many of the radionuclides, simulations are unnecessary because they never appear in the release and subsequent dose calculations. Also, including all of the inventory introduces a wide variation in the numerical criteria for time and space increments in the numerical solution algorithm in NEFTRAN. As a result, the inclusion of short-lived parents along with longer half-life daughters can seriously degrade the results of the transport calculation. It is beneficial to identify these radionuclides and to account for their presence through increased daughter activity. The criterion used to identify the radionuclides in question was the retarded transport distance during one half life. If, for a given radionuclide, the transport distance was only a few meters or less, then the parent activity was adjusted and added to that of the daughter. Here it is assumed that N parent atoms are instantly transformed into N daughter radionuclei. Since the activity A is approximately equal to the product of the decay constant λ and N, the additional daughter activity may be written as follows:

$$\Delta A_d = \lambda_d \, A_p/\lambda_p$$

Where the subscripts d and p refer to the daughter and parent respectively.

After simplification, the inventory chosen for analysis consists of seven fission products (^{14}C, ^{135}Cs, ^{129}I, ^{94}Nb, ^{79}Se, ^{126}Sn, and ^{99}Tc) and fourteen actinides (^{242}Pu, ^{238}U, ^{237}Np, ^{236}U, ^{235}U, ^{234}U, ^{233}U, ^{232}Th, ^{231}Pa, ^{230}Th, ^{229}Th, ^{227}Ac, ^{226}Ra, and ^{210}Pb). The fourteen actinides are included in the following four chains:

Thorium Chain

$$^{236}U \rightarrow {}^{232}Th$$

Neptunium Chain

$$^{237}Np \rightarrow {}^{233}U \rightarrow {}^{229}Th$$

Uranium Chain

$$^{242}Pu \rightarrow {}^{238}U \rightarrow {}^{234}U \rightarrow {}^{230}Th \rightarrow {}^{226}Ra \rightarrow {}^{210}Pb$$

Actinium Chain

$$^{235}U \rightarrow {}^{231}Pa \rightarrow {}^{227}Ac$$

The simplified inventory for the seven fission products and the fourteen actinides is presented in Table 4-3.

The baseline case was simulated using NEFTRAN-S with the same parameters (including dose conversion factors) that were used for simulations using UCBNE-41 and RIP. As was the case for these other analyses, ^{14}C was restricted to the aqueous pathway, thereby exaggerating the ^{14}C doses. The baseline case was conducted as a single leg problem, a dispersivity of 240 m, and the parameters shown in Tables 2-2 through 2-4. This approach produced the dose results that are presented in Figure 4-1 for the major dose radionuclides and in Figures 4-2 and Figure 4-3 for the actinide chains. Figures 4-2 and 4-3 show the relatively low doses for the actinides in comparison with ^{237}Np. These results are directly comparable to those using UCBNE-41 (Figure 2-2) and are equivalent to those using RIP (Figure 3-1) which were conducted assuming a two leg problem. The difference between assuming a one leg or a two leg problem are discussed below in Section 4.3.

4.3 EFFECT OF DISPERSIVITY ON DOSE

The effect of different values of dispersivity is significant on the radionuclide first arrival time and on the peak height of the dose curve. The dose to an individual at the accessible environment for a range of dispersivities (alpha) for ^{99}Tc is presented in Figure 4-4 where the transport path is assumed as an equivalent single leg. Both the first arrival and the peak height of the dose curve change by about an order of magnitude for a range of dispersivities of nearly two orders of magnitude (alpha = 15 to 1,000 m). Figure 4-4 also shows the difference in solving the single leg transport problem using the analytical or numerical options that are included in NEFTRAN-S.

As discussed above, NEFTRAN-S has the capability to approximate multi-dimensional problems with a series of one-dimensional legs. The conceptual Yucca Mountain problem (shown in Figure 2-1) can be approached as a two leg problem, one leg for the 200 m unsaturated zone and a second leg for the 5,000 m saturated zone. The effects of assuming two legs as compared to assuming one equivalent leg on dose to an individual at the accessible environment for ^{99}Tc and ^{129}I are presented in Figure 4-5. These results show that the peak doses are identical and the arrival time for the single leg problem is slightly ahead of that of the double leg problem. From these results the conclusion can be drawn that for the porous media assumption of flow the results are equally good from the single and double leg assumptions.

The effect of varying dispersivity in the unsaturated zone on the dose to an individual at the accessible environment from ^{99}Tc for the two leg problem is presented in Figure 4-6. Here the dispersivity (alpha) is varied over a factor of about four with an equivalent change in peak dose and arrival time. A change in dispersivity in the unsaturated zone has a large effect because a significant portion of the ground-water travel time is in the unsaturated zone (20,000 yrs of the 25,000 yr ground-water travel time). Figure 4-7 presents the effects of varying the dispersivity in the saturated zone on ^{99}Tc dose to an individual at the accessible environment. As would be expected, a change in dispersivity of more than an order of magnitude has little effect on either

the peak dose or the first arrival. Based on the results shown in Figures 4-6 and 4-7 the difference between the dose from a one leg and a two leg problem in Figure 4-5 can be attributed to dispersion in the unsaturated zone (i.e., a dispersivity of 15 m for the two leg problem as compared to 240 m for a one leg problem).

4.4 EFFECT OF WASTE PACKAGE LIFETIME ON RELEASE AND DOSE

The effect of waste package lifetime can be simulated by using a source-release time in NEFTRAN-S, the time at which all packages fail. The discharge to the accessible environment (for each assumed release time) can be plotted as a weighted (i.e., normalized to the Table 1 values of 40 CFR 191) discharge (release) or a peak dose over the time of the simulation. If the weighted discharge is equal to 1.0 for a 10,000 year simulation the discharge to the accessible environment integrated over 10,000 years (cumulative release over 10,000 years) is equal to the EPA Standard (Table 1, Appendix A, Subpart B of 40 CFR 191, EPA, 1993b). For these calculations, a dispersivity of 0.2 m was used for advection in the waste package, 15 m was used in the unsaturated zone, and 50 m was used in the saturated zone. The remaining parameters remain the same as the baseline case. Here it should be noted that NEFTRAN-S only has the capability of assuming advective release from the waste package (i.e., the case of diffusion-controlled release from the waste package could not be investigated as was done using RIP in Chapter 3). However, as will be seen in Section 4.6, diffusion through an engineered barrier can be approximated through manipulation of the advection-dispersion parameters.

The weighted discharge over a simulation period of 10,000 years as a function of the source-release time is presented in Figure 4-8. Only ^{14}C and ^{129}I are shown in Figure 4-8 because significant quantities of ^{99}Tc have not yet arrived at the accessible environment and all other radionuclides are retarded more than ^{99}Tc. The solid dots on Figure 4-8 represent the cumulative release normalized to the Table 1 values of 40 CFR 191 (^{14}C and ^{129}I in this case). Figure 4-8 illustrates that the waste package lifetime must be on the order of 1,000 years to have any appreciable effect on the 10,000 year integrated normalized release. When the time of the simulation (and integration) is increased by an order of magnitude to 100,000 years, ^{99}Tc appears along with ^{14}C and ^{129}I (Figure 4-9). Figure 4-9 illustrates that the waste package lifetime must be on the order of 80,000 years to have any appreciable effect on the 100,000 year integrated normalized release. It bears noting that in the 100,000 year simulation, the normalized releases were normalized to the Table 1 values in 40 CFR 191, and these Table 1 values were not multiplied by a factor of 10 which would seem appropriate. Had a factor of 10 been applied to the EPA Table 1 release limits on Figure 4-9 (the normalized releases) would have been 10^{-1}. The radionuclides ^{237}Np, ^{227}Ac, ^{135}Cs, and ^{79}Se all arrive at the accessible environment beyond 100,000 years.

The peak dose over a 10,000 year period is shown as a function of source-term release time on Figure 4-10. This figure shows the dose to an individual at the accessible environment for ^{14}C and ^{129}I and the total dose (indicated by the solid dots). The maximum total peak dose is 3×10^{-7} Sv/yr which indicates that there is little release over the 10,000 years because of the 25,000 year ground-water travel time (the release which does occur is caused by dispersion).

When the simulation period is extended by an order of magnitude to 100,000 years, the total peak doses increase significantly to about 4 x 10^{-3} Sv/yr and ^{99}Tc now appears as the largest component of the total dose (Figure 4-11). As discussed earlier, the radionuclides ^{237}Np, ^{227}Ac, ^{135}Cs, and ^{79}Se have not reached the accessible environment because of retardation. In examining Figure 4-11 (and subsequent figures of peak dose) it should be remembered that the plot represents the peak dose associated with each radionuclide and is plotted independently of the time the peak occurs. However, the summed dose of all radionuclides is first calculated as a function of time, and then the peak of the sum is plotted. Therefore, one shouldn't sum the values on Figure 4-11, because they may occur at different times.

4.5 EFFECT OF UNSATURATED ZONE VELOCITY ON RELEASE AND DOSE

The effect of the unsaturated zone pore velocity on the cumulative release of radionuclides to the accessible environment was determined for a pore velocity range of 10^{-3} to 10^{-1} m/yr which is equivalent to 10^{-4} to 10^{-2} m/yr (0.1 to 10 mm/yr) percolation flux. This range in pore velocities corresponds to a water travel time range of 200,000 to 2,000 years through the 200 m thick unsaturated zone. For these calculations the waste package lifetime (the source-release time) was assumed to be zero. The weighted discharge (weighted cumulative release) for 10,000 years, plotted as a function of unsaturated zone pore velocity is presented in Figure 4-12. This figure indicates that the EPA cumulative release limits are only exceeded as pore velocities approach 10^{-1} m/yr (travel times through the unsaturated zone approach 2,000 yrs.). Figure 4-12 also shows that only ^{14}C and ^{129}I reach the accessible environment in significant quantities even at the higher pore velocities. The ^{14}C dose in these analyses is unrealistically high because it was included in the aqueous phase when in reality much of it would be released as a gas. When the period is increased to 100,000 years, ^{99}Tc is added to the radionuclides released (Figure 4-13). This figure indicates that the EPA release limit is exceeded at pore velocities above 10^{-2} (travel times below 2,000 yrs). As discussed above for Figure 4-13, the EPA integrated release limit for 10,000 years was not increased by a factor of ten. Had it been, then the release limit would not have been exceeded even at a travel time of 2,000 years through the unsaturated zone.

The effect of unsaturated zone pore velocity on the peak dose to an individual at the accessible environment for 10,000 years is significant in that doses of about 10^{-2} Sv/yr are calculated for unsaturated-zone velocities on the order of 3 x 10^{-2} m/yr (Figure 4-14), although the peak dose decreases to about 2 x 10^{-7} Sv/yr at unsaturated zone pore velocities of 10^{-2} m/yr. When the simulation time is increased to 100,000 years, the peak dose to an individual also increases (Figure 4-15). This increase is due almost entirely to the arrival of ^{99}Tc over the longer period (Figure 4-15).

4.6 EFFECT OF DIFFUSIVE RELEASE FROM THE ENGINEERED BARRIER SYSTEM ON RELEASE AND DOSE

For analyses of the potential effects of a diffusive near field barrier, the waste packages are assumed to be placed horizontally in the drift and are surrounded by an engineered capillary barrier, consisting of coarse-grained material overlain by fine-grained material. The porosity contrast between the two materials (as well as between the engineered material and the rock matrix) causes the advective flow to go around the waste package in the fine-grained material or the still finer grained welded tuff rock matrix. The advective flow in the coarse grained

material is negligibly small and diffusion becomes the only mechanism for transporting radionuclides from the waste package through the coarse grained barrier to the advective flow in the fine-grained material and the host rock. The length of the diffusion path (thickness of the coarse grained material) is assumed to be one meter.

Implementation of the diffusion barrier in NEFTRAN-S presents a problem because the code does not offer a diffusive transport option. This means that the effects of diffusion must be approximated by the advection and dispersion options. The diffusion model assumes two constant-concentration boundary conditions, one at each end of the diffusion path. The relatively small amount of water available inside the coarse grained material suggests that solubility limited concentrations (C_s) will govern releases for most radionuclides. At the point of contact with the fine-grained material, the contact between the capillary barrier and flowing ground water, the diffusion model assumes that dilution would reduce concentrations to relatively small levels.

Subject to these boundary conditions and an initial condition $C(0<x<L, t=0) = 0$, the steady-state flux across the barrier becomes $F = DC_s/L$. Defining the diffusive velocity $v = D/L$, the diffusive flux is approximated by the advective flux $F = vC_s$. It should be noted here that $D = \beta D_0$ denotes the net coefficient of diffusion. As such, it contains both the free-water diffusion D_0 and the tortuosity β. Since capillary barriers contain near-residual levels of liquid saturation, tortuosity values are expected to be quite small relative to the values $\beta \sim 10^{-1}$ which would apply to the same medium if it were completely saturated. This report assumes that, because of its dependance on liquid saturation, the net coefficient of diffusion D is an uncertain parameter.

The transient effects of linear diffusion between two constant-concentration boundary conditions are characterized by an infinite sum $1 \leq n \leq \infty$ over terms of the form $A_n \exp(-t/\tau_n)\sin(n\pi x/L)$ (Carslaw and Jaeger, 1959, p. 99ff). Each term of the series thus subdivides into three additional terms. The first (A_n) is a constant determined by boundary and initial conditions, and the third is a variable depending only on spatial location x. The approximate treatment, however, focuses on the second term. This term depends only on time t, and contains a time constant $\tau_n = L^2/(n^2\pi^2 D)$. Comparing the values for various values of n, it is observed that $\tau_1 > \tau_2 > \tau_3 \ldots$. This means that τ_1 controls the final approach to equilibrium. Recognizing that, like flux reduction, time delay represents an important effect of the diffusion barrier, the approximate treatment seeks to approximate τ_1 with a combination of advection and dispersion.

During the period $0 \leq t \leq \tau_1$, the advective velocity defined above would advance a concentration front by a distance $x_0 = (D/L)\tau_1$. In order to define the dispersion process, the model used here assumes that, during this same period, dispersion would advance the front to the full length of the capillary barrier L, i.e., $L = x_0 + \gamma\sigma$. Here the dispersion process is characterized by the standard deviation $\sigma = (2\alpha x_0)^{1/2}$, where α is the dispersivity. The value γ depends on the concentration level used to define the time of arrival of the front at the boundary between the two materials at x = L. For simplicity, γ is taken as 1.0, corresponding to a concentration level of approximately 0.16 C_s. These assumptions yield an expression for the dispersivity, $\alpha = (\pi-1)^2 L/2$.

Based on this analysis, the model in NEFTRAN-S approximates diffusion within the engineered capillary-diffusion barrier as a combined advective-dispersive process with interstitial velocity $v = D/L$ and dispersivity $\alpha = (\pi-1)^2 L/2$.

The weighted (normalized to EPA Table 1 release limits) release to the accessible environment as a function of diffusion coefficient through a 1.0 meter capillary-diffusion barrier over a 10,000 year period is presented in Figure 4-16. The effect of the diffusion barrier can be seen by comparing Figure 4-16 with Figure 4-12. This comparison indicates that the barrier provides a several order of magnitude reduction in the releases over advective flux through the waste package. The advective velocity in the unsaturated zone beyond the diffusion barrier was a constant 1.0×10^{-2} m/yr (i.e., the value of the baseline case) for this and the calculations that follow. When the period is extended to 100,000 years the release limit is approached at the higher values of diffusion (Figure 4-17). In Figure 4-17 it should be noted that the EPA release limits were not multiplied by a factor of ten which could be justified by the longer period.

The effect of the coefficient of diffusion through the diffusion barrier on dose to an individual at the accessible environment from ^{129}I is presented in Figure 4-18 which appears only as a time delay in the arrival and peak dose because of the relatively long half-life of ^{129}I. Here it should be noted that because the release of ^{129}I is alteration controlled, too much of the radionuclide may be dissolved in a very small amount of water within the capillary barrier. In order to produce more realistic results, solubilities for the alteration-controlled radionuclides should be used. This approach would reduce the releases and doses that appear in Figure 4-16 through 4-20 for these radionuclides.

Figure 4-19 shows the peak dose to an individual at the accessible environment over a 10,000 year period as a function of the coefficient of diffusion through a capillary barrier. The doses that occur in the first 10,000 years are low even for the higher values of diffusion primarily because the assumed unsaturated-zone velocity of 10^{-2} m/yr implies that only the initial dispersion-controlled arrivals occur over the first 10,000 years. For a longer period (100,000 yrs), the doses increase (Figure 4-20), but are considerably lower than those for advective flow through the waste package (Figure 4-15). Based on these analyses it appears that a capillary barrier could be very effective in reducing the peak doses over long periods of time.

4.7 SUMMARY

The model NEFTRAN-S was used to simulate the baseline case of dose to an individual at the accessible environment as a comparison to the same simulation using UCBNE-41 and RIP. These simulations were conducted using a single transport leg (pathway) with an appropriate dispersivity (240 m). The comparison with the results of UCBNE-41 is good (compare Figure 4-1 with Figure 2-2), and the comparison with RIP is acceptable (compare Figure 4-1 with Figure 3-1). The difference in the shape of the ^{237}Np peak between NEFTRAN-S and UCBNE-41 is caused by the approximate way in which competing isotopes and fractional releases are handled in the UCBNE-41 model. Both RIP and NEFTRAN-S have this capability and the shape of the ^{237}Np peaks from both models are the same.

The effect of simulating the baseline case using one average leg to represent both the unsaturated zone and saturated zone was compared to a two leg solution where both the unsaturated zone and the saturated zone were explicitly represented. The results were nearly identical (Figure 4-5) which indicates that the more complex model (the two-leg representation) can be simplified to a one leg problem with no loss in accuracy. This result gives more credibility to the results using UCBNE-41 which only accommodates a single leg simulation.

The effects of dispersivity on dose to an individual at the accessible environment was investigated in both the unsaturated zone and the saturated zone. This work indicated that the range of results for a range of dispersivities in the saturated zone is small as compared to the range of results for a range of dispersivities in the unsaturated zone. This is caused by the fact that the majority of the transport time is in the unsaturated zone.

The effect of various waste package lifetimes was investigated using NEFTRAN-S and the results for dose at the accessible environment were similar to those produced by both UCBNE-41 and RIP. Package lifetimes of up to 10,000 years have little effect on peak doses at the accessible environment, the peaks only arrive 10,000 years later for the longer lived waste packages. Longer lived packages (100,000 yrs) reduce the dose from ^{14}C because there has been significant time for decay. Doses from longer lived radionuclides are not affected significantly, only delayed in arrival. The effects on cumulative release are similar.

The effect of a distribution of waste package failures was not investigated because NEFTRAN-S only has the capability of failing the waste packages all at the same time. Likewise the effects of diffusion within the waste package was not investigated because NEFTRAN-S only has the capability for advective release from the waste package. However, the diffusion through an engineered barrier was approximated using an appropriate relationship treating the diffusive flux as an advective velocity. The effects of a diffusion barrier, an engineered capillary barrier with a diffusion path length of 1.0 m, are significant to both cumulative release and to dose to an individual at the accessible environment. Cumulative release to the accessible environment is decreased significantly and peak doses for ^{14}C and ^{99}Tc are significantly reduced because of the time delay provided by diffusion through the 1.0 m barrier. At the lower diffusion coefficients (10^{-6} m^2/yr) a one meter barrier can provide about a one million year delay. This delay provides significant decay time for ^{14}C and ^{99}Tc and delays the arrival of ^{237}Np and ^{129}I by the travel time through the diffusion barrier.

Table 4-1. Description of NEFTRAN Output Options

OPTION	DESCRIPTION
Weighted discharge	Radionuclide discharge rates are integrated over the problem time then summed after being divided by weighting factors entered in the radionuclide inventory array.
Peak dose	The peak dose is determined for each trial.
Dose at specified time	Dose is determined at each time specified by the user.
Concentration at specified time	Radionuclide concentrations in ground water at the discharge point are determined at the time specified.
Discharge rate versus time	This option provides radionuclide time dependent discharge rates (breakthrough curves).
Concentration versus time	This option provides radionuclide concentration in ground water at the discharge point as a function of time.
Dose versus time	This option provides dose from ingestion as a function of time.

Table 4-2. Description of Source Term Options for NEFTRAN

OPTION	DESCRIPTION
Inventory access fraction	The fraction of the radionuclide inventory that is to be released in the calculation. The ranges must be specified so that sampled values will be within the limits of zero and one.
Mixing cell	The pore volume at the source area. This entry is used only if the mixing-cell source option is applied.
Source flow rate	The fluid flow rate through the source area. This entry is needed if the flow network is not solved and solubilities are used or the mixing cell model is used.
Discharge flow	The fluid flow rate at the radionuclide discharge point. This quantity is used to calculate radionuclide concentrations and is needed only if the flow network is not solved and concentrations are desired or required for dose calculations.
Leach rate	If the constant-leach-rate option is chosen, this entry is the reciprocal of the leach time. If the exponential leach rate option is chosen, this entry is the reciprocal of the time required to reduce the waste matrix mass to 1/e of the original mass. This is required only if leaching is modeled and ranges must be established so that sampled values will be greater than zero.
Onset of leaching	The time at which waste matrix leaching begins. Leaching may begin earlier but no later than the release time.
Time of release	The time at which radionuclide migration begins. This entry must be coordinated with onset of leaching because nothing can be transported before the onset of leaching.

Table 4-3. Simplified Inventory Used for Analyses Using the Model NEFTRAN-S

RADIONUCLIDE	INVENTORY (Ci)	RADIONUCLIDE	INVENTORY (Ci)
^{14}C	9.32×10^{4}	^{232}Th	2.97×10^{-5}
^{79}Se	3.02×10^{4}	^{237}Np	7.57×10^{4}
^{94}Nb	5.61×10^{4}	^{233}U	4.93×10^{0}
^{99}Tc	9.51×10^{5}	^{229}Th	2.72×10^{-2}
^{126}Sn	5.83×10^{4}	^{242}Pu	1.37×10^{5}
^{129}I	2.34×10^{3}	^{238}U	1.99×10^{4}
^{135}Cs	3.57×10^{4}	^{234}U	1.72×10^{5}
^{235}U	1.88×10^{3}	^{230}Th	2.39×10^{1}
^{231}Pa	2.26×10^{0}	^{226}Ra	1.66×10^{-1}
^{227}Ac	1.24×10^{0}	^{210}Pb	4.51×10^{-2}
^{236}U	2.86×10^{4}		

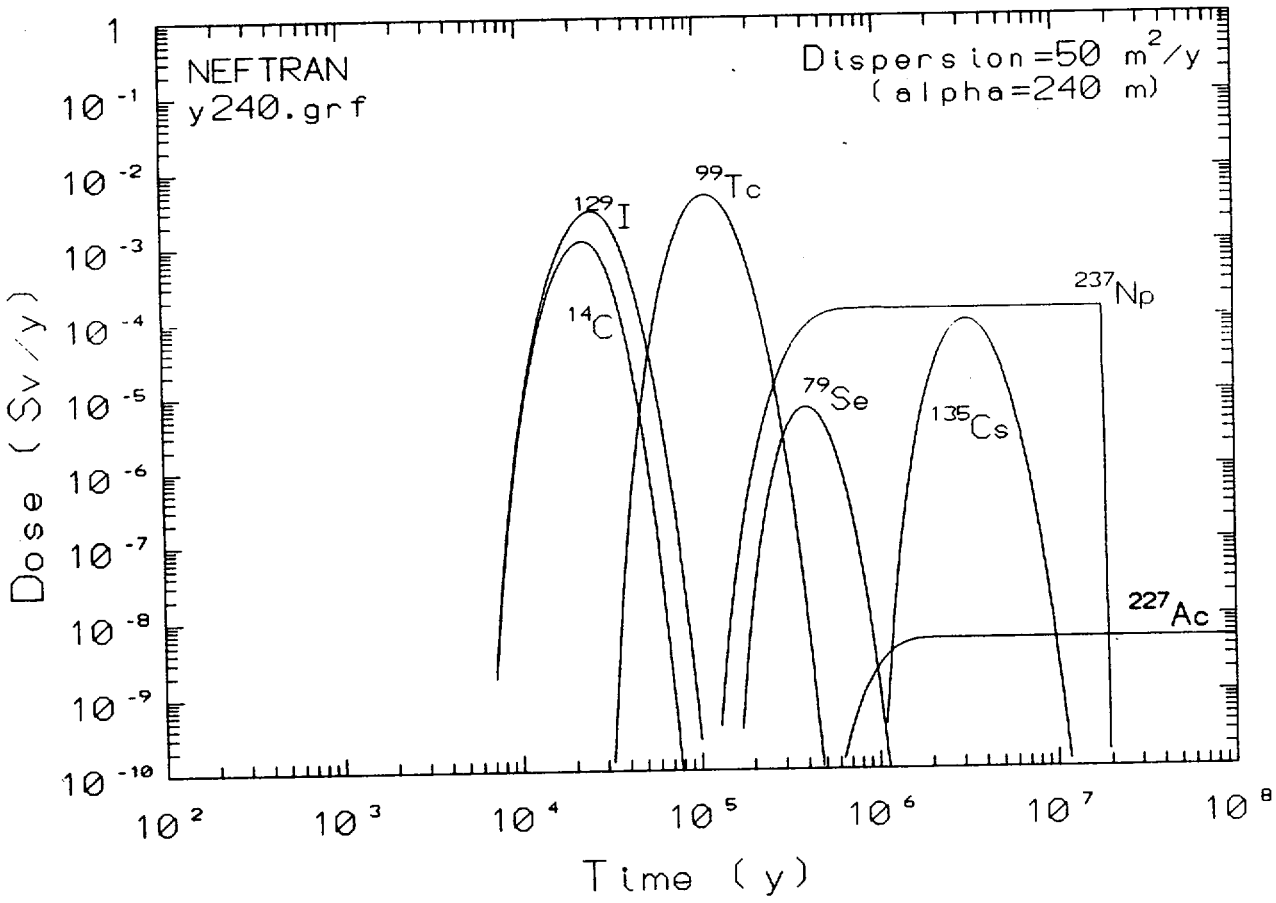


Figure 4-1. Dose to an Individual at the Accessible Environment for the Baseline Case Using NEFTRAN-S

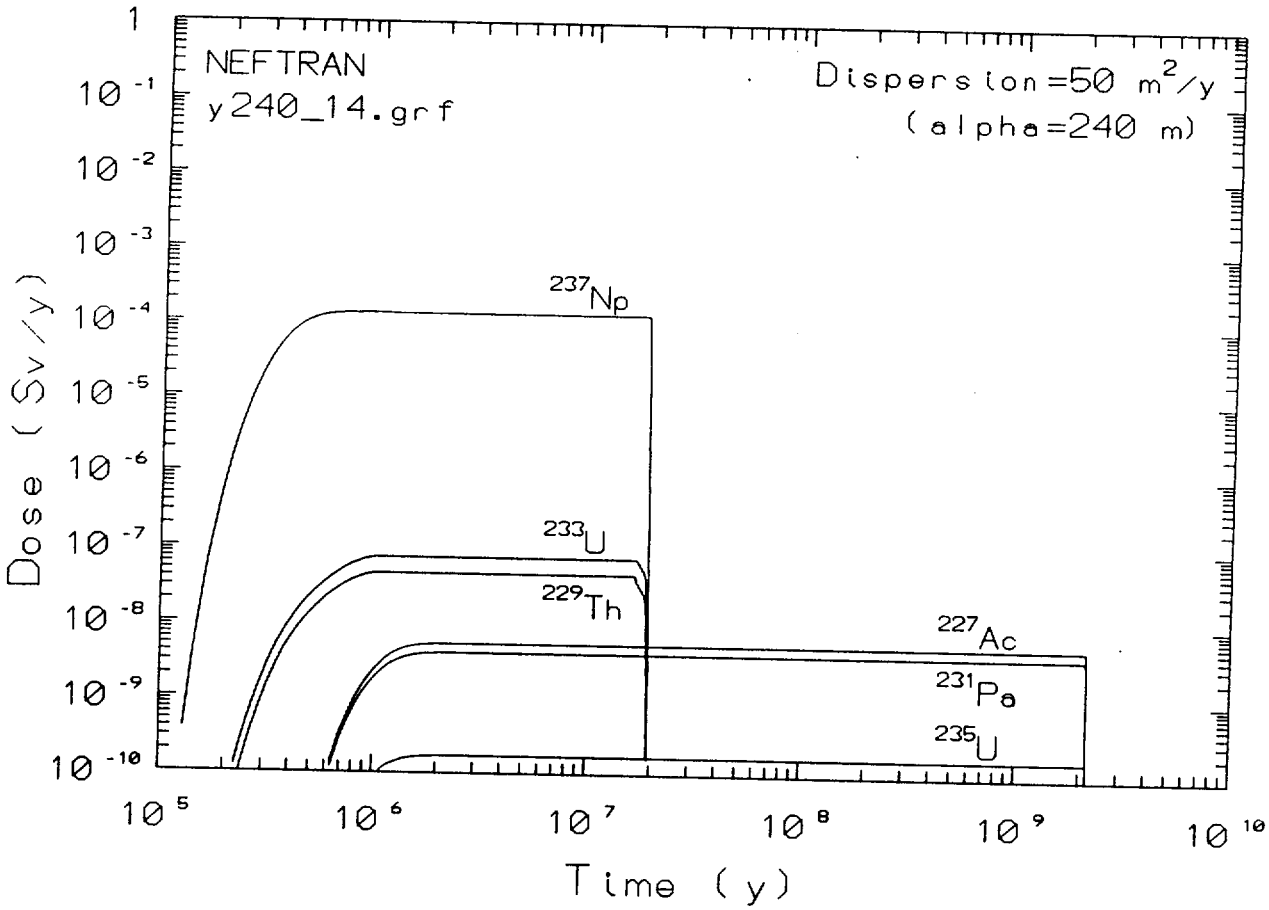


Figure 4-2. Dose to an Individual at the Accessible Environment from the ^{237}Np and ^{235}U Chains for the Baseline Case

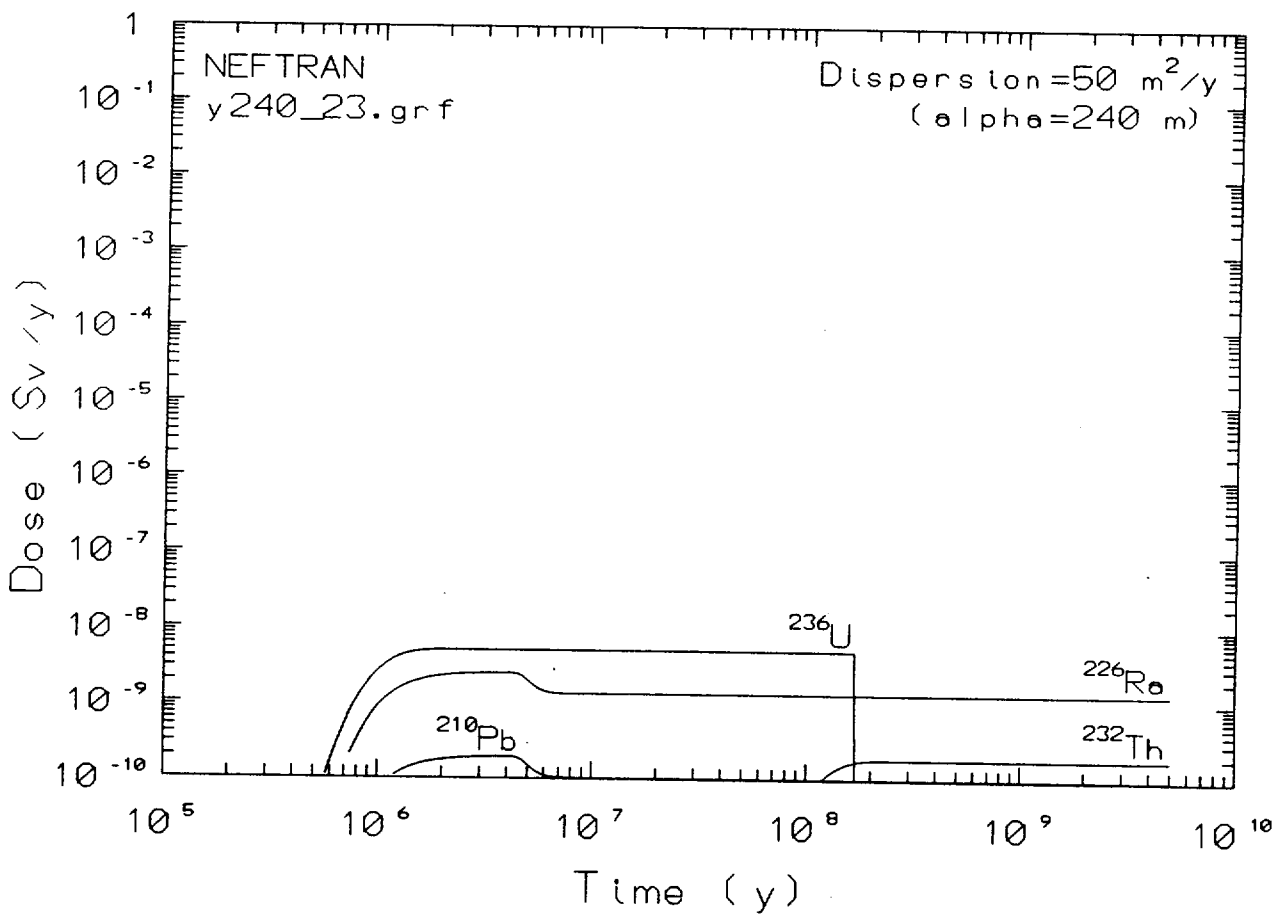


Figure 4-3. Dose to an Individual at the Accessible Environment from the ^{238}U and ^{232}Th Chains for the Baseline Case

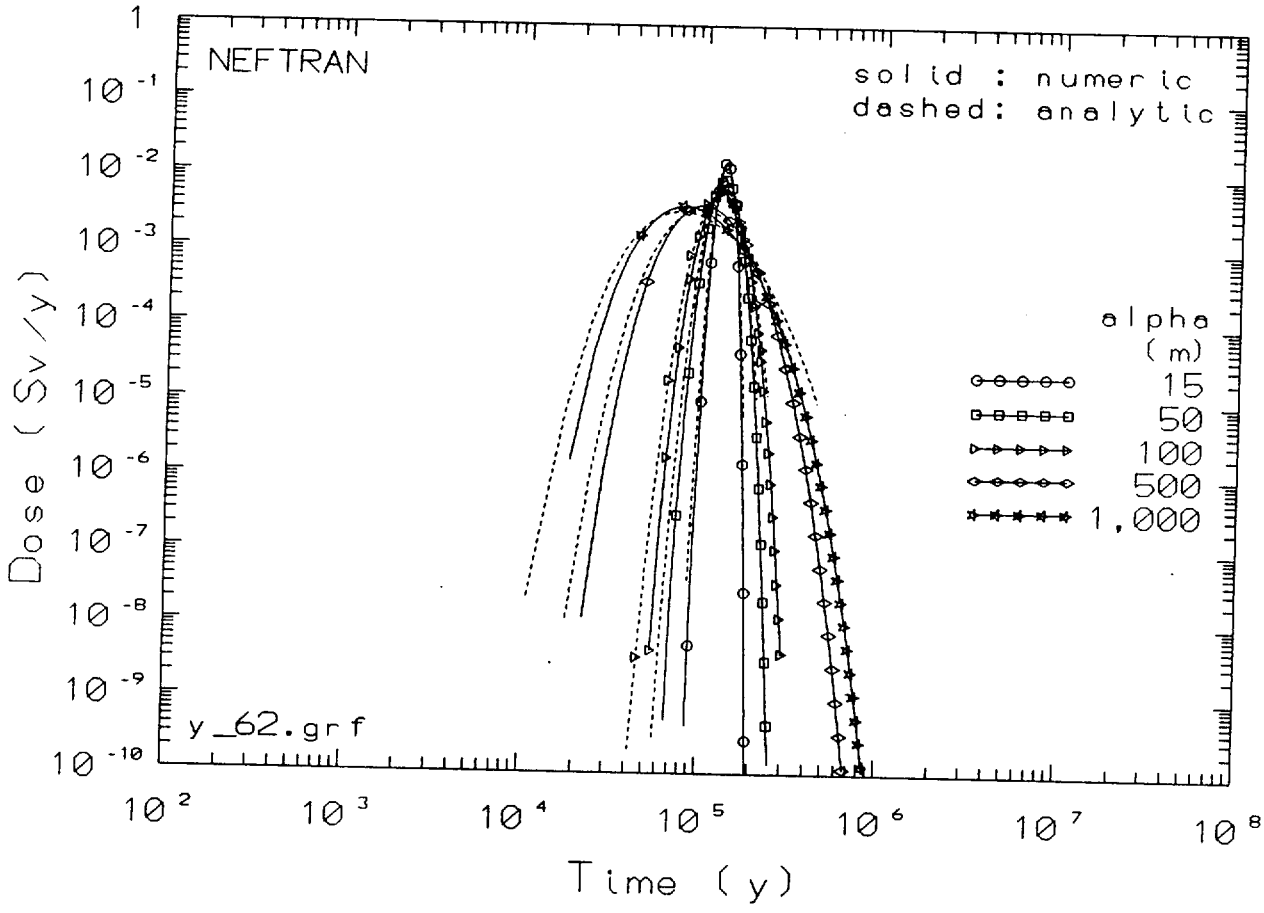


Figure 4-4. The Effects of a Range of Dispersivities (alpha) on ^{99}Tc Dose to an Individual at the Accessible Environment

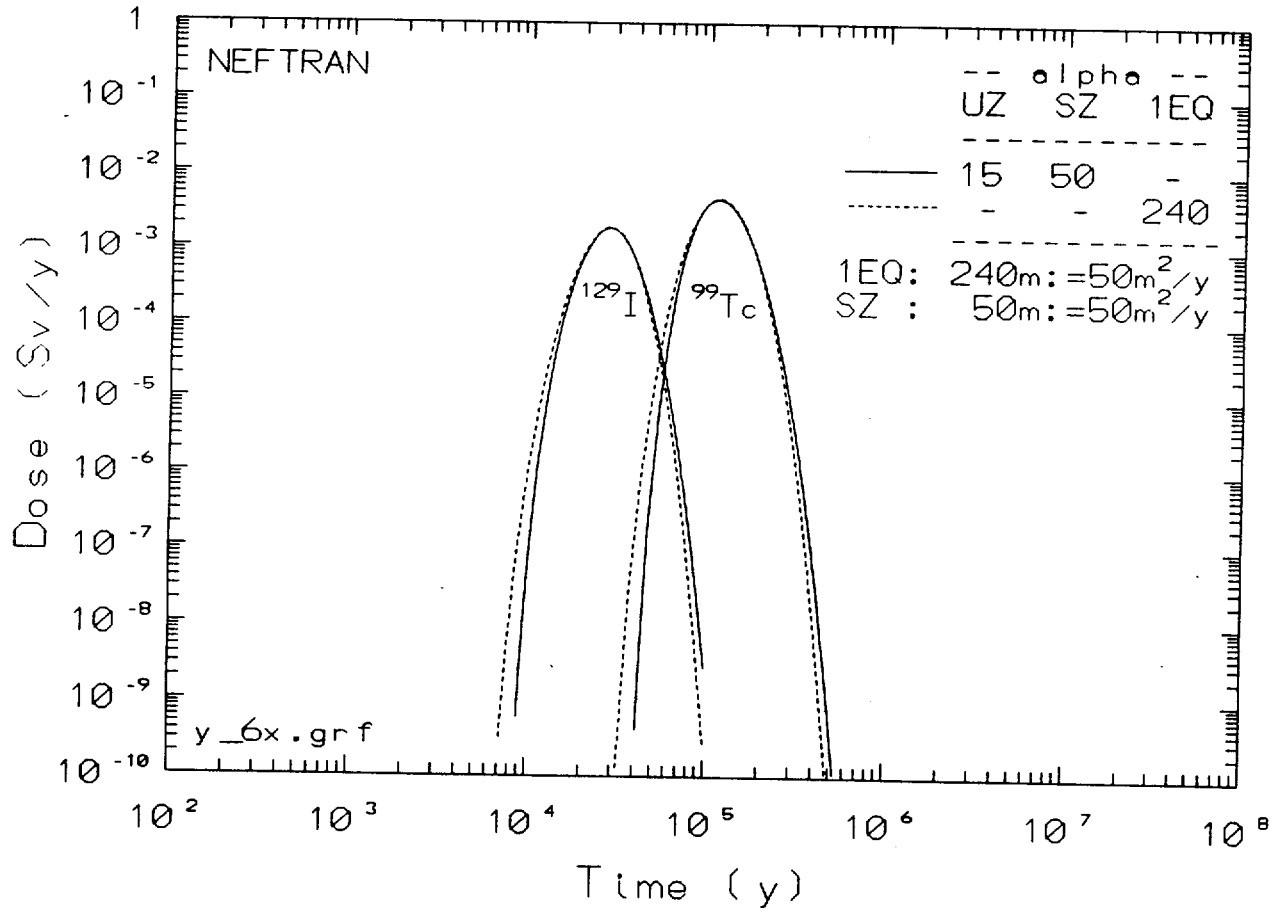


Figure 4-5. Comparison of Dose to an Individual at the Accessible Environment from ^{99}Tc and ^{129}I Assuming a Single Leg and a Double Leg Transport Path

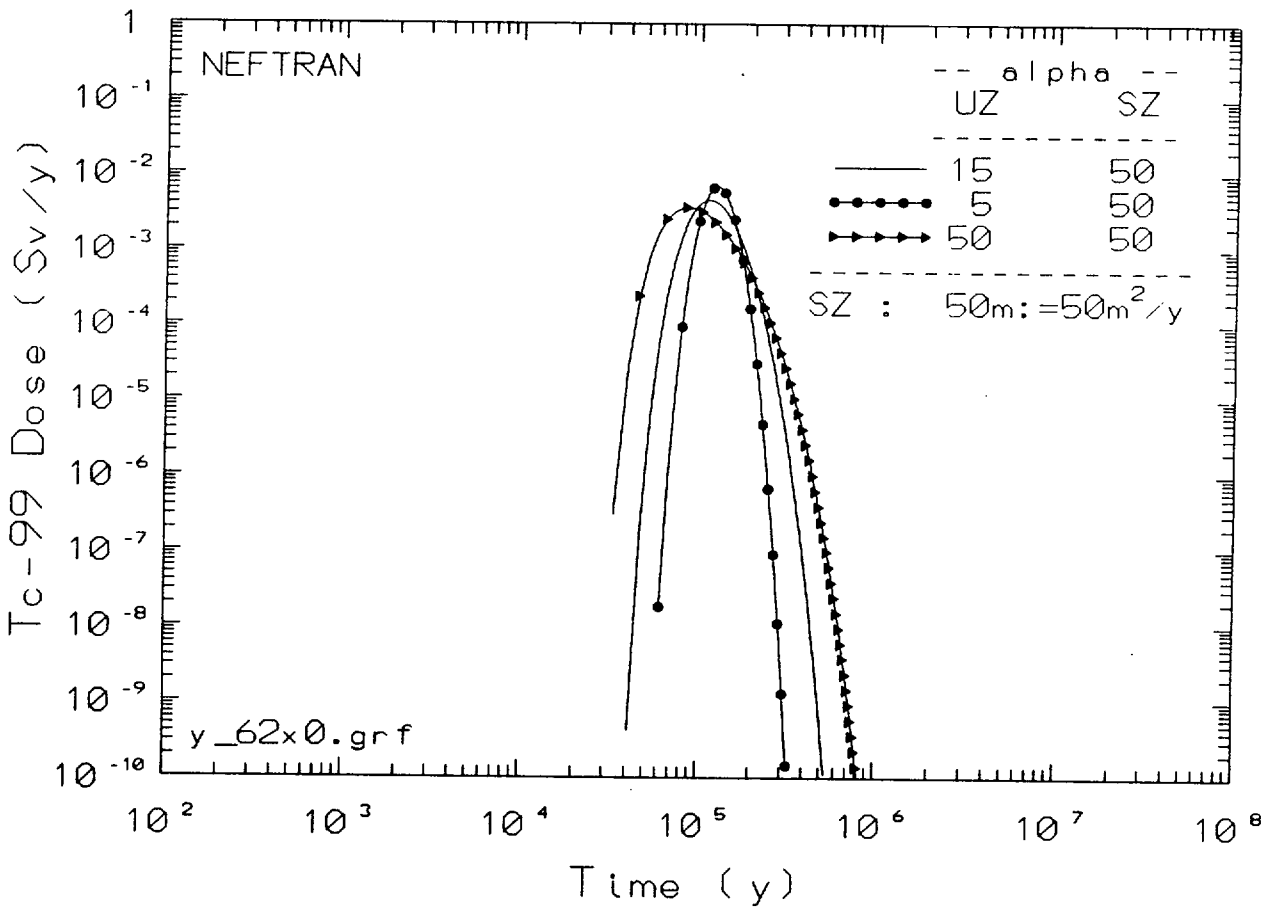


Figure 4-6. Effect of Variation of Unsaturated Zone Dispersivity on ^{99}Tc Dose to an Individual at the Accessible Environment

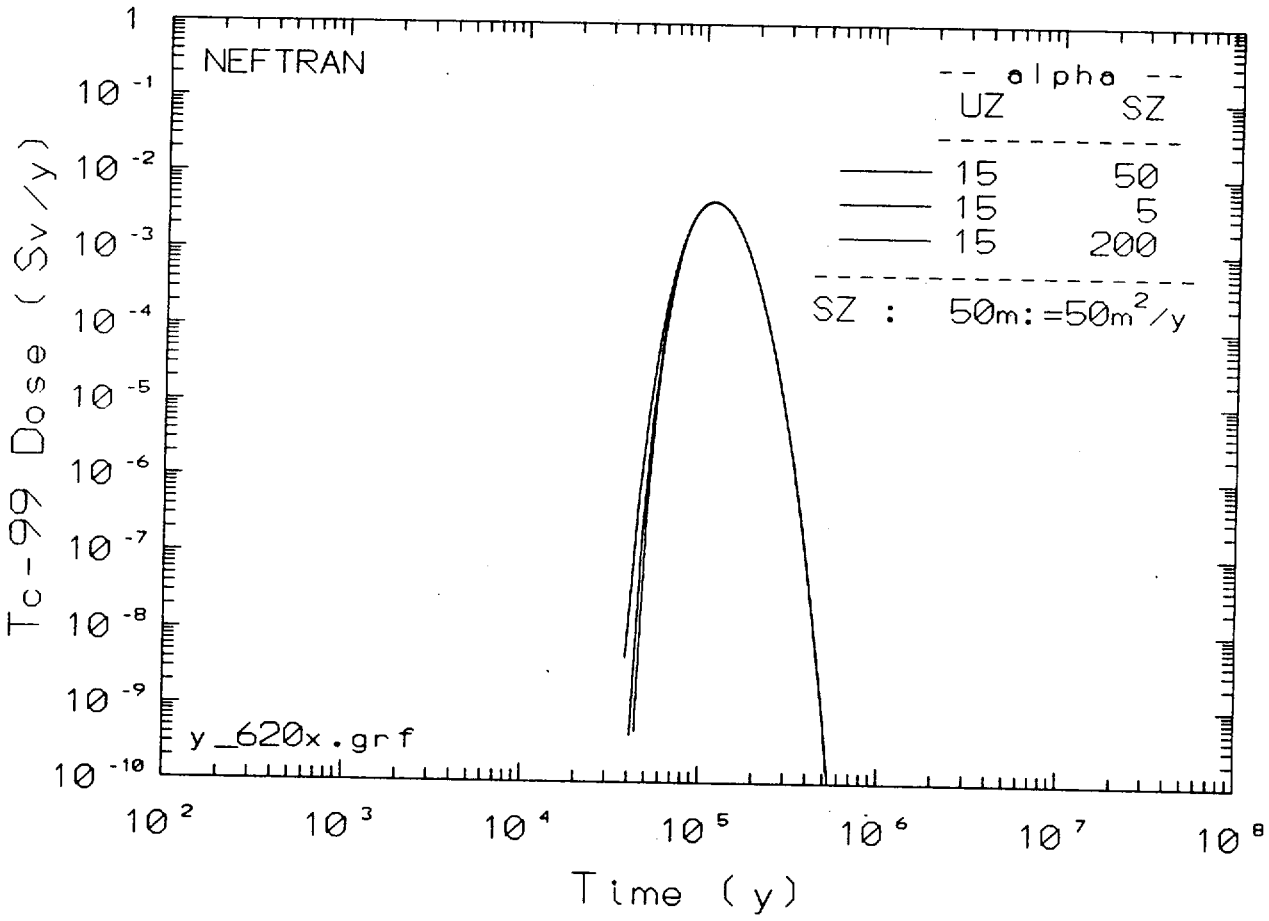


Figure 4-7. Effect of Variation of Saturated Zone Dispersivity on ^{99}Tc Dose to an Individual at the Accessible Environment

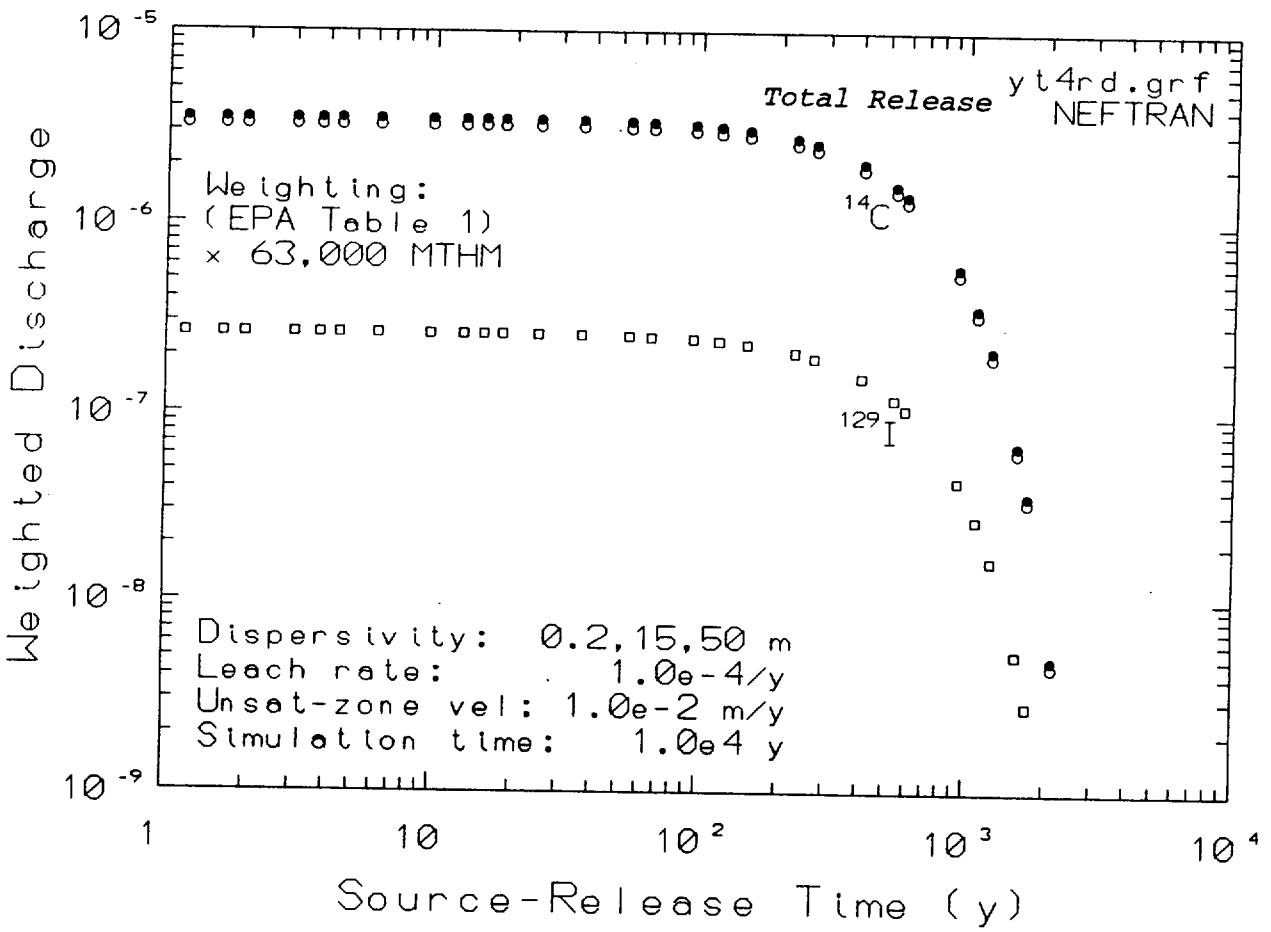


Figure 4-8. Normalized Release to the Accessible Environment as a Function of Source-Term Release Over 10,000 Years

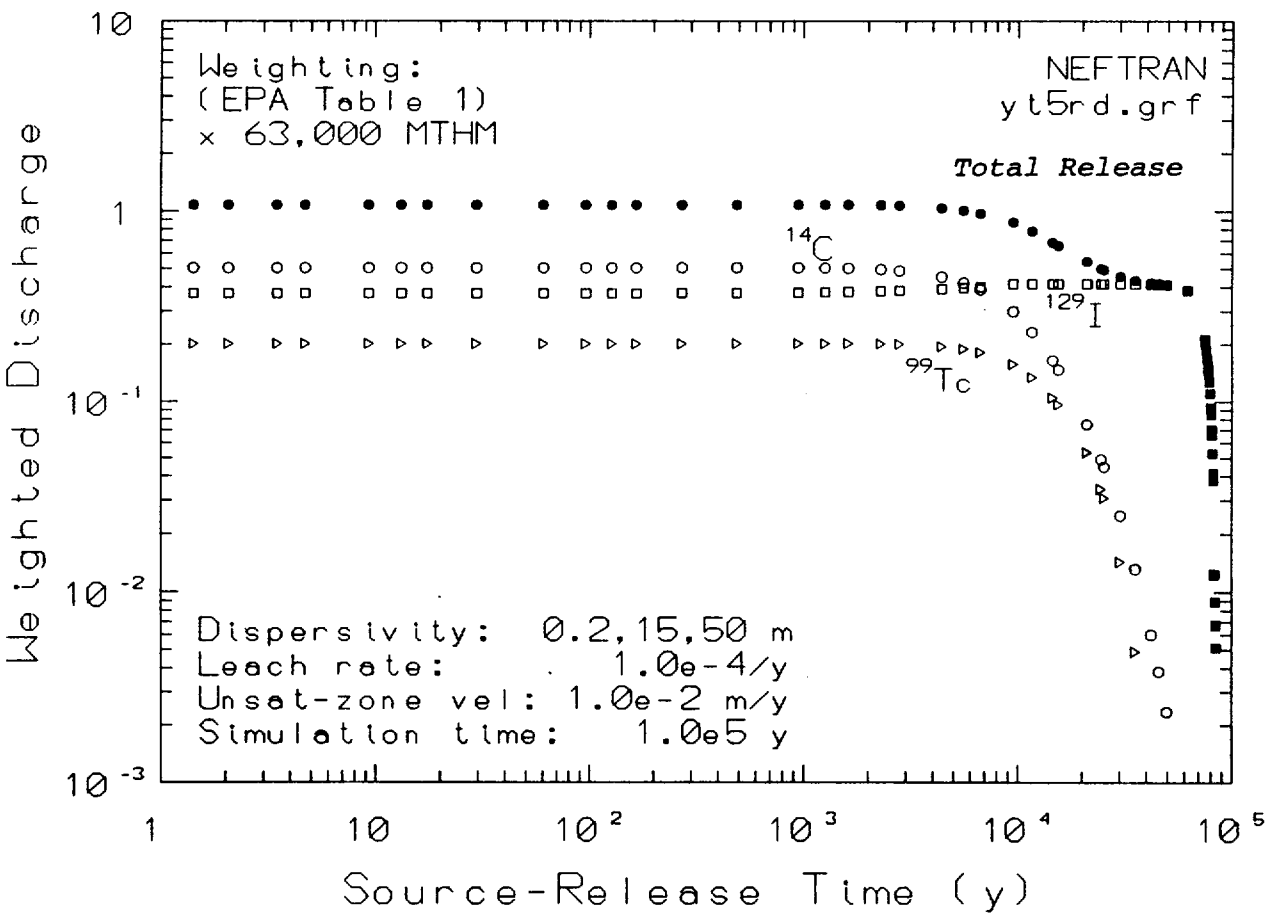


Figure 4-9. Normalized Release to the Accessible Environment as a Function of Source-Term Release Over 100,000 Years

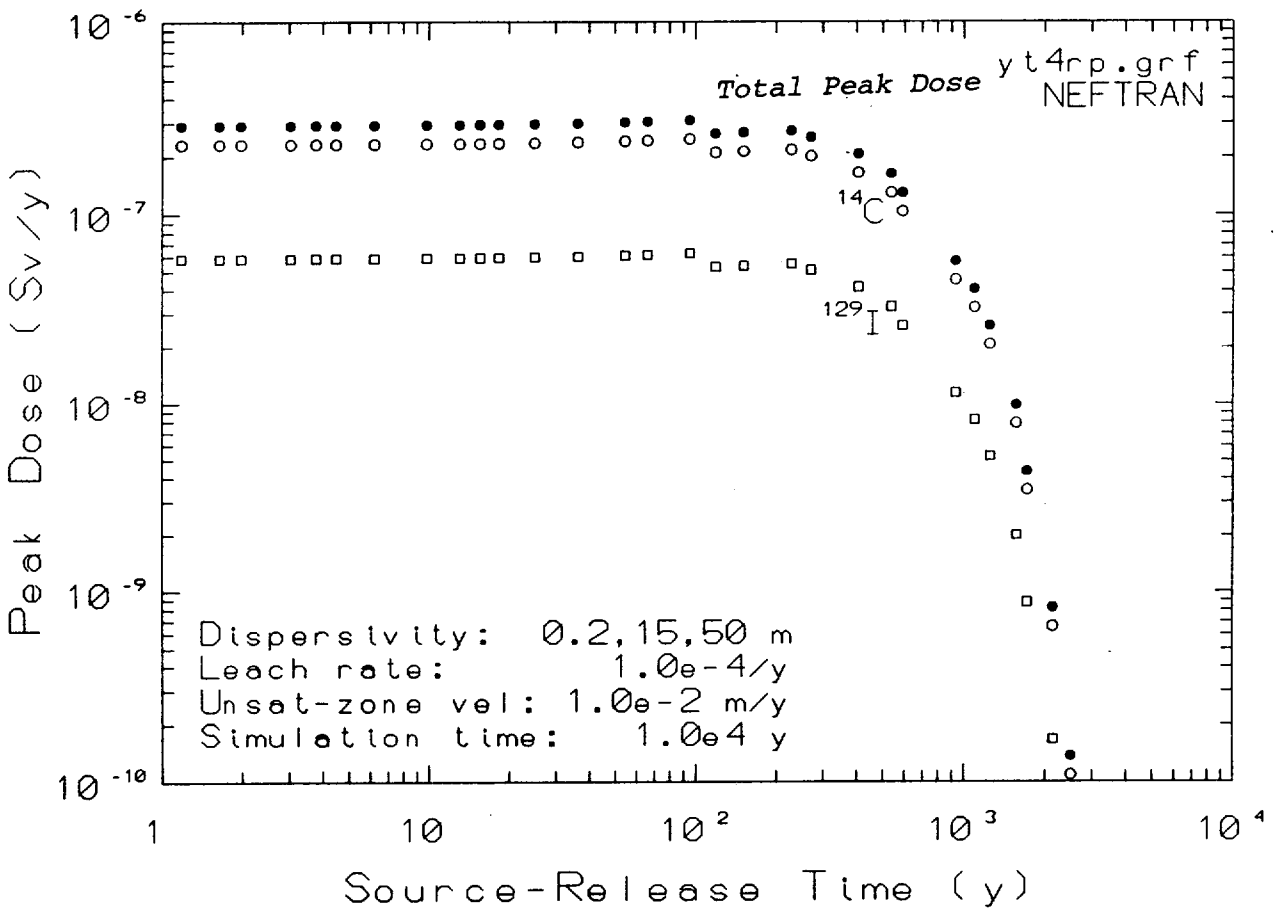


Figure 4-10. Peak Dose to an Individual Over 10,000 Years as a Function of Source-Release Time

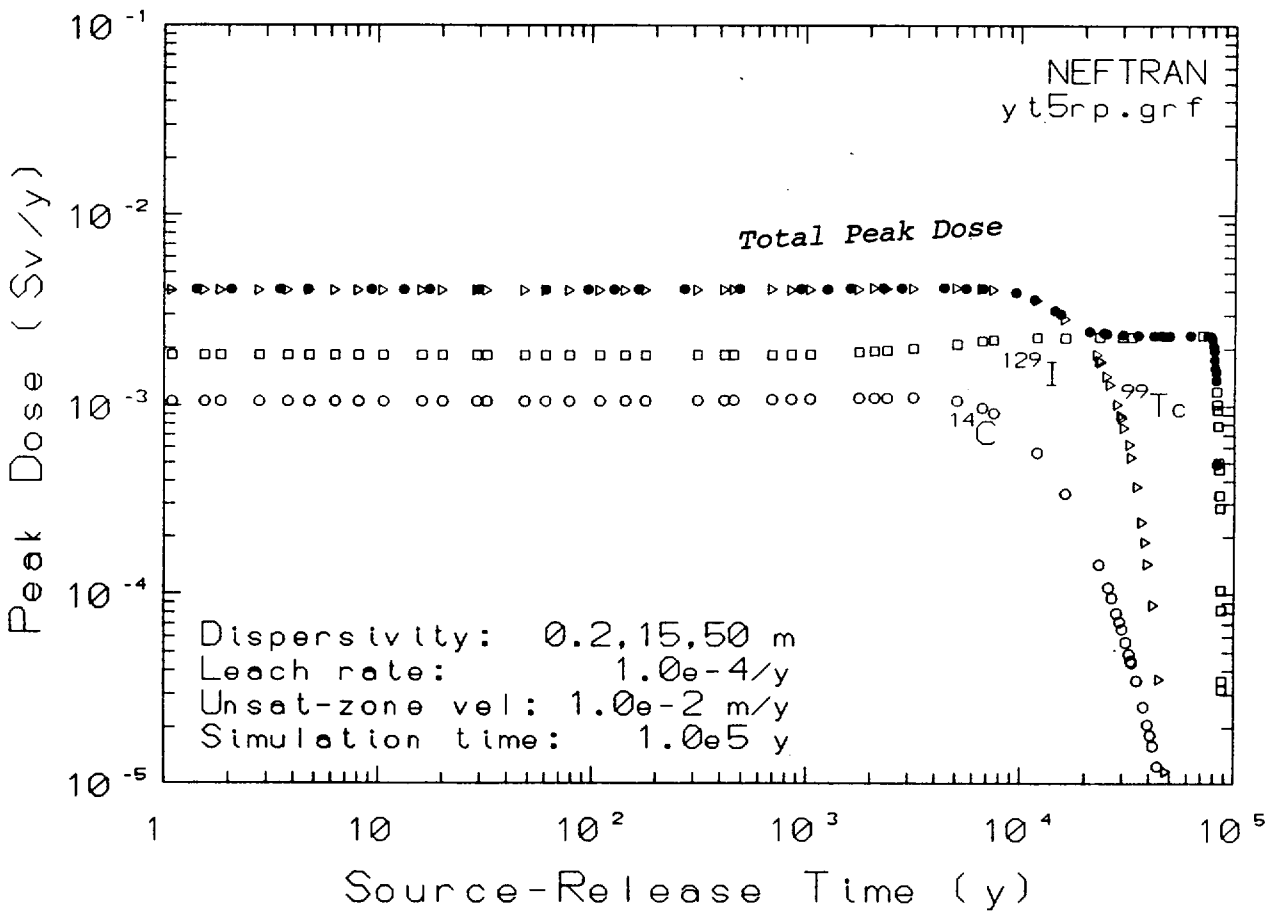


Figure 4-11. Peak Dose to an Individual Over 100,000 Years as a Function of Source-Release Time

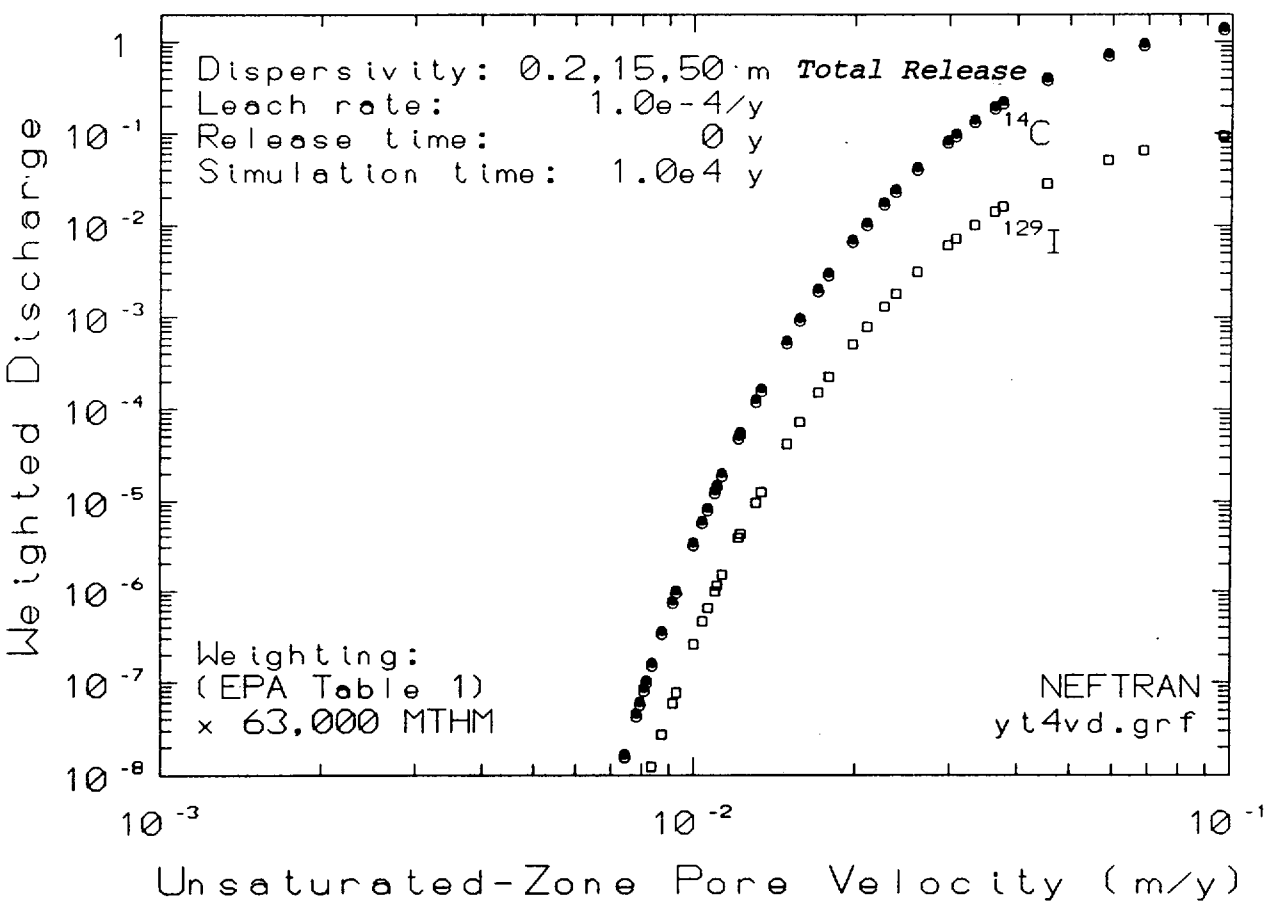


Figure 4-12. Normalized Release to the Accessible Environment Over 10,000 Years as a Function of Unsaturated Zone Pore Velocity

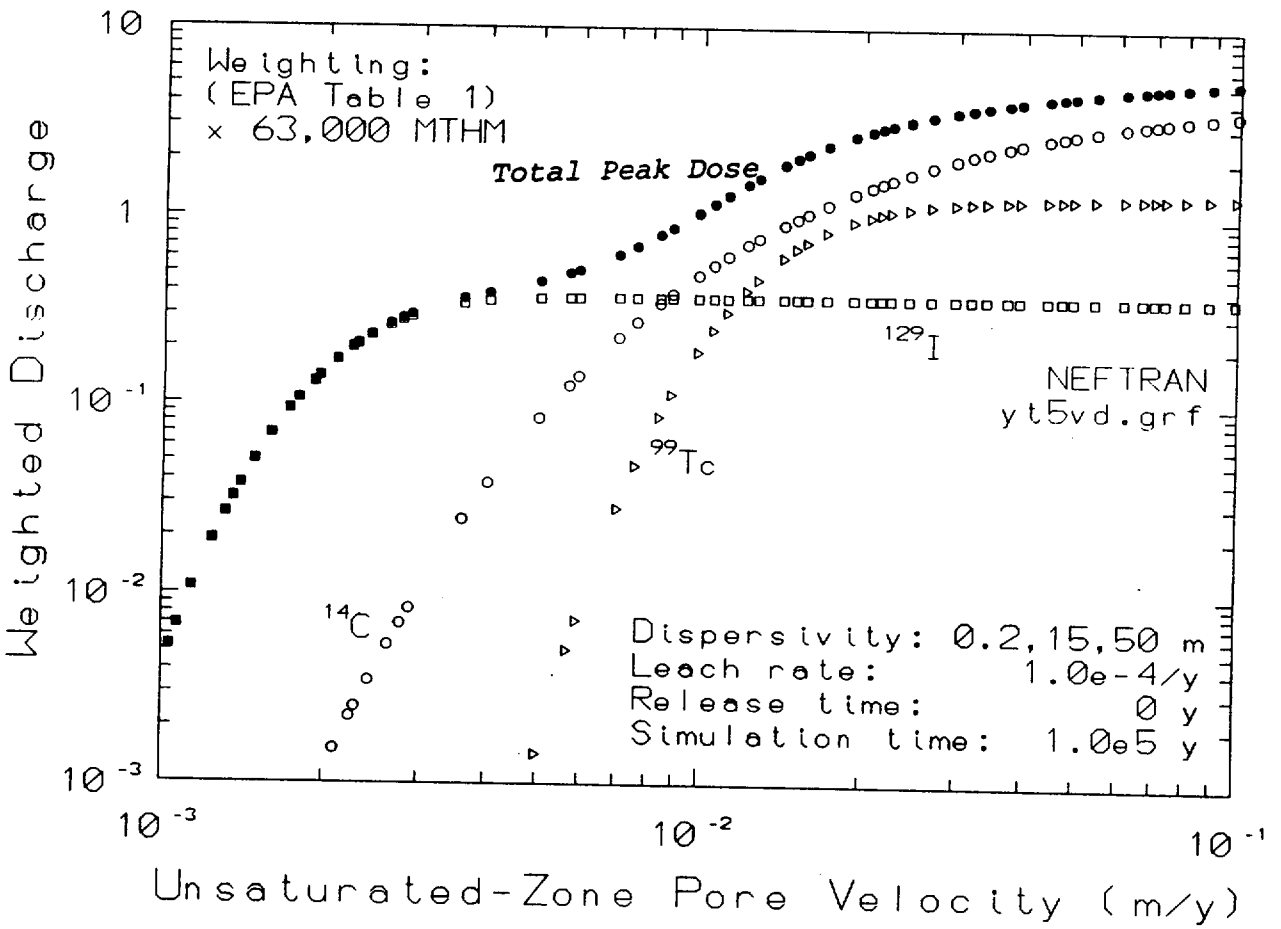


Figure 4-13. Normalized Release to the Accessible Environment Over 100,000 Years as a Function of Unsaturated Zone Pore Velocity

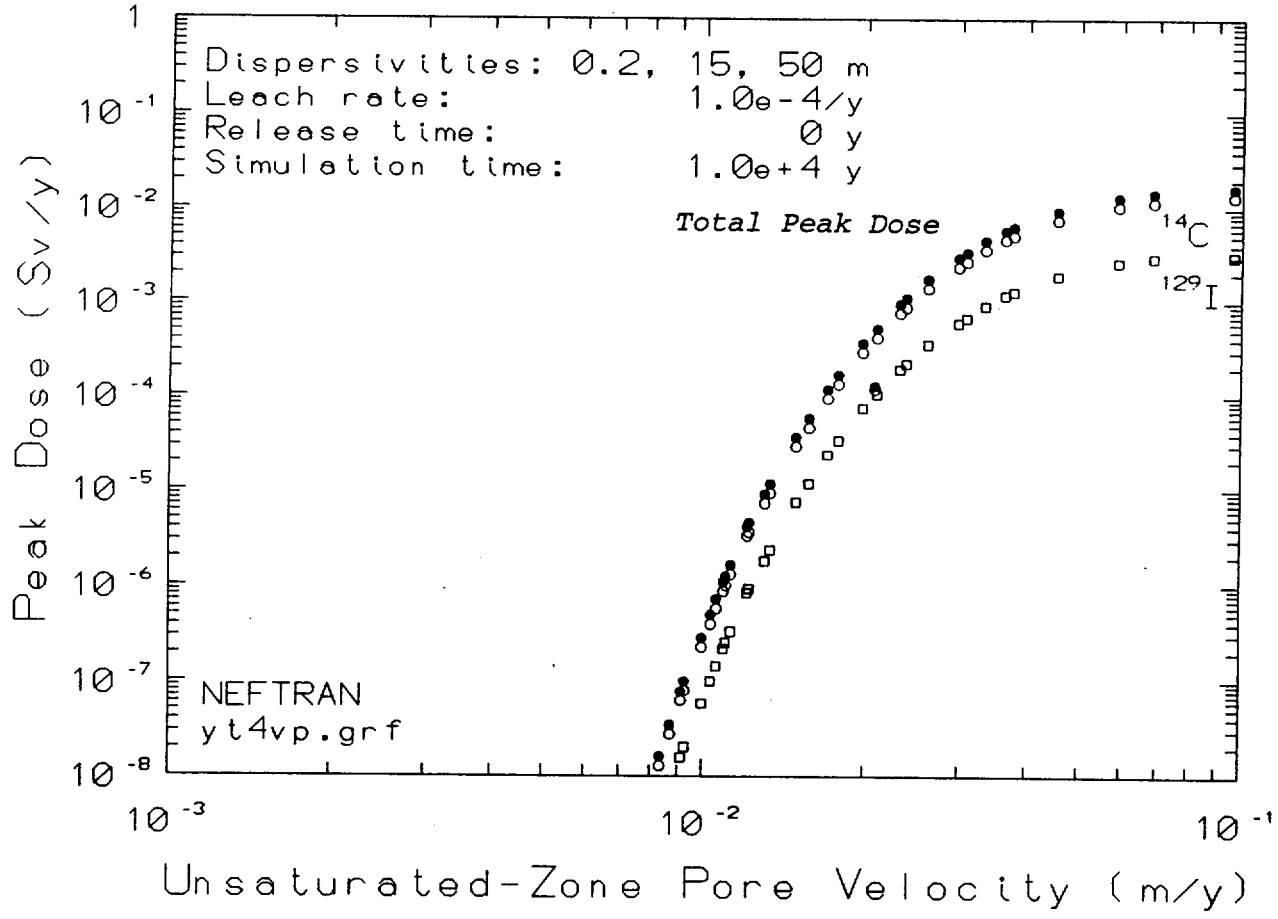


Figure 4-14. Peak Dose to an Individual at the Accessible Environment Over 10,000 Years as a Function of Pore Velocity in the Unsaturated Zone

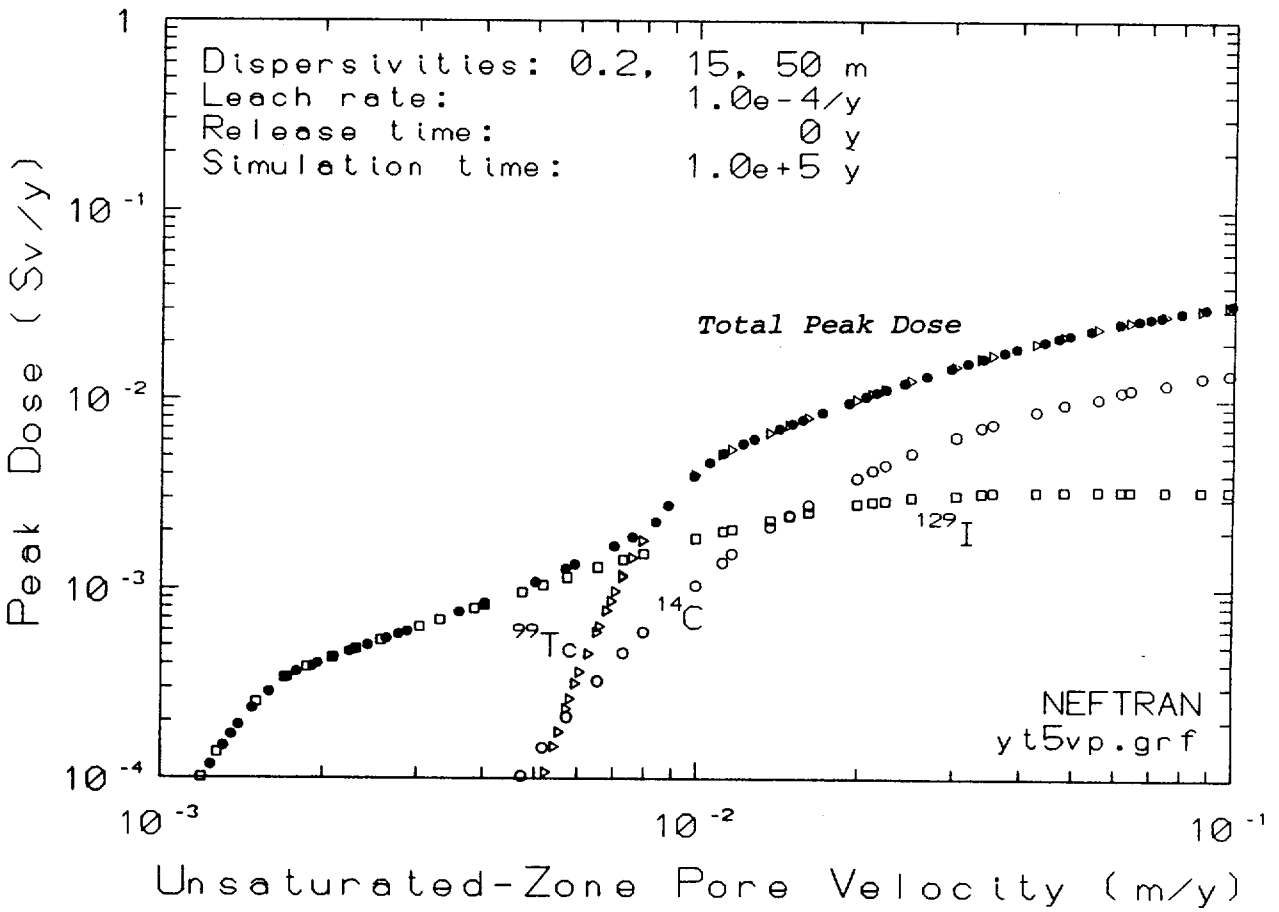


Figure 4-15. Peak Dose to an Individual at the Accessible Environment Over 100,000 Years as a Function of Pore Velocity in the Unsaturated Zone

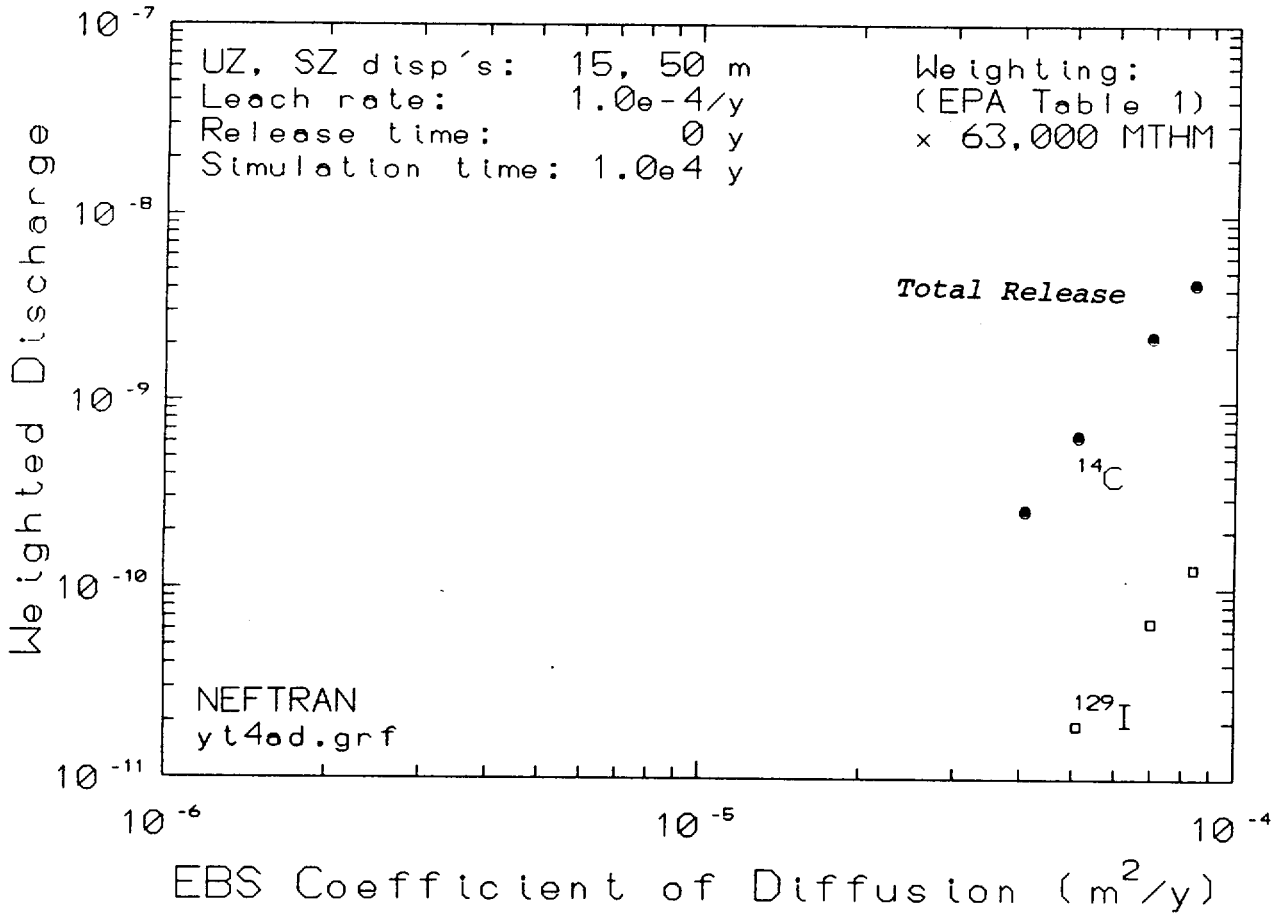


Figure 4-16. Normalized Release to the Accessible Environment Over 10,000 Years as a Function of Diffusive Release from the Engineered Barrier System

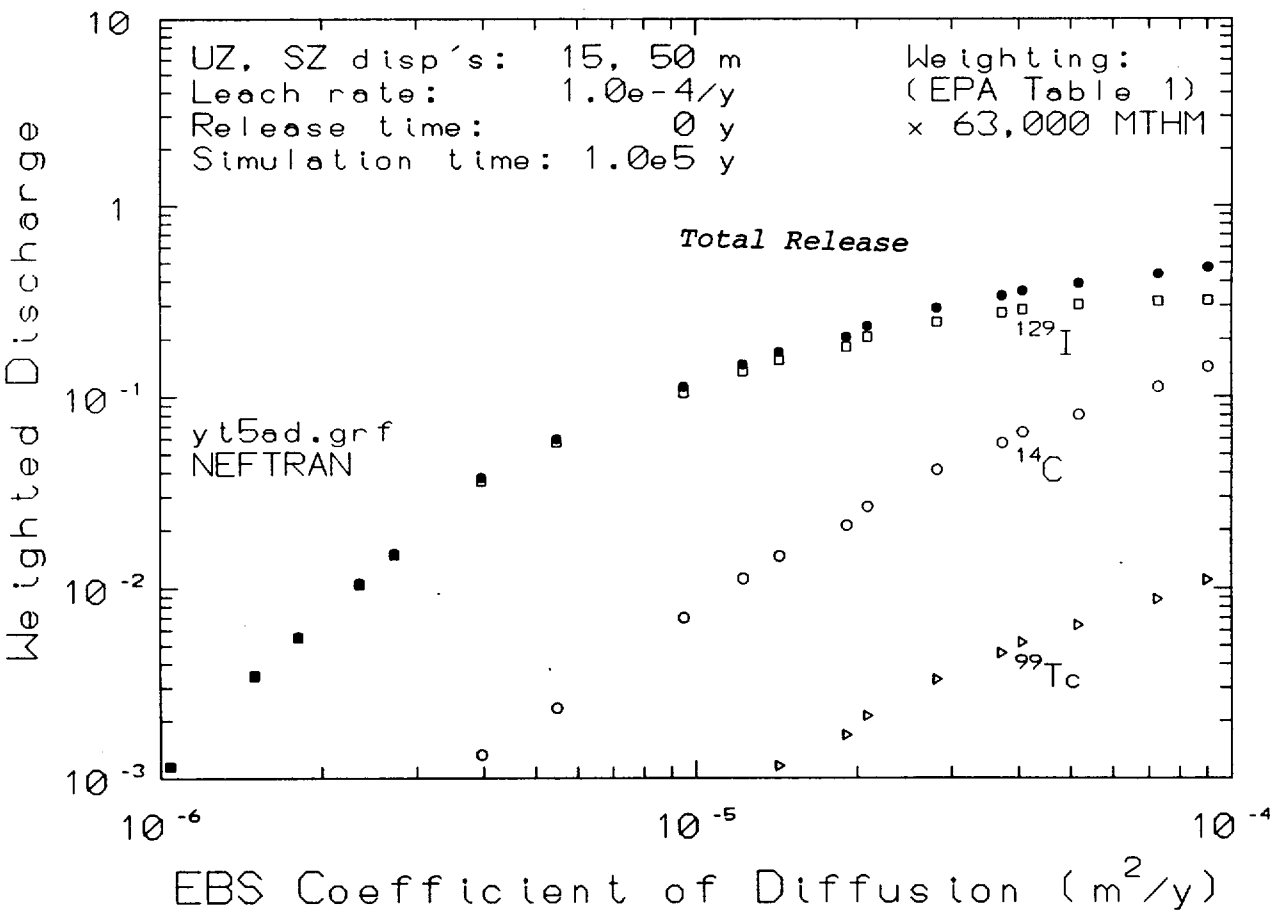


Figure 4-17. Normalized Release to the Accessible Environment Over 100,000 Years as a Function of Diffusive Release from the Engineered Barrier System

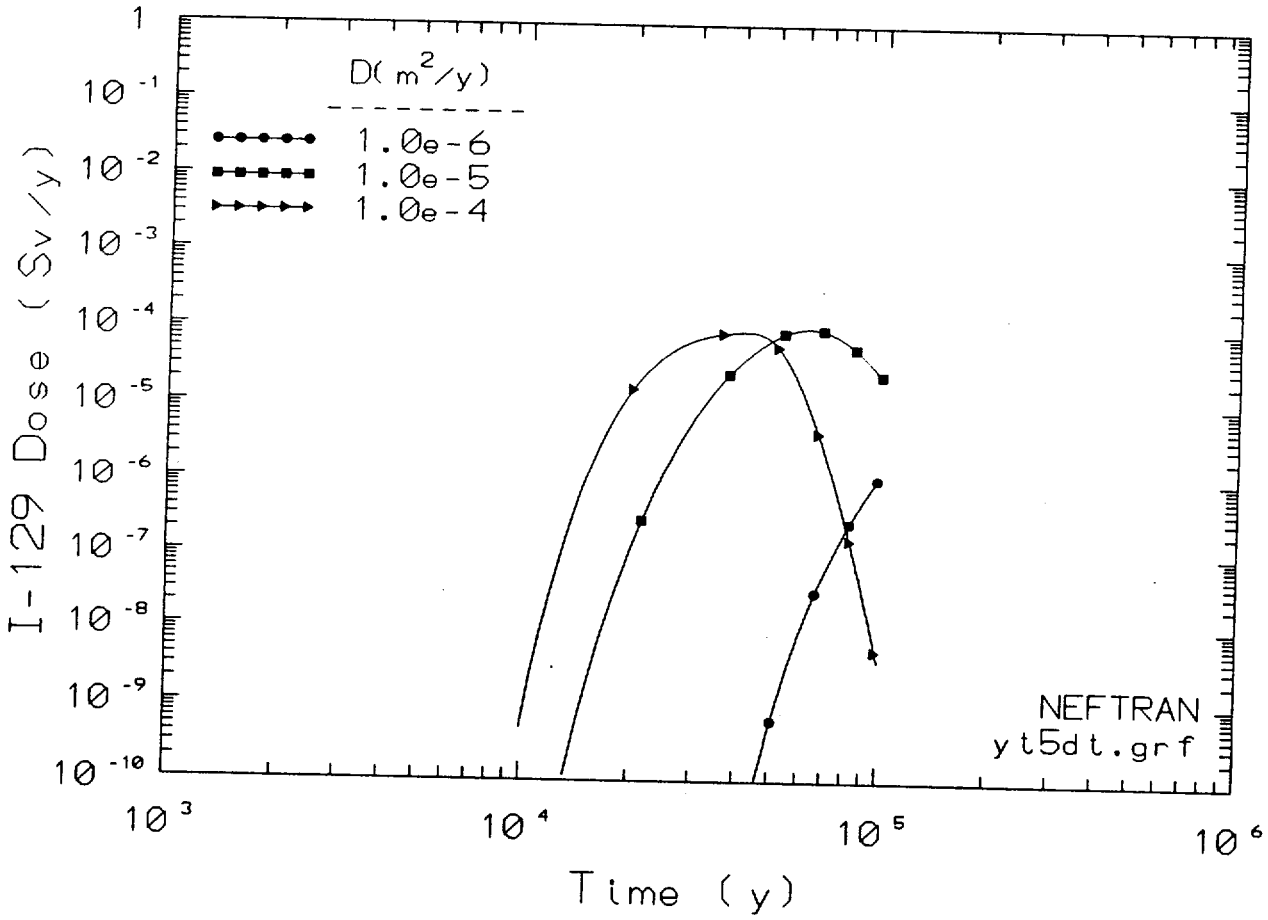


Figure 4-18. Dose to an Individual at the Accessible Environment from ^{129}I as a Function of Time for a Range of Values of Diffusion Through the Engineered Barrier System

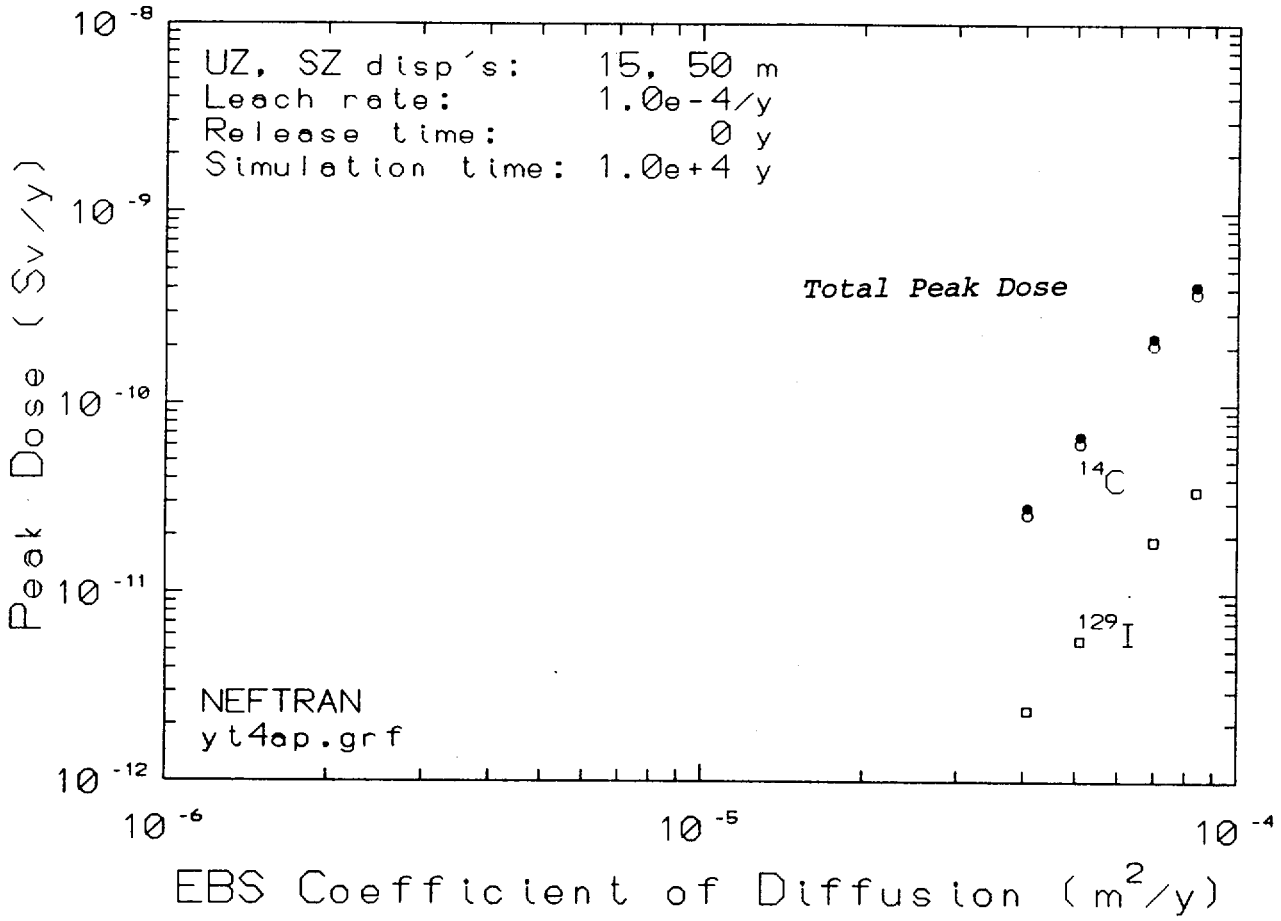


Figure 4-19. Peak Dose to an Individual at the Accessible Environment Over 10,000 Years as a Function of Diffusive Release from the Engineered Barrier System

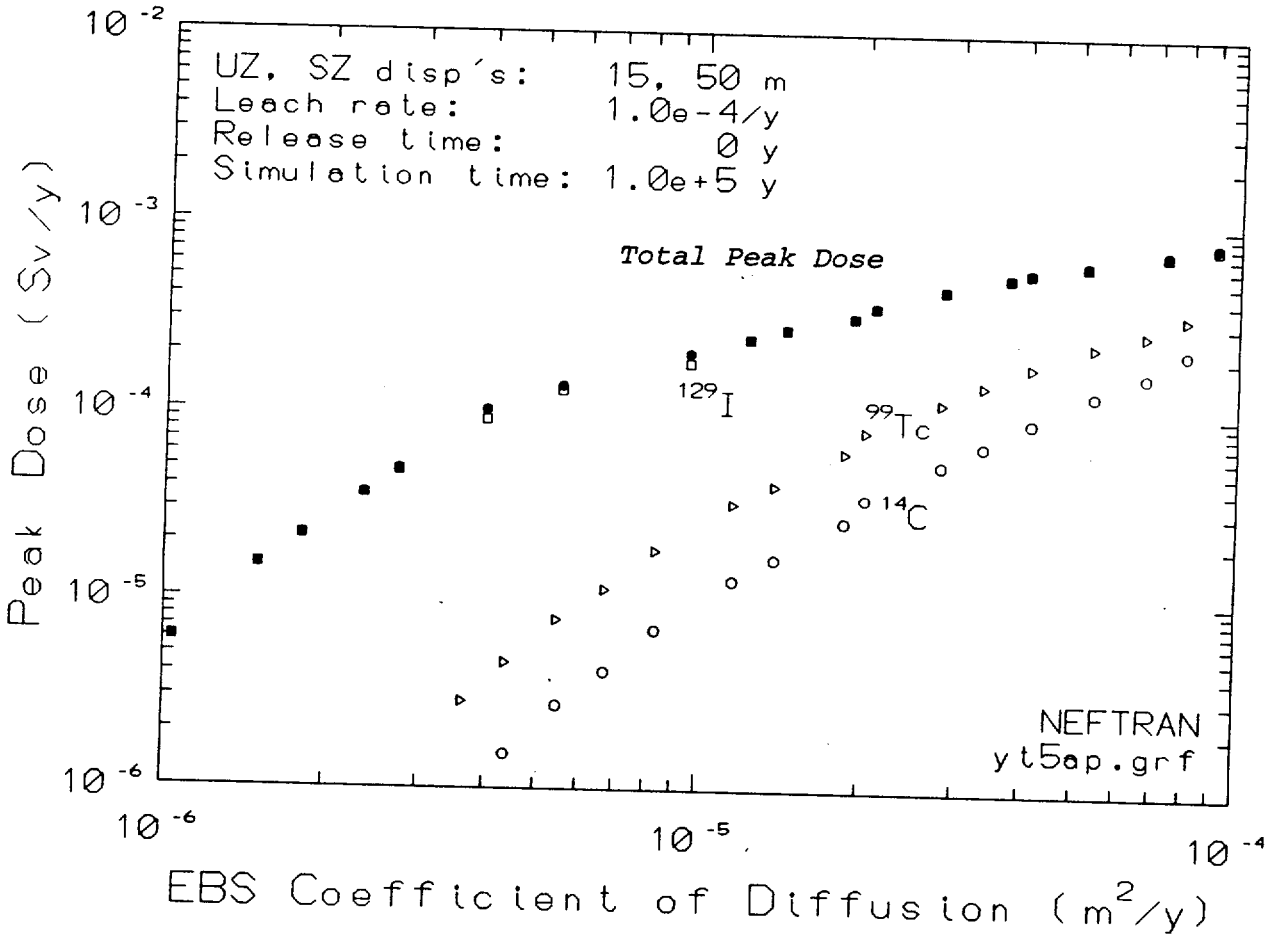


Figure 4-20. Peak Dose to an Individual at the Accessible Environment Over 100,000 Years as a Function of Diffusive Release from the Engineered Barrier System

5. SUMMARY AND CONCLUSIONS

5.1 INTRODUCTION

This document has presented a number of different analyses that should be useful to those responsible for setting appropriate environmental standards for the Yucca Mountain site in Nevada. In addition, in Appendix A, analyses are presented to quantify the range of the number of people possible to be supported on the available ground-water resources in the vicinity of Yucca Mountain. These populations are useful to bear in mind when selecting the critical population group which may be exposed to any aqueous release from the repository under the current climatological conditions. That is not to say that this critical population group will be exposed, because that depends on the long-term performance of the site and engineered barriers, which is the focus of the analyses presented in Chapters 2 to 4, as well as the probability that the population exists and uses ground water pumped from the tuff aquifer for the its individual use.

The authors have conducted some calculations of the possible performance of a potential repository at the Yucca Mountain site considering some of the key uncertainties affecting the release of radionuclides to the accessible environment. The performance measures addressed are the integrated release of radionuclides at the accessible environment normalized by the Table 1 values presented in 40 CFR 191, and the individual dose associated with a maximally exposed individual who extracts water for personal use and irrigation for a small farm from the saturated zone tuff aquifer at the accessible environment. The authors did not extend the calculations to the next level of performance (i.e., the individual or cumulative health effects associated with the release or dose, or the corresponding risk(s) that the individual or population may be subject to at the time of release, several tens to hundreds of thousands of years in the future). This extension was not made because of the subjectivity related to determining the probability that the individual or population actually exists at the location defined as the "accessible environment" as well as the consumption habits of those individuals at the time when any release may potentially be ingested. It should also be noted that even the translation from concentration to dose is fraught with considerable uncertainty which is not addressed in the analyses presented in Chapters 2 to 4.

All potential environmental standards have not been evaluated that could be applied to assure the protection of individuals and population groups that may be exposed to radionuclides that may be released over the millennia after closure of a potential repository at Yucca Mountain. A comprehensive evaluation would need to examine alternative assumptions about the future of the biosphere, as well as considering the full range of possible conversion factors from dose to risk (i.e., health effects), in order to ascertain the likely ranges of possible performance. The authors stress their belief that if dose, health effects, or risk are stipulated as performance measure(s) to be regulated, the future states of the biosphere must be specified explicitly by the regulatory bodies, or compliance will not be demonstrable. That is, the individuals or populations that our society chooses to protect (whose actual number, habits, and locations are unknowable) will have to be stipulated in order to allow a determination of the consequences and risks associated with the potential repository that is defensible as a regulatory compliance calculation.

The "simple" post-closure total system performance assessments that were conducted and described in this document are useful in assisting those responsible in determining an appropriate measure of post-closure performance as well as those who have to evaluate site and design information needs. Each of the processes included in the analyses has been simplified to capture the primary significance of that process in containing or isolating the waste from the biosphere. For example, the primary role of the waste package is to delay the initial exposure and the rate of exposure of the waste form and spent fuel matrix to water. The degradation of the waste package is a very complex function of the very near field thermal-hydrologic-chemical-mechanical environment and the behavior of the different engineered materials comprising the package and any backfill. In these simplified analyses, these complex interactions were approximated by a simple waste package "failure" distribution which is defined by the initiation and rate of waste package "failures". Similarly, releases from the waste package and the engineered barrier system are controlled by the very near field environment and the presence and rate of advective and diffusive release processes. Release was approximated by either a simple alteration-controlled release for the high solubility radionuclides and either a diffusive or advective-controlled release for the solubility-limited radionuclides. In general, the term "simple" performance assessment implies that the complexities associated with process coupling and process interactions have been neglected. This allows the analyses to be more transparent so that the relative importance of a particular component of the system (whether the waste package, or engineered barrier, or geosphere) is more illustrated. In Section 5.4 the results of the "simple" total system performance assessments presented in this document are compared with the more representative (in that more process interactions and dependencies are incorporated) performance assessment results conducted as part of the total system performance assessment (TSPA-1993) documented in Andrews et al. (1994).

5.2 COMPARISON OF RESULTS FROM UCBNE-41, RIP, AND NEFTRAN-S

In addition to addressing alternate performance measures and the effects of alternate parameter distributions, three different computational tools have been used, UCBNE-41, RIP, and NEFTRAN-S. These tools have different strengths that were exploited in the analyses. The model UCBNE-41 has an excellent transport module which has been widely used to evaluate the transport of radionuclide chains in the geosphere, in particular during the course of the WISP analyses documented in NAS (1983). Using UCBNE-41, the results were first compared to those presented in NAS (1983) to check for consistency (see Appendix C). The disadvantage of the UCBNE-41 model is that the source term and biosphere had to be treated by the use of spreadsheets external to the model itself, which made data manipulation cumbersome and did not readily allow the use of stochastic sampling to evaluate the uncertainty in the predicted performance. The effect of assuming advective flow through the waste package was also compared to the assumption that the release was through diffusion out to the boundary of an equivalent spherical package where it enters the advective flow system as was the case in the WISP analyses. This assumption yields the same peak doses for the alteration-controlled radionuclides and produces similar peak doses for ^{237}Np and ^{227}Ac.

The RIP model has been used in TSPA-1993 and allows for a very flexible definition of the relevant processes according to the objectives of the analyses. In the present use of RIP, none of the process dependencies were incorporated. An advantage of using RIP or NEFTRAN-S is that the geosphere can be approximated using multiple one-dimensional legs or segments

connected in parallel or in series. In the present analyses, two legs are used, one leg for the unsaturated zone and one for the saturated zone instead of the one leg used in the UCBNE-41 analyses. The primary advantage of using RIP is that it can be operated in the probabilistic mode with distributions of parameters over the range of their uncertainty. This allowed for the use of parameter distributions to capture the effects of a range of pore velocities in the unsaturated zone, neptunium solubilities, and waste package lifetimes. The RIP model has the capability of incorporating a relatively sophisticated waste package with the package release being either through advection or diffusion or a combination of both. For most of the simulation runs, advection through the waste package was held to zero and the release was assumed to be through diffusion either as a fixed value or as a distribution of values. This approach can be thought of as the failed packages being surrounded by a capillary barrier which allows no advective flux but can have a thin water film through which radionuclides can diffuse to the rock once the packages have been breached. No attempt was made to relate the range of diffusion coefficients to the saturation of the capillary barrier (which would relate to the continuity of the water film layer for continuous diffusion). The results using RIP can be plotted as dose or release as a function of time, as scatter plots of either dose or release as a function of a variable over its range (pore velocity, solubility, release time, or diffusion), or as CCDFs of release or peak dose. This flexibility allows the user a series of plots from a given analysis for interpretation of the effects of specific variables.

The attributes of NEFTRAN-S are similar in many aspects to RIP with the exception of the versatility of the waste package. Advection from the waste package is the only mechanism of release and all packages fail instantaneously at a given time. This limits the investigation of waste package failure that can be done using NEFTRAN-S. One major advantage of NEFTRAN-S is that it is designed to approximate a multi-dimensional problem using a series of one-dimensional legs, allowing the user to simulate barriers to release as independent legs (i.e., a capillary diffusion barrier). The difficulty with doing this in the case of a diffusion barrier is that diffusion mechanisms were not incorporated explicitly and had to be approximated as an advective flow with appropriate parameters. The multiple-leg capability was used to investigate the effects of a single leg conceptualization of the Yucca Mountain geosphere as compared to the two leg unsaturated-/saturated-zone representation. The results indicated that there is very little difference between the two representations when appropriate values of dispersivity are used.

A disadvantage of both RIP and NEFTRAN-S is a conceptual difficulty for alteration-controlled radionuclides when a diffusion barrier is present. The alteration allows a large amount of a particular radionuclide to be dissolved in the very small amount of water that is in the diffusion barrier when physically there should be a limit (some rather high solubility limit) to how much of the radionuclide the water film can hold. This conceptual problem produces higher releases of alteration-controlled radionuclides than might be expected to occur. A correction to this problem would require data for the solubility of radionuclides that have previously been considered to be alteration controlled.

The application of UCBNE-41, RIP, and NEFTRAN-S to the baseline case produced very similar results. The magnitude of the peak doses to an individual for the key radionuclides (^{14}C, ^{129}I, ^{99}Tc, and ^{237}Np) from the three models are close enough that the differences are indistinguishable on the plots. In addition to the peak doses being analogous, the time of arrival of the peaks was also the same. These observations indicate that these models, as implemented in these analyses,

are each applicable to assessing the relative performance of a conceptually simplified potential repository at Yucca Mountain.

5.3 SUMMARY OF SENSITIVITY ANALYSES

The sensitivity analyses were conducted using a source term that contained 39 radionuclides for simulations using UCBNE-41 and RIP. The source term was reduced to 21 radionuclides in simulations using NEFTRAN-S with no loss in the applicability of the results. In all of the simulations conducted in this study, the primary radionuclides responsible for dose to an individual at the accessible environment are ^{14}C, ^{99}Tc, ^{129}I, and ^{237}Np. All of the simulations assumed that ^{14}C was released by way of the ground-water pathway when in fact it is expected to be released by way of the gaseous pathway at an unsaturated site such as Yucca Mountain.

In the baseline case, a ground-water travel time of 25,000 years was assumed, with 20,000 years in the unsaturated zone and 5,000 years in the saturated zone. This is equivalent to a 1.0 mm/yr percolation flux in the unsaturated zone. The waste packages were assumed to fail immediately (at time zero), the release from the waste package was assumed to be from advection through the package and the release to the saturated zone was assumed to be mixed to a depth of 2400 m. For these assumptions, the dose peaks from ^{14}C and ^{129}I are in the 10^{-3} Sv/yr range and arrive at about 25,000 yrs because they are not retarded. The ^{99}Tc peak arrives just beyond 100,000 years and is somewhat greater than the ^{129}I peak. The peak dose from ^{237}Np arrives between 200,000 and 500,000 years at about 10^{-4} Sv/yr (assuming a solubility of 10^{-3} g/m^3). Had the assumption been made that the mixing depth in the saturated zone was 1,000 m or 100 m the peak doses in the baseline case would have been increased by a factor of 2.4 or 25, respectively.

Dose to an individual at the accessible environment was found to be relatively insensitive to waste package lifetime for the radionuclides ^{129}I, ^{99}Tc, and ^{237}Np because of their long half lives. The time of occurrence of the peak doses for waste package lifetimes varying from 1,000 to 100,000 years are only shifted in time by the lifetime of the package. This is also true when a linear rate of package failures (1% per 1,000 years) is assumed. As the time for the initial waste package failure increases from zero to 10,000 to 100,000 years, the time for the peak doses to occur at the accessible environment also increases from 160,000 to 180,000 to 260,000 years; while the peak dose (due to ^{99}Tc) changes from about 1.5×10^{-3} to 1.1×10^{-3} to 1.0×10^{-3} Sv/yr. The slight decrease in peak dose is due to the relatively long half life of ^{99}Tc (about 200,000 years). For slower failure rates (1.0 % per 10,000 years), the effect of decay is more pronounced, reducing the peak dose to about 10^{-4} Sv/yr.

The effect of waste package lifetime on release to the accessible environment is largely a function of the period over which the cumulative release is integrated. If one determines the 10,000 year integrated release, then the package lifetime would have to be on the order of several thousand years to have an effect; while for the 100,000 year release, the package lifetime would have to be on the order of 80,000 years to have an appreciable effect.

The effects of percolation flux (or interstitial velocity) in the unsaturated zone on dose to an individual at the accessible environment is dependent on the model for release from the waste package. For advective flow through the failed waste package, the doses for higher fluxes (shorter travel times) are higher because of increased flux through the waste package and less

radioactive decay prior to the release at the accessible environment. For a range of unsaturated zone pore velocities from 10^{-3} to 10^{-1} m/yr (percolation fluxes from 0.1 to 10 mm/yr) the normalized cumulative release over 10,000 years approaches the EPA Table 1 release limit as the pore velocity approaches 10^{-1} m/yr (which is much higher than the estimated pore velocity at the Yucca Mountain site). The peak dose over 10,000 years approaches 10^{-2} Sv/yr at the higher end of the pore velocity range. The doses are from ^{14}C and ^{129}I because ^{99}Tc has not arrived at significant concentrations in this period, even at the highest pore velocities. When the time is increased to 100,000 years, the normalized cumulative release increases to more than one, but less than a factor of ten times the EPA release limit specified in 40 CFR 191. The peak dose to an individual increases due to the arrival of ^{99}Tc and the increased release of ^{14}C and ^{129}I. The unsaturated-zone average linear velocity is still the dominant controlling factor in the magnitude of the dose exposure. This is especially true for the cases where only the dispersive fronts of some of the significant dose contributors are estimated to reach the accessible environment for simulation times of 10^4 and 10^5 years. For these cases the dose exposures are so sensitive to the uncertainty in the unsaturated-zone average linear velocity that it tends to obscure the possible sensitivity of the releases from the waste package due to the effective diffusion coefficient and the neptunium solubility.

The effects of diffusion from the waste package is most easily observed in the few runs conducted using UCBNE-41 with an advective/diffusive release from an approximate sphere. The diffusion does not effect the release rate of alteration-controlled radionuclides and only effects the solubility-controlled radionuclides. Scatter plots of peak dose for a range of diffusion coefficients within the waste package show that the dose rate to an individual is reduced significantly at the lower values of diffusion (dose is reduced about three orders of magnitude over a range of four orders of magnitude in diffusion). The results of diffusive release from the waste package appear to indicate that a significant dose reduction could be achieved for ^{129}I and ^{99}Tc as long as it can be assured that no advection through the waste package can occur.

Using UCBNE-41 it was shown that the very-long-term doses to an individual at the accessible environment are strongly related to the solubility of ^{237}Np which may range from 0.001 to 100 g/m^3 depending on the geochemical environment in the vicinity of the waste package, in particular the degree of oxidation. In fact, neptunium releases become alteration controlled at the higher values of solubility. Scatter plots of 100,000 to 1,000,000 year normalized release and peak dose as a function of neptunium solubility indicate that peak doses and releases increase as solubility increases. However, at times less than 100,000 years, the releases and doses are insensitive to neptunium solubility due to the relatively long transport time of this radionuclide due to its sorption in the geosphere. For this reason, for many of the runs the ^{237}Np solubility was fixed at 1.0 g/m^3, the point where it appears the release becomes essentially alteration limited.

The effects of a capillary barrier that produces a 1.0 m diffusion path length was investigated using both RIP and NEFTRAN-S. The results indicate that providing a diffusive-release barrier significantly reduces the doses and releases. If the diffusion coefficient could be shown to be lower than 1.0×10^{-6} m^2/yr it would allow time for significant decay of ^{237}Np which has a half-life of 2.14×10^6 yrs.

Based on the results of this study the following conclusions can be made:

- Doses to an individual at the accessible environment occur well beyond 10,000 years and do not decline significantly for hundreds-of-thousands of years;
- Peak doses and releases at the accessible environment over the 1,000,000 year period are generally unaffected by waste package lifetimes of up to 100,000 years;
- Dose and release are significantly affected by unsaturated-zone pore velocity;
- Dispersion in the unsaturated zone is more important than that in the saturated zone;
- The wide range of ^{237}Np solubility has a significant effect on the magnitude of the peak dose at times approaching several hundred thousand years; and
- The effects of a diffusion path length formed by a capillary barrier has a significant effect on dose and release even at long times (of the order of hundreds-of thousands of years).

5.4 COMPARISON OF RESULTS TO TSPA-1993

Numerous differences exist between the "simple" analyses reported in this document and the more complete analyses conducted as part of TSPA-1993 (see Andrews et al., 1994). A primary difference is that the TSPA-1993 analyses attempt to present representative analyses based on the best site-specific and design-specific information available at the time, accounting for the uncertainty in the parameters. In addition, the TSPA-1993 analyses attempt to incorporate as much realism in the individual processes potentially affecting the containment and isolation of radioactive wastes in a potential repository at Yucca Mountain, including the thermo-hydrologic regime in the vicinity of the repository as a function of repository loading, time, and location and the effects of the thermo-hydrologic regime on the initiation and rate of waste package degradation, waste form alteration, and release from the engineered barrier system. The TSPA-1993 analyses were performed with a range of possible thermal loads, a range of possible waste package designs, two possible criteria for the initiation of aqueous corrosion, and two conceptual representations of the aqueous corrosion rates. As a result of these differences, making direct comparisons of the results presented herein to the TSPA-1993 analyses is not possible. However, some useful insights as described below can be gained.

The 10,000 year integrated release to the accessible environment determined in TSPA-1993 is totally determined by the release of gaseous ^{14}C. As a result, these results are not at all comparable to the results in this document where it was assumed the ^{14}C was transported in the aqueous phase. In TSPA-1993, for the baseline thermal load and waste package design, the normalized integrated aqueous ^{99}Tc release at the accessible environment varied from 10^{-16} to 10^{-4} for the range of possible percolation fluxes from 5×10^{-4} to 2×10^{-3} m/yr. This is analogous to the results presented here although the dominant radionuclide over this period is the aqueous ^{14}C (because the technetium is retarded by a factor of five). The 100,000 year integrated normalized releases determined in TSPA-1993 range from 10^{-4} to 1 for percolation fluxes which range from

10^{-4} to 2 x 10^{-3} m/yr, which is again very similar to the range of normalized releases determined in the "simple" assessment of post-closure performance presented herein.

The peak doses over a 1,000,000 year period determined in the TSPA-1993 analyses ranged from 10^{-4} to 10 Sv/yr for a range of percolation fluxes from 10^{-4} to 2 x 10^{-3} m/yr. The low doses correspond to low advective velocities in which the peak dose becomes controlled by technetium rather than neptunium. The peak doses calculated in the present "simple" analyses over this same period are on the order of between 0.1 to 0.5 Sv/yr even for the highest neptunium solubilities and advective velocities. These relatively high doses correspond to advective velocities in the unsaturated zone of greater than 0.01 m/yr, neptunium solubilities greater than about 1 g/m^3, and effective diffusion coefficients greater than about 10^{-4} m^2/yr (with or without the presence of a diffusive near field barrier). The difference in the peak doses determined in the two sets of analyses is due to the difference in the mixing depth assumed for the saturated zone transport. In TSPA-1993, 50 m of mixing was assumed to occur, while in the present analyses a 2400 m mixing depth was assumed (corresponding to the entire thickness of the saturated zone aquifer). This factor of 50 difference in the mixing depth relates to a factor of 50 difference in concentration and dose. For low neptunium solubilities or low advective velocities in the unsaturated zone or diffusion releases through the engineered barrier, the peak doses in the 1,000,000 year period become controlled by technetium rather than neptunium.

In summary, although significant differences exist in these two sets of analyses, for the closest comparable thermal load and waste package design, the results are very similar. Even for the more robust packages assumed in the TSPA-1993 analyses (such as the 45 cm overpack of corrosion allowance material), the very long term peak doses are analogous to the results presented herein for comparable alternate waste package failure distributions.

5.5 SIGNIFICANCE OF RESULTS TO STANDARD SETTING

The results of the calculations of potential population in the vicinity of Yucca Mountain based on available ground-water supplies in the sub-basins that encompass the site indicate that populations may be small (Appendix A). When it is assumed that the available ground water is used for household use only, the size of the population ranges from about 1,200 to 13,000 persons. When a farming scenario is assumed (i.e., a farm family that uses ground water for household use and for raising all their food supply) the population ranges from about 14 to 150 persons. It is unlikely that such a farm family exists in the vicinity of Yucca Mountain (the maximally exposed individual). Under other scenarios, such as importing water for household use or farming, the population could be larger, but the imported water would dilute the local ground water so individual doses would be lower, and population doses would be about the same.

The results of analyses of two representative uranium ore bodies, in oxidizing and reducing environments, indicates that doses from drinking water 5,000 m down gradient from the ore bodies range from 400 to 30 mrem/yr (Appendix B). The number of health effects integrated over 10,000 years from drinking water for the two repository equivalent (100,000 MTHM) ore bodies ranges from about 17,000 to 2,000 for oxidizing and reducing conditions, respectively. This can be compared to the basis for the EPA Standard which is 1,000 health effects integrated over 10,000 years for 100,000 MTHM. The study also found that the integrated releases from the ore bodies is lower than the EPA release limit. In evaluating this apparently contradictory

result, the scenario upon which the EPA Standard is based must be considered (i.e., ground-water flow to an average world river which flows to the ocean). If a new standard were to be based on the premise that a repository should produce no more risk than that from the unmined uranium from which the waste was derived it appears that dose to an individual could be 100 mrem/yr or higher and that integrated health effects over 10,000 years could be considerably higher than the 1,000 used by the EPA as the basis for 40 CFR 191.

One of the important observations that can be drawn from the sensitivity analyses presented in Chapters 3 and 4, is that the relative response of the repository system appears to be the same whether one is comparing integrated releases or individual peak doses. This can best be observed by examining the sensitivity of both the integrated release and peak dose performance measures to the unsaturated zone interstitial velocity (or percolation flux) for the different periods. In the RIP analyses, the 10,000 year integrated release varies by 12 orders of magnitude (from 10^{-12} to 10^{0} normalized to the Table 1 values in 40 CFR 191) for unsaturated zone velocities which vary by two orders of magnitude (from 10^{-3} to 10^{-1} m/yr). In addition, the peak individual dose at 10,000 years varies by 12 orders of magnitude (from 10^{-14} to 10^{-2} Sv/yr) for the same two order of magnitude variation in unsaturated zone velocity. Similarly, the 100,000 year normalized integrated release varies by two orders of magnitude (from 10^{-2} to 10^{0}) as does the peak dose for this same period (from 10^{-4} to 10^{-2} Sv/yr) for the same variation in the unsaturated-zone velocity. The same trend generally holds for the 1,000,000 year period (with the normalized integrated release varying by a factor of 10 and the peak dose by a factor of 100 over the range of sampled unsaturated-zone velocities), although the results are complicated by the addition of the near-field diffusion and neptunium solubility which also affect the range in the calculated performance measures. Similar observations are made with the NEFTRAN-S model, with the 100,000 year normalized integrated release varying by three orders of magnitude (from 5×10^{-3} to 5) and the peak dose over this same period varying almost three orders of magnitude (from less that 10^{-4} to 3×10^{-2} Sv/yr) for the two order of magnitude range of sampled unsaturated-zone velocities.

It is not intuitively obvious that the above relationships should have been observed. The integrated release performance measure (based on the desire to protect potentially exposed populations over extended periods of time) effectively averages the release over time, while the peak individual dose (based on the desire to protect the potentially maximally exposed individual or some average individual of a potentially exposed "critical" group) effectively looks at the peak dose or concentration no matter when it occurs in the period of interest. It is certainly conceivable to imagine cases where there is little correlation between the desires to protect populations versus individuals. One excellent example would be the difference between the releases of ^{14}C in the gaseous phase versus the individual doses due to drinking water consumption along potential aqueous radionuclide transport pathways. Another example would be the very-long term (about 1,000,000 year) peak dose potentially associated with high solubility neptunium releases versus the integrated release prior to the arrival of this peak. In other words the correlation between these two performance measures observed in the analyses presented in Chapters 3 and 4 may more reflect the integration period rather than be a general relationship that holds under all conceivable periods of potential concern.

In order to test the hypothesis that there is a very high correlation between the peak dose and normalized cumulative release performance measures, cross plots of the two performance measures have been prepared for the three RIP analyses corresponding to the 10,000, 100,000,

and 1,000,000 year simulations where the unsaturated zone velocity, the waste package diffusion coefficient, and neptunium solubility are sampled for 100 realizations. The three cross plots are illustrated in Figures 5-1 to 5-3, for 10,000 years, 100,000 years and 1,000,000 years, respectively. Figure 5-1 shows a remarkable correlation between the two performance measures over the entire range of possible releases and peak doses for the 10,000 year period. The degree of correlation shown in Figure 5-1 appears to indicate that a standard could be written either in terms of individual dose or cumulative release. Figure 5-2 shows a very high correlation between the two performance measures for the 100,000 year period, except for the slight dip in the middle of the plot (probably due to the fact that the cumulative release can continue to increase after the peaks have passed, i.e., mass continues to be released at the accessible environment even after the peak concentration has passed the same point) and the two high dose outliers associated with very high neptunium solubilities and high unsaturated zone velocities. Figure 5-3 also shows a high correlation between the two performance measures with the few points off the log-log linear trend being the result of both a high diffusion coefficient and a high neptunium solubility. The steepness of the slope in Figure 5-3 represents the significance of the neptunium solubility and the fact that the cumulative release is dominated by releases of all radionuclides over the 1,000,000 year period, while the peak dose is controlled by the neptunium dose which generally occurs between 200,000 and 500,000 years.

It is important to remember that the above correlation between integrated release and peak dose may only hold in a relative sense, as the absolute relationship will depend entirely on the definition of the biosphere conditions, including the primary pathways by which the radionuclides are ingested and the definition of the exposed population and the maximally exposed individual or average individual of the "critical" portion of the exposed population. If the population of concern in an integrated release performance measure is the same as the "critical" population and the dose is applied to the average individual of the same "critical" population, then the results should be analogous, with the caution that the actual period selected may lead to different release integrals than dose peaks.

From the analyses of the potential populations that may be sustainable on the available ground water (Appendix A) and the sensitivity analyses of possible post-closure consequences (release or dose, Chapters 2 through 4) associated with a potential repository at Yucca Mountain, the following conclusions may be useful in the development of standards can be made:

- There is little ground-water available to support large populations in the vicinity of Yucca Mountain, and there is little water for dilution of releases from a potential repository;
- Releases from a repository at Yucca Mountain may begin after a few tens of thousand years and extend to times over one million years;
- Short-term releases (less than 10,000 years) could be influenced by long-lived waste packages (i.e., lifetimes of several thousand years);
- Mid-term releases (10,000 to 100,000 yrs) are influenced by diffusion out of the waste package or very long-lived packages; and

- Long-term doses (beyond 500,000 yrs) from ^{237}Np can only be affected by an engineered capillary barrier that creates a diffusion-controlled release from the engineered barrier system or a highly sorptive low solubility near field geochemical environment..

5.6 SIGNIFICANCE OF RESULTS TO SITE CHARACTERIZATION AND DESIGN

Based on the results of this study, the following conclusions can be related to characterization activities that would aid performance assessment. In addition, the results have implications for repository design.

- Additional research concerning the solubility of ^{237}Np under anticipated near field thermo-hydrologic-geochemical conditions would reduce the uncertainty of performance assessment calculations, if the range of measured solubilities can be narrowed;

- The pore velocity in the unsaturated zone is an important parameter to performance assessment (as was already known), and uncertainty in performance assessment calculations could be reduced if the range of percolation flux could be reduced;

- Dispersivity in the unsaturated zone is far more important than dispersivity in the saturated zone, with the dispersivity being very important if the period being considered is on the order of about half of the retarded radionuclide travel time (for example, if the time considered is 10,000 years and the retarded travel time is about 20,000 years, then the dispersivity will control the arrival of the radionuclide);

- Additional research on the coefficient of diffusion in capillary barriers could provide important information that would significantly lower the calculated doses at the accessible environment, even at very long times (beyond 1,000,000 years);

- The waste package lifetime has little effect on dose and release over very long periods (on the order of a million years).

- Waste form alteration/dissolution rate information used in the analyses that were conducted have been based on the value used in the WISP study (NAS, 1983). This value has been substantiated by recent flow-through tests conducted by DOE contractors (see Andrews et al., 1994). This value may be very conservative given the possible limitation on dissolution caused by the lack of available chemical reactants at the waste form-water contact. More realistic (quasi-static) tests may yield substantially lower alteration/dissolution rates.

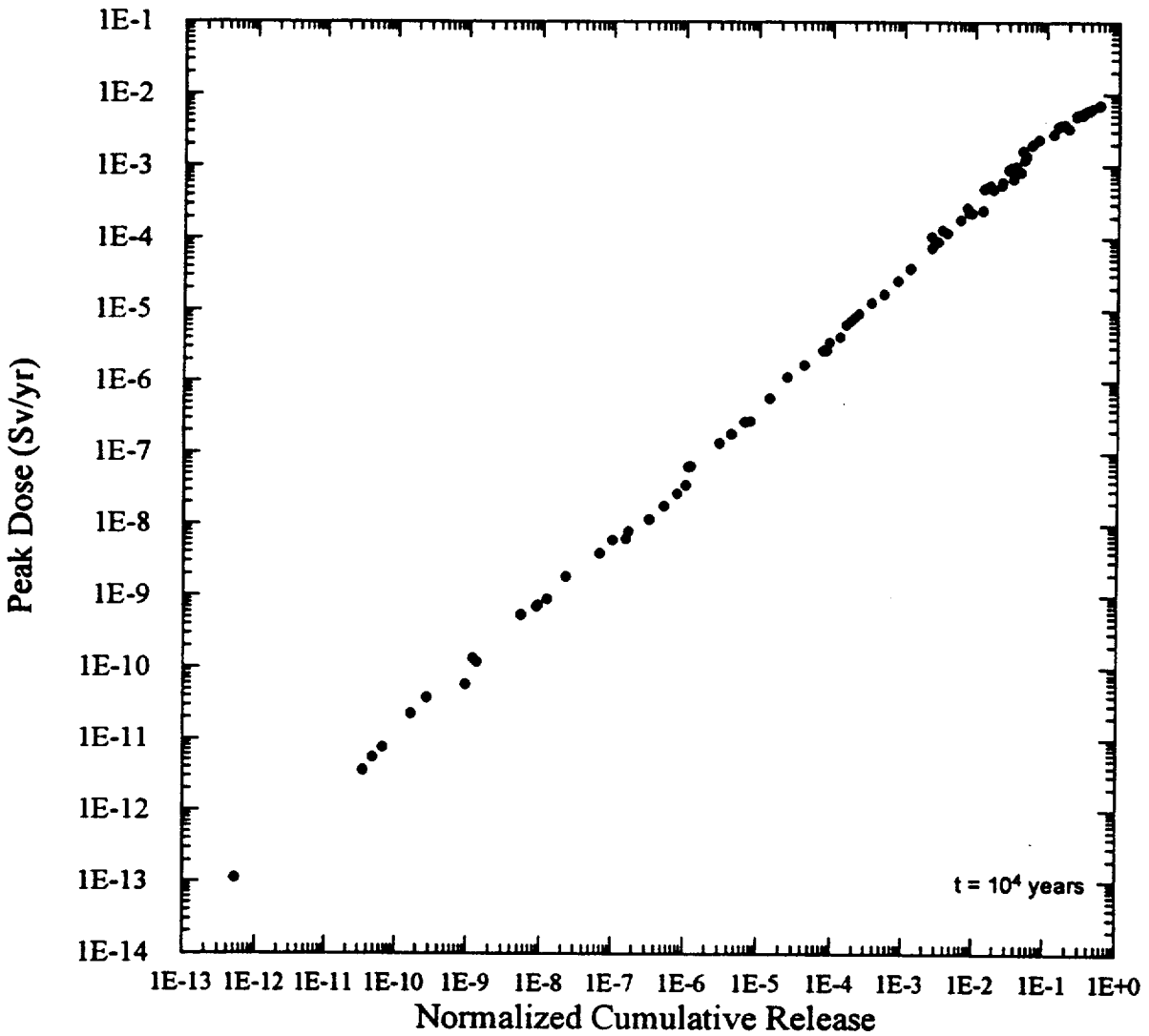


Figure 5-1. Correlation Between Peak Dose and Normalized Cumulative Release Over 10,000 Years

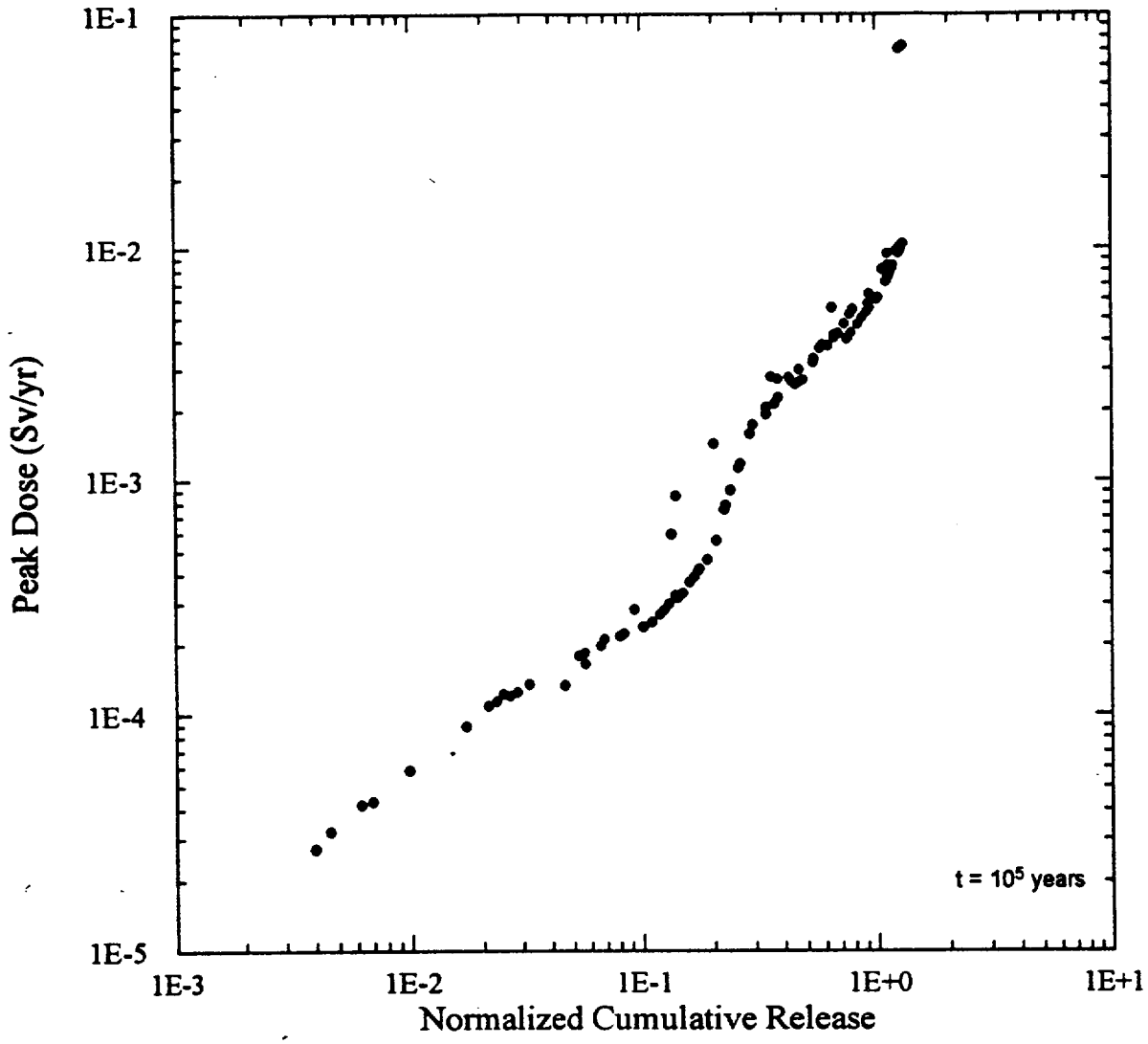


Figure 5-2. Correlation Between Peak Dose and Normalized Cumulative Release Over 100,000 Years

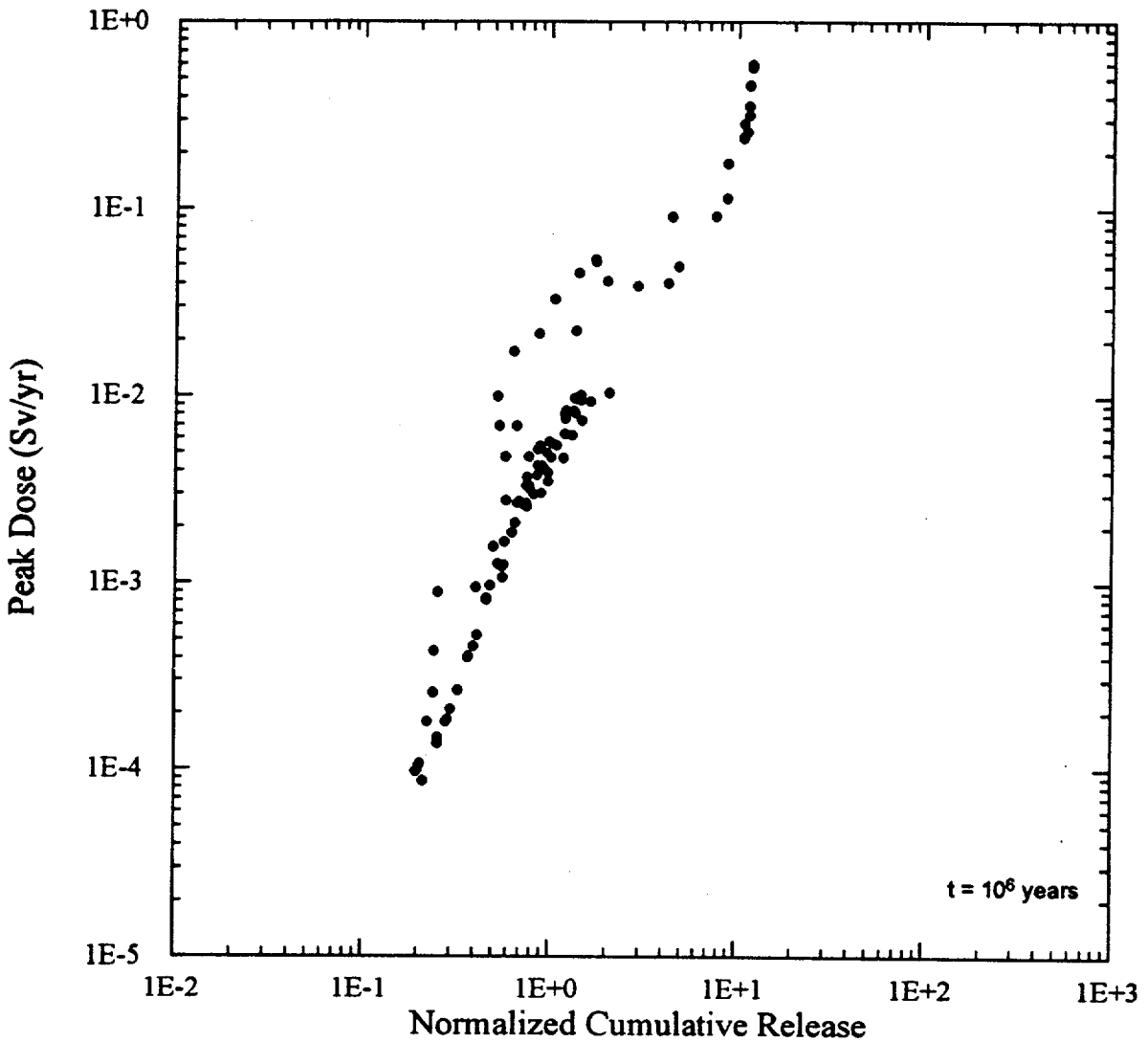


Figure 5-3. Correlation Between Peak Dose and Normalized Cumulative Release Over 1,000,000 Years

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APPENDIX A

POTENTIAL POPULATION IN THE VICINITY OF YUCCA MOUNTAIN

INTRODUCTION

The population basis for a repository standard should be the group of people who could reasonably be exposed to a radioactive release from the repository. The population exposed is directly related to site specific features that in part control the number and the location of people that are exposed to a potential release of radionuclides. For example, if the release is to a large river that discharges into the ocean the release could be to the entire world population. At a site in an arid region the population exposed is those people using ground water for household use and/or crop production. Thus, the type of site and its proximity to surface water bodies could control the number of people that can be exposed. The approximate number of people who could be exposed at a site should be a consideration in the process of standard development.

POPULATION BASED ON AVAILABLE WATER

In arid regions there are few surface water bodies and the primary source of water for household use and farming is from ground water. The exceptions to this are surface streams such as the Colorado River which originate in areas of higher precipitation. In the vicinity of the Yucca Mountain site there are only ephemeral streams and playa lakes, and these only have water after significant storm events. These surface water features could not be regarded as a source of domestic water because of their sporadic nature. Thus, the only source of water is from infiltration of precipitation that recharges the basin aquifers in the vicinity of Yucca Mountain. The amount of ground water available on a continuous basis can be used as a basis for estimating the size of a population that could exist in the vicinity of the Yucca Mountain site at a future time. This ground water in the down-gradient flow direction from the potential repository would be the primary source of dose to the population using this water if radionuclides are released to the aquifer.

The hydrologic subbasins in the vicinity of the Yucca Mountain site are shown in Figure A-1. The recharge to the Buckboard Mesa subbasin enters the ground-water flow system where it flows southward and could potentially receive radionuclides released from the potential repository which lies on the boundary between the Crater Flat and Jackass Flats subbasins (Figure A-1). The ground-water recharge to the three subbasins is presented in Table A-1. The annual recharge of the three subbasins is approximately 2200 acre-feet (Czarnecki, 1985). The southward flow from Buckboard Mesa and Jackass Flats of 8000 acre-feet (outflow, Table A-1) is thought to be due to over-pumping in support of agriculture in the Amargosa Desert which lies to the south of the two subbasins. It can be argued that the safe yield of the three subbasins (Crater Flat, Buckboard Mesa, and Jackass Flat) is approximately 200 acre-feet [lateral inflow + recharge - lateral outflow = safe yield (6000 + 2200 - 8000 = 200, see Table A-1)]. This means that the amount of water that is currently available without permanently lowering the ground-water table (depleting the resource) is 200 acre-feet, with the remaining 2000 acre-feet of recharge already being used farther to the south in the Amargosa Desert. Thus, in the vicinity of Yucca Mountain the amount of ground water available lies between the annual safe yield (200 acre-feet, 2.5×10^8 l) and the annual recharge (2200 acre-feet, 2.7×10^9 l). The latter value would be appropriate if the over-pumping in the Amargosa Desert were neglected. The higher value of available water could be tapped up-gradient from its location of current use at a location nearer the Yucca Mountain site.

Water use by a population is dependent on the type of use and the climate. In arid climates the *per capita* water use for domestic purposes tends to be higher than in a more temperate climate. For example, for daily household use the cities of Tucson and Phoenix report 160 gal/person and 267 gal/person, respectively. A conservative value, maximizing the potential population, of 150 gal/person/day (2.1×10^5 l/person/yr) for household use will be assumed. Farming requires about 20,000 m^2/person in an arid climate (similar to Richland, Washington), the irrigation rate is about 150 l/m^2/mo, and the growing season is assumed to be 6 mo in the Richland area (Eslinger et. al., 1993). This yields a water use of 1.8×10^7 l/person/yr. This value is conservative for Yucca Mountain in that it will yield a higher population (e.g., irrigated farming near Yucca Mountain would require somewhat more water).

The number of persons in the population that could potentially receive doses from releases of radioactivity from the repository can be estimated by assuming the type of ground-water use that exists, either household or household plus irrigated farming and then dividing the volumetric use rate into the annual water available. This approach produces a range of populations for each of the two use scenarios because the volume of water available ranges from the annual safe yield to the annual recharge. These ranges in population are shown in Table A-2.

AN ALTERNATIVE METHOD BASED ON CURRENT POPULATION

Another way to approach a determination of the population would be to base it on the population that currently exists near Yucca Mountain. The population would be made up of only those individuals who could potentially be affected by the repository, that is, those living in the immediate vicinity of the repository and those that use the tuff aquifer for their major source of domestic water supply. The "vicinity" of the repository can be defined by several different approaches. One means would be to define "vicinity" by a circle with a radius equal to the expected transport pathway length from the repository to either the point of ground-water discharge or to the point where radionuclides decay to levels below regulatory concern. This approach recognizes that the primary mechanism for radionuclide release from a geologic repository is by way of the ground-water pathway. The population could then be defined as the number of people that could subsist within the "vicinity" of the repository based on land use patterns that currently exist in the defined area. The dose calculations for this population would be based on today's society; with the standard individual conducting the normal activities of today's society (i.e., showering, gardening, farming, eating habits, water consumption, etc.). This produces a hypothetical individual living in the population who may receive doses in the rem range (i.e., an individual subsisting entirely on ground water for household use and food production) at long periods of time after repository closure. This is a worst case scenario for a maximally exposed individual, it yields high calculated doses because the hypothetical scenario of water use is unrealistic. In order to be more realistic, a critical group within the population could be defined based on current human activities within the "vicinity" of the repository. The representative individual for regulatory purposes, would live within this population, and would be a member of this critical group.

SUMMARY

The results of the calculations of potential population in the vicinity of Yucca Mountain based on available ground-water supplies in the sub-basins that encompass the site indicate that

populations may be small. When it is assumed that the available ground water is used for household use, the size of the population ranges from about 1,200 to 13,000 persons. When a farming scenario is assumed (i.e., a farm family that uses ground water for household use and for raising all their food supply) the population ranges from about 14 to 150 persons. It is unlikely that such a farm family exists or will exist in the vicinity of Yucca Mountain (the maximally exposed individual). Under other scenarios, such as importing water for household use or for farming the population could be larger, but the imported water would dilute the local ground water and population doses would be about the same.

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Table A-1. Annual Recharge to Hydrologic Subbasins in the Vicinity of the Yucca Mountain Site

SUBBASIN	LATERAL INFLOW (Acre-Feet)	LATERAL OUTFLOW (Acre-Feet)	VERTICAL RECHARGE [1] (Acre-Feet)	VERTICAL RECHARGE [2] (Acre-Feet)
Buckboard Mesa	} 6000	} 8000	} 2000	1400
Jackass Flats				580
Crater Flat	2000	2000	0	220

(1) Harrill et al. (1988)
(2) Czarnecki (1985)

Note: Conversion factor one acre-foot = 1234 m^3

Table A-2. Size of Population Based on Use of Available Ground Water

TYPE OF WATER USE	VOLUME (Liter/Year/Person)	WATER AVAILABLE (Acre-Feet)	POSSIBLE POPULATIONS (Persons)
Household Use	2.07×10^5	200 - 2200	1,200 - 13,000
Household & Farming Use	1.80×10^7	200 - 2200	14 - 150

Note: Conversion factor one acre-foot = 1234 m^3

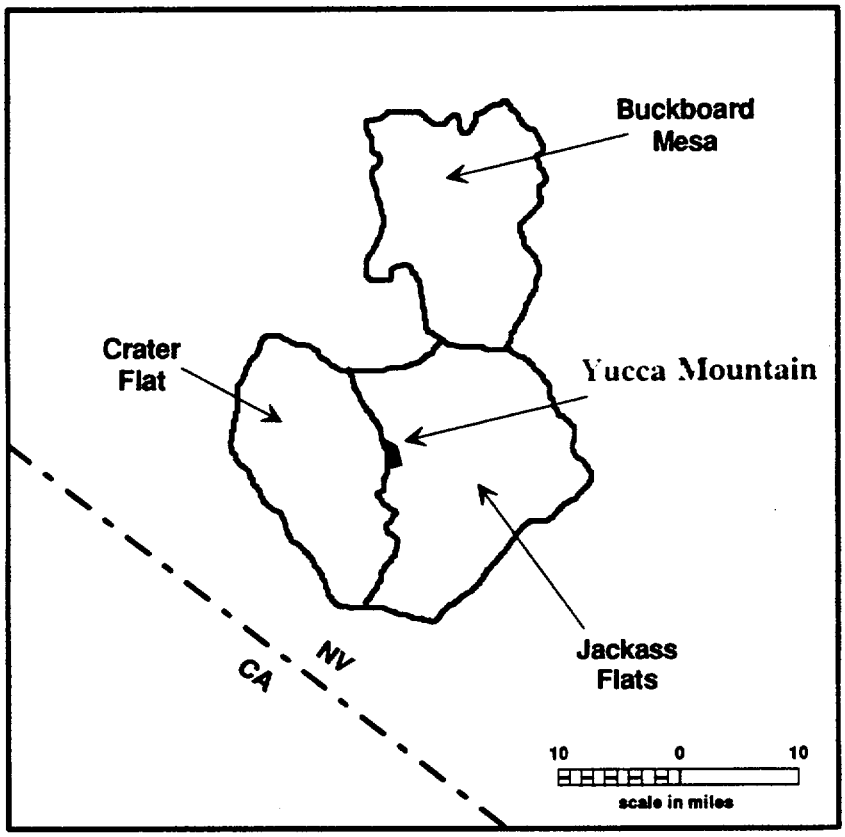

Figure A-1. Location of Hydrologic Subbasins in the Vicinity of the Yucca Mountain Repository.

APPENDIX B

POTENTIAL EFFECTS OF URANIUM ORE BODIES

INTRODUCTION

The development of regulations by the EPA for the management and disposal of high-level radioactive wastes has had a long history which extends over nearly two decades. The standard which was first promulgated in 1985 (EPA, 1985b) was begun in the late 1970's based on the premise that a geologic repository should pose no more risk to the public than that from an equivalent unmined uranium ore body. For this purpose an equivalent uranium ore body was defined as the amount of uranium ore necessary to produce the reactor fuel that would be placed in a 100,000 metric ton heavy metal (MTHM) repository (EPA, 1980). By the time the standard was promulgated, the EPA had decided not to rely on the comparison between spent fuel and the unmined uranium ore from which the spent fuel was derived, although the comparison was cited in the supporting documentation (EPA, 1985a). As a result, the Standard promulgated was more restrictive than one based on unmined ore by more than an order of magnitude (Klett, 1991).

The basic EPA premise that a geologic repository should produce no more risk than the uranium ore from which the waste came is a valid premise. For example, if the standard is more restrictive than this basis, then the repository could be viewed as a means of remediation of the effects of natural ore bodies. Therefore, it would seem prudent that the risk from the repository should be no more restrictive than that of the unmined uranium ore. This approach to the development of environmental standards would also follow the concept of not passing increased risk from the present generation to future generations.

DEVELOPMENT OF A REPRESENTATIVE ORE BODY

For the purposes of this study, two representative uranium ore bodies of equal size were developed. One was assumed to be in an oxidizing environment and the other was assumed to be in a reducing environment. Uranium and its daughter products were assumed to be in equilibrium in the ground water of both ore bodies. The concentration of dissolved uranium in each environment was derived from the literature discussed below.

Concentrations of natural radionuclides in ground water have long been recognized as a source of dose to humans. The EPA has set limits on the concentrations of natural radionuclides in public water supplies, 40 CFR 141 (EPA, 1986). The EPA states, in the preamble of this proposed rule, that the data base of the U. S. Geological Survey National Uranium Resource Evaluation (NURE) Program lists concentrations of uranium up to 600 pCi/l (approximately 1,800 ppb) in ground water with only a few samples of the 34,000 analyzed exceeding 50 pCi/l (150 ppb) (EPA, 1986). The NURE Program was conducted by sampling waters across the United States on an approximate grid system in an attempt to quantify the national uranium reserves. The higher concentrations were generally associated with uranium ore bodies.

Samples of standing water in the open pit of the Jackpile mine in New Mexico were found to contain concentrations of uranium of about 560 ppb (Kaufmann et. al., 1975). In shallow ground water and seeps Judson and Osmond (1955) reported concentrations of 100 to 460 ppb near ore deposits in Colorado. In Wyoming, Murphy (1956) reports concentrations of uranium that are up to 340 ppb in spring discharge. The target cleanup concentrations for uranium in ground water at the Irigaray solution mine in Wyoming is 1.0 ppm (1,000 ppb) (Wyoming Minerals Corporation, 1977). All of the these values for uranium concentrations are assumed to be for

oxidizing conditions because of the proximity of the waters to the land surface. In addition, the high concentration of 1,800 ppb from the NURE Program could only have been under oxidizing conditions.

Under reducing conditions, Baker and Scott (1958) report concentrations of uranium up to 46 ppb in Texas, and Harshmann (1968) reports concentrations in ground waters in Wyoming that range from 10 to 20 ppb. Judson and Osmond (1955) report concentrations in Texas of 2 to 10 ppb.

From these literature values it would appear that the higher concentrations of uranium occur under oxidizing conditions at or near the ground surface in ponds and streams, in shallow ground water, and in seeps and springs. Higher concentrations are also observed on the up-gradient side of ore bodies where dissolution is occurring under oxidizing conditions (Wyoming Minerals Corporation, 1977; Baker and Scott, 1958; Harshmann, 1968; and Capuano, 1978). Lower concentrations occur within the ore body and down-gradient from it where conditions are reducing.

In the EPA analysis of actual ore bodies, Williams et al. (EPA, 1980) show concentrations of uranium in ground water at mines in Wyoming that range from 100 to 200 ppb. However, Williams only considers the dose from radium in his analysis of health effects, and the generic ore body that he used had a uranium concentration that was significantly lower than that of the actual ore bodies (1.0 ppb). Wick and Cloninger (1980), in their comparison of geologic repositories to natural uranium ore bodies, report uranium concentrations for creeks and springs near ore deposits that range from 150 to 800 ppb. These concentrations are considered to be under oxidizing conditions. Wick and Cloninger (1980) provide an excellent review of uranium concentrations in surface and ground waters of the United States. They also provide calculations of dose to an individual from these uranium ore deposits.

Based on this review of the literature, the concentrations of uranium dissolved in the ground water under oxidizing conditions ranges from about 100 ppb to 1,800 ppb with several high occurrences that range from 800 to 1,000 ppb. The concentration of uranium under reducing conditions is more difficult to determine because it grades into values that are truly oxidizing. The range of concentrations of uranium in ground water of ore bodies under reducing conditions was found to be 2 ppb to 46 ppb. The values for dissolved ^{238}U in the ground water for oxidizing and reducing conditions selected for this study are 500 ppb and 20 ppb, respectively. These values are near the middle of the range of concentrations observed in each environment (oxidizing and reducing).

The size and hydrogeologic characteristics of the two representative uranium ore bodies were assumed to be identical. The hydrogeologic and geometric parameters as well as the uranium concentration in the ore were taken from the study by Williams et al. (EPA, 1980), because his choice of these parameters appears to be appropriate based on the literature. Williams analyzed four ore bodies located in Wyoming, which had ground-water velocities that were somewhat higher than his generic ore body. The velocity he selected was based on the analyses for developing the EPA standard. The characteristics of the ore bodies are presented in Table B-1. The pore velocity of 2.1 m/yr is similar to that in the saturated zone at Yucca Mountain which is of the order of 1.0 m/yr. The dimensions of the ore body were selected by Williams based on the four actual ore bodies he examined. The concentration of 0.09% U_3O_8 compares favorably

with the average of 0.07% for uranium ore reserves. Williams goes on to show that 620,000 metric tons of U_3O_8 are required to produce the fuel that will be disposed of in a 100,000 metric ton repository. Therefore, the factor necessary to convert the representative ore body depicted in Table B-1 to an ore body that is equivalent to a high-level waste repository is 62.

The radionuclides in the uranium decay chain are ^{238}U, ^{234}U, ^{230}Th, ^{226}Ra, and ^{210}Pb. The retardation factors for these radionuclides along the ground-water transport pathway under reducing conditions were taken from the NAS WISP report (NAS, 1983). For oxidizing conditions the retardation factor for the isotopes of uranium was reduced by a factor to approximate the difference between ore body and spring discharge concentrations (approximately a factor of eight based on the higher end of the concentrations under reducing conditions). The retardation factors for the radionuclides in the uranium decay chain are presented in Table B-2. In EPA's study of uranium ore bodies (EPA, 1980) higher values of retardation coefficients for uranium and thorium are used, and the dose from lead was not considered (Table B-2). These high retardation values allowed the EPA to neglect doses from all radionuclides except radium. These assumptions cause the calculated doses to be considerably lower than if all of the radionuclides were considered (see the discussion that follows).

DOSE TO AN INDIVIDUAL FROM DRINKING WATER

For the uranium ore body calculations, the radionuclides of the uranium decay chain that are dissolved in ground water were assumed to be in equilibrium with ^{238}U (500 ppb and 20 ppb for oxidizing and reducing environments, respectively). Based on this assumption the concentration of radionuclides in the ore body were calculated. These concentrations are shown in Table B-3.

The concentration in the direction of ground-water flow down-gradient from the uranium ore body was determined using the transport model UCBNE-41 (Lung et al, 1987). This model is based on the analytical solution of the transport equation where dispersion, retardation, and chain decay of radionuclides is considered. The concentrations presented in Table B-3 and the flow parameters presented in Table B-1 were used to calculate down-gradient concentrations of the five radionuclides in two separate model runs; one for oxidizing conditions and one for reducing conditions. The radionuclide concentrations were calculated for a point 5,000 meters down-gradient from the two ore bodies in order to yield results that can be compared to the releases from a geologic repository (e.g., the accessible environment is assumed to be 5,000 meters from a geologic repository). The concentrations of dissolved radionuclides at this point were calculated after the releases had reached steady state in order to be comparable to a natural ore body of considerable age (in which releases have taken place over geologic time scales). The concentrations in ground water 5,000 meters down-gradient in the direction of ground-water flow for oxidizing and reducing conditions are presented in Table B-4. The doses to an individual drinking 700 liters of ground water from a well (located 5,000 meters down the flow gradient from the ore deposit) are presented in Table B-5. The dose conversion factors used in the calculation and presented in Table B-5 are the highest values from among those used by the EPA (EPA, 1988), the DOE (DOE, 1988), and the NRC (NRC, 1981). These calculations indicate that the annual dose to an individual from drinking ground water at a location 5,000 meters down-gradient from a uranium ore body is 320 and 39 millirems for ore bodies located in oxidizing and reducing environments, respectively.

DOSE TO MAXIMALLY EXPOSED INDIVIDUAL

The worst case dose to an individual, as described in Appendix A, is that of a person living 5,000 meters in the direction of ground-water flow from the uranium ore body who uses well water for household use and farming. The dose to this maximally exposed individual can be obtained by multiplying the radionuclide concentrations in the well water at this location by an appropriate dose conversion factor that accounts for the household use/farming scenario. The dose conversion factors were taken from the WISP report (NAS, 1983), and were converted to the units shown in Table B-6. The dose to an individual that arise from household use and food production using ground water at a point 5,000 meters down-gradient from the ore body range form approximately 560 to 170 mrem/yr for oxidizing and reducing conditions, respectively (Table B-6).

DOSE TO A POPULATION FROM DRINKING WATER AND FARMING

The dose to a population from drinking water can be estimated once the population has been determined. The population can be calculated using two approaches. The first approach assumes that the population that could receive dose from drinking water is assumed to be the number of persons that could use all of the ground water flowing through the repository-equivalent ore body for household use. The second approach assumes that all of the ground water flowing through the repository-equivalent ore body is used for drinking water only. The second approach will yield a larger population but, it is thought to be unrealistic and the first approach will be used.

The first approach (ground water used for household use and dose from drinking water) neglects doses that would be received through bathing, inhalation, etc. The volume of water flowing through the ore body is 3.50×10^4 m^3/yr (2.1 x 0.15 x 30 x 3700), and the volume of water flowing through the repository-equivalent ore body is 2.17×10^6 m^3/yr (3.50×10^4 x 62). Assuming that the average person uses 150 gal/day (2.07×10^2 m^3/yr) the population can be estimated to be 1.05×10^4 persons. The dose to the population is calculated to be 3.36×10^3 and 4.10×10^2 person-rem/yr for oxidizing and reducing conditions, respectively.

If a farming scenario had been assumed for water use the population would have been approximately 120 persons (water use of 1.8×10^4 m^3/yr/person, Appendix A). The total dose to the farming population from household use and food production (Table B-6) is calculated as 66 and 20 person-rem/yr for oxidizing and reducing conditions, respectively.

HEALTH EFFECTS AND COMPARISON WITH THE EPA STANDARD

The EPA Standard was based on the number of health effects (1,000) that were allowed to occur over a 10,000 year period from a 100,000 MTHM repository. The release limits were derived assuming release to an average world river that transported the radionuclides to the world population. In this section the number of health effects from a uranium ore body is compared to the basis for the EPA Standard. The discharge from the ore body is assumed to be released into the EPA river scenario and the health effects are calculated. In addition the releases from the ore body are compared to the EPA release limits.

The number of health effects can be estimated using the conversion factor developed by EPA which is 500 health effects for 10^6 person-rem. In order to be comparable to a repository, the health effects are integrated over 10,000 years. For this calculation the dose is from drinking water to the population that could use all of the water flowing through the ore body for household use (a population of 10,500 persons and total dose from Table B-5). This calculation yields 1.68×10^4 and 2.05×10^3 health effects from drinking water over the 10,000 year period for oxidizing and reducing conditions, respectively. For the farming scenario the number of health effects are greatly reduced because of the significantly lower population. Had the population been calculated assuming that all of the water was consumed by drinking only, the number of health effects would have been considerably higher (as discussed earlier, this appears to be an unrealistic scenario).

The EPA assumed that the release from the repository flows into a large river (5,000 meters down-gradient from the repository) which distributes the dose to a much larger population. Using the EPA conversion factors for health effects for the EPA river scenario the number of health effects can be determined. Table B-7 shows that the approximate number of health effects for the EPA river scenario, from a repository-equivalent uranium ore body is 208 and 24 over 10,000 years for oxidizing and reducing conditions, respectively [integrated release over 10,000 years (Table B-8) x EPA dose conversion factor]. This low value of health effects is a direct result of dilution in the river.

The integrated release from a repository-equivalent uranium ore body is calculated by multiplying the radionuclide concentration (Ci/m^3) by the volume of ground water flowing through the repository-equivalent ore body over a 10,000 year period (2.17×10^{10} m^3). This calculation is presented in Table B-8 where the concentration of the uranium decay chain radionuclides at a point 5,000 meters down-gradient from the ore body are expressed in units of Ci/m^3. These releases can be compared to the EPA Standard Table 1 release limits from a repository (40 CFR 191) (EPA, 1985b), which are expressed per 1,000 MTHM. Thus, the values in Table B-8 must be divided by a factor of 100 because they are for a repository-equivalent uranium ore body (100,000 MTHM). The EPA limits for total uranium are 100 Ci; for ^{230}Th, 10 Ci; and for ^{226}Ra, 100 Ci for each 1,000 MTHM. These values are compared to 74 Ci total uranium, 0.04 Ci ^{230}Th, and 0.4 Ci ^{226} Ra released from an equivalent uranium ore body under oxidizing conditions. This indicates that the equivalent uranium ore body is releasing radionuclides slightly below the EPA limit, yet the dose to an individual is in the 300 to 500 mrem/yr range (see Tables B-5 and B-6). This difference stems from the release standard having been derived through dilution in an average world river and the uranium ore body having an assumed hydrology similar to typical ore bodies of the arid western United States.

Had the population been based on drinking water only by assuming an annual consumption of 700 liters per person, the population would be calculated to be 3.1×10^6 persons. The health effects integrated over 10,000 years from the equivalent ore body under oxidizing conditions would have been 4.96×10^6. This value is 49,600 health effects over 10,000 years as compared to the basis for 40 CFR 191 which is 1,000 health effects over 10,000 years. This is very unrealistic because it is unlikely that a population would drink all of the water flowing through the ore body. When the dose is assumed to be from drinking water and the population is based on a household use scenario (150 gal/day/person) the estimated number of health effects for the

equivalent ore body under oxidizing conditions is 16,800 over 10,000 years (2,050 for reducing conditions). This is believed to be a more realistic value than that based on drinking water only.

SUMMARY

Two representative uranium ore bodies were developed based on a review of the literature They were located in chemically oxidizing and reducing environments with 500 and 20 ppb ^{238}U dissolved in the ground water of the ore body, respectively. The daughter products of ^{238}U were assumed to be in equilibrium with the parent within the ore body. Dose to an individual was calculated and a comparison of the effects of the ore body with the EPA Standard was made for a point 5,000 m down the ground-water gradient from the ore body. This point is equivalent in position to the accessible environment defined for a geologic repository. These analyses produced the following results:

- The dose to an individual from drinking water ranges from 320 to 39 mrem/yr for oxidizing and reducing conditions, respectively.

- The dose to an individual from household use and farming ranges from 560 to 170 mrem/yr.

- If all of the ground water flowing through the ore body is used for household use and the dose is considered to be from drinking water only; the number of health effects integrated over 10,000 years ranges from 16,800 to 2,050 for oxidizing and reducing conditions, respectively. This is compared to the basis for the EPA Standard of 1,000 health effects.

- The releases integrated over 10,000 years from the repository-equivalent uranium ore body per 1,000 MTHM, for oxidizing conditions, are 74 Ci total uranium, 0.04 Ci ^{230}Th, and 0.4 Ci ^{226}Ra. The EPA release limits are 100 Ci total uranium; 10 Ci ^{230}Th; and 100 Ci ^{226}Ra.

- This indicates that the EPA Standard is more stringent in terms of health effects and less stringent in terms of release than the analog of a natural uranium ore body.

The scenario upon which the EPA Standard is based must be considered when viewing these results (i.e., ground-water flow to an average world river which flows to the ocean). this scenario is in contrast to the uranium ore bodies with assumed hydrology typical of the arid west where they occur. However even with these differences, if a new standard were to be based on the premise that a repository should produce no more risk than that from the unmined uranium from which the waste was derived it appears that dose to an individual could be 100 mrem/yr or higher, and that integrated health effects over 10,000 years could be considerably higher than the 1,000 used by the EPA for the basis for 40 CFR 191.

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Table B-1. Geohydrologic and Size Characteristics of a Representative Uranium Ore Body

Hydraulic conductivity	1.0×10^{-4} cm/sec
Gradient	0.01
Porosity	0.15
Pore velocity	2.1 m/yr
Thickness	30 m
Width	3,700 m
Length	50 m (in the direction of flow)
U_3O_8 concentration	0.09 %
Host rock density	2.0 g/cm^3
Reserve	10,000 metric tons U_3O_8
Repository conversion factor	62

Table B-2. Retardation Factors (R_d) for Radionuclides in the Uranium Decay Chain for Oxidizing and Reducing Conditions

RADIONUCLIDE	**R_d (EPA, Williams et. al.[1])**	**R_d (Oxidizing, by ratio using spring concentrations)**	**R_d (Reducing, WISP[2])**
^{238}U	14,300	5	40
^{234}U	14,300	5	40
^{230}Th	50,000	5000	5000
^{226}Ra	500	500	500
^{210}Pb	N/A	50	50

(1) See EPA (1980)
(2) See NAS (1983)

Table B-3. Concentration of Radionuclides Dissolved in Ground Water within the Ore Body

RADIO-NUCLIDE	HALF-LIFE (YEARS)	SPECIFIC ACTIVITY (Ci/g)	CONCEN-TRATION (g/m^3) (Oxidizing)	CONCEN-TRATION (g/m^3) (Reducing)
^{238}U	4.47×10^{9}	3.37×10^{-7}	5.0×10^{-1}	2.0×10^{-2}
^{234}U	2.45×10^{5}	6.26×10^{-3}	2.7×10^{-5}	1.1×10^{-6}
^{230}Th	7.70×10^{4}	2.02×10^{-2}	8.4×10^{-6}	3.4×10^{-7}
^{226}Ra	1.60×10^{3}	9.90×10^{-1}	1.7×10^{-7}	6.8×10^{-9}
^{210}Pb	2.23×10^{1}	7.64×10^{1}	2.2×10^{-9}	8.8×10^{-11}

Table B-4. Concentration of Radionuclides 5,000 Meters Down-Gradient From a Uranium Ore Body

RADIONUCLIDE	CONCENTRATION, OXIDIZING (pCi/l)	CONCENTRATION, REDUCING (pCi/l)
^{238}U	1.7×10^{2}	6.7×10^{0}
^{234}U	1.7×10^{2}	6.9×10^{0}
^{230}Th	1.8×10^{-1}	5.7×10^{-2}
^{226}Ra	1.8×10^{0}	5.6×10^{-1}
^{210}Pb	1.8×10^{1}	5.7×10^{0}

Table B-5. Dose to an Individual From Drinking Ground Water 5,000 Meters Down-Gradient From a Uranium Ore Body

RADIONUCLIDE	DOSE CONVERSION (rem/Ci)	DOSE, Oxidizing (rem/yr)	DOSE, Reducing (rem/yr)
^{238}U	8.57×10^{5}	0.10	4.0×10^{-3}
^{234}U	9.86×10^{5}	0.12	4.8×10^{-3}
^{230}Th	6.29×10^{5}	7.9×10^{-5}	2.5×10^{-5}
^{226}Ra	1.38×10^{6}	1.7×10^{-3}	5.4×10^{-4}
^{210}Pb	7.43×10^{6}	9.4×10^{-2}	3.0×10^{-2}
Total		0.32	3.9×10^{-2}

Table B-6. Dose to an Individual From Household Use and Farming Using Ground Water 5,000 Meters Down-Gradient From a Uranium Ore Body

RADIONUCLIDE	DOSE CONVERSION [1] (rem m^3/Ci yr)	DOSE, Oxidizing (rem/yr)	DOSE, Reducing (rem/yr)
^{238}U	1.08×10^{5}	1.83×10^{-2}	7.28×10^{-4}
^{234}U	1.41×10^{5}	2.43×10^{-2}	9.71×10^{-4}
^{230}Th	2.97×10^{5}	5.32×10^{-5}	1.70×10^{-5}
^{226}Ra	8.88×10^{6}	1.55×10^{-2}	4.99×10^{-3}
^{210}Pb	2.84×10^{7}	5.06×10^{-1}	1.62×10^{-1}
Total		5.64×10^{-1}	1.69×10^{-1}

[1] See NAS (1983)

Table B-7. Health Effects From a Repository-Equivalent Uranium Ore Body Over 10,000 Years for the EPA River Scenario

RADIONUCLIDE	DOSE CONVERSION[1] (Health effects/Ci)	HEALTH EFFECTS (Oxidizing)	HEALTH EFFECTS (Reducing)
^{238}U	2.08×10^{-2}	76.3	3.04
^{234}U	1.98×10^{-2}	73.9	2.97
^{230}Th	7.25×10^{-1}	2.81	0.91
^{226}Ra	1.68×10^{-1}	6.38	2.05
^{210}Pb	1.25×10^{-1}	48.3	15.4
Total		208 (207.69)	24 (24.37)

(1) See EPA (1985a)

Table B-8. Integrated Release of Radionuclides Over 10,000 Years at a Point 5,000 Meters Down-Gradient From a Repository-Equivalent Uranium Ore Body

RADIO-NUCLIDE	CONCEN-TRATION OXIDIZING (Ci/m^3)	CONCEN-TRATION REDUCING (Ci/m^3)	INTEGRATED RELEASE OXIDIZING (Ci)	INTEGRATED RELEASE REDUCING (Ci)
^{238}U	1.69×10^{-7}	6.74×10^{-9}	3670	146
^{234}U	1.72×10^{-7}	6.89×10^{-9}	3730	150
^{230}Th	1.79×10^{-10}	5.74×10^{-11}	3.88	1.25
^{226}Ra	1.75×10^{-9}	5.62×10^{-10}	38.0	12.2
^{210}Pb	1.78×10^{-8}	5.69×10^{-9}	386	123

APPENDIX C

COMPARISON WITH THE RESULTS OF THE WASTE ISOLATION SYSTEM PANEL

COMPARISON WITH THE RESULTS OF THE WASTE ISOLATION SYSTEM PANEL (WISP)

The analyses conducted by the WISP (NAS, 1983) were done prior to the decision to use the unsaturated zone at Yucca Mountain and before the distance to the accessible environment had changed from 10,000 to 5,000 meters (EPA, 1985). Also, to avoid making assumptions regarding the pore velocity for the generic sites (because that parameter was poorly known at the time), the WISP plotted their dose results as a function of ground-water travel time. In order to assure ourselves that we were producing similar results with the UBCNE-41 model (Lung et al., 1987) our first task was to compare our results with the results of the WISP.

This comparison was begun by examining the source term used in the WISP calculations. This source term is presented in Table C-1, along with the retardation and dose conversion factors. The transport model requires an initial concentration of each radionuclide (which is a function of the solubility and the flux past the waste package or the alteration rate and the flux past the package), and the radionuclide chains considered. When competing isotopes of the same element are present, the solubility must be proportioned among the isotopes. This was done on the basis of the mass of each isotope present in the inventory. For advective release the fractional release for each radionuclide was determined by the following equation (NAS, 1983):

$$f_{j,A} = \frac{8\, N_j\, D_j^{0.5}\, n_e\, U^{0.5}\left(1 + \frac{R}{L}\right)}{(\pi R)^{1.5}\, n_j} \quad , \quad \text{when } \frac{U}{R} > 4 \qquad \text{(Eq. C-1)}$$

where:

$f_{j,A}$ is the fractional release rate of radionuclide j (yr^{-1});
N_j is the radionuclide solubility (g/m^3);
D_j is the diffusion coefficient (m^2/yr);
n_e is the effective porosity;
U is the pore velocity (m/yr);
R is the package radius (m);
L is the package length (m); and
n_j is the bulk density of the radionuclide in the waste (g/m^3).

When U R / D_j is less than 4 the release is no longer advection controlled, but is controlled by diffusion. The fractional release is then given by the following equation (NAS, 1983):

$$f_{j,D} = \frac{\beta\, n_e\, D_j\, N_j}{n_j} \qquad \text{(Eq. C-2)}$$

where:

β is 3 / R_s^2; and
R_s is the radius of a sphere that has a surface area equal to that of the package.

The leach time is then calculated as:

Leach time = $1 / f_j$ (either advection or diffusion controlled; $f_j = f_{j,A}$ or $f_{j,D}$)

For radionuclides that are released through alteration of the waste (alteration limited) the leach time is simply the inverse of the alteration rate.

The WISP used the following parameters:

Porosity, n_e, = 0.01;
$f_j = 2 \times 10^{-8}$ yr^{-1} for ^{237}Np, 4×10^{-7} yr^{-1} for ^{14}C;
L = 2.46 m;
R = 0.152 m;
$D_j = 3.15 \times 10^{-2}$ m^2/yr;
U = 1.00 m /yr;
Dispersion Coeff. = 50 m^2/yr;
Ground-water flow 9.90×10^4 m^3/yr; and
Distance to the accessible environment = 10,000 meters.

For each radionuclide, the parameters and equations shown above and the source term components shown in Table C-1 were used to calculate the initial concentrations and leach times for various fluxes of ground-water flow through a repository in the saturated zone.

The initial concentration of each radionuclide was calculated as follows:

$$C_0 = \frac{\text{mass of radionuclide in inventory at } t_0) \text{ x (fractional release or alteration rate)}}{\text{(ground-water flow rate)}}$$

where t_0 is the time of waste package failure.

These concentrations and leach times were then used as input to UCBNE-41, which was used to calculate concentrations at the accessible environment. The dose to an individual from each radionuclide was calculated and plotted against its corresponding ground-water travel time. This process was used to produce the curves shown in Figure C-1, which were taken directly from Figure 9-12 of the WISP report (NAS, 1983). This approach was followed to produce the plots for ^{237}Np and ^{14}C which are shown in Figure C-2, and compare favorably with the results of the WISP (Figure C-1). The time that the peak dose occurs is a function of the retardation of each radionuclide.

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Table C-1. Source Term Information, Retardation Factors, and Dose Conversion Factors Used by the Waste Isolation System Panel (WISP) [1]

RADIO-NUCLIDE	HALF-LIFE (yrs)	INVEN-TORY (Bq)	SOLU-BILITY (g/m^3)	RETARD-ATION FACTOR	DOSE CON-VERSION ($Sv\text{-}m^3$/ Bq-yr)	DOSE RATIO (Drinking/ Total)
^{14}C	5.73×10^{3}	5.73×10^{15}		1	9.21×10^{-7}	1.22×10^{-4}
^{79}Se	6.50×10^{4}	1.46×10^{15}	1.0×10^{-3}	50	1.60×10^{-7}	4.27×10^{-4}
^{93}Zr	1.50×10^{6}	6.68×10^{15}	1.0×10^{-4}	5000	4.83×10^{-13}	3.00×10^{-1}
^{99}Tc	2.12×10^{5}	4.84×10^{16}	1.0×10^{-3}	5	7.03×10^{-10}	1.00×10^{-2}
^{126}Sn	1.00×10^{5}	2.88×10^{15}	1.0×10^{-3}	1000	2.77×10^{-8}	1.20×10^{-2}
^{129}I	1.70×10^{7}	1.16×10^{14}		1	2.04×10^{-8}	7.07×10^{-2}
^{135}Cs	3.00×10^{6}	1.28×10^{15}		500	5.26×10^{-8}	2.67×10^{-2}
^{234}U	2.47×10^{5}	4.16×10^{13}	1.0×10^{-3}	40	3.80×10^{-8}	1.88×10^{-1}
^{238}U	4.51×10^{9}	1.16×10^{13}	1.0×10^{-3}	40	2.91×10^{-8}	2.15×10^{-1}
^{237}Np	2.14×10^{6}	1.16×10^{15}	1.0×10^{-3}	100	1.29×10^{-5}	8.41×10^{-2}
^{238}Pu	8.60×10^{1}	4.26×10^{16}	1.0×10^{-3}	200		
^{239}Pu	2.44×10^{4}	1.16×10^{16}	1.0×10^{-3}	200	9.80×10^{-9}	1.94×10^{-1}
^{240}Pu	6.58×10^{3}	1.95×10^{16}	1.0×10^{-3}	200	9.80×10^{-9}	1.94×10^{-1}
^{241}Pu	1.32×10^{1}	4.57×10^{18}	1.0×10^{-3}	200		
^{242}Pu	3.79×10^{5}	6.50×10^{13}	1.0×10^{-3}	200	9.50×10^{-9}	1.85×10^{-1}
^{241}Am	4.58×10^{2}	7.03×10^{17}	1.0×10^{-4}	1000	1.26×10^{-7}	4.30×10^{-2}
^{243}Am	7.95×10^{3}	6.31×10^{16}	1.0×10^{-4}	1000	1.23×10^{-7}	4.29×10^{-2}
^{242}Cm	4.46×10^{-1}	7.29×10^{19}	1.0×10^{-3}	500		
^{244}Cm	1.76×10^{1}	5.55×10^{18}	1.0×10^{-3}	500		
^{245}Cm	9.30×10^{3}	7.68×10^{13}	1.0×10^{-3}	500		
^{246}Cm	5.50×10^{3}	1.06×10^{14}	1.0×10^{-3}	500		

(1) See NAS (1983)

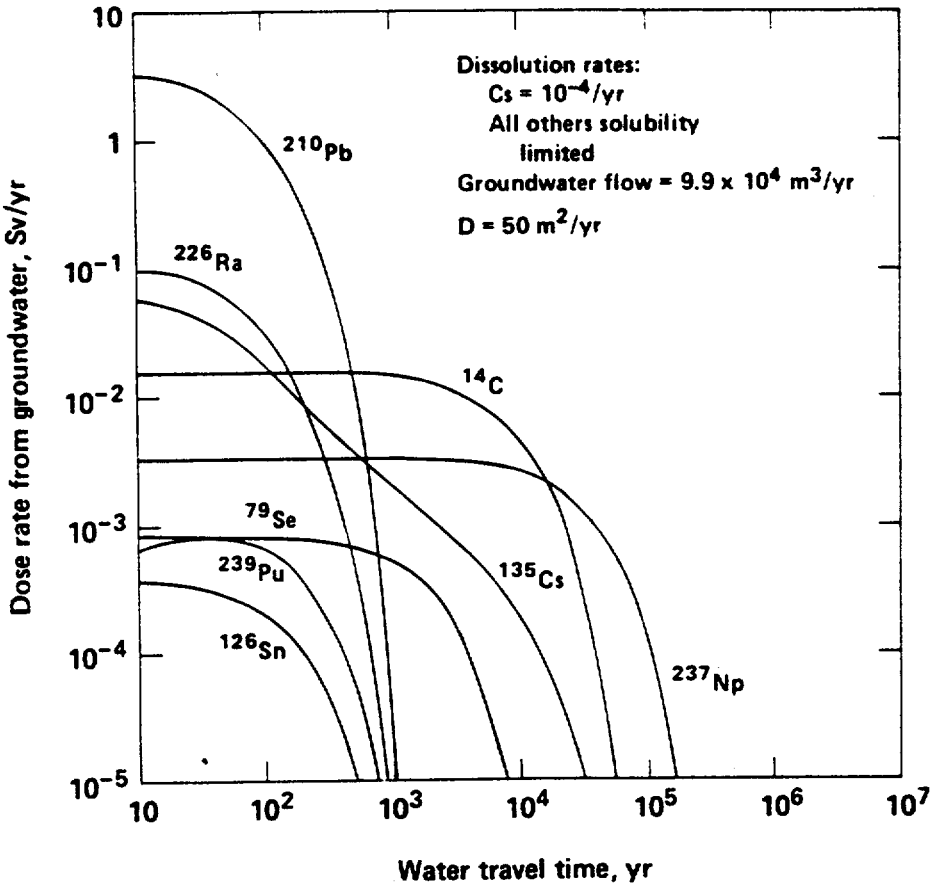

Figure C-1. Peak Individual Dose as a Function of Ground-Water Travel Time (Taken from Figure 9-12 of the WISP Report, Reference NAS, 1983).

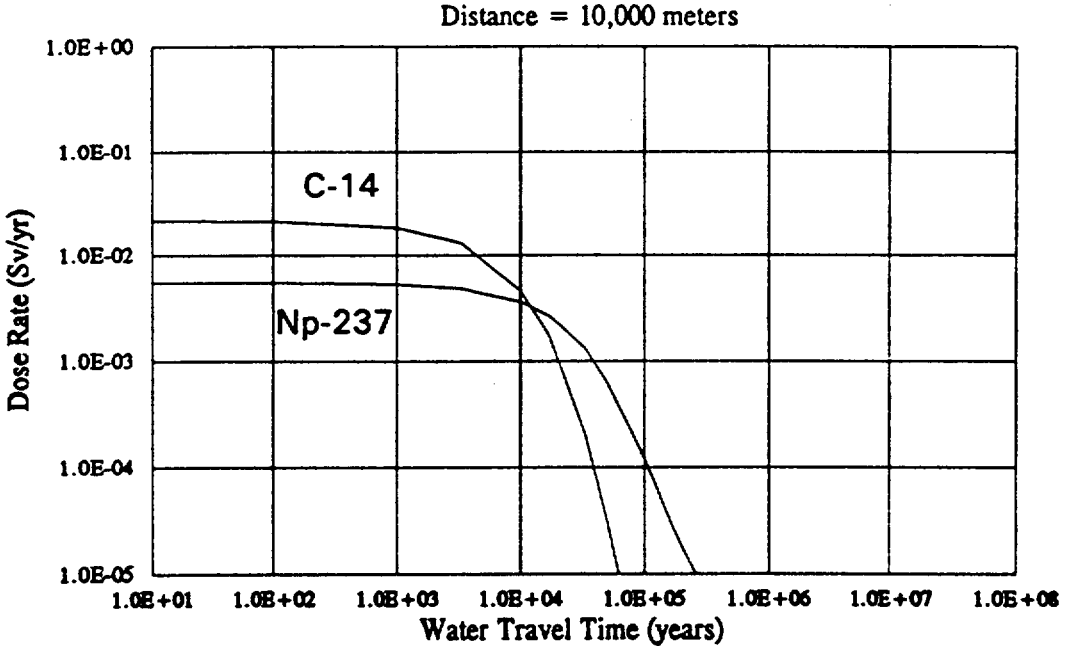

Figure C-2. Peak Individual Dose as a Function of Ground-Water Travel Time for ^{237}Np and ^{14}C for Comparison with the WISP Results.

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