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Volume 1

Methodology Manual

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PRESTO-EPA-POP: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code

Volume 1

METHODOLOGY MANUAL

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PREFACE

This two-volume PRESTO-EPA-POP model documentation provides the background information on the mathematical modeling used to generate the basic data for the Environmental Impact Statement (EIS) which is used to support EPA's rulemaking for generally applicable environmental standards for the management and disposal of low-level radioactive wastes (LLW). Volume 1 of the PRESTO-EPA-POP documentation presents the theoretical bases of the mathematical model and their implemented computer code for the assessment of the cumulative population health effects (including fatal cancer deaths and serious genetic effects) to the general population residing in the downstream regional basin of a LLW disposal site. The model simulates the leaching of radionuclides from the waste matrix, the hydrological, hydrogeological, and biological transports, the resultant human exposures, and finally the assessment of the probable health effects for the entire regional water basin population. Volume 2 of the PRESTO-EPA-POP documentation provides the information on the structure of the computer code and how it is used in the health effects assessments.

The two volumes present enough model detail so that interested persons may apply the model, using appropriate and applicable input data, for assessing the cumulative population health effects from a LLW disposal site.

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EXECUTIVE SUMMARY

The U.S. Environmental Protection Agency (EPA) is responsible for developing a generally applicable standard for the land disposal of low-level radioactive waste (LLW). As an aid in developing the standard, a family of computer codes, entitled PRESTO-EPA-POP, PRESTO-EPA-DEEP, PRESTO-EPA-CPG, PRESTO-EPA-BRC and PATHRAE-EPA, has been developed under EPA direction. The PRESTO-EPA-POP code was the first code developed and served as the basis for the other codes in the PRESTO-EPA family. PRESTO-EPA-POP estimates potential health effects to local and regional basin populations from the shallow land disposal of low-level radioactive waste under a wide variety of hydrologic, geologic, climatic, site engineering, and waste form conditions. The EPA uses the PRESTO-EPA code family to compare the potential health impacts of a broad number of LLW disposal alternatives to evaluate and support its decisions for the LLW standard.

In developing the LLW standard, EPA initially identified more than 24 distinct types of radioactive waste in various waste forms and containers which required disposal by one or more of eight alternative disposal methods within the United States. Basic requirements for the PRESTO-EPA code family included: (1) use of existing release and pathway models where possible to reduce development time and costs; (2) use of modular subroutines to allow the use of improved or more appropriate submodels when available or needed; (3) ability to rapidly and economically execute the code for several thousand scenarios or alternatives as necessary; (4) flexibility to analyze a wide range of hydrologic, climatic, waste type, and site engineering combinations; (5) use of EPA's health risk

calculational methodology; and (6) results which are reasonably realistic, reliable, and verifiable.

The user of the PRESTO-EPA-POP code, may, with appropriate input data, simulate a specific waste type (e.g., LWR decontamination resins), a specific form for the waste (e.g., solidified in cement), and special containers (e.g., 300 year high-integrity containers). The user may also simulate special site engineering and waste emplacement configurations such as sanitary landfill, use of deeper trenches, or installation of trench caps with either natural or special low permeability obtained from use of clays and compaction. Although most shallow LLW disposal facilities consist of many disposal trenches, the PRESTO-EPA-POP code treats all LLW trenches and their wastes as a single, representative, combined trench and volume of wastes.

Water, principally from precipitation, is the primary agent of release and transport of radioactivity from LLW disposed in shallow trenches. Geology, hydrology, and climate also affect not only the mode and rate of release, but also which transport pathways dominate, and the rate of transport. Therefore, the input data portion of the PRESTO-EPA-POP code and the release and transport subroutines, have been designed with sufficient flexibility to accommodate varying hydrologic and climatic conditions, ranging from typical conditions found in the humid southeastern United States to the arid southwest, with reasonable accuracy and realism.

The PRESTO-EPA-POP code can simulate the following typical water-related release and transport phenomena. Precipitation falling on the ground surface above the trench is apportioned between infiltration into the trench through the trench cap, drainage away from the site by surface

runoff, and evaporation into the atmosphere. The user can specify the time and percent of trench cap failure, and the time duration of container integrity. Radionuclides released from the waste by infiltrated water are transported away from a LLW disposal trench primarily by groundwater or surface runoff. The amount of groundcover, length and steepness of the slope, etc., and runoff percent can be specified to simulate erosion of the surface soil and the trench cap. Radioactively contaminated water exfiltrating into the subtrench soil zone may ultimately enter an aquifer. Radionuclides that reach the aquifer will be transported horizontally within the aquifer. These radionuclides will generally be transported at a slower rate than the ground water velocity in the aquifer due to hydrological and geochemical interaction with the solid materials in the aquifer which retards nuclide migration. Some of the radionuclides which enter the aquifer may eventually reach irrigation or water supply wells or surface streams, and thus become available for uptake by the local population. Residual radionuclides in the aquifer which are not consumed by the local population are assumed to undergo further transport to a large downgradient population (referred to as the "basin" population). Health effects to the basin population are calculated for a time period of up to 10,000 years.

The overflow of contaminated water from the trench onto the surface soil is simulated when conditions suitable for overflow occur. Once overflow has occurred, the radionuclides are transported by surface runoff into nearby streams and may become available for human consumption via irrigation or drinking water. Residual radionuclides in the streams which are not consumed during the "local" 1,000 year assessment are assumed to be released to the regional basin within one year.

The exposure of the local population to radionuclides transported from LLW sites by the atmospheric pathway is also simulated. The atmospheric transport calculations nominally assume that the local population resides within a single 22.5-degree sector. User specified parameters give the fraction of the year that the wind blows in that sector. A simple external model not contained in the PRESTO-EPA codes (see Appendix C) enables the user to make downwind atmospheric concentration calculations which properly distributes atmospheric contamination to population centers throughout a 360 degree circle around the site.

Radionuclides remaining on the soil surface by trench overflow, spillage during disposal operations, or erosion of the trench cap, may become suspended in the atmosphere and transported downwind where the nuclides may be inhaled or deposited on vegetation and soil. When the radioactivity is deposited, the PRESTO-EPA-POP code simulates both external exposure to humans and internal exposure from the ingestion of contaminated crops, meat, and milk.

The PRESTO-EPA-POP code allows the user to select special human exposure scenarios such as an inadvertent intruder residing on or farming the site, as well as routine migration of radionuclides from the trench through the hydrologic and atmospheric environmental pathways to crops and drinking water. Normal scenarios assume that the population resides downstream of the plume of contamination and ingests radionuclides from various hydrologic, atmospheric, and food chain pathways. Processes considered in calculating individual or population exposure include: groundwater transport under saturated or partially saturated flow conditions, surface runoff, trench water overflow and seepage, geochemical exchange, trench cap erosion, stream dilution, and resuspension and atmospheric dispersion of contaminated soil.

Average annual concentrations of each radionuclide in environmental media (e.g., well water or the atmosphere) over the assessment period are used to calculate radionuclide concentrations in foodstuffs. Information from foodstuffs and human ingestion and breathing rates are utilized to calculate the annual average intake of radionuclides per individual in the population by ingestion and inhalation. These intake data are used by the DARTAB subroutine to estimate dose rates, health effects, and genetic effects.

The PRESTO-EPA-POP code calculates doses, fatal health effects, and serious genetic effects for individuals and local populations over the period of assessment, using a lifetable approach developed by EPA. This approach assumes that each person in the population is a member of a large population cohort that is exposed to constant, averaged radionuclide concentration levels. Each member of the population is assumed to ingest regional average quantities of food (vegetables, beef, and milk) and water. It is assumed these foods are produced on contaminated fields and spray irrigated with contaminated water. Beef and milk cattle are also assumed to drink contaminated water.

As previously mentioned, the PRESTO-EPA-POP code also addresses on request, special exposure and pathway scenarios for inadvertent intruders residing on or farming the site. The inadvertent resident intruder scenario assumes that an intruder unknowingly excavates a basement in the disposal trench while building a residence. The individual is externally exposed to the buried radionuclides because the walls of the basement of the residence are assumed to be surrounded by trench material. The time of residency during the assessment period when the residence is first occupied and the

composition of the initial trench inventory which contributes to exposure are specified by the user.

Farming the site after loss of institutional control is also treated as a separate intrusion scenario. By farming the site, the farmer is affected by the transport processes previously described. In contrast to an off-site population, however, the farmer may ingest crops whose roots have penetrated into the radioactive waste in the trench. The water used by the farmer for irrigation and drinking normally would contain a much higher concentration of radioactivity than water used by the off-site population because the water is taken from a well, stream, or pond immediately adjacent to the disposal trench. Farming activities may also mechanically suspend contaminated soil into the atmosphere. The time when mechanical suspension of the surface soil is initiated by farming is specified by the user. The impact of such mechanical suspension is calculated for both the farmer and the downwind population.

The PRESTO-EPA-POP code can be used to make health effect assessments for up to 10,000 years following the end of the disposal operations for the regional basin population. These assessments are made in two stages. The first stage calculates the health effects to the local population for up to 1000 years, and the second stage calculates the health effects to the regional basin population over 10,000 years, using a synthesized model community in the basin. If necessary, the local assessment period, which is a more detailed analysis, can be extended for several thousand years or more, but with a significant increase in computer time and costs.

The health effects to the basin population are based on the cumulative residual radionuclides leaving the disposal area with the

surface water or groundwater. Residual radioactivity is calculated, taking into account the time required for the radionuclides to flow through the aquifer from the local use point to the basin stream, by considering the amount of contaminated water consumed or used by the local community during the period for local impact assessment and the dilutions to the basin stream by contributions from surface runoff and stream flow coming from upstream of the basin use point. After 1,000 years, the local community is assumed to be incorporated in the regional basin community and the uptake of radionuclides takes place at the regional basin.

The end of LLW disposal operations at a site is the starting time for the assessments made by the code. The user specifies the length of time for the local assessment period, up to 1,000 years. The regional basin assessment period is optionally set equal to the local assessment period or the local assessment period plus 9,000 years. For the local assessment, subtotals for the releases to surface water, well, and atmosphere may be calculated and printed at a number of user-specified time intervals within the assessment period. EPA normally determines releases every 100 years for a standard 1,000 year run. Health effects for the local population are averaged over the length of the entire local assessment period.

The time step for the PRESTO-EPA-POP code is fixed at one year and default parameters in the model give annual averages. The maximum concentration of each radionuclide and the year of the maximum for each nuclide is printed out for the concentrations in the atmosphere, well, and stream.

Other features in the PRESTO-EPA-POP model and code include:

- The radionuclide inventory modeled accounts for radioactive decay during disposal operations.

- The leaching of radionuclides from the trench waste materials can be delayed by containment in high integrity waste containers if specified by the user.

- In-growth of radiological daughter products is not calculated explicitly by the model, but can be estimated by considering the daughters in the initial trench inventory for some nuclides.

- The sorption and desorption characteristics of each radionuclide in the waste, the surface soil, the geologic disposal media, and the underlying aquifer can be calculated by a linear model characterized by four distribution coefficients, or K_d values.

- Groundwater flow in the host soil and disposal medium may be either saturated or partially saturated.

- The hydrologic transport of radionuclides vertically by trench water from the trench bottom to the aquifer and then horizontally through the aquifer is modeled as one-dimensional flow with a correction made for dispersion.

- Because many of the submodels in the PRESTO-EPA-POP code were initially developed for other types of assessments and have been adapted to estimate health effects from shallow land disposal of LLW, the code is modular in design to allow submodels or subroutines to be replaced when desired or necessary.

- Annual hydrologic, climatic, and meteorologic data which are used for a site can be based on the annual variations recorded in 30-year or other long-term averages from a nearby weather station and normalized for input into the PRESTO-EPA-POP code.

- There are no significant changes assumed in farming practices, demographics, water and foodstuffs usage, or climate during the period of analysis.

The PRESTO-EPA-POP code uses unit response, bookkeeping, and scheduled event types of submodels. The unit response submodels calculate the annual response for a process. For example, the INFIL unit response submodel calculates the annual infiltration through an intact trench cap. This annual infiltration is then apportioned within each year by the bookkeeping

submodels. Other unit response models calculate the atmospheric radionuclide concentration transported downwind per unit source strength and the annual average erosion of the trench cap.

Bookkeeping submodels follow the results of unit response submodels and user-supplied control options. For example, the TRENCH submodel, which maintains a water balance in the trench, calculates the maximum level of standing water in the trench and the water volume annually leaving via the trench bottom or overflow. Scheduled event submodels consider the events such as cap failure, basement construction, the failure of container integrity, and initiation of scheduled mechanical suspension of dust. The timing for these events is specified by the user.

The Environmental Protection Agency wishes to warn potential users that, like any complex computer code, the PRESTO-EPA codes can be misused. Misuse could consist of using the code to examine a site where one or more critical modeling assumptions are invalid, or where values for significant input parameters are chosen that do not accurately reflect variables such as radionuclide inventory, site meteorology, surface and subsurface hydrology and geology, and future population demographics. Certain release and transport scenarios, such as major changes in meteorology or mining of the trench contents, are not considered in the PRESTO-EPA-POP model and code. Significant changes to the existing code and the input data would be required to consider such scenarios. The PRESTO-EPA codes were developed to assess and compare alternative methods for managing and disposing of LLW at generic sites for general scenarios. The codes were not developed to analyze specific sites.

1. INTRODUCTION

The U.S. Environmental Protection Agency (EPA) is developing a generally applicable environmental standard for the disposal of low-level radioactive waste (LLW) to support the U.S. Nuclear Regulatory Commission (NRC), the U.S. Department of Energy (DOE) and others in developing a national radioactive waste management system. As an aid in developing the standard, a family of computer codes entitled PRESTO-EPA-POP, PRESTO-EPA-DEEP, PRESTO-EPA-CPG, PRESTO-EPA-BRC and PATHRAE-EPA was developed under EPA direction. The EPA uses the PRESTO-EPA code family to estimate and compare the potential health impacts of a broad number of LLW disposal alternatives for evaluation and support of the LLW standard. Table 1-1 provides a brief description of each of these EPA codes. These codes, and how the EPA uses them, have been described in detail (Gal84). Information on obtaining complete documentation and users' manuals for the PRESTO-EPA family of codes (EPA85a through EPA85i, Mey81, Mey84) is available from the EPA.

The PRESTO-EPA-POP code (Prediction of Radiation Effects from Shallow Trench Operations - EPA - Population), which is the first member of the family of PRESTO-EPA codes and the subject of this document, is designed to permit the EPA to compare the relative potential population health effects of different management and shallow land disposal alternatives for low-level radioactive waste.

1.1 DESCRIPTION OF A LOW-LEVEL WASTE DISPOSAL SITE

The description of the life cycle of a shallow land disposal site is useful in explaining the modeling approach. Following site selection and

TABLE 1-1

PRESTO-EPA CODE FAMILY

PRESTO-EPA CODE	Purpose
PRESTO-EPA-POP	Estimates cumulative population health effects to local and regional basin populations from land disposal of LLW by shallow methods; long-term analyses are modeled (generally 10,000 years).
PRESTO-EPA-DEEP	Estimates cumulative population health effects to local and regional basin populations from land disposal of LLW by deep methods.
PRESTO-EPA-CPG	Estimates maximum annual whole-body dose to a critical population group from land disposal of LLW by shallow or deep methods; dose in maximum year is determined
PRESTO-EPA-BRC	Estimates cumulative population health effects to local and regional basin populations from less restrictive disposal of BRC wastes by sanitary landfill and incineration methods.
PATHRAE-EPA	Estimates annual whole-body doses to a critical population group from less restrictive disposal of BRC wastes by sanitary landfill and incineration methods.

procurement, trenches are dug on the site. Waste materials in various types of containers ranging from plastic bags or cardboard boxes to steel drums are added to each trench. There is currently no standardized container for LLW disposal. Once a trench is filled, or at various stages of filling, the trench may be backfilled to eliminate voids and decrease the potential for subsidence and cracking of the trench cap. Following backfilling, the trench is covered with a cap of soil or clay, one to several meters thick, mounded above grade to facilitate runoff of precipitation and decrease infiltration. To minimize erosion in non-arid climates, the cap is seeded with grass or cover crop and is maintained for a number of years. Although not all burial operations are identical, most current and past LLW facilities have included some of the practices described.

In general, hydrologic transport is the major process by which the general public may become exposed to radioactivity from LLW disposed in shallow trenches. Figure 1-1 is a schematic description of the routes that water, and any transported radionuclides, may follow from a trench in a LLW disposal site. The major source of water at existing sites and, it is anticipated, at future sites is precipitation. Precipitation at a site may infiltrate the soil or trench cap, run off the surface, or evaporate. Depending on the groundcover, length and steepness of slopes, and other factors, runoff may cause erosion of soil from the surface.

Hydrologic transport of radionuclides from a LLW burial trench may be by the groundwater or via the runoff. Groundwater or precipitation entering the trench may mix with the waste material and become contaminated. This contaminated water may either overflow the top of the trench or

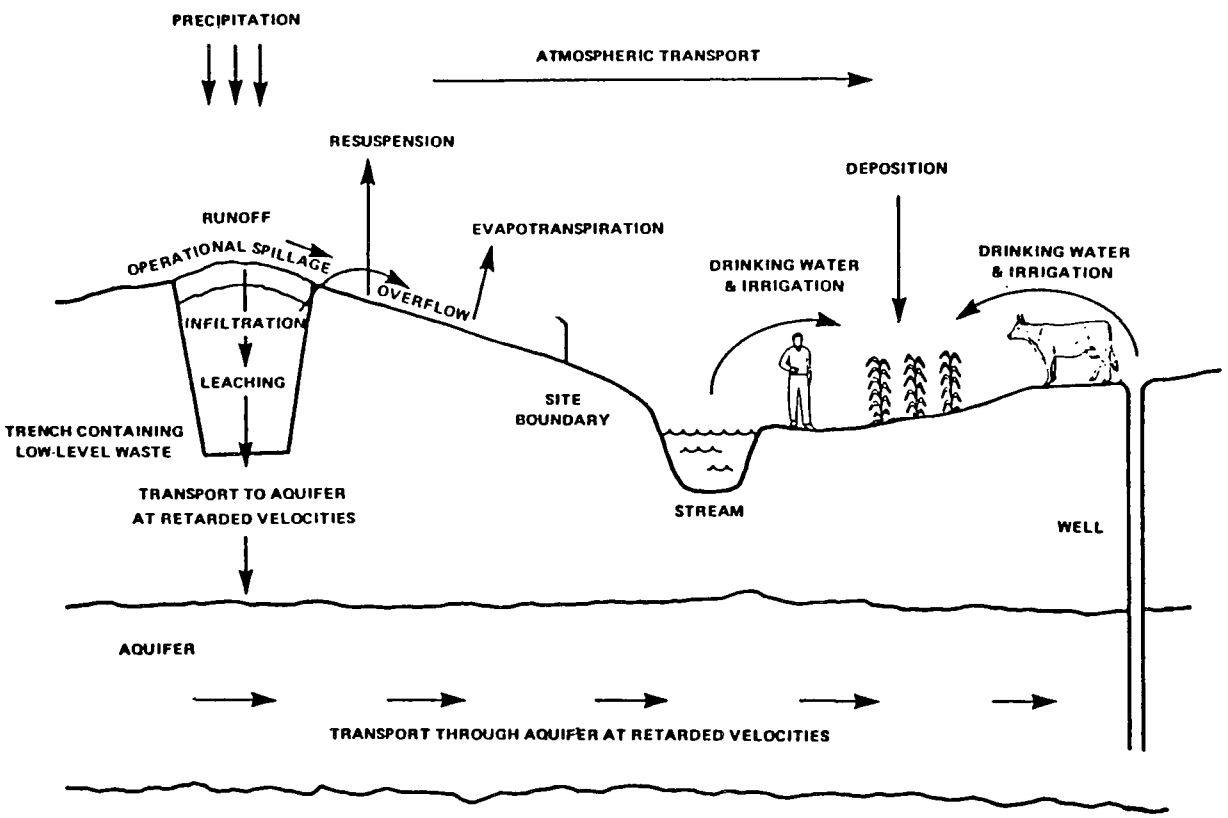

FIGURE 1-1. ENVIRONMENTAL TRANSPORT PATHWAYS USED IN PRESTO-EPA-POP.

percolate downward through the bottom of the trench to the subtrench soil zone and ultimately enter an aquifer. In most cases, particularly in the arid southwestern United States, the disposal formation and much of the geologic formations overlying the aquifer are only partially saturated. Transport velocities are different for saturated and partially saturated flow but can be calculated. The water in the aquifer will have a characteristic flow velocity from a few meters per year to hundreds of meters per year. Radionuclides that finally reach the aquifer generally will be transported at velocities much lower than the characteristic flow velocity of the water in the aquifer. This "retardation" is due to interaction of the radionuclides with the solid materials in the aquifer. Some of the radionuclides in the aquifer may eventually reach the water supplying irrigation or drinking wells or a point where the aquifer communicates with surface streams. Water infiltration from the aquifer is used by the local population. Residual radionuclides in the aquifer which are not consumed by the local population are assumed to undergo further transport to a large downgradient population (referred to as the "regional basin" population). Health effects to the regional basin population are calculated for a time period of up to 10,000 years.

A trench dug in material of low permeability may eventually overflow if rainfall is high enough and if water infiltrating into the trench is not removed. Once overflow has occurred, the radionuclides may be transported by runoff to nearby streams and become available for human consumption via irrigation or drinking, or else released to the regional basin through a basin stream.

Humans away from the burial site may be exposed to the contaminated water if this water is used for irrigation of food crops or drinking.

Humans may also be exposed to radionuclides transported from LLW sites by atmospheric processes. Radionuclides deposited on the soil surface by trench overflow, by spillage during disposal operations, or by complete erosion of the trench cap, may be suspended in the atmosphere and transported downwind where they may be inhaled or deposited on the ground and vegetation. Deposited radioactivity may contaminate crops, meat, and milk and enter the food chain. Deposition on the soil surface may also result in external radioactive exposure to humans.

Besides the transport pathways represented in Figure 1-1, there are several other methods by which humans can be exposed to radioactivity from a LLW disposal site. If the trench cap is eroded away, the wastes will be more accessible to environmental transport processes. After the end of institutional control of the site, an intruder could reside on or farm the site. It is likely the intruder would receive an external exposure from the buried radionuclides if excavation occurred near the trench. Factors such as the length of residency, length of time after burial, and initial radionuclide inventory would all contribute to the amount of exposure and subsequent dose and health effects.

Farming the site may also result in off-site exposures to radioactivity by mechanically suspending contaminated soil in the atmosphere. In contrast to an off-site population, however, a farmer may ingest crops grown over the trench where the nuclide uptake by roots will be higher, because nuclide concentrations in the soil will be higher. Also, depending on when farming started, the on-site irrigation water may contain higher concentrations of radioactivity than off-site water.

1.2 DESCRIPTION OF MODEL

The model has been designed to calculate the doses and health effects, resulting from the disposal of low-level radioactive wastes, to a local population and to the population in a hypothetical regional basin in which the disposal site is located. The impacts on the local and regional basin populations are analyzed for a period of up to 1,000 years. The regional basin analysis can be extended at the option of the user for an additional 9,000 years, with the local population assumed to become part of the regional basin population.

Many of the submodels included in PRESTO-EPA-POP were developed for other types of assessments and have been adapted for the estimation of the environmental transport of radioactive waste and ensuing health effects from LLW disposal. Because of this, PRESTO-EPA-POP is modular in construction to allow for different versions of the submodels or subroutines to be substituted, if desired.

PRESTO-EPA-POP first simulates the environmental transport of radionuclides from the low-level waste trench to the environments of the local and regional basin populations. Then the code calculates doses and health effects using the DARTAB model for the local population, and a separate health effects accounting model for the regional basin population. The calculations in DARTAB are based on average internal intakes and average external exposures. Internal intakes are due to ingestion of contaminated foods and water, and inhalation of contaminated air. External exposures are due to contaminated air and contaminated soil. For a detailed account of the DARTAB methodology, see Section 2.3 of the DARTAB documentation report (Be81).

The health effects calculations for the regional basin population are based on the cumulative amount of radionuclides released to the regional basin through a basin stream during the assessment period for the regional basin, which is usually 10,000 years. The transport portion of the regional basin submodel provides the cumulative amount of radionuclides discharged into the downstream basin. The subsequent health effects are calculated by using precalculated healtheffect conversion factors for each radionuclide and region.

The population health effect assessment begins immediately after the closure of the disposal site. The radionuclide inventories of the wastes in the trench are calculated to account for the radioactive decay during the operational period. The waste is assumed to be containerized so that the leaching of radionuclides from the waste cannot begin until the container fails. The length of the container integrity is a user specified parameter; therefore, a zero time should be specified when the waste is not physically containerized or no credit is given for container integrity.

The code was developed to handle a wide variety of hydrogeologic and climatic situations. It can also handle waste leaching and the groundwater transport of nuclides under partially saturated as well as saturated hydrogeologic conditions, while taking into account nuclide retardation due to hydrogeochemical processes. The code has features to account for the decay of the leaching process resulting from the use of waste containers; a farming scenario which simulates farming over the trench with vegetation root uptake of radionuclides from the waste; and reduction in the trench radionuclide inventory at the start of simulation because of decay during the operational period.

The ingrowth of radioactive decay products is not calculated by the model to maintain simplicity and to reduce computer time and expense. In cases where the major dose contribution is from external exposure to a short-lived progeny in equilibrium with a parent radionuclide present in the trench inventory, the user may wish to include the progeny in the trench inventory by a method described at the end of Section 2.2.3.

Operational spillage is defined as radionuclides spilled from incoming waste packages and remaining on the ground surface at the close of disposal operations. These radionuclides would subsequently be transported either by the atmospheric pathway to the local population or by the surface water pathway to the nearby stream.

The complex physical and chemical interaction between the radionuclides and the solid geologic media has been grouped into a single factor, the distribution coefficient, viz., K_d. Different K_d values can be used for soil, the mixture of soil and waste material in the trench, sub-trench soil, and the aquifer.

The subsurface transport path of radionuclides is assumed to be vertical from the trench bottom to the aquifer and then horizontal through the aquifer. The flow in the vertical strata can be calculated either as saturated or unsaturated flow, depending on the relationship between the rate of exfiltration, the degree of saturation, and the properties of the geologic media. However, flow in the aquifer is assumed to be saturated. The transport of radionuclides in the aquifer is calculated by employing Hung's "optimum groundwater transport model" (Hu81). The model also uses Hung's correction factor which was designed to compensate for the dilution effects of dispersion on the health effects evaluation.

Three types of submodels are used in the PRESTO-EPA-POP code: unit response submodels, bookkeeping submodels, and scheduled event submodels. The unit response submodels calculate the annual response of a give process. For example, the submodel INFIL calculates the annual infiltration through an intact trench cap. This annual infiltration is then apportioned among the transport processes by the bookkeeping submodels. Other unit response models calculate the annual average atmospheric dispersion coefficient and erosion from the trench cap.

Bookkeeping submodels keep track of the results of unit response submodels and user-supplied control options. For example, the TRENCH submodel maintains a water balance in the trench, calculates the level of standing water in the trench, and the volume of water leaving the trench.

Average concentrations of each radionuclide over the assessment period in environmental media, such as well water or the atmosphere, are used to calculate radionuclide concentrations in foodstuffs. Foodstuff concentrations and average human ingestion and breathing rates are utilized to calculate the annual average radionuclide intake per individual in the local population by ingestion and inhalation. These intake data are then used to estimate dose rate and health effects.

The atmospheric transport calculations made by the code assume that the total population resides within the same 22.5-degree sector. User-specified parameters give the fraction of year that the plume blows in that sector. Therefore, each member of the population, breathing at the same rate, will inhale the same quantity of each radionuclide. An external code, RADE (EPA85i), allows the user to replace the atmospheric dispersion calculation made for a 22.5 degree sector by the PRESTO-EPA-POP code with

an externally calculated equivalent dispersion coefficient for a number of population centers in a 360 degree circle enclosing the disposal site. The RADE computer code is briefly described in Appendix C.

Each person in the local population is assumed to eat the same quantities and varieties of food, all grown on the same fields, and obtains his or her drinking and crop irrigation water from the same source. This assumption simplifies the calculations and is appropriate because of the large uncertainties in predicting individual mobility, population demography. agricultural practices, geologic and hydrologic changes that might occur during the 1,000 year local analysis period. As input parameters, the user may specify the fraction of the drinking and irrigation water that is supplied by the contaminated well or stream.

Doses and health effects to populations are calculated by characterizing the population center for each site with a single geographic centroid location and the total population. In calculating the health effects, the population age distribution and size is held constant over the assessment period.

Scenarios resulting in radiological exposure from the buried waste may be modified by changing input parameters. These scenarios include: normal disposal site operations; spillage of wastes during disposal operations; a resident intruder on the site; farming of the site; and an eroded trench cap with subsequent atmospheric contamination via suspension of mixed waste material and soil. By changing the site description parameters, the user may design other scenarios of interest. To satisfy the needs of EPA for an assessment of the total impact of a disposal facility, the code is designed to make calculations for combinations of all reasonable exposure pathways

for a given scenario. The exposure pathways for each of four scenarios are listed in Table 1-2.

Additional assumptions in the model and features of the code follow:

- Annual average meteorology for the site is part of the required input data.
- All members of the population use the same sources of drinking and irrigation water contaminated from trench seepage or overflow.
- The population age distribution and size are held constant over the assessment period.
- The user can specify the fraction of the drinking and irrigation water that is supplied by the contaminated well or stream.
- Doses and health effects to populations are calculated from population centers for each site characterized by a single geographic centroid location containing the total population.
- Exposure from the eroded trench occurs when the trench cap has become completely removed by either erosion or mechanical means. Thereafter, the eroded trench scenario may provide for suspension and atmospheric transport of trench contents and resulting inhalation by an individual or the specified population.
- An individual may be directly exposed to trench contents and thus receive an external dose. The population may receive an external dose from immersion in the suspended plume.
- The code is structured to consider only one scenario per computer run. The scenario to be simulated may be designed by the input values a user chooses for such parameters as population size, location and distance to well, percent cap failure, resuspension rate, etc.

1.3 OUTLINE OF METHODOLOGY MANUAL

Chapter 1 provides an introduction to and basis for the PRESTO-EPA-POP code. The description is given of a shallow land disposal site for LLW and the potential pathways by which radiation exposures to humans might occur.

TABLE 1-2

RADIOLOGICAL EXPOSURE PATHWAYS FOR SELECTED SCENARIOS

Scenario	Local Population
Normal	Ingests off-site water Ingests off-site foods Inhales downwind air
Farming	Ingests on-site foods Ingests off-site water
Eroded Trench	Inhales suspended material at geographic centroid Direct exposure from plume

In Chapter 2 a detailed discussion is found of the methodology employed in developing the quantitative models for evaluating environmental transport pathways and calculating the radiation doses, associated risks and potential health effects to individuals and selected populations. Appendices A and B provide background information in support of the models employed in the code.

Chapter 3 provides a description of the structure of the PRESTO-EPA-POP code and its subroutines.

Chapter 4 provides a detailed description of the output data as produced by the PRESTO-EPA-POP code.

The PRESTO-EPA-POP Users Manual (EPA85b) provides detailed information and guidance in the use of the code together with sample problems and a source listing of the code.

2. DISCUSSION OF METHODOLOGY

2.1 ENVIRONMENTAL TRANSPORT PATHWAYS

Pathways for environmental transport of radionuclides considered by the model are shown in Figures 2-1 and 2-2. Pathways of environmental transport of radionuclides that involve water (Figure 2-1) include both surface water and groundwater. Water may leave the trench through the bottom or by overflowing. The user may also choose to allow the soil surface to be contaminated initially by operational spillage. Radionuclides in water near the soil surface may be transported to a surface water body or may enter the aquifer by deep seepage. Contaminated water may ultimately reach the local population by use of water from either a well or surface water. The regional basin population will be exposed to the radionuclides that enter the regional basin stream from the surface water body and groundwater. Surface water transport to the basin use point is assumed to take less than one year, while the groundwater transport takes a much longer time, depending on the distance of travel from the local to the basin use points and the distribution coefficient of the radionuclide. Radionuclides not used by the regional basin population are assumed to enter the ocean which acts as a radionuclide sink.

Atmospheric pathways for radionuclide transport considered by the model are illustrated in Figure 2-2. Material may reach the atmosphere from the site soil surface contaminated by overflow or operational spillage, or by the denuded trench following possible erosion of the total cap sometime in the future. The local population may ultimately be impacted by inhalation of or immersion in the suspended materials downwind,

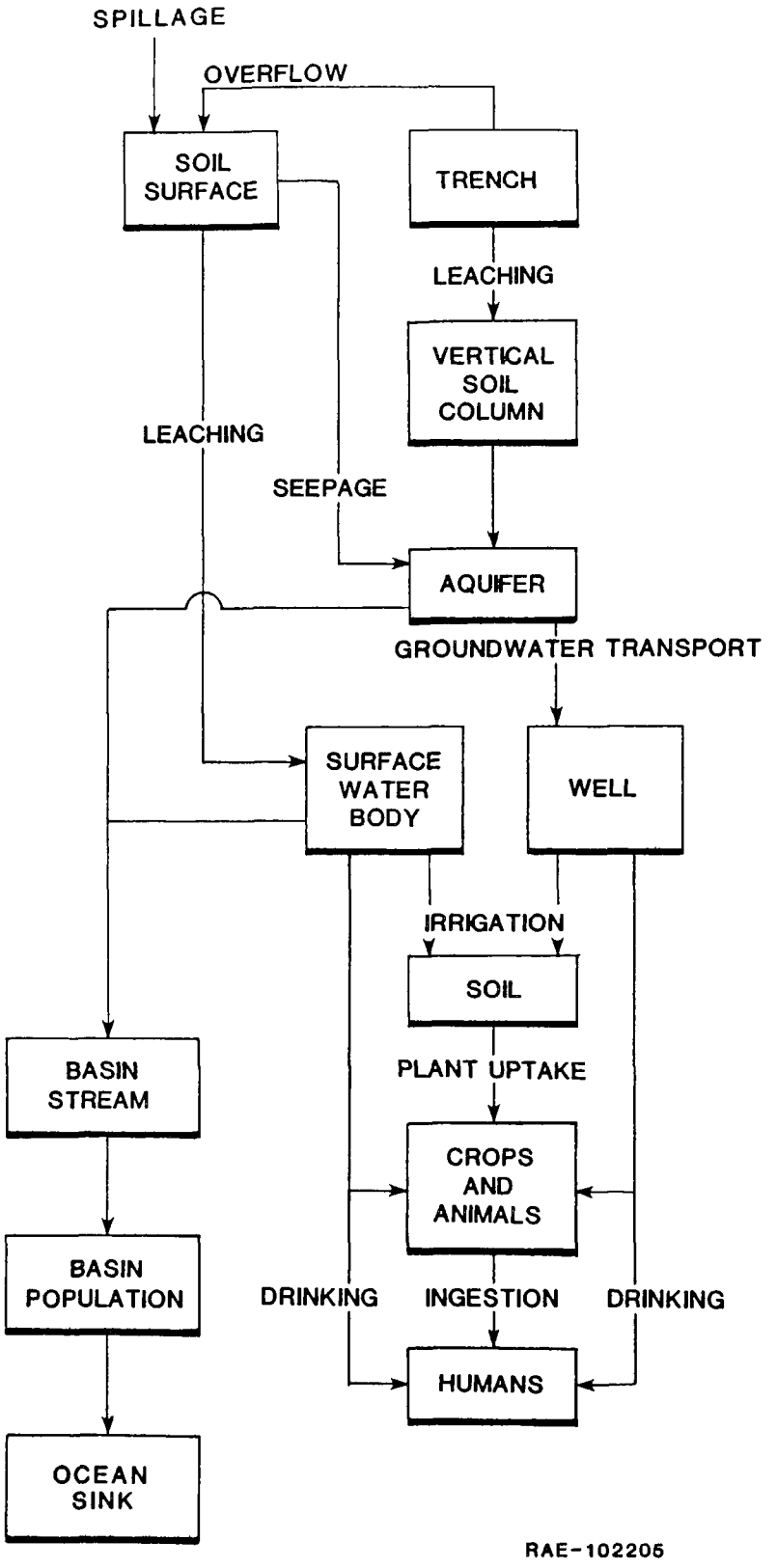

FIGURE 2-1. HYDROLOGIC ENVIRONMENTAL TRANSPORT PATHWAYS

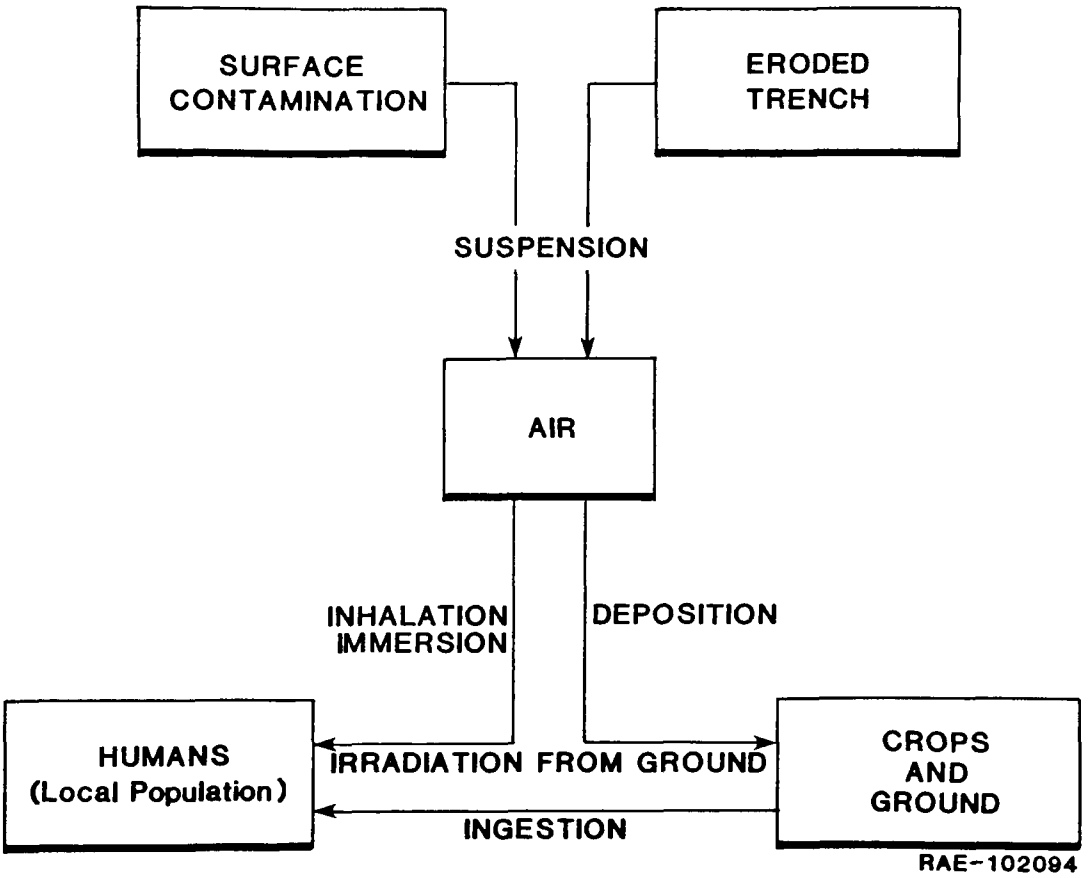

FIGURE 2-2. ATMOSPHERIC ENVIRONMENTAL TRANSPORT PATHWAYS.

by ingestion of crops contaminated following deposition on soil or crops, or by direct irradiation from ground surfaces. The impact of the atmospheric pathway on the regional basin population is considered to be negligible because of its distance from the disposal site and, therefore, no calculations are made for the atmospheric pathway.

The approach of the PRESTO-EPA-POP code to calculating radionuclide concentrations in the pertinent environmental media is described in the following two sections.

2.1.1 Transport Pathways Involving Water

At many sites, water is the most important medium for transport of radionuclides away from the trench. Whether the transport pathway is predominantly by groundwater or by overland flow of water, an important quantity is the amount of water entering the trench via infiltration through the trench cap.

The basic model for simulating the annual infiltration of precipitation assumes that it occurs in two modes: (1) annual average infiltration which is dependent upon the inherent physical properties of the trench cap and (2) infiltration through failed portions of the trench cap. Annual average infiltration depends upon soil properties, seasonal and annual temperatures, amounts of precipitation, rates of evapotranspiration, and other similar physical parameters. Infiltration through failed portions of the trench cap increases with time from zero at the beginning as the percent of trench cap failure increases. The year when failure begins, the rate of failure, and the final percent of failure are specified by the user and determine the amount of infiltration due to cap failure.

Infiltration Through Trench Cap - The basic model for simulating the annual infiltration through the trench cap assumes a portion of the trench cap will fail and allow the precipitated water to drain into the trench. The fraction of the trench which fails is assumed to vary with time.

Due to the distinct nature of the infiltration mechanism between the intact portion and the failed portion of the trench cap, the annual infiltration through the trench cap is divided into separate components.

On the intact portions of the cap, the normal infiltration rate is calculated by the method developed by Hung (Hu83b) which is described in Appendix A. For the failed portion of the cap, the infiltration is the sum of rainfall plus irrigation. Therefore, the volume of water entering the trench annually is calculated by

$$W_T = A_T[f_c(P_a+I_a)+(1-f_c)W_a] \qquad (2\text{-}1)$$

where

W_T = volume of water entering trench in current year (m^3)

A_T = area of trench (m^2)

f_c = fraction of trench cap that has failed (unitless)

P_a = annual precipitation (m)

I_a = annual irrigation (m)

W_a = annual infiltration (m)

The value of W_T is added to the standing trench water from the earlier year to calculate the maximum depth of standing water in the trench for the current year.

The component of annual infiltration through the intact portion of the trench cap, W_a, is estimated by employing the infiltration model developed by Hung (Hu83b, Appendix A). The model simulates the rate of infiltration by solving system equations which describe the dynamics of overland flow, subsurface flow, and atmospheric dispersion systems. The basic equations employed in the model are

$$Q_o = \frac{\sin^{1/2} a}{n} H^{5/3} \tag{2-2}$$

$$\frac{dH}{dt} = P - E_o - q_o - \frac{Q_o}{L} \tag{2-3}$$

$$E_o = \begin{cases} E_p & \text{when } P + \frac{H}{\Delta t} > E_p \\ P + \frac{H}{\Delta t} & \text{when } E_p > P + \frac{H}{\Delta t} > 0 \\ 0 & \text{when } P + \frac{H}{\Delta t} = 0 \end{cases} \tag{2-4}$$

$$q_o = \begin{cases} K_s & \text{when } P - E_o + \frac{H}{\Delta t} > K_s \\ P - E_o + \frac{H}{\Delta t} & \text{when } K_s > P - E_o + \frac{H}{\Delta t} > 0 \\ 0 & \text{when } P - E_o + \frac{H}{\Delta t} = 0 \end{cases} \tag{2-5}$$

$$q_i = \begin{cases} K_s & \text{when } Z_g < Z_{max} \\ 0 & \text{when } Z_g = Z_{max} \end{cases} \tag{2-6}$$

$$\frac{dZ_g}{dt} = (q_i - q_o + q_t)/W_g \tag{2-7}$$

$$\begin{aligned} q_L &= -D_e W_p / Z_p + K_e \\ q_L &< E_p - E_o \end{aligned} \tag{2-8}$$

$$q_v = -(E_p - E_o)\left[1 + \frac{0.5Z_p}{0.66(W_p + W_g)}\right]^{-1} \tag{2-9}$$

$$dZ_p/dt = -(q_p + q_t)/W_p \tag{2-10}$$

$$q_t = \begin{cases} q_o & \text{when } Z_p > 0 \\ 0 & \text{when } Z_p = 0 \end{cases} \tag{2-11}$$

$$\text{and } q_p = -\text{Max}(\,|\,q_L\,|\,,\,|\,q_v\,|\,) \tag{2-12}$$

where

Q_o = rate of overland flow per unit width of trench cover (m^3/m-hr)

H = average depth of overland flow over the entire trench cover (m)

L = length of slope or half of trench width (m)

n = Manning's coefficient of roughness

a = average inclination of the trench cover (m/m)

P = rate of precipitation (m/hr)

E_o = rate of evaporation from the overland flow (m/hr)

q_o = rate of percolation from the overland flow system (m/hr)

E_p = evaporation potential (m/hr)

q_i = flux of moisture infiltrating into the trench (m/hr)

q_L = flux of pellicular water transported in the liquid phase (m/hr)

K_s = saturated hydraulic conductivity of the soil (m/hr)

Z_g = deficit of gravity water (m)

Z_{max} = maximum deficit of gravity water, equivalent to the thickness of the trench cover (m)

W_g = component of wetness for the gravity water under a fully saturated condition, numerically identical to the porosity for the gravity water (unitless)

q_L = flux of pellicular water transported in the liquid phase (m/hr)

W_p = component of wetness for the pellicular water under fully saturated condition and is numerically identical to the porosity for pellicular water (unitless)

Z_p = deficit of the pellicular water (m)

D_e = hydraulic diffusivity at equivalent wetness (m^2/hr)

K_e = hydraulic conductivity at equivalent wetness (m/hr)

q_v = flux of moisture being transported in the vapor phase (m/hr)

q_t = flux of moisture being transformed from gravity water to pellicular water (m/hr)

q_p = flux of pellicular water (m/hr)

The annual infiltration through the trench cap is then calculated by integrating the hourly infiltration over the entire year.

Trench Cap Modifications - The trench cap may fail by erosion or mechanical disturbance. In the case of erosion, the annual thickness of material removed from the trench cap by sheet erosion is calculated using an adaption of the universal soil loss equation (USLE) (Wi65).

The annual amount of erosion is subtracted from the cap thickness for the current year of simulation. If the remaining thickness is less than 1 cm, the cap is considered to be completely failed and f_c is set to 1.0. The USLE may be written as

$$I_L = f_R f_K f_L f_S f_c f_p f_D \qquad (2\text{-}13)$$

where

I_L = yearly sediment loss from surface erosion (tons/ha)

f_R = rainfall factor (f_R unit or 100 m-tons-cm/ha)

f_K = soil erodibility factor (tons/ha/f_R-unit)

f_L = slope-length factor (unitless)

f_S = slope-steepness factor (unitless)

f_C = cover factor (unitless)

f_P = erosion control practice factor (unitless)

f_D = sediment delivery factor (unitless)

The parameterization scheme of McElroy et al. (McE76) was used to specify site-specific values of the factors in Equation (2-13). The rainfall factor, f_R, expresses the erosion potential of average annual rainfall in the locality. The soil erodibility factor, f_K, is also tabulated by McElroy et al. as a function of five soil characteristics: percent silt plus very fine sand; percent sand greater than 0.1 mm; organic matter content; soil structure; and permeability. The factors, f_L and f_S, for slope-length and steepness account for the fact that soil loss is affected by both length and degree of slope. The PRESTO-EPA-POP code usage of USLE combines both factors into a single factor that may be evaluated using charts in McElroy et al. The factor, f_C, represents the ratio of the amounts of soil eroded from land that is treated under a specified condition to that eroded from clean-tilled fallow ground under the same slope and rainfall conditions. The erosion control practice factor, f_P, allows for reduced erosion potential by the effect of practices that alter drainage patterns and lower runoff rate and intensity. The sediment delivery ratio, f_D, is defined by McElroy et al. as the fraction of the gross erosion that is delivered to a stream. Units of I_L are converted to (m/yr) within the code. See Chapter 4 for a description of input units.

The second method of trench cap failure accounts for the possibility of mechanical disturbance due to human intrusion or some other means which might completely destroy portions of the cap. This phenomenon can be termed a partial failure, but in reality is a total failure of some part of

the cap. The code user may specify some rate of cap failure as shown in Figure 2-3. By specifying appropriate values for the (x,y) pairs in Figure 2-3, the user may selectively remove the cap from a portion of the trench area. Mathematically this function is represented by

$$f_C = \begin{cases} 0 \text{ if } t<NYR_1 \\ \dfrac{(PCT2-PCT1)(t-NYR_1)}{(NYR_2-NYR_1)} + PCT1 \text{ if } NYR_1 \leq t \leq NYR_2 \\ PCT2 \text{ if } t>NYR_2 \end{cases} \tag{2-14}$$

Even though PCT2 might be less than 1.0 in year NYR_2, the cap may ultimately fail completely by virtue of erosion. As f_C changes, the amount of water added to the trench annually also changes.

The amount of water leaving the trench annually via the trench bottom is calculated by

$$V_B = \frac{(D_W+L)I_T A_T}{L} \tag{2-15}$$

and

$$D_W = V_W/A_T W_T \tag{2-16}$$

where

V_B = volume of water leaving trench bottom annually (m^3/yr)

D_W = depth of water in trench during current year (m)

I_T = permeability of material below the trench (m/yr)

A_T = trench area (m^2)

V_W = volume of water in trench (m^3)

W_T = porosity of trench contents (unitless)

L = length of saturated zone (m)

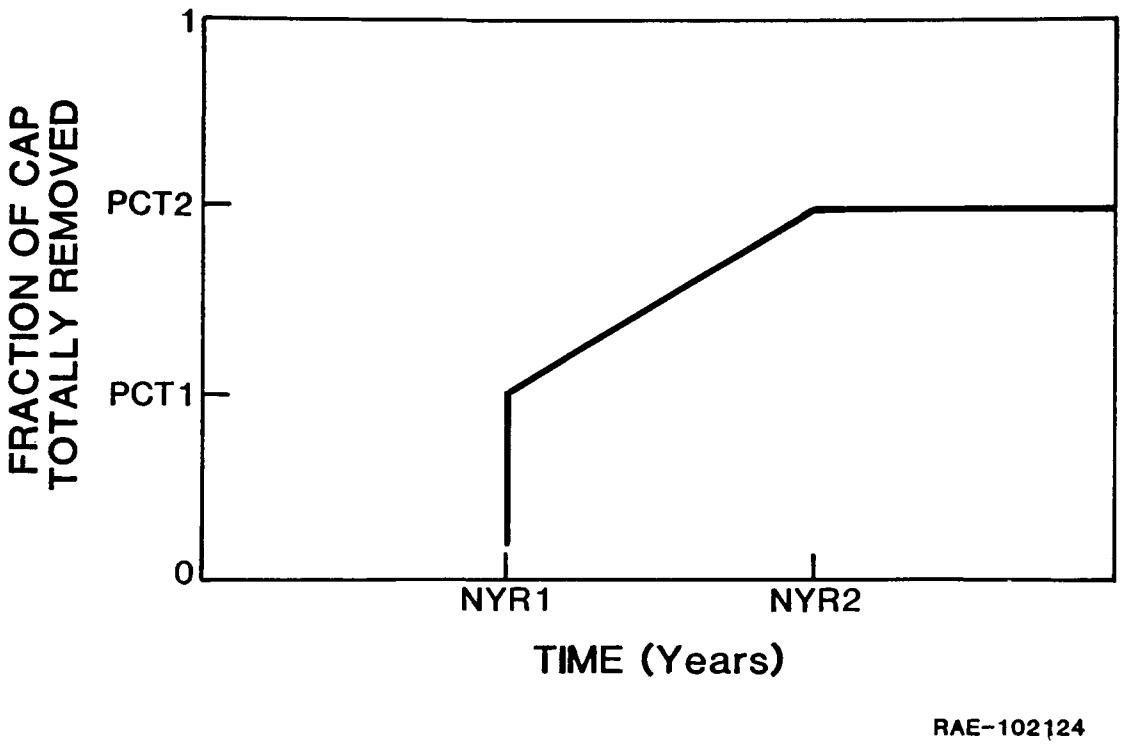

FIGURE 2-3. TRENCH CAP REMOVAL FUNCTION.

Water will overflow the trench if the maximum depth of standing water is greater than the trench depth. If this is the case, the overflow is calculated by

$$V_0 = (D_W - D_T) A_T W_T \tag{2-17}$$

where

V_0 = volume of water overflowing trench in a year (m^3)

D_W = depth of water in trench (m)

D_T = trench depth (m)

A_T = trench area (m^2)

W_T = porosity of trench material (unitless)

Water in the trench may be contaminated by contact with the waste material. The user must choose one of five methods shown in Table 2-1 to calculate the concentration of radionuclides in the trench water.

Options 1-4 are combinations of two factors: the amount of waste material contacted by water and the method of partitioning contamination between waste and water. In options 1 and 3, where the waste is in total contact with the trench water, it is assumed that the total volume of wastes in the trench is wetted each year as water trickles from top to bottom in the trench. In options 2 and 4, wherein the wastes are not totally immersed in trench water during the entire year, the submodel (immersed fraction) calculates the wetted fraction as the ratio of maximum water depth to trench depth. Leaching options, 1 and 2, utilize a distribution coefficient, K_d, to estimate the radionuclide concentration C_W^T in the trench water based on chemical exchange (Equation 2-18a)

TABLE 2-1

LEACHING OPTIONS SPECIFIED BY LEAOPT

Option	Leach Calculation Method
1	Total contact, distribution coefficient
2	Immersed fraction, distribution coefficient
3	Total contact, solubility
4	Immersed fraction, solubility
5	Release fraction

$$C_W^T = \frac{I_T f_W \quad TCON}{A_T(D_{max}W_T + D_T f_W K_{d2} \rho_W)} \quad \text{(Chemical Exchange Option)} \quad (2\text{-}18a)$$

$$C_W^T = \text{Min}\left[\frac{SN_c N_v}{M}, \frac{I_T f_W}{D_{max} A_T W_T}\right] \text{(Solubility Option)} \quad (2\text{-}18b)$$

and

$$TCON = \text{Min}\ [TINFL/PERMT,\ 1]$$

where

TCON = estimates the average annual fraction of the time that the waste is in contact with the trench water

TINFL = annual infiltration rate (m/yr)

PERMT = trench hydraulic conductivity (m/yr)

C_W^T = concentration of radionuclide in trench water (Ci/cm^3)

I_T = amount of nuclide in trench (Ci)

f_W = fraction of total waste contacted by water (=1 for LEAOPT = 1; <1 for LEAOPT = 2)

A_T = trench area (m^2)

D_{max} = maximum depth of standing water (m)

W_T = porosity within trench (unitless)

D_T = trench depth (m)

K_{d2} = distribution coefficient within waste for radionuclide (ml/g)

ρ_W = density of waste material (g/cm^3)

S = elemental solubility (g/ml)

M = mass of radionuclide (g/mole)

N_c = ratio (Ci/mole)

N_v = ratio (ml/m^3)

Leaching options 3 and 4 use a solubility factor to estimate the maximum concentrations of radionuclides in the trench water. The solubility option may be used when information concerning K_d values is not available.

Leaching option 5 allows the user to input an average annual fractional release of the total radionuclide inventory. This fraction is applied to each radionuclide and does not consider either K_d or solubility. Leaching option 5 is normally used for a solidified waste form. The model calculates the primary release of radionuclide from the waste form by using a user specified constant fractional leach rate. To accommodate the hydrodynamic effects, the released radionuclides are then adsorbed by the waste form according to Equation 2-18 to calculate the actual rate of release out of the trench. This calculation accounts for the adsorption effects inside and outside of the waste form.

Waste containers can inhibit nuclide leaching until they loose their integrity. This period of time may be controlled by their design, structural strength, and material. In PRESTO-EPA-POP, this effect is achieved by multiplying the radionuclide concentration in the trench water by the fractional container fracture factor (CFF) that is time dependent. The fraction CFF is set to zero while all of the containers are intact. Once the container starts to fracture, CFF is assumed to increase linearly to a maximum value of 1, which represents failure of all of the containers.

Transport Below Trench - Once radionuclides have been leached out of the waste in the trench and migrated through the surrounding trench walls or bottom, they are transported vertically to the aquifer and then horizontally through the aquifer to a well. The velocity of radionuclide

transport is retarded, relative to the movement of water, by vertical and horizontal retardation factors, R_V and R_H, as explained below:

The groundwater flow in the vertical reach is assumed to be saturated or partially saturated. The degree of saturation is used to calculate the water velocity, V_v and the vertical retardation factor, R_V. The degree of saturation, SSAT, is either read in as an input parameter, or is calculated from the equation

$$SSAT = RESAT + (1-RESAT) \left(\frac{ATINFL}{PERMV}\right)^{0.25} \qquad (2\text{-}19a)$$

where

RESAT = residual moisture content, expressed in a fraction of total water content when saturated (unitless)

ATINFL = average exfiltration rate (m/yr)

PERMV = vertical saturated hydraulic conductivity (m/yr)

Equation (2-19a) is based on approximate expressions for the fraction of saturation (Cla78, McW79). The exponent, 0.25, is generally a function of soil type, but has been assigned a conservative fixed value for simplicity. The residual moisture content, RESAT, is an input parameter that is generally identical to the input parameter W_p of the INFIL submodel. The parameter ATINFL is the average trench exfiltration rate. When there is no overflow of trench water, the rate is calculated by the expression

$$ATINFL = [PCT2\ (PPN+XIRR)+(2-PCT2)\ XINFL]0.5 \qquad (2\text{-}19b)$$

where

PCT2 = maximum fraction of trench cap failure (unitless)

PPN = annual precipitation rate (m/yr)

XIRR = annual irrigation rate (m/yr)

XINFL = infiltration rate through the intact trench cap (m/yr) (calculated by the INFIL subroutine)

Vertical water velocity V_V (m/yr), and the vertical retardation factor R_V (unitless) are calculated as follows:

$$V_V = ATINFL/(PORV \cdot SSAT) \tag{2-19c}$$

$$R_V = 1 + (BDENS \cdot XKD_3)/(PORV \cdot SSAT) \tag{2-19d}$$

where

BDENS = host formation bulk density (g/cm^3)

XKD_3 = distribution coefficients for the host formation (ml/g)

PORV = subsurface porosity (unitless)

On the other hand, the horizontal retardation factor, R_H, is calculated as

$$R_H = 1 + (BDENS \cdot XKD_4)/PORA \tag{2-19e}$$

where

XKD_4 = distribution coefficient of the aquifer (ml/g)

PORA = aquifer porosity (unitless)

Finally, the vertical horizontal transit time, t_V (yr), and t_H (yr), are calculated according to

$$t_V = \frac{D_V R_V}{V_V}, \qquad t_H = \frac{D_H R_H}{V_H} \tag{2-20}$$

where

D_V = distance from trench to aquifer (m)

D_H = length of aquifer flow from trench to well (m)

V_V = vertical water velocity (m/yr)

V_H = water velocity in aquifer (m/yr)

Retardation, R_V and R_H, are as previously defined.

The breakthrough time, which is the time required for a radionuclide to travel from the bottom of the trench to the well, is the sum of the vertical and horizontal transit times. A second horizontal transit time is also calculated by the model, which is the time it takes for a radionuclide to migrate in the aquifer from the well point to the point of release to the basin stream. This transit time is calculated using the corresponding horizontal distance from the well to the basin stream.

The transport of radionuclides in the aquifer is evaluated by employing Hung's groundwater transport model (Hu81, Hu86, Appendix B). The basic equations for the model as adopted from Hung are

$$\dot{Q} = zQ_0 \ (T-RL/V) \ \mathrm{Exp}(-\lambda t) \qquad (2\text{-}21)$$

and

$$z = \int_0^\infty \frac{0.5(RP/\pi W^3)^{1/2} \ \mathrm{Exp}\ [-N_d W-(PW/4R)(R/W-1)^2]\ dW}{\mathrm{Exp}(-RN_d)} \qquad (2\text{-}22)$$

$$= \frac{\mathrm{Exp}\left[\frac{P}{2} - \frac{P}{2}\sqrt{1 + \frac{4RL\lambda}{PV}}\right]}{\mathrm{Exp}\ (-\frac{RL\lambda}{V})}$$

where

z = a correction factor to compensate for the dispersion effect

R = retardation factor, R_V or R_H

P = Peclet number, $V_V D_V/d$ or $V_H D_H/d$

W = dimensionless time, yV/L

N_d = decay number, $\lambda L/V$

L = flow length, D_V or D_H (m)

V = water flow velocity, V_V or V_H (m/yr)

t = transit time, t_V or t_H (yr)

d = dispersion coefficient (m^2/yr)

= radiological decay constant (yr^{-1})

y = dummy time variable (yr)

$\dot{Q}$ = rate of radionuclide transport at distance L from the source (Ci/yr)

$\dot{Q}_o$ = rate of radionuclide released at the source (Ci/yr)

T = time of simulation (yr)

To calculate the radionuclide concentration at the well point, the rate of groundwater flow in the plume of contamination at the well point is calculated by

$$W_A = V_A P_A D_A \left[\sqrt{A_T} + 2 \cdot \tan(a/2) D_H \right] \qquad (2\text{-}23a)$$

where

W_A = the rate of contaminated water available for removal from well (m^3/yr)

V_A = groundwater velocity (m/yr)

P_A = porosity of aquifer material (unitless)

D_A = thickness of the aquifer (m)

a = constant angle of spread of the contaminant plume in the aquifer (radians)

A_T = trench area (m^2)

D_H = trench to well distance (m)

The angle "a" is the dispersion angle of a contaminated plume in the water in an aquifer. This dispersion angle may be empirically determined (e.g., by field dispersion tests wherein the angle of dispersion is determined from measurements of chemical, conductivity, or radioactivity tracers in water from a series of boreholes downstream across the plume) or it may be estimated. The use of a dispersion angle is consistent with published characterizations of the horizontally projected profile of a chemical contamination front as it moves through an aquifer (Sy81).

The radionuclide concentration in the well water, C_W (Ci/m^3), is calculated by

$$C_W = \frac{\dot{Q}}{W_A} \tag{2-23b}$$

The total water demand, V_U, including drinking water, cattle feed, and crop irrigation, is calculated by

$$V_U = [3.9X10^7 W_I f_I L_I + U_W L_H + 1.5X10^4 L_A] N_p \tag{2-24}$$

where

V_U = annual well water demand in liters (l/person yr)

$3.9X10^7$ = 4492 m^2 irrigated per person X 8760 hr/yr

W_I = irrigation rate per unit area (l/m^2 hr)

f_I = fraction of year when irrigating (unitless)

U_W = individual annual water consumption (l/person yr)

L_H = fraction of drinking water obtained from well water

$1.5x10^4$ = water fed to cattle consumed by humans (l/person yr)

L_A = fraction of cattle feed water obtained from well water

N_p = size of the population (persons)

L_I = fraction of irrigation water obtained from well water

If the calculated total water demand, V_U, exceeds the flow rate of the contaminated plume, W_A, the concentration of radionuclides in the pumped out water is recalculated using the actual volume of pumping to correct for the dilution effect from the non-contaminated groundwater. Units of V_U are converted to cubic meters within the code.

The calculated concentrations of radionuclides in well water are averaged over the length of the simulation and used by the food chain and human exposure parts of the code for the drinking water and cattle feed pathways.

Trench Overflow Transport and Stream Contamination - As previously mentioned, water will overflow the trench onto the soil surface when the maximum depth of standing water is greater than the trench depth. If this occurs, radionuclides will be added to the surface inventory of radionuclides deposited by initial operational spillage. The surface soil will then have a component adsorbed by the soil with concentration C_S^S (Ci/kg) and a component of contaminated water in the surface soil of C_W^S (Ci/m^3). The material adsorbed by the soil will remain in the soil and becomes a source term for resuspension and atmospheric transport (this process is discussed in Section 2.1.2). The contaminated water in the surface soil is available to enter nearby surface water bodies via overland flow, or percolate down to the aquifer.

Radionuclides dissolved in the soil water may either be transported to the stream by overland flow or to the deep soil layers by percolation. The amount of each radionuclide added to the stream is represented by the product of C_W^S, the radionuclide concentration in the surface soil water, and the annual volume of runoff from the contaminated soil surface, WSTREM. The value of C_W^S for each radionuclide is calculated by

$$C_W^S = \frac{10^3 I_S}{K_{d1} M_S + M_{W2}/\rho_a} \tag{2-25}$$

where

C_W^S = radionuclide concentration in surface soil water (Ci/m^3)

I_S = amount of radionuclide on surface (Ci)

K_{d1} = distribution coefficient for surface soil region (ml/g)

M_S = mass of soil in contaminated region (kg)

M_{W2} = mass of water in contaminated soil region (kg),

ρ_a = density of water (1 g/cm^3)

10^3 = conversion factors used for K_d(1 ml/g = 1 $m^3/10^3$ kg) and for d_W (1 g/cm^3 = 10^3 kg/m^3).

Equation 2-25 is used to compute the concentration of radionuclides in the surface soil interstitial water.

The radionuclide concentration in the contaminated surface soil region, C_S^S, is calculated using

$$C_S^S = 10^{-3} C_W^S K_{d1} \tag{2-26}$$

The contaminated region of surface soil is defined by the user in terms of length, S_L (m), width, S_W (m), and depth, S_D (m). These parameters allow the calculation of soil mass (M_S) and the water mass (M_W) in the contaminated soil region by

$$M_S = 10^3 \rho_s S_W S_L S_D, \quad M_W = 10^3 W_S S_W S_L S_D \tag{2-27}$$

where

ρ_s = soil bulk density (g/cm^3)

W_S = soil porosity (unitless)

10^3 = (cm^3 kg/m^3 g) for soil and (kg/m^3) for water

Water falling on the contaminated soil region may either evaporate, run off, or infiltrate. Of the liquid, a certain fraction of the total precipitation, f_r, will enter the stream annually.

The amount of water that enters the stream from runoff of the contaminated region is given by

$$W_S = f_r P S_W S_L \tag{2-28}$$

The amount of water that enters deep soil layers and eventually the aquifer is given by

$$W_D = W_a S_W S_L \tag{2-29}$$

where W_a is the yearly farmland infiltration parameter.

The annual contribution of radionuclides from the contaminated surface soil region to the stream, R_S, is then the product of W_S and the radionuclide concentration in the surface soil water C_W^S (Equation 2-25). The amount of each radionuclide annually entering the deep soil layers from the contaminated surface soil region is the product of W_D and C_W^S. The concentration of radionuclides in the stream is the quotient of R_S and the annual flow rate of the stream.

As with water removal from the well, the amount of each radionuclide removed from the stream is conserved by using

$$I_r = [3.9 \times 10^7 W_I f_I S_I + U_W S_H + 1.5 \times 10^4 S_A] N_p \quad C_W^R \tag{2-30}$$

where

I_r = annual amount of nuclide removed from stream (Ci)

C_W^R = radionuclide concentration in stream (Ci/m3)

S_I = fraction of irrigation water obtained from stream

S_H = fraction of drinking water obtained from stream

S_A = fraction of cattle feed water obtained from stream

Other parameters are the same as defined in Equation (2-24).

If I_r is larger than the annual input of that nuclide to the stream, R_S, then the radionuclide concentration in the stream is recalculated referencing the water volume removed from the stream rather than the stream flow by

$$C_W^R = \frac{R_S}{V_U} \tag{2-31}$$

Mean concentrations of each radionuclide in well water and stream water are calculated for the appropriate number of simulation years by dividing the sum of the annual radionuclide concentrations in the well water and the stream water by the length of the simulation.

2.1.2 Atmospheric Transport Sources and Pathways

For some sites, atmospheric transport of radionuclides may be a major transport mechanism. Therefore, careful consideration must be given in the formulation of the atmospheric transport model. Another factor which was considered in generating PRESTO-EPA-POP was the desire to minimize the amount and complexity of input data. The compromise solution to these conflicting considerations was to employ in the code a simplified, compact algorithm suitable for those sites where the population is concentrated into a single, small community, and to make provision in the input data set to allow the code user to enter an externally computed population average value for the air concentration, X, to source strength, Q, ratio. An externally computed X/Q ratio should be used for complex population

distributions. An example of a code which could be used for determining X/Q in such cases is AIRDOS-EPA (Moo79).

In most cases, the uncertainties in the computed atmospheric source strength for contaminated areas are larger than the differences between the internally computed and externally determined (using a code such as AIRDOS-EPA) X/Q ratios. Use of an external code has several advantages, however, the most salient being that explicit specification of complex population distributions and the site wind rose removes the possibility of the code user making errors of judgement in determining population centroids.

Internal Model Capability and Formulation - The atmospheric transport portion of the PRESTO-EPA-POP code will be discussed in two parts: (a) a description of source strength computation and (b) a discussion of the calculation of atmospheric concentration at the residence site of the specified at-risk population. For most applications, PRESTO-EPA-POP is expected to be applied to a site of known population distribution, and the user must input geographical and meteorological parameters characterizing the population site and its relationship to the low-level waste disposal area. The formulation of atmospheric transport discussed herein is not intended to automatically identify regions of high risk; rather, it is formulated to calculate risk-related parameters for a particular, identified site.

Where population health effects are to be determined, the geometric population centroid specified by the user is the point for which a 22.5-degree sector average ground level air concentration is determined. Where the population distribution subtends from the waste disposal area an

angle significantly greater than 22.5 degrees, the user should run the code separately for each sub-population. A mean yearly value for the sector-averaged atmospheric concentration is computed by PRESTO-EPA-POP and is input to DARTAB for use in computing external exposures.

The most common approach used for estimating the atmospheric concentration and deposition of material downwind from its point of release is the Gaussian plume atmospheric transport model (Sl68). This approach is versatile and well documented. We have chosen to implement a Gaussian plume transport code called DWNWND (Fi80a) as a module, in subroutine form, in the PRESTO-EPA-POP code.

User inputs for the atmospheric transport simulation allow specification of a surface radionuclide concentration at the waste disposal site. Parameters used here include the initial surface radionuclide inventory and the chemical exchange coefficient for surface soils. The portion of radionuclide sorbed onto soil particles is considered available for transport. A source strength is computed based either on a time-dependent (monotonically decreasing) resuspension factor or a process-dependent mechanical suspension variable. The given LLW site is described by meteorological variables including:

F_W = fraction of the year wind blows toward at-risk individuals

H = source height (m)

H_L = lid height (m)

S = stability class

T_d = type of dispersion formulation

H_r = Hosker roughness parameter (m) (about .01 of the actual physical roughness)

u = wind velocity (m/s)

V_d = deposition velocity (m/s)

V_g = gravitational fall velocity (m/s)

x = distance from source to receptor (m)

Source Term Characterization - The release rate for atmospheric transport is termed the source strength. In PRESTO-EPA-POP, the source strength is directly dependent on the surface soil-sorbed radionuclide concentrations from operational spillage and trench overflow, C_G (Ci/m^2). The source strength is the arithmetic sum of two parts: that parameterized by a time-dependent resuspension factor, R_e, (An75) and that parameterized by a resuspension rate, R_r, (He80).

First, the wind-driven suspension component is described. If the time-dependent resuspension factor is defined as

$$R_e = R_{e1}\exp(R_{e2}\sqrt{T})+R_{e3} \tag{2-32}$$

where T is elapsed time (days) and R_e has units of inverse meters, then the atmospheric concentration above the site, C_A, is given by

$$C_A = R_e C_G \tag{2-33}$$

and

$$C_G = C^S \rho_s S_D 10^3 \tag{2-33a}$$

Using Anspaugh's values of $1x10^{-4}$, -0.15, and $1x10^{-9}$ for R_{e1}, R_{e2}, and R_{e3}, respectively, the value of R_e calculated as above is probably conservative for humid sites. As additional data from humid sites become available, model users may wish to update the equation used for computing R_e.

The value of elapsed time appearing in Equation (2-32) is computed from the start of the simulation. It is, therefore, correct for the initial surface inventory, but not for incremental additions thereto, which may occur at later times. However, when later additions result from trench overflow, they will likely consist of dissolved material and would likely act as surface depositions of mobile particulates. It is, therefore, assumed that a steady-state asymptotic value of R_e is for most sites appropriate for later additions to the surface inventory.

The user wishing to specify a time independent windblown resuspension factor may do so by setting the values of R_{e1} and R_{e2} to zero. When this is done, determination of windblown suspension of all contributions to the surface inventory will be treated identically, regardless of time of occurrence.

In the above expression, C_A is the atmospheric concentration of radionuclide immediately above the site at a height of about 1 m (Shi76), for a site of large upwind extent. Large upwind extent may be interpreted as exceeding the atmospheric build-up length, given by $u\ H_D/V_g$, where u is wind velocity in m/s, H_D is the mixing height (=1 m), and V_g is the gravitational fall velocity. The representative site extent used in the PRESTO-EPA-POP code is the square root of the site area, A (which is characterized by $S_L S_W$), and a tentative correction fraction, F. The correction factor is computed using the equation

$$F = \frac{V_g \sqrt{S_L S_W}}{u H_D} \qquad (2\text{-}34)$$

With the stipulation that the value used for F may not exceed unity, the source term component (Ci/s) for windblown suspension is given by

$$Q_r = C_A H_D u F \sqrt{A} \tag{2-35}$$

The second source component results from mechanical disturbance of site surface soil. Mechanical disturbance occurs during a user-specified interval. Within this interval, the fraction of time per year that the disturbance occurs is F_{mech}. The source term component for mechanical disturbance is parameterized by the resuspension rate, R_r, having units of inverse seconds, as

$$Q_{mech} = C_G A R_r F_{mech} \tag{2-36}$$

The net source strength for the site is the sum of these components:

$$Q = Q_r + Q_{mech} \tag{2-37}$$

Transport Formulation - The PRESTO-EPA-POP code uses a Gaussian plume atmospheric transport model, which is an extension of an equation of the form (S168)

$$X = \frac{Q}{2\pi u \sigma_y \sigma_z} \exp\left[-\frac{y^2}{2\sigma_y^2}\right]\left\{\exp\left[-\frac{1}{2}\left(\frac{z-H}{\sigma_z}\right)^2\right] + \exp\left[-\frac{1}{2}\left(\frac{z+H}{\sigma_z}\right)^2\right]\right\} \tag{2-38}$$

This equation describes Gaussian distribution, where X represents the radionuclide concentration, Q the source strength, and H the corrected source release height to be discussed later. Dispersion parameters, σ_y and σ_z, are the standard deviations of the plume concentration in the horizontal and vertical directions, respectively. The aerosol is assumed to be transported at a wind speed (height-independent) u to a sampling position located at surface elevation z and transverse horizontal distance y from the plume center. Mass conservation within the plume is insured by assuming perfect reflection at the ground surface. This is accomplished by

the use of an image source at an elevation -H, which leads to the presence of two terms within the braces, and to the factor 1/2. A correction for plume depletion will be discussed later. Equation (2-38) may be obtained from any of several reasonable conceptual transport and dispersion models.

Atmospheric transport at several sites of possible interest to individuals evaluating consequences of low-level waste transport have also been considered elsewhere. These include Hanford, Washington (Fi81, Mi81), Savannah River, South Carolina (Fi80b), and Brookhaven, New York (Si66).

Implicit in Equation (2-38) is the assumption that the plume centerline height is the same as the release height H. In practice, the plume may be considered to originate at some height, H, with respect to the population at risk. Some situations, such as the existence of a ridge between the disposal site and the population centroid, may dictate use of an effective height greater than H (e.g., the ridge height). The plume thus has an effective height, H_{eff}, at which the plume may be considered to originate. This effective value should be used instead of the actual stack height as the starting point of Gaussian plume calculations. If the particulates in the effluent have an average gravitational fall velocity, V_g, the plume centerline will tilt downward with an angle from the horizontal, the tangent of which is V_g/u. The elevation of the plume centerline at a distance x downwind is then

$$H = H_{eff} - xV_g/u \quad \text{for} \quad H \geq 0 \tag{2-39}$$

and it is this corrected value that is used to compute the aerosol concentration at a distant point.

Effects of a Stable Air Layer on Transport - The Gaussian plume formulation has been modified for use in PRESTO-EPA-POP to account for the presence of a stable air layer at high altitudes. Upward dispersion of the plume subsequent to release is eventually restricted when the plume encounters an elevated stable air layer or lid at some height H_L. Pasquill has summarized some reasonable approximations to the modified vertical concentration profile for various ranges downwind which are used here (Pa76). The limiting value of σ_z may be defined as

$$\sigma_z(\text{limit}) = 2(H_L - \frac{H}{2})/2.15 \qquad (2\text{-}40)$$

This equation follows from setting the ground-level contribution to the plume from an image source located above the stable air layer to one tenth the value of the plume concentration. It is assumed that the limiting value of σ_z calculated in this manner is correct for distances beyond this point. For shorter downwind distances where the vertical dispersion coefficient σ_z is less than σ_z(limit), the Pasquill-Gifford value of σ_z is used. For greater downwind distances where σ_z is greater than or equal to σ_z(limit), the value of σ_z(limit) given in Equation (2-40) is used instead. The lid height is a user-specified value in the PRESTO-EPA-POP code. For LLW applications, the source height will usually be sufficiently low that the influence of H_L will be small. For some sites, however, the influence of an intervening ridge may necessitate a larger effective source height.

Effects of Plume Depletion - The plume is depleted at ground level during travel as the particulates are deposited. Both fallout and electrochemical deposition may be important considerations, and ground

cover characteristics are of major importance. Under certain obvious conditions, washout is also of importance, but those conditions have not been included within this model. Fallout is partially quantified in the V_g term defined earlier. Near ground level the deposition process is often characterized by a deposition velocity V_d (Gif62, Mu76a, Mu76b). The deposition rate W is defined by

$$W = V_d X \tag{2-41}$$

Where

X = radionuclide concentration in air (Ci/m^3).

The magnitude of the plume depletion within the downwind sector may be found by integrating across the plume. Using Equation (2-38) and setting z = 0 it is found that

$$\frac{dQ}{dx} = \int_{-\infty}^{\infty} V_d \, X dy$$
$$= \int_{-\infty}^{\infty} \frac{V_d}{u\pi} \frac{Q}{\sigma_y \sigma_z} \exp\left[- \frac{y^2}{2\sigma_y^2} - \frac{H^2}{2\sigma_z^2} \right] dx \tag{2-42}$$

By performing the indicated quadrature across the plume and further integrating along the longitudinal direction to express the loss of release agent as a multiplicative factor, it can be shown (Mi78) that the ratio of the air concentration considering deposition processes, X_d, to the air concentration without regarding deposition, X, is

$$F_d = \frac{X_d}{X} = \exp\left\{ -\sqrt{\frac{2}{\pi}} \frac{V_d}{u} \int_0^x \frac{1}{\sigma_z} \exp\left[- \frac{H^2}{2\sigma_z^2} \right] dx \right\} \tag{2-43}$$

Since σ_z is a complicated empirical function of x, Equation (2-43) must be evaluated numerically.

In the PRESTO-EPA-POP applications, the average value of radionuclide concentration X, across a 22.5-degree downwind sector is the desired quantity. In this case, the trans-sector integration leads to the value 2.032 in the air concentration equation (Cu76). This value includes the $1/2\pi$ factor in Equation (2-38).

In conclusion, assuming that the radionuclide distribution is that of a Gaussian plume, we may compute the mean radionuclide concentration, X, at ground level for the 22.5-degree downwind sector by

$$X = \frac{2.032 F_d F_w Q}{u x \sigma_z} \exp\left[- \frac{H^2}{2\sigma_z^{\,2}} \right] \qquad (2\text{-}44)$$

The value of H in Equation (2-44) must be an effective source height. This value is corrected in PRESTO-EPA-POP for plume tilt as in Equation (2-39) and the accompanying discussion. For use in the PRESTO-EPA-POP code, H is on the order of 1 m for reasonably flat sites, but in many cases other values should be used to account for local site characteristics, e.g., for the presence of updrafts.

It has been noted that the choice of plume dispersion parameter σ_z is a user option in the PRESTO-EPA-POP code. Choice of appropriate parameterization depends on site meteorology, topography, and release conditions. The DWNWND code (Fi80a), which has been included as part of PRESTO-EPA-POP, includes a choice of eight parameterization schemes for plume dispersion and a choice of six stability classifications. The most often used dispersion parameterization scheme for the Gaussian plume is the Pasquill-Gifford

model. This is the approach most appropriate for long-term LLW assessment calculations. Likewise, unless site-specific meteorology dictates otherwise, the D stability category, denoting a neutral atmosphere, should be used.

Pasquill (Pa61, Pa74) considered ground level emission tracer studies and wind-direction fluctuation data and developed dispersion parameterizations for six atmospheric stability classes ranging from A, most unstable, through F, most stable. Pasquill's values are approximate for ground level emissions of low surface roughness (Vo77). These values were devised for small source distances (<1 km). The so-called Pasquill-Gifford form of this parameterization (Hi62) has been tabulated by Culkowski and Patterson (Cu76), and are used in this model.

2.1.3 Food Chain Calculations

Mean concentrations of radionuclides in air, stream water, and well water are calculated by the equations listed in Sections 2.1.1 and 2.1.2. This section describes how radionuclides in those and other environmental media are used to calculate human internal exposure and potential health effects.

Radionuclides in water may impact humans by internal exposure, directly from use of drinking water or indirectly from use of irrigation water used for crops. Radionuclides in air may impact humans by either external or internal radiological doses. External doses may result from immersion in a plume of contaminated air or by exposure to soil surfaces contaminated by deposition from the plume. Internal doses may result from inhalation of contaminated air or ingestion of food products contaminated

by deposition from the plume. Dose and health effects calculations are made by the DARTAB program (Be81) which is utilized as a subroutine in PRESTO-EPA-POP. DARTAB will be discussed in some detail later. Radionuclide input to DARTAB consists of the constant concentrations in air (person-Ci/m^3), constant concentrations on ground surface (person-Ci/m^2), constant collective ingestion rate (person-pCi/yr) and constant collective inhalation rate (person-pCi/yr). Calculation of each of these by PRESTO-EPA-POP will be discussed next.

Concentrations of radionuclides in air which affect the population or an individual are calculated as described in Section 2.1.2. It is assumed that the mean nuclide concentrations in air are constant during the total period of the simulation, as required, for input to DARTAB.

Concentration of each radionuclide on the ground surface, Q_S(pCi/m^2) is calculated using

$$Q_S = CSP + CSPO \tag{2-45}$$

where

Q_S = concentration of radionuclide on the ground surface at the populated area of interest (pCi/m^2)

CSP = radionuclide concentration in the soil used for farming due to atmospheric deposition (pCi/m^2)

CSPO = radionuclide concentration in the soil used for farming due to irrigation (pCi/m^2)

Appropriate unit conversions are made within the code.

The collective inhalation rate is calculated by multiplying the population size by the generic individual inhalation rate of radionuclides.

$$Q_{inh} = U_a \overline{C}_A \tag{2-46}$$

where

Q_{inh} = Ci/yr

U_a = inhalation rate (m^3/yr)

$\overline{C}_A$ = mean ground level radionuclide concentration at a point of interest (Ci/m^3)

The units of Q_{inh} are converted to person pCi/yr for input to the DARTAB subroutine.

The collective ingestion rate is the input to DARTAB that requires the most calculations by PRESTO-EPA-POP. Ingestion includes intake of drinking water, beef, milk, and crops. Except for drinking water, all of these media may be contaminated by either atmospheric processes or by irrigation.

The atmospheric deposition rate onto food surfaces or soil that is used in subsequent calculation of radionuclide content in the food chain is

$$d = 3.6X10^{15}\ \overline{C}_A V_d \tag{2-47}$$

where

d = mean rate of radionuclide deposition onto ground or plant surfaces (pCi/m^2 hr)

$\overline{C}_A$ = mean ground level radionuclide concentration at point of interest (Ci/m^3)

$3.6X10^{15}$ = sec-pCi/hr-Ci

V_d = deposition velocity (m/sec)

The following equation estimates the concentration $\overline{C}_v$ of a given nuclide in and on vegetation at the deposited location (except for H-3 and C-14):

$$\overline{C}_v = \frac{d\ R[1-\exp(-\lambda_e t_e)]}{Y_v \lambda_e} + \frac{B\ CSP}{P} \exp(-\lambda t_h) \qquad (2\text{-}48)$$

where $\overline{C}_v$ is measured in pCi/kg, d is defined as above, and

- R = the fraction of deposited activity retained on crops (unitless)
- λ_e = effective removal rate constant for the radionuclide from crops (hr^{-1}), where $\lambda_e = \lambda + \lambda_w$, is the radioactive decay constant and λ_w is the removal rate constant for physical loss by weathering
- t_e = the time period that crops are exposed to contamination during the growing season (hr)
- Y_v = the agricultural productivity or yield [kg(wet weight)/m^2]
- B = the radionuclide concentration factor for uptake from soil by edible parts of crops, [pCi/kg (dry weight) per pCi/kg dry soil]
- CSP = soil radionuclide concentration updated yearly (pCi/m^2)
- P = the effective surface density for topsoil [kg(dry soil)/m^2]
- t_h = time interval between harvest and consumption of the food (hr)

$$CSP = (CSPL + d\Delta t)\exp[\Delta t(-\lambda - \lambda_s)]$$

where

- CSP = soil radionuclide concentration for this year (pCi/m^2)
- $CSPL$ = soil radionuclide concentration for last year (pCi/m^2)
- d = mean rate of radionuclide deposition (pCi/m^2-yr)
- λ = radioactive decay constant (yr^{-1})
- λ_s = rate constant for contaminant removal (yr^{-1})
- Δt = time increment, equal to one year in PRESTO model

If farming is performed on the trench site, then the soil radionuclide concentration is calculated as

$$SOCON = 10^{12} S_D (C_W^S W_S + 10^3 C_S^S \rho_s)$$

where

SOCON = soil radionuclide concentration (pCi/m^2)

S_D = depth of contaminated surface region (m)

C_W^S = radionuclide concentration in interstitial water of contaminated surface region (Ci/m^3)

W_S = porosity of surface soil (unitless)

C_S^S = radionuclide concentration in soil of contaminated surface region (Ci/kg)

ρ_s = bulk density of surface soil (g/cm^3)

10^{12} = pCi/Ci

10^3 = $\frac{kg}{g} \cdot \frac{cm^3}{m^3}$

The rate constant for contaminant removal from the soil, R_S, is estimated using

$$\lambda_S = \frac{r_S}{(0.15)(8760)\left(\frac{1+\rho_S K_d}{W_S}\right)} \qquad (2\text{-}49)$$

where

λ_S = removal rate coefficient (hr^{-1})

r_S = watershed infiltration (m/yr)

ρ_s = soil bulk density (g/cm^3)

K_d = distribution coefficient (ml/g)

W_S = porosity (unitless)

0.15 = depth of soil layer (m)

8760 = hr/yr

Equation (2-48) is used to estimate radionuclide concentrations in produce and leafy vegetables consumed by humans and in forage (pasture grass or stored feed) consumed by dairy cows, beef cattle, or goats.

The concentration of each radionuclide in animal forage is calculated by use of the equation

$$C_f = f_p f_s C_p + (1-f_p f_s) C_s \tag{2-50}$$

where

C_f = the radionuclide concentration in the animal's feed (pCi/kg)

C_p = the radionuclide concentration on pasture grass (pCi/kg), calculated using Equation (2-48) with $t_h = 0$

C_s = the radionuclide concentration in stored feeds in pCi/kg, calculated using Equation (2-48) with t_h = 2160 hr or 90 days

f_p = the fraction of the year that animals graze on pasture (unitless)

f_s = the fraction of daily feed that is pasture grass when the animals graze on pasture (unitless)

The concentration of each radionuclide in milk is estimated as

$$C_m = F_m C_f Q_f \exp(-\lambda t_f) \tag{2-51}$$

where

C_m = the radionuclide concentration per liter in milk (pCi/l)

C_f = the radionuclide concentration in the animal's feed (pCi/kg)

F_m = the average fraction of the animal's daily intake of a given radionuclide which appears in each liter of milk (d/l)

Q_f = the amount of feed consumed by the animal per day (wet kg/d)

t_f = the average transport time of the activity from the feed into the milk and to the receptor (hr)

λ = the radiological decay constant (hr^{-1})

The radionuclide concentration in meat from atmospheric deposition depends, as with milk, on the amount of feed consumed and its level of contamination. The radionuclide concentration in meat is estimated using

$$C_F = F_f C_f Q_f \exp(-\lambda t_s) \tag{2-52}$$

where

C_F = the nuclide concentration in animal flesh (pCi/kg)

F_f = the fraction of the animal's daily intake of a given radionuclide which appears in each kilogram of flesh (d/kg)

C_f = the concentration of radionuclide in the animal's feed (pCi/kg)

Q_f = the amount of feed consumed by the animal per day (kg/d)

t_s = the average time from slaughter to consumption (hr)

Concentrations of radionuclides in foodstuffs that result from spray irrigation with contaminated water are estimated using essentially the same equations as for atmospheric deposition with the following differences: the concentration in vegetation, $\overline{C}_v$ is estimated using Equation (2-48), but a different value of the retention fraction, R, is used. For irrigation, the second term of Equation (2-48) is modified by a factor of f_I, fraction of the year during which irrigation occurs and the t_e in the exponent becomes t_w, equivalent to f_I in hours. For irrigation calculations, the deposition rate, d, in Equation (2-48) becomes the irrigation rate, I_r, expressed as

$$I_r = C_w W_I \tag{2-53}$$

where

I_r = radionuclide application rate (pCi/m^2 hr)

C_w = radionuclide concentration in irrigation water (pCi/l)

W_I = irrigation rate (l/m^2 hr)

The concentration in water, C_w, is an average of well and stream water weighted by the respective amounts of each that are used.

Another modification introduced for irrigation calculations is related to the radionuclide concentration in milk and meat where animal's intake of water was added to Equation (2-51) and (2-52), respectively. This becomes

$$C_m = F_m(C_f Q_f + C_w\ Q_w)\exp(-\lambda t_f) \tag{2-54}$$

$$C_F = F_f(C_f Q_f + C_w\ Q_w)\exp(-\lambda t_s) \tag{2-55}$$

where

Q_w = the amount of water consumed by the animal each day (l/d)

Once radionuclide concentrations in all the various foodstuffs are calculated, the annual ingestion rate for each radionuclide is estimated by

$$Q_{ing} = Q_v + Q_{milk} + Q_{meat} + Q_w \tag{2-56}$$

where the variables represent individual annual intakes of a given radionuclide via total ingestion, Q_{ing}, and ingestion of vegetation, Q_v, milk, Q_{milk}, meat, Q_{meat}, and drinking water, Q_w, respectively, in pCi/yr. The annual intakes via each type of food, Q_v for instance, are calculated as

$$Q_v = (C_I^v + C_A^v)\ U_v \tag{2-57}$$

where

Q_v = annual radionuclide intake from vegetation (pCi/yr)

C_I^v = radionuclide concentration in vegetation from irrigation (pCi/kg)

C_A^v = radionuclide concentration in vegetation from atmospheric deposition (pCi/kg)

U_v = individual annual intake of vegetation (kg/yr)

To satisfy the input requirements for DARTAB, the annual individual intakes are multiplied by the size of the population to calculate the collective ingestion annually.

As mentioned earlier, Equations (2-47) through (2-55) do not apply directly to calculations of concentrations of H-3 or C-14 in foodstuffs. For application of tritium in irrigation water, it is assumed that the concentration in all vegetation C_v is the same as the tritium concentration in drinking water; therefore

$$C_v = C_w \tag{2-58}$$

where C_v and C_w are in pCi/kg and pCi/l, respectively. The concentration of H-3 in animal's feed, C_f, is therefore also equal to C_w. Then, from Equations (2-54) and (2-55), the concentration of tritium in animal's milk and meat can be written as

$$C_m = F_m C_w (Q_f + Q_w) \tag{2-59}$$

$$C_F = F_f C_w (Q_f + Q_w) \tag{2-60}$$

where

C_m = concentration of tritium in milk (pCi/l)

F_m = fraction of the animal's daily intake of H-3 that appears in each liter of milk (days/l)

C_w = H-3 concentration in animal drinking water (pCi/l)

Q_f = animal's daily intake of forage (kg/d)

Q_w = cow's daily intake of water (l/d)

C_F = concentration of tritium in animal meat (pCi/kg)

F_f = fraction of the animal's daily intake of H-3 that appears in each kg of meat (d/kg)

The exponential term is neglected due to the relatively long half life of tritium as compared to transit times in the food chain.

The root uptake of C-14 from irrigation water is considered negligible and, therefore, has been set equal to zero.

For vegetation contaminated by atmospheric deposition of tritium, H-3 concentrations are calculated by

$$C_V = \frac{C_A}{h} (0.75)(0.5)(1x10^{15}) \qquad (2\text{-}61)$$

where

C_V = H-3 concentration in vegetation (pCi/kg)

C_A = concentration of H-3 in air (Ci/m^3)

h = absolute humidity of the atmosphere (g/m^3)

0.75 = ratio of H-3 concentration in plant water to that in atmospheric water

0.5 = ratio of H-3 concentration in atmospheric water to total H-3 concentration in atmosphere.

$1x10^{15}$ = $(1x10^{12}pCi/Ci)x(10^3g/kg)$

The mean ground level air concentration of H-3, C_A, is calculated using equations in Section 2.1.2.

For C-14, the concentration in vegetation is calculated assuming that the ratio of C-14 to be the natural carbon in vegetation is the same as that ratio in the surrounding atmosphere. The concentration of C-14 is given by

$$C_V = \frac{C_{Ar}}{0.16} (0.11)(1x1015) \qquad (2\text{-}62)$$

where

C_V = C-14 concentration in vegetation (pCi/kg)

C_A = mean ground-level concentration of C-14 in air (Ci/m^3), also calculated from equations given in Section 2.1.2

r = ratio of the total release time of C-14 to the total annual time during which photosynthesis occurs, $r \leq 1$

0.11 = fraction of the plant mass that is natural carbon

0.16 = concentration of natural carbon in the atmosphere (g/m^3)

$1x10^{15}$ = $(1x10^2 pCi/Ci)x(10^3 g/kg)$

2.2 DOSE AND HEALTH EFFECT CALCULATIONS

2.2.1 DARTAB Calculations

In Equations (2-49) through (2-59) it was shown how calculations are made of radionuclide concentrations in air, ground concentration, and annual ingestion and inhalation rates. These concentrations and rates are utilized by the DARTAB portion of the PRESTO-EPA-POP code to generate tables of radiological dose and resulting health effects. This section describes the mathematical calculations made within DARTAB. For the most part, the equations and text have been taken from the DARTAB documentation report (Be81), Section 2.3 entitled, "General Equations."

Radiological Doses - The annual dose committed to an individual at location k for the lth organ, ith nuclide, and jth exposure pathway is given by

$$D_{ijl}(k) = \frac{K_j E_{ij}(k)\ DF_{ijl}}{P(k)} \qquad (2\text{-}63)$$

where K_j contains any numerical factors introduced by the units of $E_{ij}(k)$, the exposure to the ith radionuclide in the jth pathway, DF_{ijl} is the dose rate factor of the ith radionuclide, the jth pathway and the lth organ, and P(k) is the exposed population at location k. Note that all E_{ij} and DF_{ijl} for various nuclides (index i) and organs (index l) have consistent units.

DARTAB performs three calculations and tabulations for dose rate and dose: (1) dose rate to an individual at a selected location, (2) dose rate to a mean or average individual, and (3) collective population dose rate. Table 2-2 lists units of DF_{ijl} and E_{ij} for each of the four pathways for selected individual dose calculations. Dose rates, D_{ijl}, are in mrad/yr.

Mean individual dose rates are calculated using

$$D_{ijl} = \frac{\sum_k P(k)\ D_{ijl}(k)}{\sum_k P(k)} \tag{2-64}$$

Note that in PRESTO-EPA-POP the impacted population is considered to reside at only one location (k = 1). Hence, calculations of mean individual dose rate are numerically equivalent to the sum of pathway doses for the selected individual dose rate. The collective dose rate for the exposed population is the product of D_{ijl} and the number of persons exposed. Units of the collective dose rate are person rad/yr.

The above dose rates may be expressed in a number of different combinations. The doses can be summed directly over pathways:

$$D_{il}(k) = \sum_j D_{ijl}(k) \tag{2-65}$$

or over all nuclides:

TABLE 2-2

UNITS OF EXPOSURE FACTOR E_{ij}, AND DOSE RATE FACTOR, DF_{ijl}, FOR SELECTED INDIVIDUAL DOSE RATE CALCULATIONS BY DARTAB

Pathway	Unit of Factor	
	E_{ij}	DF_{ijl}
Ingestion	(person-pCi)/yr	(mrad/yr)/(pCi/yr)
Inhalation	(person-pCi)/yr	(mrad/yr)/(pCi/yr)
Air Immersion	(person-pCi)/m^3	(mrad/yr)/(pCi/m^3)
Ground Surface Exposure	(person-pCi)/m^2	(mrad/yr)

$$D_{jl}(k) = \sum_i D_{ijl}(k) \tag{2-66}$$

The total dose to the lth organ at location k, D (k), is then

$$D_l(k) = \sum_j \sum_i D_{ijl}(k) \tag{2-67}$$

The dose equivalent (mrem), H, for the lth organ is given as

$$H_l(k) = QF_{(low\text{-}LET)}D_l(low\text{-}LET) + QF_{(high\text{-}LET)}D_l(k,high\text{-}LET) \tag{2-68}$$

where QF denotes the relative biological effect factor. The factor is defined for each organ or health effect.

To combine dose rates to different organs, a weighted sum is used

$$D_{ij}(k) = \sum_l W_l D_{ijl}(k) \tag{2-69}$$

where W_l are weighting factors for the various organ doses supplied by the user where

$$\sum_l W_l = 1 \tag{2-70}$$

Weighting factors developed by EPA for the various organs were used as input into DARTAB. The International Commission on Radiological Protection (ICRP79) has proposed a similar approach to adding organ doses.

Health Effects Estimates - The health effects and risk equivalent are computed in a manner similar to the dose calculations. The health effects or individual risk of premature death to an individual at location k for the lth cancer, ith radionuclide, and jth exposure pathway is given by

$$R_{ijl}(k) = 10^{-5}K_jE_{ij}(k)RF_{ijl}/P(k) \tag{2-71}$$

where K_j again serves to reconcile the units of $E_{ij}(k)$ and RF_{ijl}. The total individual risk represented by the exposure and intakes of all radionuclides through all pathways is given as

$$R(k) = 10^{-5} \sum_j K_j \sum_i E_{ij}(k) \sum_i RF_{ijl}/P(k) \tag{2-72}$$

and the health effects can be summed over pathways, radionuclides, or cancers. The mean or average individual risk is estimated in a similar way.

The collective health effects are expressed as the health effects rate. For example, the total equilibrium fatal cancer rate in an exposed population is

$$HE = \frac{10^{-5}}{T_e} \sum_j K_j \sum_k \sum_l E_{ij}(k) \sum_l RF_{ijl} \tag{2-73}$$

where T_e is the mean individual lifetime (70.7y)

In DARTAB, life loss (years) per premature death is calculated by

$$Y_l(k) = \frac{\sum_j K_j \sum_i E_{ij}(k) YL_{ijl}}{\sum_j K_j \sum_i E_{ij}(k) RF_{ijl}} \tag{2-74}$$

where

$Y_l(k)$ = average life lost (years) per premature death from cancer l at location k

YL_{ijl} = total life lost (years) for unit exposure to nuclide i, pathway j, and cancer l

$E_{ij}(k)$ = is the exposure to or intake rate of the ith radionuclide through the jth exposure or intake mode at location k in the environment

RF_{ijl} = the mortality risk factor per unit exposure or intake rate of the ith radionuclide in the jth exposure or intake mode for the lth cancer site

The factor K_j converts any pathway specific units to the required units. Note then that the numerator is just the total years of life lost by those experiencing a cancer of the lth organ, while the denominator is the total number of deaths due to radiation induced cancers of the lth organ.

DARTAB performs two calculations and tabulations of life loss per premature death (1) life loss per premature death for an individual at a selected location and (2) life loss per premature death for a mean or average individual. Table 2-3 lists units of the exposure, loss of life, and risk mortality factors.

The mean individual life loss per premature death estimate, Y_l, is merely the sum over all locations, k, of $Y_l(k)$ calculated using Equation (2-74). As with the dose calculations, it should be noted that PRESTO-EPA-POP assumes that the total population resides at one location, k = 1. Therefore, mean individual premature death values are equivalent to those for the selected individual.

Readers desiring a complete discussion of the development of the dose, health effect, or risk equivalent factors utilized by DARTAB should consult the DARTAB documentation report (Be81, pp. 5-10) or the supporting report (Du80). The latter describes the theory and development of the RADRISK code that generates the risk factors utilized by DARTAB.

2.2.2 Estimation of Basement Dose to Resident Intruder

The DARTAB subroutine of the PRESTO-EPA-POP model contains algorithms to compute the dose rate per unit radionuclide surface concentration to an

TABLE 2-3

UNITS OF YEARS LOST FACTOR, YL_{ijl}, AND MORTALITY RISK FACTOR, RF_{ijl}, FOR HEALTH RISK CALCULATIONS BY DARTAB

(Corresponding to Exposure Factors Listed in Table 2-2)

Pathway	Unit of Factor	
	YL_{ijl}	RF_{ijl}
Ingestion	yr life loss/(pCi/yr)	(deaths/10^5 persons)/(pCi/yr)
Inhalation	yr life loss/(pCi/yr)	(deaths/10^5 persons)/(pCi/yr)
Air Immersion	yr life loss/(pCi/m^3)	(deaths/10^5 persons)/(pCi/m^3)
Ground Surface Exposure	yr life loss/(pCi/m^2)	(deaths/10^5 persons)/(pCi/m^2)

individual standing on a contaminated, infinite plane. This section describes the calculation of a factor which is used to convert the input to this infinite plane computation so that the calculation computes a value appropriate for an individual spending part of his time in a basement. In this calculation it is assumed that the basement actually extends into and is surrounded by the trench contents. Furthermore, it is assumed that most of the individual's time is spent at the center of the basement, that the basement radius is three meters, and that the radiation attenuation coefficient of the trench may be approximated by that of soil, with attenuation coefficients taken from literature published by the British Standard Institute (BSI66). The elapsed time between closure of the waste disposal area and construction of the basement is an input parameter for the model.

A conversion factor F is defined which is used to convert the radionuclide concentration in the trench surrounding the basement to a value appropriate for an input parameter to the infinite plane calculation. Provided the basement is continuously occupied, this conversion factor is defined by the equation

$$F = \frac{D_b/N}{D_p/A} \tag{2-75}$$

where

D_b/N = dose rate in basement per unit of radionuclide concentration in trench [(mrad/yr)/(pCi/m^3)]

D_p/A = infinite plane dose rate per unit of surface concentration on ground [(mrad/yr)/(pCi/m^2)]

In Equation (2-75) A represents the radionuclide concentration per unit surface area on the infinite plane and N represents the radionuclide concentration per unit volume in the trench material. If the value of the factor F, is known the radionuclide dose rate to an individual within the basement may be found by using a modified form of the above equation

$$D_b = \frac{D_p}{A} FN \text{ (mrad/yr)} \tag{2-76}$$

The basement whole body gamma dose rate per unit of radionuclide concentration at a distance one meter above the basement floor is found by integrating the radiation flux from each volume element of the trench material over the trench volume v

$$\frac{D_b}{NC} = \int_v \frac{B(\mu_T r_T)}{r^2} \exp[-(\mu_a r_a + \mu_T r_T)] dv \tag{2-77}$$

where

C = units transformation constant [(mrad/yr)/(pCi/m^2)]

$B(\mu_T r_t)$ = build up factor, using formulas by Eisenhauer and Simmons for energies up to 200 kev and Taylor's formula for energies above 200 kev. Coefficients for the Eisenhauer and Simmons equation are taken from Eisenhauer and Simmons (Ei75) and for Taylor's formula are taken from Morgan and Turner (Mor67)

r = distance from point of interest to element of volume of the trench dv (m)

μ_a = linear attenuation coefficient of air (m^{-1})

μ_T = linear attenuation coefficient of trench (m^{-1})

r_a = distance in air from point of interest to element of volume dv (m)

r_T = distance in trench from point of interest to element of volume dv (m)

v = trench volume (m^3)

The basement may be considered circular, so that Equation (2-77) becomes

$$\frac{D_b}{NC} = \int_{v(floor)} \frac{B(\mu_T r_T)}{r^2} \exp[-(\mu_a r_a + \mu_T r_T)] dv + \int_{v(wall)} \frac{B(\mu_T r_T)}{r^2} \exp[-(\mu_a r_a + \mu_T r_T)] dv$$

$$= 2\pi \int_0^{R+d} \int_H^{H+d} r' \frac{B(\mu_T r_T)}{r^2} \exp[-(\mu_a r_a + \mu_T r_T)] dZ dr' \qquad (2\text{-}78)$$

$$+ 2\pi \int_R^{R+d} \int_0^{H} r' \frac{B(\mu_T r_T)}{r^2} \exp[-(\mu_a r_a + \mu_T r_T)] dZ dr'$$

where

R = basement radius (m)

h = distance of point of interest from floor (h = 1m)

H = basement height (m)

d = cut-off thickness of trench, chosen to be 10 mean free paths (or $10/\mu_T$)

The first integrand refers to the section of the trench immediately below the basement floor, while the second integrand refers to the trench material outside the walls of the basement. For this calculation, the basement is assumed circular, and a two-dimensional Simpson's rule method (McC64) is used to numerically evaluate the integrals.

Equation (2-78) has been evaluated to determine values of the ratio D_b/NC, and we have found that as the assumed basement radius varies from 3 to 6 m, the completed value of D_b/NC changes by only 30 percent (being greater for the smaller basement radius) for radiation energies ranging from 20 keV through 10 MeV. Tabulated values of the linear attenuation coefficient for air (Ko79) and for earth are used (BSI66).

The dose rate at a height of one meter per unit surface concentration from an infinite plane is given by the equation

$$\frac{D_p}{AC} = \int_S \frac{1}{r^2} e^{-\mu_a r} ds = 2\pi \int_1^\infty \frac{1}{r} e^{-\mu_a r} dr = 2\pi I \qquad (2\text{-}79)$$

where

C = units transformations constant [(mrad/yr)/(pCi/m^2)]

I = $\int_1^\infty (1/r)\exp(-\mu_a r)dr$ (dimensionless)

μ_a = linear attenuation coefficient of air (m^{-1})

z = height of point of interest (z = 1 m)

In this transformation, the incremental area element dS is $2\pi XdX$, where X is the radius projection onto the plane; and since $R^2 = X^2 + 1$, then have XdX=RdR. The value of the integral, I, in this equation, may be computed numerically using a polynomial approximation (Ga64) for values of μ_a corresponding to different values of gamma energies. The results of these calculations are summarized in Table 2-4.

The value of the ratio F as defined by Equation (2-75) may be obtained for a given energy by dividing the results of the basement calculation by the results of the infinite plane calculation. Values of this ratio for energies between 10 keV and 10 MeV are given for a basement radius of 3.0 m in Table 2-4. A very conservative average value of F may be chosen to be 0.1 m.

If the basement is occupied one-third of each day, then the radionuclide concentration within the trench is one-third. Therefore, the basement exposure dose rate in the infinite plane dose rate calculation of the DARTAB subroutine is found by multiplying the average radionuclide concen-

TABLE 2-4

RESULTS OF BASEMENT AND INFINITE PLANE
UNIT DOSE RATE COMPUTATIONS

Energy MeV	F(m)
0.05	0.015
0.10	0.045
0.20	0.061
0.50	0.087
1.0	0.087
2.0	0.088
4.0	0.092
6.0	0.098
8.0	0.099
10.0	0.101

tration by the volume within the trench during the basement occupancy period by the volume to the surface correction term F and the fraction of time the basement is assumed to be occupied. Thus, the value of A is augmented by the quantity 0.033N to yield a value that corresponds to the plane dose plus the basement dose. In the computer code, the time at which the basement is constructed is a user input parameter, and the average radionuclide concentration by volume for that period between basement construction until the end of the simulation period is computed by the code. This incremental concentration is added to the computed average surface concentration if the code user has elected to include the basement exposure mode.

2.2.3 Accounting for Radioactive Decay Products

The code does not account for ingrowth of radioactive decay products of materials during storage in the trench or during environmental transport following release from the trench. The RADRISK data files accessed by DARTAB do, however, include dose and risk factors that account for such ingrowth and subsequent exposure after materials have entered the human body. In cases where the radioactive decay product is short-lived relative to the parent, the radionuclide concentrations calculated for an environmental media may be in error because of this ingrowth.

For example, when cesium-137 (half-life = 30 yr) is included in trench inventory, barium-137m (half-life = 2.55 min) should be in secular equilibrium with the parent nuclide. If Ba-137m is not listed as part of the trench inventory, then no environmental concentrations will be calculated. Therefore, external exposures will be underestimated because of the absence of the 0.662 MeV barium-137m gamma rays. The internal exposures will include the exposure to these gamma rays as noted above.

In situations where external exposures may be important, the user may include the radioactive decay products in the initial trench inventory. In such cases, the radioactive decay products should be entered with the same activity, decay coefficient, and environmental transport parameters as the parent. The effect of secular equilibrium will be achieved throughout transport in the environment. The fact that cesium and barium probably do not have identical chemical and environmental behavior will be relatively unimportant because of the short half-life of the progeny. Entering the decay product in the source term in such a manner will not grossly overestimate internal exposures because the dynamics of the decay products inside a human body tend to result in low doses. In particular, this is true because the RADRISK data base, which contains risk factors used by DARTAB, will not be affected by the decay rate of the decay product as indicated in the environmental transport portion of the input data set (K. F. Eckerman, Oak Ridge National Laboratory, personal communication).

2.3 HEALTH EFFECTS TO REGIONAL BASIN POPULATION

The PRESTO-EPA-POP model evaluates the cumulative health effects to the population of a regional basin downstream from the disposal site for a period of 10,000 years after site closure. Because of the uncertainties in predicting health effects over long periods of time, and to reduce the computer costs, the analysis is divided in two parts, the primary analysis and the basin analysis. The primary analysis, for 1000 years, simulates the health effects to the local community as described in the previous sections.

Those radionuclides not used or consumed by the local community during the first 1000 years are considered to be residual radionuclides. In addition, after the 1000 year local assessment period has ended, all radionuclides that leave the disposal site and enter the regional basin (which now incorporates the local area, see Figure 2-4) are considered residual radionuclides. In the regional basin analysis, health effects to a regional basin population are calculated for 10,000 years for all residual radionuclides entering the downstream regional basin, using health effects conversion factors. The total health effects are equal to the sum of the health effects obtained from the primary analysis and the regional basin analysis (see Figure 2-4).

The regional basin analysis assumes that all of the communities located in a regional water basin downstream from the disposal site, including the community analyzed in the primary analysis, can be combined into a single composite community. The transport of radionuclides from the disposal site, through the hydrologic pathway, continues as described for the primary analysis. The atmospheric transport pathway is not included, since it is assumed that the health effects to a more distant regional basin from this pathway will be negligible.

Instead of performing lengthy food chain simulations and health effects analyses for 10,000 years, the model calculates the impact on the basin based on the "residual radionuclides" released downstream. Residual radionuclides are the sum of those nuclides released from the LLW site which are not used by the local community during the primary analysis and the radionuclides released to the downstream basin during the period of regional basin analysis. The radionuclides considered are those that enter

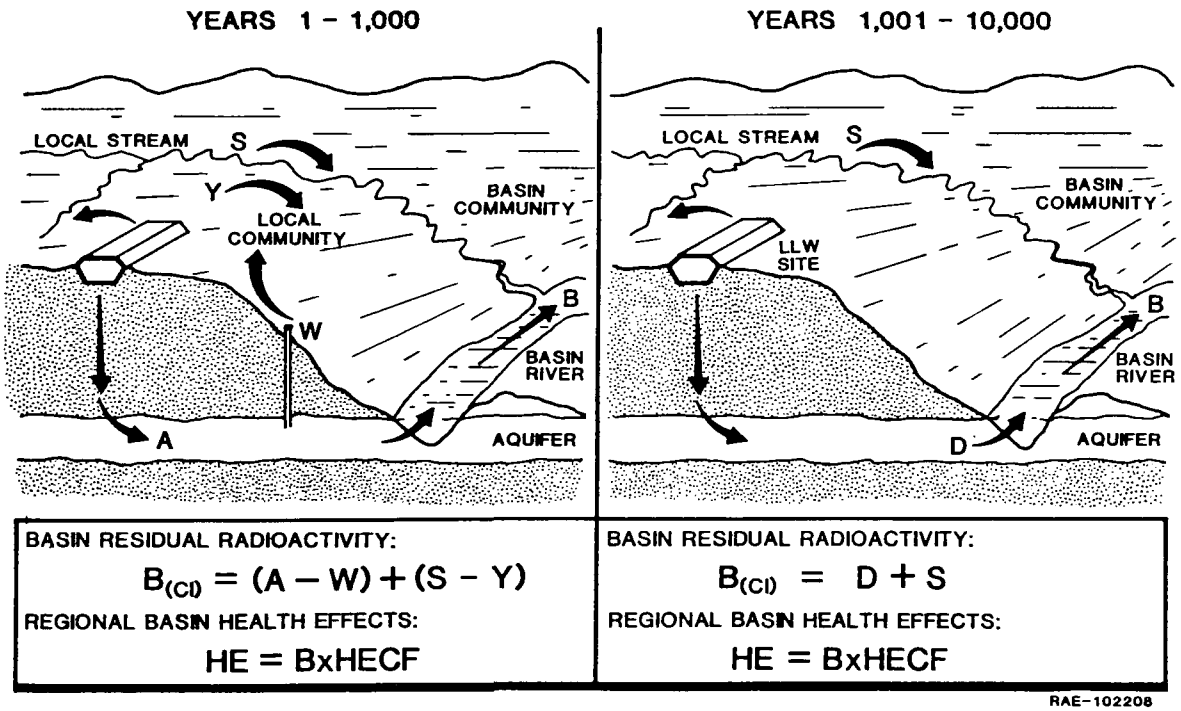

FIGURE 2-4. REGIONAL BASIN HEALTH EFFECTS PATHWAY.

the aquifer through the trench bottom and those that enter the regional stream by way of runoff.

In order to determine the health effects to the regional basin population, the residual radionuclide activity released to the regional basin is multiplied by a conversion factor precalculated for each radionuclide within PRESTO-EPA-POP code. The conversion factors, which are nuclide dependent, are based on local water use characteristics and the hydrologic pathway.

2.3.1 Calculations of Regional Basin Health Effects

The code uses health effects conversion factors (see Section 2.3.2) to calculate the health effects to the regional basin population from the basin residual radionuclides. The basin residual radionuclides consist of all the radionuclides that leave the disposal site area through the surface and underground water pathways but are not used by the local population and go on to enter a major basin stream. Once the radionuclides arrive at the basin stream, they are released to the basin within the same year of arrival. Radionuclides not used by the regional basin community are assumed to travel to the ocean where they contribute no health effects.

Releases of radionuclides to the regional basin can be simulated for up to 10,000 years. The annual nuclide releases to the regional basin are collected in the model in ten periods of 1,000 years each. The annual releases of each nuclide from surface runoff and the aquifer to the regional basin during the first millenium are collected in the array variable QDWSB. Nuclides that leave the bottom of the trench during the first millenium and arrive at the well beyond year 1000 are not neglected; they are also considered in the yearly loop.

In each of the first thousand years of simulation, the amount released to the basin, SSTREM(N), is decreased by the amount of nuclides removed from the nearby stream(s) with water used by the local population. This correction is accomplished using the stream flow rate, STFLOW; the hypothetical volume of water withdrawn from the stream, VOLUSS; and the stream water concentration, STCON(N). The nuclide release to the basin in each year of the first millenium is

$$\begin{cases} (\text{STFLOW} - \text{VOLUSS})\ \text{STCON(N)} & \text{if VOLUSS} < \text{STFLOW} \\ 0 & \text{if VOLUSS} \geq \text{STFLOW} \end{cases} \tag{2-80}$$

To calculate radionuclide contributions from the aquifer to the basin, it is necessary to evaluate additional radionuclide transit times as well as Hung's correction factors for the reach from the well to the stream. The user can also control the percentage of well water that flows to the basin stream by specifying the input parameter CPRJ as the fraction of groundwater that bypasses the basin stream.

The surface soil and surface water concentrations are calculated every year according to Equations (2-25) and (2-26). These concentrations also change from year to year as a result of wind resuspension of radionuclides, water runoff, trench water overflow, the seepage of water from the disposal site surface to the aquifer. The amount of radionuclides released from the surface soil of the disposal site to the surrounding surface streams is calculated using the current year surface water nuclide concentration, C_W (Ci/m^3); the area of the disposal site, $S_W S_1$ (m^2); the annual precipitation rate, P_a (m/yr); the current year amount of trench water overflow, V_0 (m^3/yr); and a transfer factor, f_r, as follows

$$W_S = f_r(P_a\ S_W S_1 + V_O) \tag{2-81}$$

$$SSTREM = W_S\ C_W \tag{2-82}$$

where W_S (m^3/yr) is the amount of water that enters the surrounding streams from runoff and trench overflow during the current year of simulation, and SSTREM (Ci/yr) is the amount of nuclides going along with that volume of water. The surrounding streams in turn will transport the nuclides to the regional basin stream within the same current year of simulation.

The total release of radionuclide, QLBTTH, to the basin is the sum of the 10,000 yr releases, i.e.,

$$QLBTTH = QDWSB + \sum_{J=1}^{9} QLB(J) \tag{2-83}$$

The population health effects in the regional basin due to the release of residual radionuclides to the basin are calculated by multiplying the total release of the radionuclide to the regional basin by a conversion factor, i.e.,

$$HE = QLBTTH \cdot CON \tag{2-84}$$

When a unit response approach is used, the total health effects for the actual concentration of the waste are calculated in a separate health effects accounting model. Since the water usage for each water basin is different from one site to another, different health effect conversion factors should be used. A detailed description of the health effects accounting model is discussed in a separate report on the characterization of health risks and disposal costs associated with alternative methods for

land disposal (EEI84). The health effects conversion factors, which are used to determine the health effects to the regional basin population from residual radionuclides, are calculated independently of PRESTO-EPA-POP. However, the methodology associated with their derivation is included in the following section.

2.3.2 Conversion Factors for Regional Basin Health Effects

The health effect and genetic effect conversion factors are used to calculate the impacts of residual radioactivity entering a regional surface water drainage basin system. A number of assumptions must be made concerning the quantity of water contaminated with residual radioactivity which will be withdrawn by downstream communities, the population using the stream water, and the uses to which the water will be put (i.e., drinking, irrigation, etc.). These conversion factors are applied to the residual radioactivity entering the basin to obtain the number of health effect and genetic effect.

Per capita water consumption is calculated for each site, taking into account the local irrigation requirements and the fraction of the year during which irrigation takes place (Equation (2-85)). For regions where multiple water sources may be used, fractional correction factors are applied. For example, a portion of the requirement for the irrigation water may be met by using a well or stream, while the remainder may be withdrawn from a farm pond with no contamination. This is handled by including "switches" in the water consumption equation. Thus if half of the irrigation water at a given site is withdrawn from a stream while the rest is gathered from precipitation-fed farm ponds, the irrigation pathway switch will equal 0.5.

$$V_u = [3.9\text{x}10^7 W_i f_i L_i + U_w L_h + 1.5\text{x}10^4 L_a] N_p \qquad (2\text{-}85)$$

where

V_u = water used (l)

$3.9\text{x}10^7$ = 4492 m^2 irrigated land per person x 8760 hr/yr

W_i = irrigation rate (l/m^2-yr)

f_i = fraction of year when irrigating

L_i = irrigating pathway switch

U_w = individual water consumption per year (l/person-yr)

L_h = human pathway switch

$1.5\text{x}10^4$ = annual water fed to cattle consumed by humans (l/person-yr)

L_a = animal pathway switch

N_p = size of population

In order to match the per capita water use patterns of the downstream, basin population to those of the local population, fractional usage parameters must be included. For the actual PRESTO-EPA-POP analyses, fractional usages are modeled for sites where no water use comes from the well or stream. The annual per capita water usage by the regional basin population is given by

$$(V_u/N_p)/1000 = \text{per capita water consumption } (m^3/\text{person-yr})$$

It is assumed that the regional basin population downstream of the disposal site is directly related to the volume flow rate in the stream. An average river basin generally has an annual river flow-to-population ratio of 3000 m^3/person. The site-specific water usage rate (per person) is then compared to the standard rate of 3000 cubic meters/person-yr to determine the fraction of stream flow that will be used by the downstream populations.

$$\begin{bmatrix}\text{Surface Water} \\ \text{Utilization Factor}\end{bmatrix} = \frac{\text{per person site-specific water consumption rate}}{\text{standard rate (3000 m}^3\text{/person-yr)}} \quad (2\text{-}86)$$

When the water utilization fraction has been computed, it is used to compute the number of health effects occurring in the downstream populations. For each disposal site, output data from PRESTO-EPA-POP are analyzed to determine the number of health effects and genetic effects occurring in the local population per curie of each radionuclide pumped from the well or stream. The water pathway conversion factors are defined as follows:

$$HECF_W = \left[\frac{HE}{WW + SW}\right] F \quad (2\text{-}87)$$

$$GECF_W = \left[\frac{GE}{WW + SW}\right] F \quad (2\text{-}88)$$

where

$HECF_W$ = health effect conversion factor for the water pathway (deaths/Ci)

$GECF_W$ = genetic effect conversion factor for the water pathway (effects/Ci)

HE = number of fatal cancers in the local population over 1000 years

GE = number of genetic effects occurring in the local population over 1000 years

WW = radioactivity pumped from the well over 1000 years (Ci)

SW = radioactivity pumped from the stream over 1000 years (Ci)

F = surface water utilization factor

For the health effects conversion factor analysis, the air pathway sources were shut off for the regional basin by setting spillage equal to zero. No on-site farming or basement exposures are included, either. Therefore, the health effects in the exposed population are due to use of contaminated water only.

In addition to water usage, there is another exposure pathway for the downstream communities. The possibility exists that contaminated fish will be consumed by the downstream regional basin population. Conversion factors for fish consumption (health effects/Ci) were added to the water pathway conversion factor, as follows:

$$HECF = HECF_W + HEF \tag{2-89}$$

$$GECF = GECF_W + GEF$$

where

HECF = combined health effect conversion factor (effects/Ci)

HEF = health effect conversion factor for consumption of fish (effects/Ci)

GECF = combined genetic effect conversion factor (effects/Ci)

GEF = genetic effect conversion factor for consumption of fish (effects/Ci)

The health effect conversion factors for consumption of fish are calculated in the following manner:

$$HEF_i = \left(\frac{P}{R}\right) B_{fi} U_f \left(\frac{D}{C}\right)_i \tag{2-90}$$

$$GEF_i = \left(\frac{P}{R}\right) B_{fi} U_f \left(\frac{G}{C}\right)_i \tag{2-91}$$

where

HEF_i = health risk from fish consumption per curie of nuclide i released to the river

P = population utilizing the river

R = river flow rate (l/yr)

B_{fi} = nuclide transfer factor - water to fish $\left(\frac{pCi/kg\ fish}{pCi/l\ water}\right)$

U_f = annual fish consumption rate per person

$(D/C)_i$ = cancer risks per curie of nuclide i ingested by the population

GEF_i = genetic effects from fish consumption per curie of nuclide i released to the river

$(G/C)_i$ = genetic effects per curie of nuclide i ingested by the population

The regional basin health/genetic effects for each nuclide are calculated by multiplying the residual for the radionuclide released to the downstream basin by the appropriate health effect or genetic effect conversion factor, as follows:

$$\text{Basin Health Effects}_i = [\text{Residual}_i\ (\text{1st 1000 years}) + \text{Residual}_i\ (\text{Years 1001 - 10,000})]\ HECF_i \quad (2\text{-}92)$$

$$\text{Basin Genetic Effects}_i = [\text{Residual}_i\ (\text{1st 1000 years}) + \text{Residual}_i\ (\text{Years 1001 - 10,000})]\ GECF_i \quad (2\text{-}93)$$

The total regional basin health/genetic effects resulting from all residual nuclides are calculated by summing over all nuclides (i) as follows:

$$\text{Total Basin Health Effects} = \sum_{i=1}^{N} \text{Basin Health Effects}_i$$

$$\text{Total Basin Health Effects} = \sum_{i=1}^{N} \text{Basin Genetic Effects}_i$$

3. DEVELOPMENT OF PRESTO-EPA-POP CODE

3.1 STRUCTURE AND INFORMATION FLOW

The PRESTO-EPA-POP code is written in FORTRAN IV for an IBM 3081 or comparable computer system and requires 850K bytes of memory. It is designed to process up to 40 nuclides for a maximum of 10,000 years. Evaluation of local and regional basin health effects of 31 radionuclides over 10,000 years takes approximately 7 minutes to execute. A shorter test of ten nuclides takes less than two minutes to execute. The program should be easily transferrable to other IBM installations. It has run correctly on another non-EPA IBM computer system after installation directly from tape. Non-IBM users may have to modify the job control language (JCL), the NAMELIST inputs and other program segments where character manipulations are used.

The PRESTO-EPA-POP code is structured in a modular form to permit simple upgrading or replacement of given submodels without rewriting the entire code. The subroutine structure of the code is shown in Figure 3-1.

There are three classes of submodels: unit response, scheduled event, and bookkeeping submodels. Unit response submodels simulate processes such as rainwater infiltration through the intact portion of the trench cap, erosion of soil overburden from the trench cover, and atmospheric transport. Such submodels are usually accessed only once during a model run and generate parameters and rates used elsewhere in the simulation.

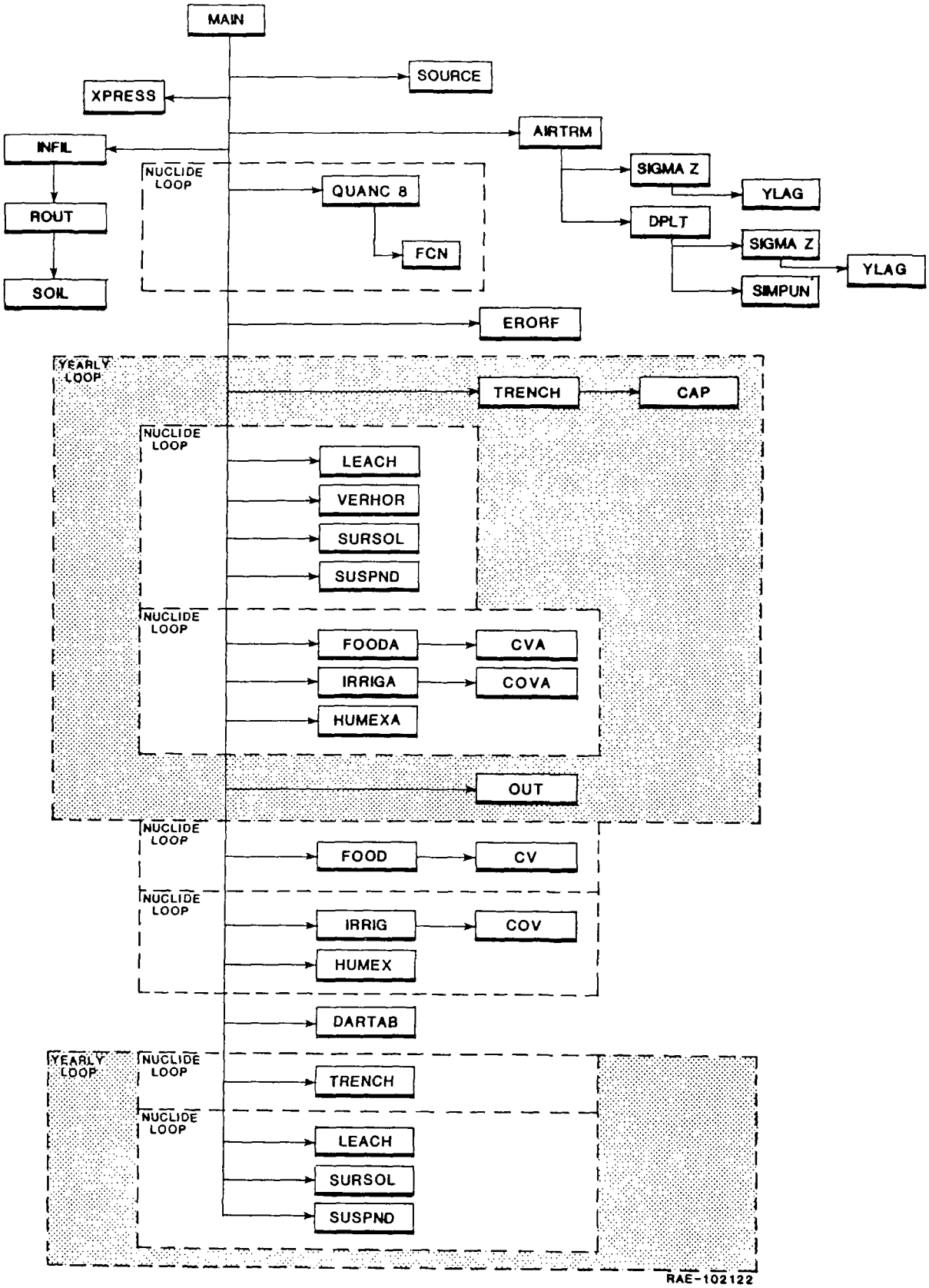

FIGURE 3-1. PRESTO-EPA-POP SUBROUTINE STRUCTURE.

Scheduled event submodels estimate events such as the time of trench cap failure, while bookkeeping submodels determine the water balance in the trench and radionuclide concentrations in the trench outflow and the aquifer. Output from the bookkeeping submodels is iterated annually over the simulation period. Risk evaluation bookkeeping submodels accept the cumulative or mean output from the transport portion of the code and generate doses and population risks, based on a life-table approach.

3.2 SUBROUTINE DESCRIPTION

An alphabetical listing and description of the subroutines and main program found in PRESTO-EPA-POP is given below.

MAIN - This routine is the main calling program of PRESTO-EPA-POP and defines the most commonly used variables of the code, specifies dimension and common areas, and initializes variables and input control parameters. The input and output subroutines, SOURCE and OUT, are called directly by MAIN (Figure 3-1), as are the unit response model subroutines AIRTRM, and ERORF. MAIN also calculates: the vertical water velocity; retardation factors; vertical, horizontal and total transit times in groundwater (the transfers from trench to vertical soil column to aquifer in Figure 2-1); and the basement exposure correction factor (Section 2.2.2). The decay-dispersion correction factor, DDETA (Hu81), is calculated for each radionuclide in MAIN (factor DDETA adjusts the activity output of the aquifer for the combined interactions of longitudinal dispersion and radioactive decay.) QUANC8, which is based on an eight panel Newton-Cotes rule, performs the integration necessary to obtain the correction factor.

MAIN calls the bookkeeping subroutines to calculate quantities associated with trench water balance, trench cap status, changes in land use and basement occupancy. Other subroutines called by MAIN compute the amount of leaching from trench, transport of soluble surface components, atmospheric concentrations, and well concentrations. In addition, aquifer volume, hypothetical radionuclide withdrawal from well, and material balances for water in the aquifer are calculated in MAIN.

Risk evaluation submodels called from MAIN account for radionuclide concentrations in food due to atmospheric deposition and water irrigation, and radionuclide intake by man. These subroutines are IRRIG, FOOD, HUMEX, CV, COV, IRRIGA, FOODA, HUMEXA, CVA, AND COVA. Finally DARTAB, which creates tables of predicted health effects from radioactive effluents is called from MAIN.

The annual simulation loop and the radionuclide loop are executed a selected number of times. During a model run, MAIN may access any or all of the subroutines or functions which are listed below in alphabetical order.

Average concentration values printed in the concentration tables are computed in the MAIN routine using a summation calculation within the MAXYR loop (the principal yearly iteration loop). After this loop is completed, the summed concentration values are divided by the number of years considered in the simulation and the results are printed. Maximum concentration values are identified by comparing stored concentration values to the values determined in each iteration of the yearly simulation loop. If the new concentration is greater, then the concentration and corresponding year number are stored for printing in the concentration tables.

AIRTRM - This subroutine is the main calling program for the atmospheric transport submodel. AIRTRM calculates sector-averaged (22.5 degree) atmospheric exposures normalized to the source strength. AIRTRM and all its supporting subroutines are adaptations of the interactive Gaussian plume atmospheric model, DWNWND (Fi80a). AIRTRM also calculates the deposition rate onto surfaces per unit source strength. To make these calculations, AIRTRM accesses four other subroutines, SIGMAZ, DPLT, YLAG, and SIMPUN, and utilizes a number of user-input parameters including source height, lid height, stability class, type of stability class formulation, Hosker roughness parameter, wind velocity, deposition velocity, gravitational fall velocity, and source to receptor distance. The normalized atmospheric exposures are returned to the main program and are used in later dose and risk calculations.

CAP - This function calculates and returns to both MAIN and TRENCH, the fraction of the trench cap that has failed. Cap failure may be either partial or total. Total failure may be caused by erosion of all overburden as calculated by ERORF. Partial failure indicates that a portion of the cap has been completely removed; the remainder of the cap is still subject to erosion. Partial failure may be caused by user input of the end points of a linear function to selectively remove all overburden from a fraction of the trench.

COV, COVA - These functions are called by subroutine IRRIG and IRRIGA to calculate radionuclide concentrations in vegetables, milk, and meat that may be contaminated by irrigation. The radionuclide concentrations in food depend on such quantities as the agricultural productivity of vegetation, the period of irrigation annually, the storage delay period between harvest

and use for pasture grass, feed, leafy vegetables and produce, and the radionuclide decay constant.

CV, CVA - These functions are utilized by subroutines FOOD and FOODA to calculate radionuclide concentrations in pasture grass and stored feed consumed by animals, and in leafy vegetables and produce consumed by humans. CV is essentially the same as function COV, except that CV is used for atmospherically deposited radionuclides and COV accounts for radionuclides deposited by spray irrigation. Pertinent input data include agricultural productivity. fraction of the year vegetation is exposed to depositing radionuclides, and the delay time between harvest and consumption for stored feed, pasture grass, leafy vegetables, and produce.

DARTAB - The original DARTAB code is a self-contained program which combines radionuclide environmental exposure data with dosimetric and health effects data to create tables of predicted impacts of radioactive effluents. DARTAB has eleven subroutines and contains over 3000 FORTRAN source statements. DARTAB subroutines are RDSTOR, FACOUT, CHLOC, PREPDR, PREPRF, PREPHR, MULT, DRTAB, ORGFAC, SUMMRY, and SUMMR2. These are not discussed specifically in this report. For information on the original DARTAB consult the document describing the code (Be81). DARTAB has been modified for PRESTO-EPA-POP so that the program is treated as a subroutine. Environmental exposure data are now passed in COMMON from MAIN to DARTAB's subroutines.

DARTAB uses dosimetric and health effects data from the methodologies of RADRISK (Du80). RADRISK uses a life-table model to calculate the human health risk to a cohort of 100,000 people from a constant input of 1 pCi/yr (0.037 Bq/yr) via ingestion and inhalation over a lifetime (70.7 yr).

These intake conditions are approximated in PRESTO-EPA-POP by calculating an average intake over the span of the assessment of each type of intake. RADRISK data files are accessed directly by DARTAB.

DPLT - The subroutine DPLT is called by AIRTRM and computes a correction factor for plume depletion. To make this calculation, DPLT calls subroutines SIGMAZ and SIMPUN.

ERORF - This subroutine uses the universal soil loss equation, USLE, developed by the U.S. Department of Agriculture (USDA61) to determine sediment loading for rain-driven surface erosion. Estimation methods and tabulations for factors used in USLE have been published (McE76). The code user inputs all six of these factor values. The calculated erosion rate is returned to MAIN where it is converted to an annual erosion rate in meters. This erosion rate is utilized by MAIN to determine the thickness of the cap.

FCN - This function subprogram returns to QUANC8 a functional evaluation of the integral used in calculation of the aquifer decay-dispersion correction factor. The routine is written in double precision to facilitate interaction with the double precision routine QUANC8.

FOOD, FOODA - Subroutine FOOD is called only once per simulation and calculates the average concentration of each radionuclide in foods contaminated by atmospheric deposition and root uptake. The deposition input to FOOD is calculated in subroutine AIRTRM. The equations and

internal parameters used by FOOD are those in AIRDOS-EPA (Moo79). Output from FOOD is used by the subroutine HUMEX to calculate the human exposure via ingestion of these contaminated foodstuffs. Subroutine FOODA is called from MAIN each simulation year.

HUMEX, HUMEXA - Subroutine HUMEX accepts user input and receives averaged data from subroutines AIRTRM, FOOD, IRRIG, and VERHOR to calculate the average annual human exposures via ingestion and inhalation. Output from HUMEX supplies the input to the DARTAB subroutines for calculations of risk and dose and tabulation of health results. Subroutine HUMEXA is called from MAIN each simulation year.

INFIL - The subroutine INFIL is based on a model by Hung (Hu83b) and calculates annual infiltration through the trench cap. INFIL calls subroutine SOIL and ROUT. Inputs to INFIL include hourly precipitation, daily temperature, and various trench cap characteristics.

IRRIG, IRRIGA - Foods may be irrigated with contaminated water from either surface or groundwater sources. Input to IRRIG, which is called only once per simulation, includes the time-averaged radionuclide concentrations in well or surface water calculated by VERHOR or subroutine SURSOL, respectively. IRRIG calls the function COV and uses the equations in AIRDOS-EPA (Moo79, Fi80b) to calculate the time-averaged concentration of each radionuclide from direct deposition by irrigation and subsequent root uptake in foodcrops. Subroutine IRRIGA is called from MAIN each simulation year.

LEACH - Subroutine LEACH calculates the amount of each radionuclide from the homogeneous trench contents that leaves the trench each year.

Losses may be via transport through the trench bottom or by overflow from the trench. There are five independent user-specified methods that may be used to calculate these amounts: the option is chosen by specifying a value from one through five for parameter LEAOPT. Table 2-2 lists the calculational methods corresponding to values of LEAOPT. The total-contact options, 1 and 3, assume that all of the trench contents have been in contact with water during the previous year. The immersed-fraction options, 2 and 4, assume that the wetted fraction of the waste equals the ratio of maximum water level to the trench depth. The distribution coefficient options, 1 and 2, utilize a K_d approach to calculate the radionuclide concentrations released from the wastes to the water, while options 3 and 4 use a solubility estimate rather than K_d. If the user selects LEAOPT = 5, then a user-specified fraction of the total radionuclide concentration in the waste is lost through the trench bottom annually. Output from LEACH is the activity leaving the trench annually for each radionuclide through the bottom of the trench and by overflowing.

OUT - This subroutine produces annual summaries for the trench cap status, trench water balance, amount of water leaving trench, and radionuclide contents in trench water, trench overflow water, aquifer, well water, and on the ground surface. The user may choose to print these summaries for every year or less frequently by specifying the appropriate values of IPRT1, IPRT2, and IDELT in the input data file.

QUANC8 - This subprogram employs a Newton-Cotes 8-panel quadrature formula (For77). The integral to be evaluated is specified by the function FCN.

ROUT This subroutine is called by INFIL.

SIGMAZ - This subroutine is called by both AIRTRM and DPLT to compute the vertical atmospheric dispersion parameters. Depending on the choice of parameterization specified in the input data set, SIGMAZ will calculate the dispersion parameters by one of eight schemes. Necessary input data include the downwind distance, stability class, Hosker roughness parameter, and lid height. Other data necessary for Lagrangian interpolations (by function YLAG) are contained internally in SIZMAZ and need not be input by the user.

SIMPUN - This subroutine, originally written by Barish (Bar70), uses Simpson's rule to integrate along the ground level centerline of the atmospheric plume to compute the depletion fraction. All input to SIMPUN is supplied by DPLT, the subroutine that calls SIMPUN and to which the results are returned.

SOIL - This is a subroutine called by ROUT.

SOURCE - Subroutine SOURCE reads the input required to initialize and quantify transport parameters, except those required for subroutine INFIL. Data concerning program control, climatic description, trench description, aquifer description, atmospheric description, site-surface description, and radionuclide description are read in by SOURCE. SOURCE also prints out these data before any calculated results are output.

SURSOL - Subroutine SURSOL computes the amount of soluble radionuclide that enters the stream annually. Input variables to SURSOL include the

average depth of active exchange in the soil, the average downslope distance to the stream, the cross slope extent of the spillage, the average annual infiltration, the bulk density of soil, the amount of spillage, and the surface soil distribution coefficients. Variables output from SURSOL include the amounts of radionuclide going to the stream and the deep soil layers and the radionuclide concentration in the interstitial water of the contaminated surface region.

SUSPND - This subroutine calculates the above trench atmospheric source term from the ground surface by two methods, a time-dependent resuspension factor and a resuspension rate due to mechanical disturbance. Input variables include the current year of simulation, the spatial area of the contaminated surface, the radionuclide concentration on the ground surface, the beginning and ending years of mechanical disturbances, the resuspension rate, and the wind velocity. SUSPND assumes that all radionuclides to be resuspended are deposited on the soil surface at a simulation time zero. The resuspension factor calculated uses the empirical equation of Anspaugh et al. (An75).

The atmospheric source term is returned to MAIN and is used along with X/Q to calculate the air concentration of each radionuclide available for deposition onto foodstuffs and for inhalation by the general population. The value of X/Q is calculated by AIRTRM.

TRENCH - This subroutine determines the trench water balance. Input variables include trench dimensions, porosity and permeability of trench contents, trench water volume from the previous year, length of the saturated zone, and annual precipitation and infiltration. Output from

TRENCH includes the maximum depth of water in the trench, the volume of water in the trench, volume of water overflowing the trench, and water volume lost from the bottom of the trench.

The amount of water which overflows the trench, is calculated by comparing the maximum water depth to the trench depth and overflowing any amount greater than the trench volume. The variables VOLO, VOLB, OLDWAT, and DMAX that quantify overflow, bottom loss, water level during previous year, and maximum water depth in trench, respectively, are used by the subroutine LEACH, discussed previously.

VERHOR - This subroutine calculates and decays the amount of each radionuclide, that reaches the irrigation/drinking water well in a given year. Variables evaluated elsewhere in the code and input to VERHOR include the current year of the simulation, transit time from the trench to the well, the volume of water leaving the trench bottom, the amount of each radionuclide leaving the bottom of the trench, the amount of radionuclide reaching the aquifer from the contaminated surface region, and the radioactive decay constant.

YLAG - This function performs a Langragian interpolation as part of the atmospheric transport calculations. The original program was written by Brooks and Long (Br70) and adapted for use here. All input data are supplied by subroutine SIGMAZ.

XPRESS - This subroutine computes and stores exponential decay factors to be used repetitively in the nuclide loops. XPRESS saves a substantial amount of computing time.

4. DESCRIPTION OF OUTPUT OF THE PRESTO-EPA-POP CODE

The output of the PRESTO-EPA-POP code is designed to be self-explanatory and provide descriptive comments, definitions, and intermediate tabulations for a comprehensive description of the model simulation. The output from PRESTO-EPA-POP is described in the following sections which correspond to the output sections contained in the printout for a run. Full details on operation of the PRESTO-EPA-POP code together with sample problem runs and a source code listing are given in the PRESTO-EPA-POP Users Manual (EPA85b).

4.1 REPLICATION OF INPUT DATA

The first section of the code output prints all input data, with the exception of the input data for the subroutine, INFIL. Input data are read and then written on a temporary storage device and then on the output device. This record of the input data set is printed out for reference identification of the run and verification of the input data. Input data is subsequently read from the temporary storage device for use by the code. INFIL input data is treated separately as shown in Section 4.4. This approach preserves the modularity of the INFIL subroutine.

4.2 ORGANIZATION OF INPUT DATA

In the second section of the output of the PRESTO-EPA-POP code, the input data are organized and summarized according to data type and transport subsystem or pathway and then printed out. A detailed description of the

input data together with sample computer runs can be found in the PRESTO-EPA-POP User's Manual (EPA85b).

The first part of this output section consists of the computer run identification and the user-supplied identification from the first input card. The control information output identifies the site and interprets the run control data entered. These control data include radionuclide leaching options, trench cap failure data, and water use parameters. The trench information output describes the trench area and depth, the porosity of the trench contents, and the annual infiltration for the watershed.

The aquifer information output defines the groundwater velocity, the trench to aquifer distance, the trench to well distance, the aquifer thickness and contamination plume dispersion angle, the porosities of the sub-trench area and the aquifer, the sub-trench permeability.

The atmospheric information output describes the effective source height for the wind-blown mechanically mobilized contamination plume from the site, the gravitational fall velocity of suspended soil particles, the site-to-population distance, the lid height, the Hosker surface roughness factor, the atmospheric stability class and dispersion formulation, the fraction of the time wind blows toward the population, and the parameters specifying the resuspension factor and resuspension rate.

The surface information output consists of the universal soil loss equation parameters, the surface soil porosity and bulk density, the runoff fraction for rainfall, the stream or river flow rate, the cross-slope extent of spillage, the active soil depth, and the average distance from the trench to the stream. The air and food chain information output describes

productivity data for grass and vegetation, timing data for computation of radioactive decay for the ingestion exposure pathway, and nuclide weathering and carbon-14 equilibrium data. The water-food-chain output describes data which characterize water use by milk cattle, goats, and beef cattle and water use for crop irrigation.

Finally, the human ingestion and inhalation rate information is printed.

4.3 RADIONUCLIDE SUMMARY TABLES

A set of three tables under the nuclide information heading summarizes radionuclide data used for the transport calculations. First, an inventory table specifies the initial inventory in the trench, on the soil surface, in the stream, and in the atmosphere. Included in this table are radioactive decay constants and the solubility constants. The second table summarizes the chemical distribution coefficients for the surface soil, the trench contents, the vertical soil column, and the aquifer. The third radionuclide table summarizes the seven radionuclide-specific food chain parameters used by the FOOD, IRRIG, HUMEX, CV, and COV subroutines. These parameters are also used by the FOODA, IRRIGA, HUMEXA, CVA, and COVA subroutines. Radionuclide atomic mass numbers are extracted from the radionuclide names and used in the calculations. Results from the mass extraction algorithm are printed in the initial calculations table.

4.4 INFIL INPUT/OUTPUT

The fourth section of output of the code consists of the input data and results produced by the subroutine INFIL. INFIL control data are given, followed by monthly average values for hours of sunshine, daily average temperature levels, and hourly rainfall amounts.

The trench characteristics presented are the snowmelt coefficient, trench cover thickness, width, slope, permeability, porosity for gravity and pellicular water, equivalent upward diffusivity, and equivalent upward hydraulic conductivity. The INFIL time step is also printed. Results are annual precipitation, evaporation, runoff, and cap infiltration.

4.5 UNIT RESPONSE CALCULATIONS

This output section includes results of nuclide-specific annual transport calculations which will be used by the bookkeeping submodels for each simulation year. The outputs also includes soil loss calculation results and the calculated or input value for the ratio of atmospheric radionuclide concentration per unit release rate at the radionuclide disposal site.

4.6 ANNUAL SUMMARY TABLES FOR SPECIFIED YEARS

Control variables determine the years for which results will be given. For those years, a number of hydrologic and transport variables are output. Included are trench cap status, maximum possible water depth in the trench, water loss by overflow and drainage from the trench, and trench inventory. Radionuclide concentration values and flux values are presented for key pathways and regions of interest.

4.7 RADIONUCLIDE CONCENTRATION TABLES

The radionuclide concentration tables present, by nuclide, the average concentration over the entire assessment period, the year of maximum concentration, and the maximum concentrations for the atmosphere, well water, and stream water. Because human exposures may commence and peak many years after closure of a disposal site, the year and level of maximum concentration are important since they are related to the year at which population exposures are most significant.

Separate average radionuclide concentration tables for the user-specified averaging time period are included. These tables present the radionuclide concentration in five types of foods resulting from atmospheric nuclide deposition and irrigation.

4.8 RADIONUCLIDE EXPOSURE TABLES

Annual population intakes of radionuclides by ingestion and by inhalation are output and the fraction of ingestion dose from drinking water is given. The radionuclide exposure tables list by nuclide the year of maximum exposure and the corresponding level of the maximum, mean population exposure from the atmosphere, the ground surface and for ingestion and for inhalation. Although the exposure terms are dependent on the concentrations summarized in the concentration tables, the times of maximum exposures will in general differ from the times of maximum concentrations. This difference occurs because of the dependence of

exposure on soil concentration and the nuclide-specific uptake and concentration mechanism of the food chain which are considered in computing exposures from ingestion.

4.9 DARTAB CONTROL INFORMATION

DARTAB control information includes run identification data, summaries of output table control information, lists of critical organs and cancers to be considered in the run, dose equivalent factors for low and high linear energy transfer (LET) radiation, radionuclide uptake and clearance data, and lists of nuclides and organ dose or cancer risk factors not found in the RADRISK data sets which were accessed by DARTAB. The location of the population at risk, the lifetime fatal cancer risk at that location, and the organ dose weighting factors are also given.

4.10 DARTAB DOSE TABLES

DARTAB dose tables are produced in the code output and present individual and collective dose summary rates by low and high LET radiation and organ, by low and high LET radiation and exposure pathway, and by low and high LET radiation and radionuclide. Both absolute dose and percentage of total dose are included in these tables.

4.11 DARTAB FATAL CANCER RISK TABLES

DARTAB fatal cancer risk tables produced in the output present individual and collective fatal cancer risk, premature death, and lifetime

fatal cancer risk exposure equivalent. Genetic risks are also summarized. Values are summarized by low and high LET radiation, by organ, and also by pathway.

4.12 RESIDUAL RADIOACTIVITY RELEASED TO THE BASIN AND HEALTH EFFECTS

This output section gives that amount of radionuclides released to the basin each millenium during the 10,000 year simulation. The aggregated total release of each radionuclide, the health effect conversion factor, and the basin population health effects by radionuclide are given.

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APPENDIX A

A MODEL TO SIMULATE INFILTRATION OF RAINWATER THROUGH THE COVER OF A RADIOACTIVE WASTE TRENCH UNDER SATURATED AND UNSATURATED CONDITIONS

APPENDIX A

This appendix contains the background theory of the infiltration submodel (subroutine INFIL) used in PRESTO-EPA-POP as developed by Hung (Hu83). Hung's analysis follows.

A MODEL TO SIMULATE INFILTRATION OF RAINWATER THROUGH THE COVER OF A RADIOACTIVE WASTE TRENCH UNDER SATURATED AND UNSATURATED CONDITIONS

INTRODUCTION

The disposal of low-level radioactive wastes by the shallow land burial method has been used for decades. An important consideration in evaluating the performance of a waste disposal site is the potential effects on the health of nearby populations. Since one of the major driving forces in causing the release of radionuclides is the rainwater which infiltrates through the cover of a disposal trench, simulating the infiltration process is an important part of any model for evaluating health effects.

An accurate model for simulating the infiltration of rainwater through a trench cover involves the modeling of three flow systems: an overland flow system, a subsurface flow system, and an atmospheric diffusion system. The overland flow system receives the rainwater and diverts the excess water from percolation and evaporation into the receiving drainage system. The subsurface flow system receives the percolated water from the overland flow system and transports the water either downward as infiltration into the trench and/or upward as evaporation into the atmospheric diffusion system. The atmospheric diffusion systems receives water/vapor from the overland flow system or subsurface flow system and transports the vapor to

the atmosphere. One of the early efforts in simulating this system for storm runoff was the Stanford Watershed Model conceived by Crawford and Linsley (Cr66). The same model was significantly improved by Moore and and Claborn (Mo71) and Hung and Keifer (Hu77). However, these models were developed for water resources planning purposes with emphasis on the overland flow system. Significant improvements are required to apply the same models to the simulation of trench cover infiltration. This study presents an efficient and yet accurate infiltration model based on the dynamic equations governing the above systems.

BASIC EQUATIONS

The simulation of rainwater movement through a homogeneous trench cover, defined here as the "total infiltration system," involves analyzing the hydrologic processes of overland flow, subsurface flow, and atmospheric diffusion systems. The basic momentum and continuity equations governing these three systems have been studied thoroughly by other investigators. They are discussed below.

Overland Flow System - The one-dimensional momentum and continuity equations governing an overland flow system are expressed by (Iw64),

$$\frac{1}{g}\frac{\partial u}{\partial t} + \frac{u}{g}\frac{\partial u}{\partial x} + \frac{\partial h}{\partial x} - \sin a + n^2\frac{u^2}{h^{4/3}} = 0 \qquad \text{(A-1)}$$

and

$$\frac{\partial h}{\partial t} + h\frac{\partial u}{\partial x} + u\frac{\partial h}{\partial x} = P - E_o - q_o \qquad \text{(A-2)}$$

where

u = velocity of overland flow

h = depth of flow

a = average inclination of the trench cover

n = Manning's coefficient of roughness

P = rate of precipitation

E_o = rate of evaporation from the overland flow

q_o = rate of percolation from the overland flow system

g = acceleration potential due to gravitational force

x = space coordinate along the slope of the trench cover

t = time

The above equation is derived from the assumptions that Corioli's energy correction factor, Boussinesq's momentum correction factor, and Jaeger's hydrostatic pressure correction factor are all unity; the flow is one dimensional, i.e., the velocity component in the longitudinal direction predominates the velocity component in lateral directions, and the flow is a gradually varied flow. The above system equations cannot be solved independently and must be coupled with the basic equations governing the other two systems.

Because Manning's law of resistance is an empirical formula developed for open-channel flow, it is conceivable that the application of the same law to a thin overland flow may be questionable. However, comprehensive laboratory and field studies conducted by Takasao and Kishimoto (Ta61) and Ishihara and Takasao (Is62,Is63) indicated that the Manning Law can also reasonably be applied to an overland flow system. This finding was confirmed by Foster, Huggins, and Meyer (Fo68) and Hjemlfelt (Hj81).

Subsurface Flow System - The movement of soil moisture in a subsurface flow system can be either upward or downward depending on the direction of the potential head gradient. In general, the moisture in the soil is simultaneously transported in both liquid and vapor phases. The basic equation governing this system has been derived by Currie (Cu61) and Hillel (Hi80). The results of their studies for the momentum and continuity equations are summarized as follows

$$q = -D_L (W)\frac{\partial W}{\partial z} + K(W) - bD \frac{\partial C}{\partial z} \qquad \text{(A-3)}$$

and

$$\frac{\partial W}{\partial t} = - \frac{\partial q}{\partial z} \qquad \text{(A-4)}$$

where

- q = flux of moisture in the vertical direction
- W = volumetric wetness of soil
- K = hydraulic conductivity
- D_L = hydraulic diffusivity
- b = conversion factor for transforming the vapor flux into liquid water flux
- D = diffusivity for water vapor
- C = concentration of water vapor in the air-filled void

The above equations were based on an isothermal condition and assumed that both viscous flow in the liquid phase and diffusion of vapor are impelled by the force field of capillarity and gravity. No explicit account has been taken of osmotic or solute effects on vapor pressure. In reality, the upper layer of the soil surface is warmer during the daytime and cooler

than the deeper layer during night time. The movement of vapor due to the thermal gradient effect tends to be downward during the day and upward during the night. Therefore, the error due to the assumption that the system is isothermal may compensate each other within the same day and will not significantly affect the accuracy of simulation.

Atmospheric Diffusion System - The atmospheric diffusion system transports the water vapor through the turbulent boundary layer into the atmosphere. The vapor transport through an unsteady turbulent boundary layer involves the analysis of the time-averaged boundary layer equations for the air stream and the continuity equation for the water vapor. This analysis is extremely complicated. However, by assuming the system is quasi-steady, the solution of the system equation is obtainable. The "quasi-steady" technique has been commonly used in solving unsteady mass transport systems. It assumes that within a small time increment the flow of the carrying fluid is steady and that the effect of the change from one flow state to the other due to the change in time step is negligible. By imposing the above assumption, the water vapor flux within the fully developed boundary layer was expressed by employing Fick's law for diffusion (Bi66) as

$$J = -k^2 y^2 \left| \frac{dv}{dy} \right| \frac{dC_b}{dy} \qquad \text{(A-5)}$$

where

J = water vapor flux

k = Prandtl's mixing length coefficient

v = time overaged wind speed

C_b = concentration of water vapor in the boundary layer

y = distance from ground surface

BASIC EQUATIONS FOR THE PRACTICAL METHOD

Theoretically, the basic equations compiled in the previous section can be solved numerically. However, the processes of analysis are complicated and consume excessive computation time. Several attempts have been made to solve a simplified system. For example, a conjunctive overland-subsurface flow model without atmospheric diffusion system has been developed by Akan and Yen (Ak81), and a subsurface flow-atmospheric diffusion model without overland flow system was reported by Hillel and van Bavel (Hi76a) and Hillel (Hi76b). The above models solve the dependent variables as functions of space and time and involve long and costly calculations if the real time of simulation is prolonged. For example, the time increment used in Akan and Yen's overland flow system was 30 seconds which would require excessive computation costs if the simulation were for a year or more.

One may simplify the above system equations by transforming all of the space-dependent variables into space-independent variables to avoid time-consuming simulation. The transformation for each system is described in the following sections.

Atmospheric Diffusion System - The solution of Equation (A-5) depends on many complicated fluid dynamic and boundary conditions. One of the simplest solutions is obtained by assuming that the vapor flux will not be limited by the availability of vapor transmitted from the subsurface system or the overland flow system. The vapor flux under this condition is known as the evaporation potential and is normally used as the upper bound of the evaporation rate from the overland flow and subsurface flow systems. The actual evaporation rate from these systems may then be calculated from the

conjunctive system to be discussed later. The solution of Equation (A-6) under the above condition was proposed by Rohwer (Ro31) as

$$E_p = 0.372(1-0.000374p_a)(1. + 0.6V_w)(e_s - e_a) \qquad \text{(A-6)}$$

where

E_p = evaporation potential

p_a = atmospheric pressure

V_w = wind speed

e_s = saturated vapor pressure

e_a = vapor pressure in the atmosphere

Equation (A-6) requires time-dependent input for atmospheric pressure, wind velocity, and vapor pressure to compute the diurnal variation of evaporation potential. The above information is normally not readily available from existing records. Therefore, from a practical viewpoint, the computation of evaporation potential may be reduced to a daily average level by using the Hamon Equation (Ha61)

$$E_p = Nr^2s \qquad \text{(A-7)}$$

where

N = coefficient of daily evaporation potential

r = duration of daylight in 12-hour unit/day

s = saturated absolute humidity corresponding to the daily average temperature

Equation (A-7) is used to calculate the evaporation potential which is the upper bound of the evaporation rate.

Overland Flow System - It is well known that the local and convective acceleration terms in Equation (A-1) govern the wave form and the celerity of flood waves. Kichikawa (Ki59) indicated that these terms are predominated by the friction term and suggested that they may be ignored. This conclusion was also confirmed by Takasao and Kishimoto (Ta61) and Ishihara and Takasao (Is62) of overland flow systems through laboratory and field observations. Takasao further simplified the system equations by replacing the space dependent depth of flow h by an average flow depth. After this modification, Equations (A-1) and (A-2) become

$$Q_o = \frac{\sin^{1/2} a}{n} H^{5/3} \tag{A-8}$$

and

$$\frac{dH}{dt} = P - E_o - q_o - \frac{Q_o}{L} \tag{A-9}$$

where

Q_o = rate of overland flow per unit width of trench cover

H = average depth of overland flow over the entire trench cover

L = length of slope (or half of trench width)

Furthermore, it is assumed that the rate of evaporation from the total infiltration system will be preferentially obtained from the overland flow system, then the component of evaporation rate may be written as

$$E_o = \begin{cases} E_p & \text{when } P + \frac{H}{\Delta t} > E_p \\ P + \frac{H}{t} & \text{when } E_p > P + \frac{H}{\Delta t} > 0 \\ 0 & \text{when } P + \frac{H}{\Delta t} = 0 \end{cases} \tag{A-10}$$

where

E_o = component of evaporation rate contributed by the overland flow system

On the other hand, since the maximum rate of percolation from the overland flow system cannot exceed the saturated hydraulic conductivity of the trench cap, the percolation rate can be written as

$$q_o = \begin{cases} K_s & \text{when } P - E_o + \dfrac{H}{\Delta t} > K_s \\ P - E_o + \dfrac{H}{\Delta t} & \text{when } K_s > P - E_o + \dfrac{H}{\Delta t} > 0 \\ 0 & \text{when } P - E_o + \dfrac{H}{\Delta t} = 0 \end{cases} \tag{A-11}$$

The above expressions show that there are four equations available for solving the four dependent variables, Q_o, H, E_o, and q_o. Therefore, the system can be solved independently.

Subsurface Flow System - For the purpose of transforming the space dependent variables into space independent variables, the moisture contained in the system may be divded into three components: gravity, pellicular, and hygroscopic waters. Gravity water is the moisture in a soil which can be drained by gravity force; pellicular water is the moisture in a soil which cannot be drained by gravity force but can be lost to the atmosphere through natural evaporation; and hygroscopic water is the moisture which will never be lost through the above natural processes. Furthermore, it is assumed that soil wetness can be mathematically approximated by a step-wise distribution composed of these three components. Figure A-1 shows a comparison of the step-wise wetness distribution and its corresponding original wetness distribution.

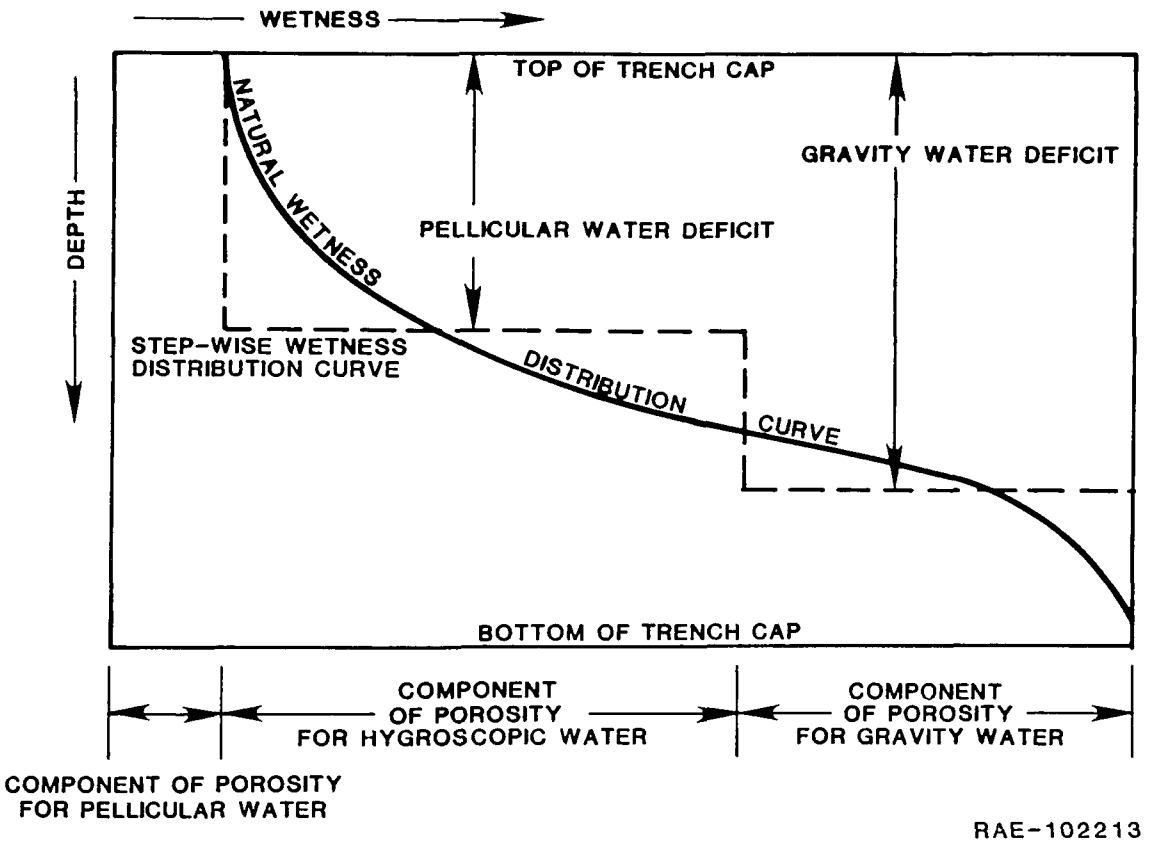

FIGURE A-1. COMPARISON OF A SCHEMATIC NATURAL WETNESS DISTRIBUTION CURVE AND ITS CORRESPONDING STEP-WISE WETNESS DISTRIBUTION CURVE.

Based on the above concept, the dependent variables appearing in the basic differential equations may be divided into three components, i.e.,

$$W = W_1 + W_2 + W_3$$

and $$q = q_1 + (q_2 + q_t) + (q_3 - q_t) \tag{A-12}$$

where

W_1 = component of wetness for the hygroscopic water

W_2 = component of wetness for the pellicular water

W_3 = component of wetness for the gravity water

q_1 = component of flux for the hygroscopic water

q_2 = component of flux for the pellicular water

q_3 = component of flux for the gravity water

q_t = flux of moisture being transformed from gravity water to pellicular water

Substituting Equation (A-12) into Equations (A-3) and (A-4) gives

$$q_3 = -D_L(W)\frac{\partial W_3}{\partial z} + K(W) \tag{A-13}$$

$$K(W) = 0, \text{ when } W_3 = 0$$

and $$\frac{\partial W_3}{\partial t} = -\frac{\partial(q_3 - q_t)}{\partial z} \tag{A-14}$$

for the component of gravity water and

$$q_2 = -D_L(W)\frac{\partial W_2}{\partial z} + K(W) - bD_v\frac{\partial C}{\partial z} \tag{A-15}$$

$$K(W) > 0 \quad \text{when } W_3 > 0$$

and $$\frac{\partial W_2}{\partial t} = -\frac{\partial(q_2 + q_t)}{\partial z} \tag{A-16}$$

for the component of pellicular water.

Applying the step-wise wetness distribution concept to the above system equations, Equations (A-13) and (A-14) may be rewritten by substituting the proper difference forms of the dependent variables. The result is

$$q_i = \begin{cases} K_s & \text{when } Z_g < Z_{max} \\ 0 & \text{when } Z_g = Z_{max} \end{cases} \tag{A-17}$$

and

$$\frac{dZ_g}{dt} = (q_1 - q_o + q_t)/W_g \tag{A-18}$$

where

q_i = flux of moisture infiltrating into the trench

K_s = saturated hydraulic conductivity of the soil

Z_g = deficit of gravity water

Z_{max} = maximum of deficit of gravity water (equivalent to the thickness of the trench cover)

W_g = component of wetness for the gravity water under a fully saturated conditions and is numerically identical to the porosity for the gravity water

To simplify the solution of Equation (A-15), the component of moisture flux for the pellicular water, q_2, is computed based on the assumptions that (1) the flow is predominated by liquid phase transport or (2) the flow is predominated by vapor phase transport. The moisture flux for the pellicular water at any instant is then chosen from the larger flux calculated from the above two assumptions.

When the flow is predominated by a liquid phase transport, the third term in Equation (A-15) vanishes. Substituting the proper difference form of the dependent variables gives

$$q_L = -D_e W_p / Z_p + K_e$$

$$\text{and} \quad q_L < E_p - E_o \tag{A-19}$$

where

q_L = flux of pellicular water transported in the liquid phase

W_p = component of wetness for the pellicular water under fully saturated condition and is numerically identical to the porosity for pellicular water

Z_p = deficit of the pellicular water

D_e = hydraulic diffusivity at equivalent wetness

K_e = hydraulic conductivity at equivalent wetness

The equivalent wetness for the purpose of this study is calculated by

$$W_e = W_h + SW_p \tag{A-20}$$

where

W_h = component of wetness for the hygroscopic water under fully saturated condition and is numerically identical to the porosity for hygroscopic water

S = coefficient of equivalent wetness for pellicular water which should be greater than 0.5 and smaller than 1.0

When the flow is predominated by vapor phase transport, the first and second terms on the righthand side of Equation (A-15) disappear. Substituting the proper difference form of the dependent variables, Equation (A-15) becomes

$$q_v = -bD_v(C_s - C_o)/Z_p \tag{A-21}$$

where

q_v = flux of moisture being transported in the vapor phase

D_v = diffusivity of water vapor in the trench cap

Z_p = deficit of pellicular water

C_s = water vapor concentration at the front of pellicular water

C_o = water vapor concentrations at the top of trench cover

The diffusion of water vapor from the top of the trench cover to the atmosphere can be written based on Fick's Law as

$$q_v' = -bD_a(C_o - C_a)/T_b \tag{A-22}$$

where

q_v' = flux of water vapor in the boundary layer of diffusion

D_a = diffusivity of vapor in the boundary layer

T_b = equivalent thickness of boundary layer

C_a = water vapor concentration in the atmosphere outside the boundary layer

Since the vapor flux in the trench cover and in the boundary layer should be identical, Equations (A-21) and (A-22) may be combined to yield

$$q_v = -\frac{bD_a(C_s - C_o)/T_b}{1 + \frac{D_a}{T_b D_v} Z_p} \tag{A-23}$$

Furthermore, the ratio of diffusivities, D_a/D_v, may be obtained from Penman's (Pe40) study on the diffusion of vapor through a porous solid and Rohwer's (Ro31) study on evaporation potentials. Substituting their results into Equation (A-23) and replacing the numerator with the evaporation potential yields

$$q_v = -\frac{E_p - E_o}{1 + \frac{1 + 0.6V_w}{T_b} \frac{Z_p}{0.66(W_p + W_g)}} \tag{A-24}$$

For the purpose of this study, the coefficient, $(1+0.6V_W)/T_b$, is assumed to be a constant and equal to $0.5m^{-10}$ (see section on Model Testing). When this coefficient is used, Equation (A-24) becomes

$$q_v = -\frac{E_p - E_o}{1 + \frac{0.5z_p}{0.66(W_p + W_g)}} \tag{A-25}$$

Now, the flux of moisture being lost through evaporation q_p can be computed by

$$q_p = -\mathrm{Max}(\,|\,q_L\,|\,,\,|\,q_v\,|\,) \tag{A-26}$$

where the Max function implies the selection of the largest number among those values designated in the parentheses.

Again, by substituting the proper difference form of the dependent variables, Equation (A-16) becomes

$$dZ_p/dt = -(q_p + q_t)/W_p \tag{A-27}$$

where q_t may be computed by

$$q_t = \begin{cases} q_o & \text{when } Z_p > 0 \\ 0 & \text{when } Z_p = 0 \end{cases} \tag{A-28}$$

Equations (A-8 through A-10), (A-11), (A-17 through A-19), and (A-25 through A-28) are the basic equations for the practical infiltration model. There are eleven equations and eleven dependent variables in the system

equations. Since all of the dependent variables are space independent and each pair of momentum and continuity equations can be solved in sequence, the computer coding for the mathematical model is greatly simplified from that of the original system equations. A computer model has been developed for testing the proposed system equations and for the application of the model to the evaluation of a low-level waste disposal site.

MODEL TESTING

The overland flow system using Equations (A-8) and (A-9) as basic equations, has been studied through laboratory studies and field observations by Takasao (Ta61). They concluded that the space independent system equations, as represented by Equations (A-8) and (A-9), can reasonably be used to characterize the nature of overland flow for flood routing purposes. Since the primary purpose of this study is the simulation of infiltration rates, the proposed overland flow system equations are judged to be adequate for the infiltration model. Testing of the model was therefore concentrated on the subsurface flow system. To test the subsurface flow system submodel, three special cases with simplified boundary conditions were selected. The results of simulation were compared with existing studies having the similar boundary conditions. They are described in the following sections.

Gravity Water Drainage Without Evaporation - A computer simulation of gravity water drainage by gravitational force without evaporation loss was conducted by Hillel and van Bavel (Hi76a). This simulation was conducted by solving equations similar to the basic differential equations expressed in Equations (A-3) and (A-4) without the term representing the vapor phase transport. The results of simulation for a sandy soil (1.16 m thick) were

used to calculate the cumulative water volume being drained from the soil (see Figure A-2).

By applying the same boundary conditions used in Hillel and van Bavel's study to the model developed for this study, the cumulative water volume drained from the soil is simulated and plotted in Figure A-2. The comparison of the two results indicates that the proposed model does not simulate the time variation of the drainage precisely, but the calculation of the cumulative water volume drained from the soil is reasonably close to the results obtained from Hillel and van Bavel's simulation. Since the main purpose of the simulation described in this study is to obtain the cumulative volume of water infiltrated into the waste trench, the proposed model simulates the rate of infiltration with acceptable accuracy.

Steady State Evaporation Rate - The steady state evaporation rate from a clay soil system with a relatively high groundwater table has been analyzed by Ripple (Ri72). This study analyzed the evaporation rate by solving the system equation characterizing the water flux transported in the subsurface flow system and the vapor flux diffused into the atmosphere. The results of his analysis for the groundwater table at depths of 0.6, 0.9, 1.2, and 1.8 meters are shown in Figure A-3.

By applying similar parameters and boundary conditions to the proposed model and reorganizing the results into the form corresponding to Ripple's results, the results as shown in Figure A-3. The results of simulation from the proposed model fit reasonably well with those obtained by Ripple.

The above evaporation rate was simulated for a relatively shallow groundwater table. Therefore, the transport of soil moisture is dominated

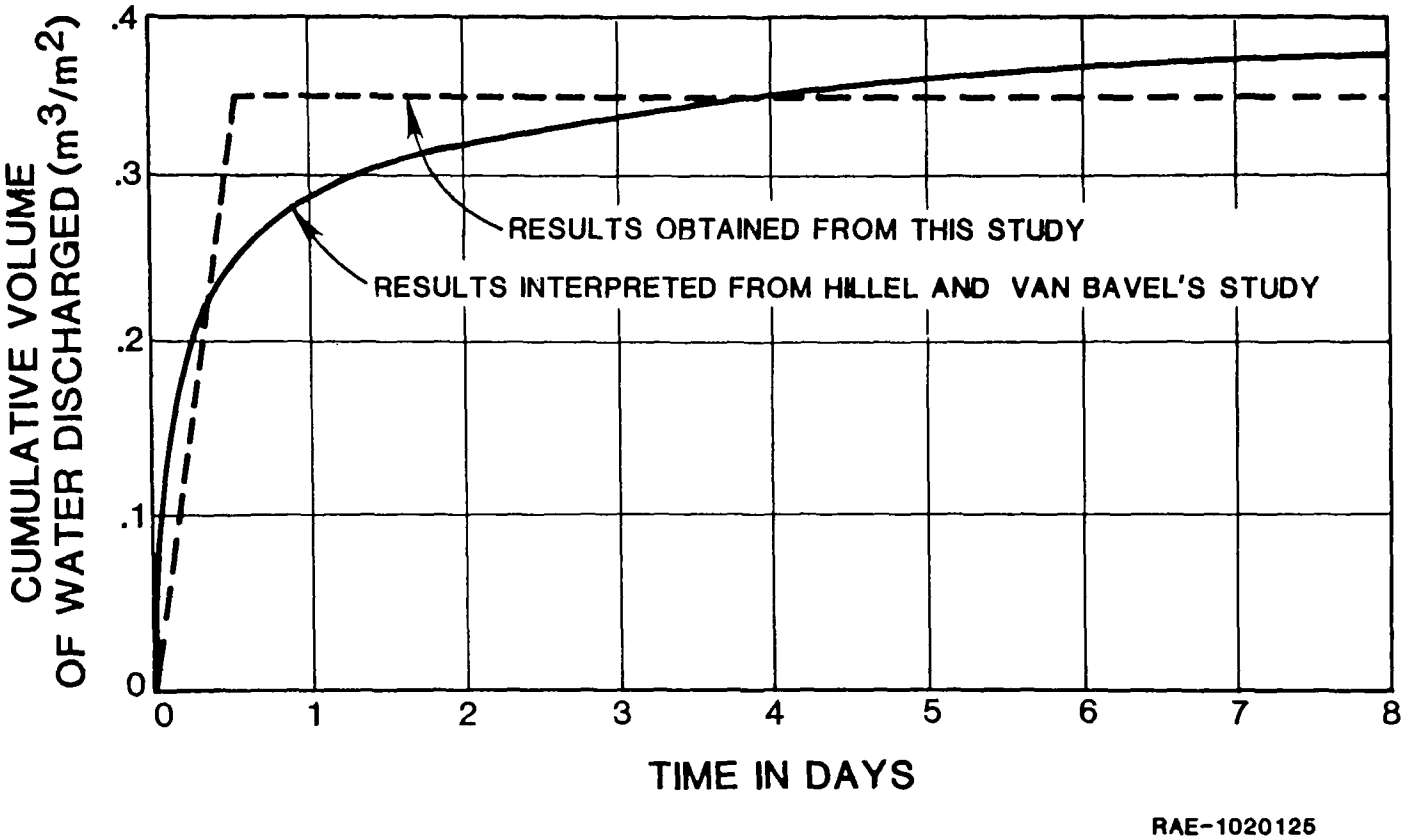

FIGURE A-2. COMPARISON OF THE RESULTS OF SIMULATIONS FOR GRAVITY DRAINAGE OF A TYPICAL SAND USING THE PROPOSED MODEL AND RESULTS INTERPRETED FROM HILLEL AND VAN BAVEL's STUDY.

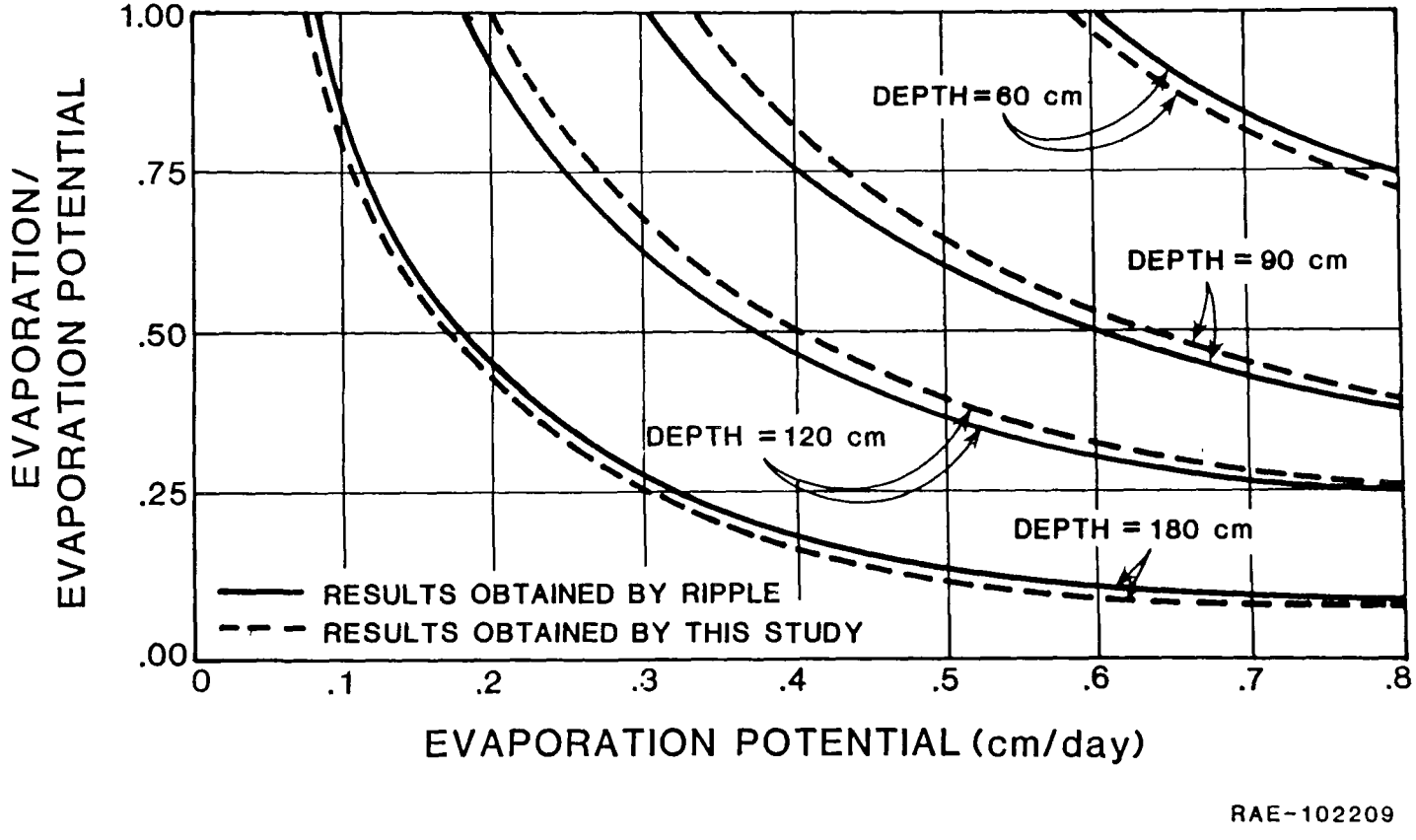

FIGURE A-3. COMPARISON OF THE RESULTS OF SIMULATION FOR THE STEADY STATE EVAPORATION OF CHINO CLAY BY USING THE PROPOSED MODEL AND THAT OBTAINED BY RIPPLE.

by liquid phase transport. However, when the groundwater table is very deep, the steady state rate of evaporation may be dominated by the vapor phase transport. The computation of vapor transport in Equation (A-25) becomes important under this condition.

Vapor Phase Transport - There are no data available for evaluating the transport of pellicular water in a vapor phase. To compare the characteristics of vapor phase transport and liquid phase transport, the flux of moisture being transported by these two phases are computed for various depths of pellicular water deficit based on Equations (A-19), (A-25), and (A-26), using evaporation potentials of 0.3 and 0.6 cm/d. The results of analysis for both sandy and clay soils are plotted and presented in Figure A-4. The coefficient represented by $(1.+0.6V_w)/T_b$ as $0.5m^{-1}$ was selected to obtain the best transition from liquid phase transport to vapor phase transport for both soils and for the evaporation potentials which ranged from 0.3 to 0.6 cm/d. The trend of pellicular water flux variations with the increase in the pellicular water deficit agrees, in general, with the trend described by Philip (Ph74).

APPLICATION OF MODEL TO AN ACTUAL WASTE DISPOSAL SITE

The low-level radioactive waste disposal site located at Barnwell, South Carolina, was selected for an application of the model to an actual waste disposal site. The trench cover is constructed in two soil layers, loamy sand (120 cm thick) at the top and clay soil (80 cm thick) at the bottom. First, the analysis simulated the infiltration rates for a homogeneous trench cover for each of the soil types constituting the trench cover; then, the rate of infiltration for the composite trench section was interpreted from the results.

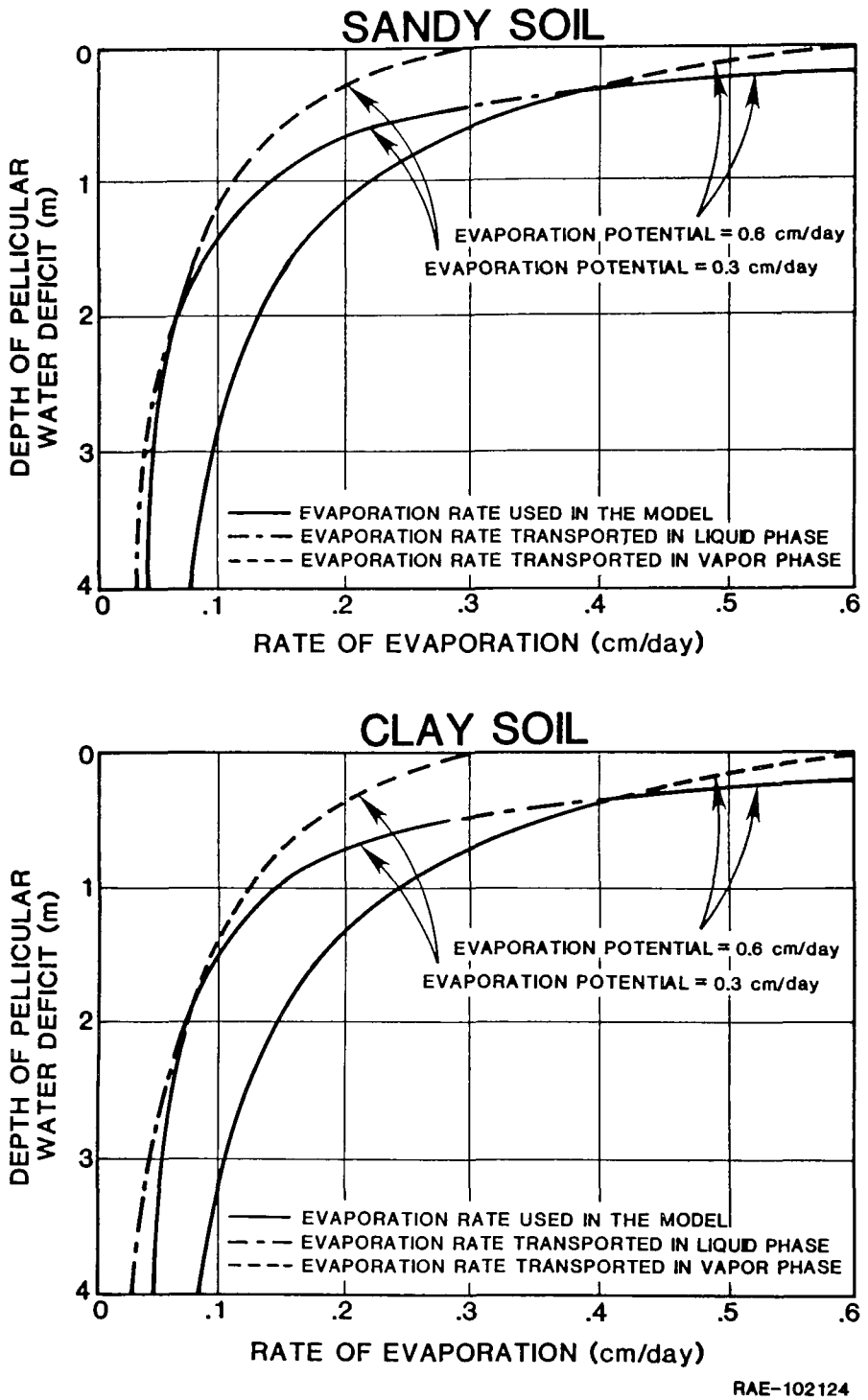

FIGURE A-4. TRANSITION IN PELLICULAR WATER TRANSPORT FROM LIQUID PHASE TO VAPOR PHASE CALCULATED FROM EQNS. A.19, A.25, AND A.26 FOR SANDY AND CLAY SOILS.

The soil characteristics for both layers were interpreted from basic data reported by Chem-Nuclear Systems, Inc. (CNS80), and the results used for the computer simulation are listed in Table A-1.

Using the annual rainfall of 118 cm/yr and the rainfall distribution pattern observed in Augusta, Georgia, the simulation was conducted for the loamy sand and the clay soil. The results of simulation, presented in Figure A-5, indicate that the infiltration rates are 45 cm/yr for the loamy sand and 7 cm/yr for the clay soil. It is obvious that the rate of infiltration will not be affected by the soil characteristics below the level of annual maximum deficit for pellicular water plus the allowance for gravity water storage. The pellicular water deficit simulated for the loamy sand (Figure A-6) indicated that the maximum pellicular water deficit reached the level of approximately 50 cm which is far above the upper boundary of the clay soil layer (120 cm below the top of trench cap). Thus, one may conclude that the infiltration rate for the Barnwell site is controlled by characteristics of the top layer, and the rate of infiltration is 45 cm/yr or 17.7 in/yr.

The above analyses assumed a uniform rainfall distribution within the hourly period when the rainfall was recorded. However, the rainfall distribution within any hour in a real case is not necessarily uniform. By assuming a peak factor of two, i.e., all of the rainfall being recorded hourly is assumed to be concentrated in the second half of the recorded period, and reapplying the same input data to the infiltration model to rerun the case for the loamy sand (top layer), the rate of infiltration is 35 cm/yr or 13.8 in/yr. Therefore, the infiltration rate for the Barnwell site may vary between 13.8 and 17.7 in/yr due to the nonuniform hourly rainfall distribution.

TABLE A-1

SOIL CHARACTERISTICS OF THE TRENCH COVER USED IN SIMULATING THE ANNUAL RAINFALL INFILTRATION, BARNWELL, RADIOACTIVE WASTE DISPOSAL SITE

Type	Saturated Hydraulic Conductivity (m/hr)	Equivalent Upward Diffusivity (m^2/hr)	Equivalent Upward Conductivity (m/hr)	Porosity Total	Porosity Pellicular Water	Porosity Gravity Water
Top Layer Loamy Sand	0.02	0.02×10^{-2}	0.12×10^{-5}	0.51	0.24	0.25
Bottom Layer Clay Soil	0.005	0.13×10^{-3}	0.01×10^{-4}	0.54	0.44	0.08

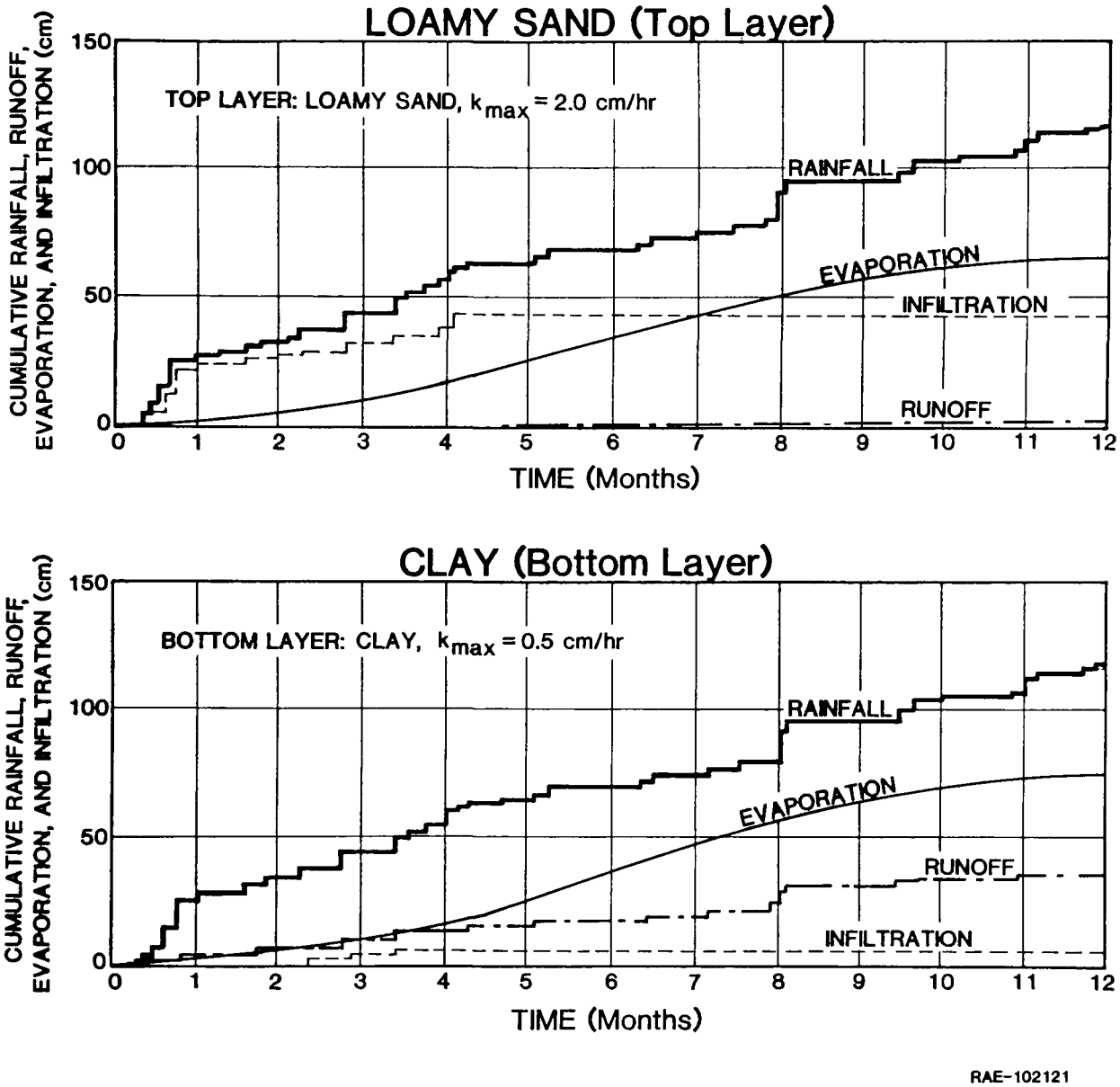

FIGURE A-5. THE RESULTS OF SIMULATIONS USING THE PROPOSED MODEL FOR LOAMY SAND AND CLAY SOIL, BARNWELL, S.C., RADIOACTIVE WASTE DISPOSAL SITE.

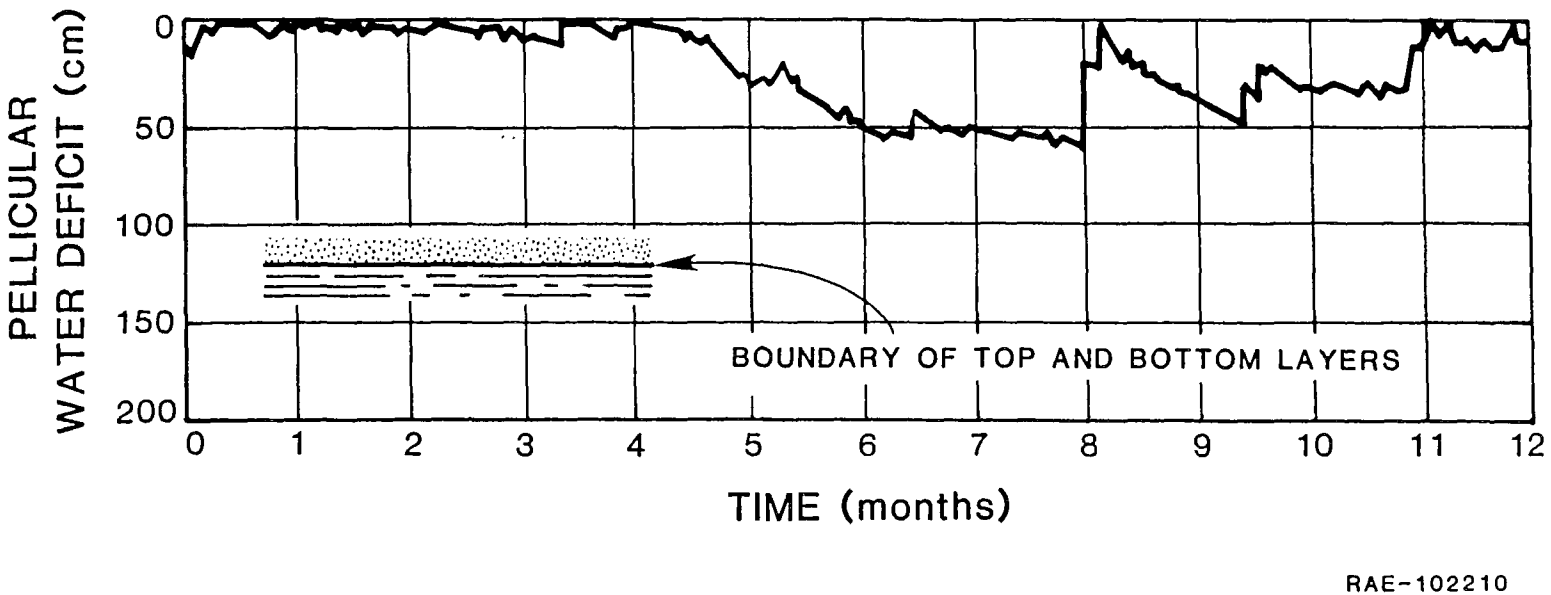

FIGURE A-6. THE RESULTS OF SIMULATING THE VARIATION OF PELLICULAR WATER DEFICIT, LOAMY SAND, BARNWELL, S.C. RADIOACTIVE WASTE DISPOSAL SITE.

Independent studies have been conducted by the U.S. NRC (NRC82) and the U.S. Geological Survey on the rate of infiltration for the Barnwell site, based on an analysis of groundwater being discharged to a nearby creek. NRC's analysis indicated a rate of 14 in/yr; USGS's, 14 to 17 in/yr. The results of simulation using the infiltration model are in agreement with these two studies.

CONCLUSIONS

1. The space-dependent variables in the momentum and continuity equations of a subsurface flow system can be transformed into space-independent variables by breaking down the soil moisture into gravity water, pellicular water, and hygroscopic water components.

2. The results of transformations for the overland flow system, the subsurface flow system, and the atmospheric diffusion system have greatly simplified the computational procedures for simulating these systems and have improved the stability and efficiency of the numerical analysis.

3. Although the transformed system equations cannot simulate the dynamic response of soil moisture as precisely as the original partial differential equations, the derived system equations can be used to simulate the long-term infiltration rates with reasonable accuracy.

4. When the model was applied to the Barnwell radioactive waste disposal site, the results of simulation fit very well with the results of analysis conducted by other investigators

using the other methods. This fact implies that the proposed model can be employed to simulate the infiltration of rain-water through the cap of a waste disposal trench.

5. The proposed practical infiltration model is suitable for integrating into a larger system model for risk assessment of a low-level radioactive waste disposal site.

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APPENDIX B

AN OPTIMUM GROUNDWATER TRANSPORT MODEL FOR APPLICATION TO THE ASSESSMENT OF HEALTH EFFECTS DUE TO LAND DISPOSAL OF RADIOACTIVE WASTES

APPENDIX B

This appendix presents background and theory for the groundwater transport model and DDETA correction factor used by the MAIN routine and VERHOR subroutine in the PRESTO-EPA-POP code. The model was developed by Hung and the original publication is included here.

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AN OPTIMUM GROUNDWATER TRANSPORT MODEL FOR APPLICATION TO THE ASSESSMENT OF HEALTH EFFECTS DUE TO LAND DISPOSAL OF RADIOACTIVE WASTES

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ABSTRACT. This paper presents a groundwater transport model for simulating radionuclide transport in an aquifer, using an approximate solution of the basic transport equation. The model is designed to avoid (1) relatively high computer simulation costs normally experienced in numerical models and (2) the large errors sometimes introduced when the physical boundary conditions are converted to a mathematical form suitable for the analytical model. The model neglects initially the effect of radionuclide transport through dispersion and compensates for this effect subsequently with a health effects correction factor. This correction factor is found to be a function of the Peclet number, a dimensionless parameter expressing the relative importance of diffusion and convective transportation, and the "transport number," which has been defined and can be determined by using the parameters of the groundwater transport system. The model has a low cost of simulation and yet maintains reasonable accuracy of predicting the cumulative radionuclide flowing through a section. This is the primary output expected from a groundwater transport model for health effects assessments. The model has been integrated into the PRESTO–EPA model designed for the prediction of radiation effects due to a shallow-trench operation.

INTRODUCTION

Comprehensive studies on the evaluation of an optimum method of diposing radioactive wastes have been conducted by various government agencies (1,2) and by an Interagency Review Group (3). All of these studies have unanimously concluded that the geological disposal method is one of the most viable alternatives. Because the transport of radionuclides through an aquifer to the biosphere is a primary pathway, the groundwater transport model, which simulates the migration of radionuclides in an aquifer, becomes one of the important submodels required in a health effects assessment model.

To date, there are more than 108 groundwater transport models (4) available. However, some of the models fail to consider the process of radionuclide decay and/or sorption and, therefore, are not suitable for the assessment of a radioactive waste disposal site. The rest of the models, which may be suitable for health effects evaluation, can be subdivided into two groups: the analytical model and the numerical model. In general, the application of an analytical model is limited by its specific form of boundary conditions. Therefore, when the actual boundary conditions do not match the form that the model called for, some approximation of its actual boundary conditions would be required before the analytical solution could be applied. As a result, these models may suffer from considerable error due to the approximation of boundary conditions.

The application of a numerical model generally requires many more tedious computations than the analytical approach. For example, the accuracy of simulation depends greatly on the adjustments of time and space increments; severe errors may result if they are improperly adjusted. On the other hand, a proper adjustment of these increments may result in consuming excessive computer time for the simulation, in some cases. Therefore it is conceivable that the cost of simulation may become prohibitive when the model is applied to long-term simulations such as health effect evaluations.

The basic groundwater transport model aiming for health effects assessment was developed by Hung (8) and was integrated into the PRESTO-EPA model (9), a model to predict the radiation effects from

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Acknowledgements—The author is grateful to Mr. G. Lewis Meyer of EPA for his valuable suggestions and continuous encouragement throughout the development of this model. Special acknowledgements should be made to Dr. Akio Ogata of USGS and Professor Donald S. Cohen of the California Institute of Technology for their suggestions and criticisms during the course of model review.

shallow trench operation. Since then, the same model has been improved in theoretical background and in practical application.

The purpose of this paper is to present the groundwater transport model used in the PRESTO-EPA model and its characteristics.

APPLICATION OF EXISTING MODEL IN HEALTH EFFECTS EVALUATION

The basic equations for a groundwater transport system include the momentum, the energy, and the continuity equations for the hydrodynamic system and for the solute. Using tensor notations, the equations take the form (10)

$$\underline{v} = -(k/\mu)(\nabla p - \varrho g \nabla z) \quad (1)$$

$$\nabla \cdot [(\varrho k/\mu)H(\nabla p - \varrho g \nabla z)] + \nabla \cdot \underline{D}_h \cdot \nabla T - q_L - q'H = (\partial/\partial t)[n\varrho U + (1 - n)(\varrho c_p T)] \quad (2)$$

$$\nabla \varrho \underline{v} + q' = -(\partial/\partial t)(n\varrho) \quad (3)$$

$$\nabla \cdot [\varrho C(k/\mu)(\nabla p - \varrho g \nabla z)] + n\nabla \cdot \varrho \underline{D}_c \cdot \nabla C - q'C - \varrho n \lambda_d RC = (\partial/\partial t)(\varrho n RC), \quad (4)$$

in which C is the concentration of the radionuclide in the fluid phase; $\underline{v}$ is the velocity vector; k is the permeability; μ is the viscosity of the fluid; p is the pressure; ϱ is the mass density; g is the gravitational acceleration; $\underline{D}$ is the dispersivity tensor; T is the temperature; q_L is the rate of heat loss; H is the fluid enthalpy; n is the porosity; z is the height above reference plane; λ_d is the radionuclide decay constant; R is the retardation factor; U is the internal energy; c_p is the specific heat; q' is the rate of fluid withdrawal; and subscripts h and c are the heat energy and component of mass, respectively.

The preceding non-linear equations characterize the transport of radionuclides in a groundwater system. Since each of the preceding equations are related through dependent variables, the direct solution of the system equation for any boundary and initial conditions is extremely difficult. A commonly used practice in solving the system equation is to assume that the flow of groundwater is steady and that there is no heat energy being generated or absorbed in the system. The system equation then reduces to a single equation:

$$R(\partial C/\partial t) - \nabla \cdot (\underline{D} \cdot \nabla C) + V\nabla C + \lambda_d RC = O, \quad (5)$$

in which V is the interstitial velocity ($V = \underline{v}/n$). Equation 9 has been further simplified and solved by numerous investigators (10,11,12, and 13) employing numerical or analytical approaches. However, some difficulties are often encountered in applying these models for health effects assessments. They are described in the following sections.

Numerical Models

The existing multidimensional models are solved either by the finite-difference method (10) or by the finite-element method (11,12, and 13). A model employing the finite-element method is, in general, found to be more efficient than one employing the finite-difference method (12). However, the time required to execute a computer model employing the finite-element method is still far beyond the limitations of a normal project budget for risk assessment when numerous cases with long durations of simulation must be considered.

Analytical Models

Lester, Jansen, and Burkholder (6) developed an analytical groundwater transport model for a one-dimensional, semi-finite aquifer system having impulse release and decaying band release boundary conditions. These boundary conditions at x = 0 were expressed as

$$C = C_o\delta(t) \quad (6)$$

and

$$C = (C_o/t_d)\exp(-\lambda_d t) \quad (7)$$

for the impulse release and the decaying band release, respectively. In the preceding equation, x is the space coordinate, t is the time, t_d is the duration of radionuclide leaching, and C_o is the concentration of radionuclides at t = 0.

Ford, Bacon, and Davis, Inc. employed an analytical model that assumed that the rate of radionuclide release at any time is proportional to the inventory of radionuclides remaining at the source point (7). Mathematically, it is expressed as

$$C = (\lambda_L I_m/Q_w)\mathrm{Exp}[-(\lambda_d + \lambda_L)t], \quad (8)$$

in which λ_L is the leaching constant, I_m is the initial inventory of the radionuclides in the waste depository, and Q_w is the rate of groundwater flow.

These analytical models have made considerable contributions in groundwater transport modeling by simplifying the simulation procedures. However, the application of these models to health effects assessments is limited, because it requires the approximation of converting the actual boundary condition to a form meeting the requirements of the analytical model. This may result in considerable simulation error in some cases.

The preceding discussion implies that existing numerical and analytical groundwater models may either be too costly to compute or may introduce large errors when applied to health effect assessments. Therefore, a more accurate and more economic groundwater transport model has been developed for health risk assessments and is presented in this paper.

THEORETICAL BACKGROUND OF THE OPTIMUM GROUNDWATER TRANSPORT MODEL FOR HEALTH EFFECTS ASSESSMENTS

Derivation of Basic Equation

The groundwater transport model to be discussed in this section simulates the transport of radionuclides from a disposal site to a point where radionuclides are pumped out to the biosphere for human uptake or discharged into a surface stream for another mode of transportation. To simplify the model, it is assumed that the flow of the fluid carrying the radionuclides is steady, uniform, and one-dimensional. It is also assumed that the dissolved radionuclides are in equilibrium with those adsorbed by the solids in the aquifer formation and that decay is in progress for both dissolved and absorbed radionuclides.

The basic one-dimensional groundwater transport equation for the model simulating radionuclide migration in an aquifer may be reduced from Eq. 5 to

$$D(\partial^2 C/\partial x^2) - V(\partial C/\partial x) - R(\partial C/\partial t) - \lambda_d RC = 0 \quad (9)$$

which is to be solved by the following initial and boundary conditions:

$$C = 0, \quad \text{at all } x, \quad \text{when } t = 0, \quad (10)$$

$$C = C_o(t), \quad \text{at } x = 0, \quad \text{when } t < 0, \text{ and}$$
$$C = \text{finite}, \quad \text{at } x = \infty \quad \text{when } t > 0. \quad (11)$$

For the convenience of analysis, one may transform the dependent variable from radionuclide concentration, C, into the rate of radionuclide transport by multiplying Eqs. 9, 10, and 11 by the rate of groundwater flow. When this transformation is completed, the preceding equations become:

$$D(\partial^2 Q/\partial x^2) - V(\partial Q/\partial x) - R(\partial Q/\partial t) - \lambda_d RQ = 0 \quad (12)$$

$$Q = 0, \quad \text{at all } x, \quad \text{when } t = 0,$$

$$Q = Q_o(t), \quad \text{at } x = 0, \quad \text{when } t > 0, \text{ and} \quad (13)$$

$$Q = \text{finite}, \quad \text{at } x = \infty \quad \text{when } t > 0,$$

where Q denotes the rate of radionuclide transport.

Since Eq. 13 is an undefined boundary condition, the analytical solution for Eq. 12 cannot be obtained. However, one may express its solution into a convolution form expressed by:

$$Q(t) = \int_0^t Q_o(t - \tau)u(\tau)d\tau, \quad (14)$$

in which u denotes the radionuclide release rate at the discharge end and, $x = L$, which responds to the unit release of a radionuclide at $x = 0$ and when $\tau = 0$. This function is normally known as unit response function.

The preceding unit response function, $u(\tau)$, has been thoroughly studied by Burkholder et al. (6), with the following results:

$$u(\tau) = (V/2L)\sqrt{(RP/\pi\theta^3)}\,\mathrm{Exp}\{-N_d\theta - (P\theta/4R)[(R/\theta) - 1]^2\}. \quad (15)$$

in which P is the Peclet number ($= VL/D$); θ is the dimensionless time ($= \tau V/L$); N_d is the decay number ($= \lambda_d L/V$). By substituting Equation 15 into Eq. 14, one obtains:

$$Q(t) = \int_0^t Q_o(t - \tau) \cdot (V/2L)\sqrt{(RP/\pi\theta^3)}\,\mathrm{Exp}\{-N_d\theta - (P\theta/4R)[(R/\theta) - 1]^2\}d\tau. \quad (16)$$

Equation 16 cannot be integrated because $Q_o(t - \tau)$ is undefined. However, if the dispersion term, $D(\partial^2 Q/\partial x^2)$, in Eq. 12 is neglected, then the unit response function, $u'(\tau)$, for the modified Eq. 12 becomes (6):

$$u'(\tau) = \mathrm{Exp}(-RL\lambda_d/V)\delta(\tau - RL/V) \quad (17)$$

As a consequence, Eq. 14 can be integrated, and the result is

$$Q'(t) = Q_o(t - RL/V)\mathrm{Exp}(-\lambda_d RL/V). \quad (18)$$

In Eqs. 17 and 18, δ denotes the delta function, and the prime on u' and Q' denotes variables responding to the system with the dispersion term being neglected.

Although Eq. 18 is easier to calculate than Eq. 16, the results obtained from Eq. 18 include an error resulting from neglecting the dispersion term in the basic transport equation. To compensate for this error, a correction factor defined by

$$\xi(P, \lambda_d R/V, t) = Q(t)/Q'(t) \quad (19)$$

has been introduced. After this correction factor has been characterized, the rate of radionuclide transport $Q(t)$ can be obtained by combining Eq. 18 and 19. The result is:

$$Q(t) = \xi Q_o(t - RL/V)\mathrm{Exp}(-RL\lambda_d/V). \quad (20)$$

In Eq. 20, the correction factor ξ is expected to be a function of t, P, and $\lambda_d RL/V$; the complexity of this term prohibits its application. This correction factor is further transformed into time independent factor based on the ground that the trasport model is primarily for the purpose of risk assessment. The transformation is derived as follows:

The assessment of health effects due to direct and/or indirect ingestion of groundwater contaminated with radionuclides involves complicated simulation of the history of organ doses. Such an assessment also requires the probabilistic analysis of the occurrence of fatal cancer effects following the simulation of radionuclide transport in the acquifer. Due to the complexity of the dose and health effects simulation, a linear model is generally employed in vari-

ous risk assessment models. An example of this application is the PRESTO-EPA model (9).

When a linear model is employed, the annual health effects due to the ingestion of a specific radionuclide, E_i, for a community with a population of P' can be expressed by (14):

$$(H_e)_i = (P'/T_e) \sum_{l=1}^{M} E_i(R_f)_{i,l}, \tag{21}$$

in which R_f is the health risk conversion factor per unit rate of chronic ingestion, T_e is the number of years of life expectancy, E is the rate of radionuclide exposure, and i and l denote the order of radionuclide and human organs. Furthermore, since a risk assessment involves the estimates of health effects for infinite generations residing at the community of interest, the total health effect due to the exposure rate of E_i pCi/yr is calculated by the following generalized equation:

$$(H_t)_i = \int_0^\infty (P'/T_e) \cdot \sum_{l=1}^{M} E_i(R_f)_{i,l} \mathrm{d}t$$

$$= (P'/T_e) \cdot \sum_{l=1}^{M} (R_f)_{i,l} \int_0^\infty E_i dt, \tag{22}$$

where H_t is the total health effects and t is the time.

Since the annual rate of pumping radionuclides that will be ingested by humans is directly proportional to the rate of radionuclides being transported to the wellpoint, one may write:

$$E_i = rQ_i/P' \tag{23}$$

where r is the ratio of the rates of radioactivity expected to be uptaken by the inhabitants of the community to the rate of radioactivity reaching the wellpoint, and Q_i is the rate of radionuclides being transported to the wellpoint.

Substituting Eq. 23 to Eq. 22, one obtains

$$(H_t)_i = (r/T_e) \sum_{l=1}^{M} (R_f)_{i,l} \int_0^\infty Q_i dt. \tag{24}$$

Equation 24 indicates that the total health effects depend on the total cumulative activity of radionuclides reaching the wellpoint and are independent of the time variation of the transport rate. Therefore, for the purpose of health effects assessment, *the key input required from a groundwater transport model is the cumulative radionuclides reaching the wellpoint over an infinite period of time.* This is the key concept used in developing the groundwater model presented in this paper.

Since the goal of health effects assessments is estimating fatal health effects, one may also introduce a long-term health effects correction factor, η, defined by:

$$\eta = H_t/H_t', \tag{25}$$

where H_t and H_t' represent the total health effects obtained from the groundwater pathway at the discharge end with and without considering the dispersion term, respectively.

For the purpose of developing a ground water transport model, one may assume that the population affected by the well water remain constant and that the total health effects due to the released radionuclides are so small that they would not alter the size of the population at risk and the other factors affecting the health effects. If this is the case, there will be a linear relationship between the cumulative radionuclides reached at that point and the expected total health effects. Substituting Eq. 24 into Eq. 25 and lumping together all terms affecting the health effects conversion factors into a single health effects conversion factor, then Eq. 25 can be rewritten as:

$$\eta = [(F_h)\int_0^\infty Q(t)dt]/[(F_h)\int_0^\infty Q'(t)dt]. \tag{26}$$

where F_h denotes the conversion factor for health effect from cumulative radionuclide release.

Subsequent substitution of Eq. 14 into Eq. 26 yields:

$$\eta = [\int_0^\infty \int_0^t Q_0(t-\tau)u(\tau)d\tau dt]/[\int_0^\infty \int_0^t Q_0(t-\tau)u'(\tau)d\tau dt. \tag{27}$$

When the order of integration of the double integral for both numerator and denominator is reversed, Eq. 27 can be rewritten as:

$$\eta = [\int_0^\infty \int_\tau^\infty Q_0(t-\tau)u(\tau)dtd\tau]/[\int_0^\infty \int_\tau^\infty Q_0(t-\tau)u'(\tau)dtd\tau]. \tag{28}$$

Furthermore, when the independent variable, t, is transformed to σ such that $\sigma = t - \tau$, Eq. 28 becomes

$$\eta = [\int_0^\infty \int_0^\infty Q_0(\sigma)u(\tau)d\sigma d\tau]/[\int_0^\infty \int_0^\infty Q_0(\sigma)u'(\tau)d\sigma d\tau]. \tag{29}$$

Since Q_0 is independent of τ and u and u' are independent of σ, the double integrals in the denominator and numerator can now be separated into products of single integrals and yield:

$$\eta = [\int_0^\infty Q_0(\sigma)d\sigma \cdot \int_0^\infty u(\tau)d\tau]/[\int_0^\infty Q_0(\sigma)d\sigma \cdot \int_0^\infty u'(\tau)d\tau],$$

which can be simplied as:

$$\eta = [\int_0^\infty u(\tau)d\tau]/[\int_0^\infty u'(\tau)d\tau]. \tag{30}$$

Equation 30 implies that the health effects correction factor, η, is independent of environmental time and, hence, can be characterized independent of time. Once the health effects correction factor is determined, the rate of radionuclides passing the discharge end may be approximated by:

$$Q(t) = \eta Q_0(t - RL/V)\mathrm{Exp}(-RL\lambda_d/V). \tag{31}$$

Equation 31 is the basic equation of this groundwater transport model.

Characterization of Health Effects Correction Factor
To determine the characteristics of the health effects correction factor, one may first substitute Eqs. 15 and 17 into Eq. 30 and then complete the integration of the denominator. This yields

$$\eta = \frac{\int_0^\infty (1/2)\sqrt{RP/\pi\theta^3} \cdot \mathrm{Exp}[-N_d\theta - (P\theta/4R)(R/\theta - 1)^2]d\theta}{\mathrm{Exp}(-RL\lambda_d/V)}. \quad (32)$$

The numerator of Eq. 32 can also be integrated into a modified Bessel function of the second kind (15); when the result of integration is substituted, Eq. 32 becomes:

$$\eta = \frac{\mathrm{Exp}\left(\frac{P}{2} - \frac{P}{2}\sqrt{1 + \frac{4RL\lambda d}{PV}}\right)}{\mathrm{Exp}\left(-\frac{RL\lambda_d}{V}\right)}. \quad (33)$$

Equation 33 indicates that the health effect correction factor is a primary function of the Peclet number and the parameter expressed by $\lambda_d RL/V$. This parameter represents the ratio of the radioactive decay constant, λ_d, and the "transport constant", V/RL, and is designated as "transport number" for this study. The preceding results were plotted on a Peclet number vs. transport number plane, as shown in Fig. 1. This figure indicates that the health effects conversion factor, η, increases with the increase in transport number and decreases with the increase in Peclet number. Equation 33 also indicates that the health effects correction factor is always greater than 1.

Proposed Groundwater Transport Model for Health Effects Assessment
Based on the previous discussions, a groundwater system with initial and boundary conditions as shown in Eq. 13 may be simulated by Eq. 31, which is duplicated as:

$$Q(t) = \eta \cdot Q_0(t - RL/V) \cdot \mathrm{Exp}(-RL\lambda_d/V). \quad (34)$$

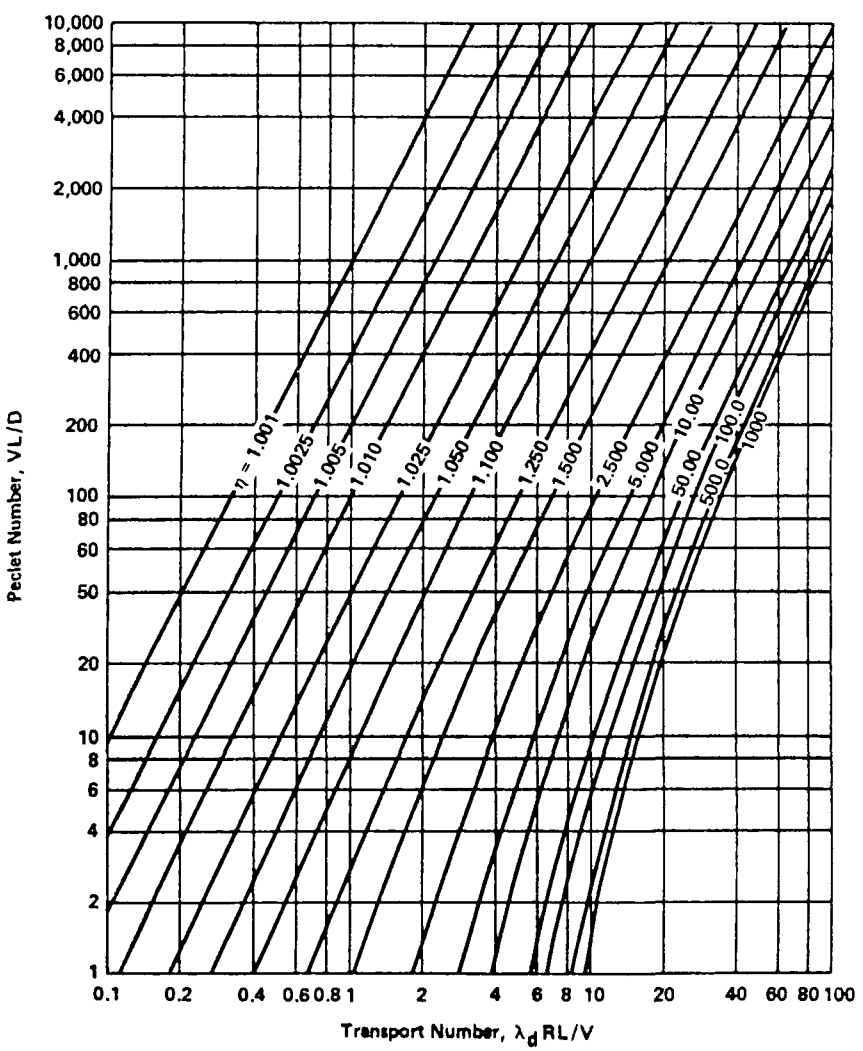

FIGURE 1. Results of health effect collection factor analysis, Equation 33.

Equation 34 represents an algebraic equation, in which constant, η, is determined from Eq. 33; and the terms RL/V and λ_d are also known constants determined from the characteristics of the groundwater system. Therefore, the proposed model described by Eq. 34 is the simplest and most economical to process when it is integrated into a health effects assessment model.

It should be noted that when this model is employed in a health effects assessment model, there may be a slight error in the health effects for each individual generation, but the total health effects for all generations should remain the same as that obtained from the direct integration of Eq. 14. The nature of this error is characterized in the following section.

CHARACTERIZATION OF THE POTENTIAL ERROR OF THE PROPOSED MODEL

Hypothetical Groundwater System

To characterize the nature of error incurred by the proposed model, a hypothetical groundwater transport system was established. In the hypothetical groundwater system, it was assumed that: the velocity of groundwater flow is 100 m/yr; the location of radionuclide discharge is 500 meters downstream from the source point where the radionuclide is released; the coefficient of dispersion is 300 m²/yr; the rate of radionuclide release at the source point, $x = 0$, varies with time and is represented by

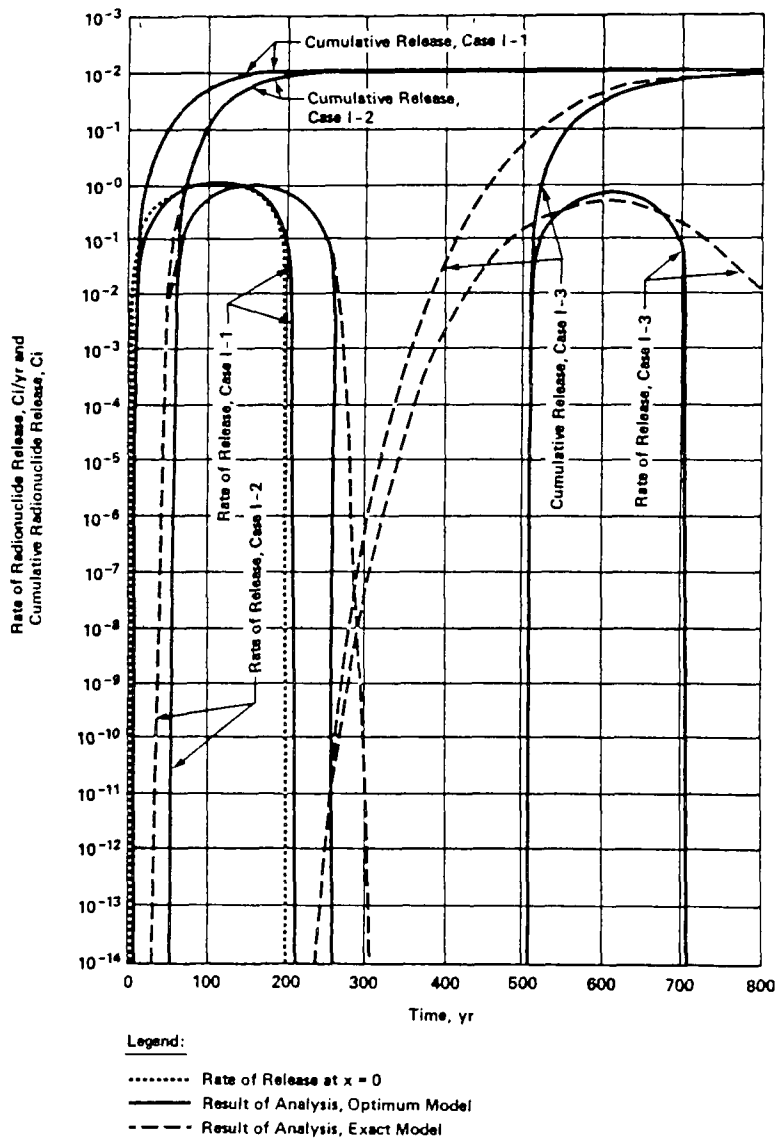

FIGURE 2. Comparison of the results of analyses obtained from the Optimum Model and the Exact Model, Case I, $\lambda_d = 0.000693$, $t_d =$ 200 years.

$$Q_0(t) = \mathrm{Sin}[(\pi/t_d)t] \qquad \text{for } 0 < t < t_d$$

$$Q_0(t) = 0 \qquad \text{for } t > t_d, \tag{35}$$

where t_d is the duration of radionuclide release.

Three cases are assumed for the analysis. Case I represents a fast release, t_d = 200 yr, and long radionuclide half-life, λ_d = 0.000693. Case II represents a fast release, t_d = 200 yr, and short radionuclide half-life, λ_d = 0.0693. Case III represents a slower release, t_d = 400 yr, and long radionuclide half-life, λ_d = 0.000693. Each case is also subdivided into three subcases with the retardation factors equal to 1, 10, and 100, respectively.

Method and Results of Analysis

Two models were developed for the analysis. One of them was developed on a numerical integration of Eq. 16 and was designated as the "Exact Model." The other model was developed based on Eq. 34 and was designated as the "Optimum Model." Each of these models was designed to compute the radionuclide discharge rate at the downstream end of the aquifer and the cumulative radionuclides released at the discharging point over infinite time. The results of com-

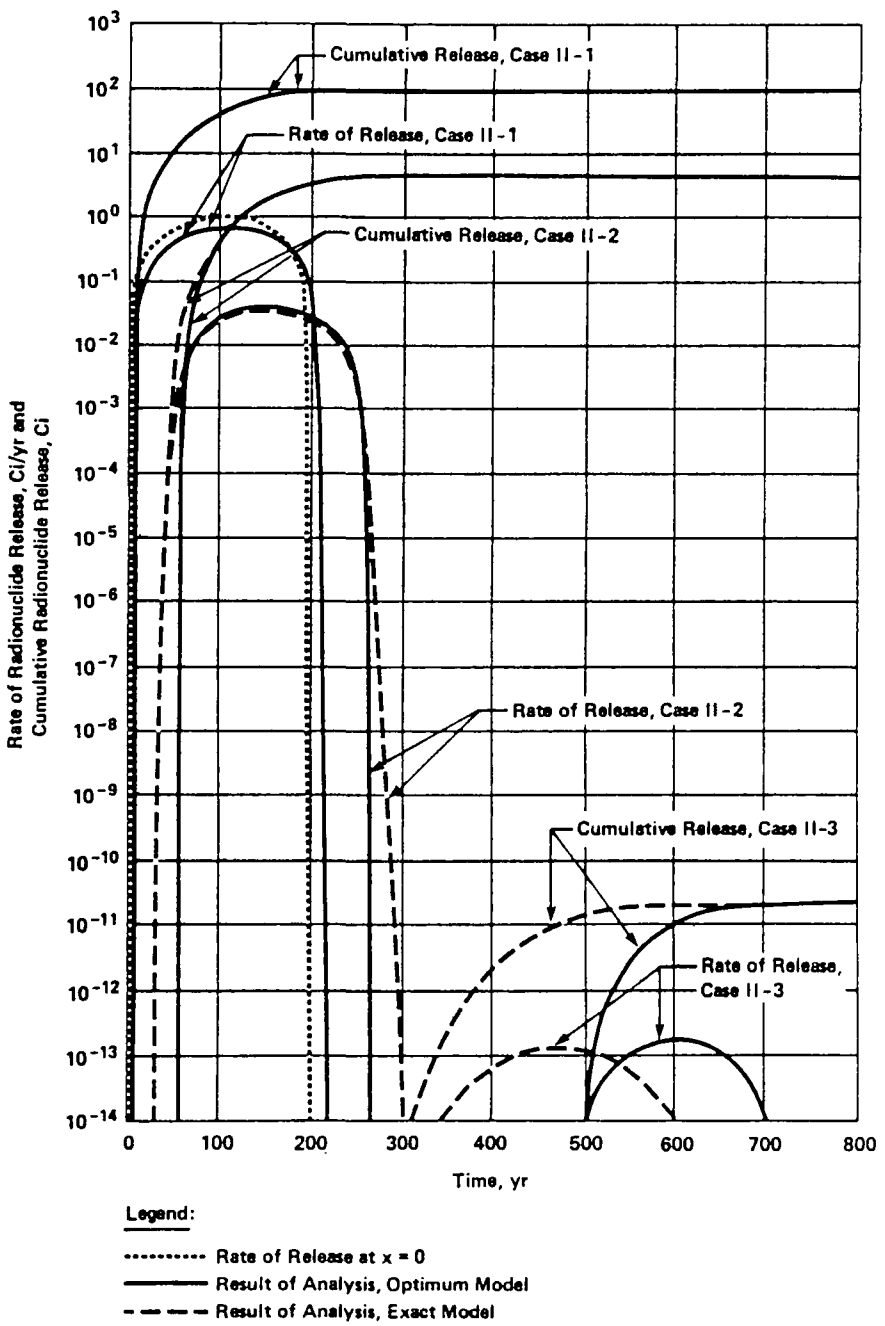

FIGURE 3. *Comparison of the Results of analyses obtained from the Optimum Model and the Exact Model Case II, λ_d = 0.0693, t_d = 200 years.*

puter analyses using these two models are presented in Figs. 2, 3, and 4.

DISCUSSION OF RESULTS

For Cases I and III with long radionuclide half-life, the rate of radionuclide release and the cumulative radionuclide release computed from the exact model and the optimum model agree very well for all three subcases assumed. However, the results of the Case II analysis, where the radionuclide half-life is short, indicate, that the rate of radionuclide discharge obtained from the optimum model deviates more and more from that obtained from the exact model as the retardation factor increases. Nevertheless, the major deviations are merely in the time-lag of radionuclide discharge, whereas the peak discharge showed little deviation (Fig. 3). The results of this latter analysis also indicated that the above deviation will be mitigated for those radionuclides with longer half-life; i.e., with smaller transport number (Fig. 2). Fortunately, a system with larger transport numbers results in a limited quantity of radionuclide discharge at its discharging point, as can be seen by comparing case II-3 with cases II-2 or II-1. This result implies that the discharge of radionuclides under a high transport

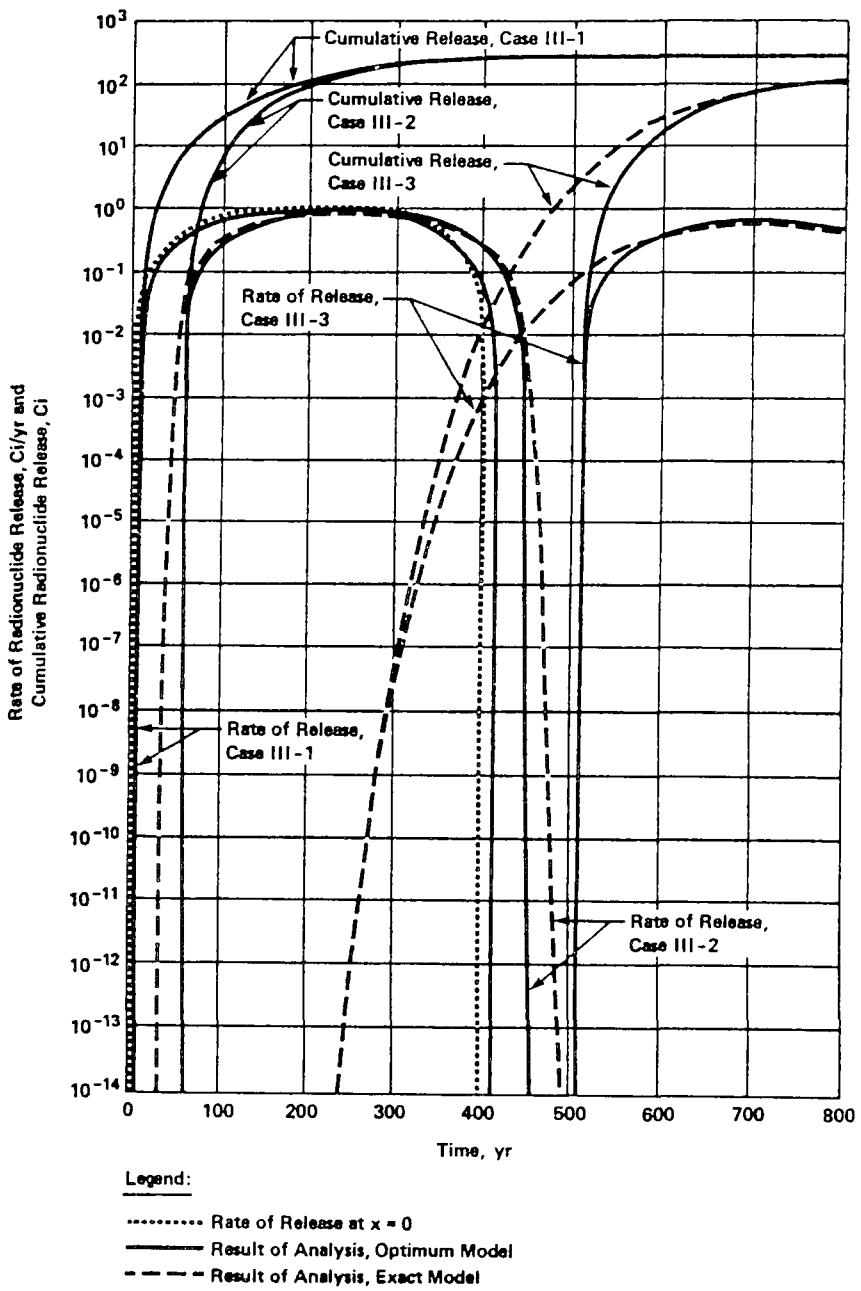

FIGURE 4. Comparison of the results of analyses obtained from the Optimum Model and the Exact Model, Case II, $\lambda_d = 0.000693$, $t_d =$ 400 years.

number, such as Case II-3, would not be a major concern from the point of view of health effects assessment. Besides, this deviation should further decrease with the increase in the duration of radionuclide release at the radionuclide source point, as shown in Figs. 2 and 4.

Nevertheless, the cumulative radioactive discharge, which is the primary parameter in determining total health effects from a radioactive waste disposal site, remains the same for those obtained from the exact model and from the optimum model in all cases. This agreement implies that the proposed optimum groundwater transport model can be integrated into the health effects assessment model without introducing any significant error in the assessment of health effects.

APPLICATION TO A RISK ASSESSMENT MODEL

The optimum groundwater model has been integrated into the PRESTO-EPA model, a model designed to predict the radiological effects due to the disposal of radioactive wastes by a shallow trench operation. The PRESTO-EPA model is a complete risk assessment model. It includes the simulation of radionuclide transport from the wastes through hydrologic and atmospheric pathways to environmental receptors and from the environmental receptors to human organs through food-chain pathways and the evaluation of fatal health effects from the history of organ dosimetries. The model was developed by the Oak Ridge National Laboratory for EPA.

The groundwater transport submodel receives the output from the leaching submodel as its boundary conditions and calculates the rate of radionuclide transport at the wellpoint. These outputs are then fed into the food-chain submodels for human exposure evaluations and, subsequently, for health effects assessments.

The PRESTO-EPA model has been used to assess the health effects for over 400 different combinations of radionuclide source terms with various hydrogeological, demographical, and disposal conditions. The model interacts with other submodels smoothly, implying that the proposed optimum groundwater transport model can be integrated into a risk assessment model.

CONCLUSION

An optimum groundwater transport model for health effects assessment has been derived and characterized. The processing time required to simulate this model is significantly less than that for other comparable models, because it requires only algebraic calculations.

The error from simulating the peak discharge of radionuclides using the "optimum" model, as compared with the results from that of the "exact" model (i.e., the numerical model represented by Eq. 16 and 35), may be noticeable when the groundwater transport number is greater than 5. However, the cumulative radionuclides released under the same conditions is normally only a small fraction of the cumulative radionuclides released at the source point. Therefore, the effect of this error should not be as important as that for those radionuclides with lower transport numbers.

The predicted cumulative radionuclides passing the point of interest obtained from the proposed "optimum" model agree very well with that obtained from the "exact" model in all cases. This implies that the proposed model can be integrated into a health effects assessment model with confidence, if one accepts a one-dimensional flow model.

The application of the proposed optimum groundwater transport model to the PRESTO-EPA model indicated that the model interacts with other submodels very smoothly, implying that the model can be easily integrated into a risk assessment model.

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NOMENCLATURE

C: concentration of radionuclide disolved in fluid.
C_0: concentration of radionuclide at t = 0.
c_p: specific heat of fluid.
D: dispersivity of fluid.
E: rate of radionuclide exposure.
F_h: fatal health effect conversion factor.
g: acceleration of gravitational force.
H: enthalpy of fluid.
H_s: number of fatal health effects.
h: subscript for heat energy.
I_m: initial inventory of radionuclide.
i: subscript for radionuclide.
k: permeability of porous medium.
L: distance from disposal site to wellpoint.
l: subscript for organs.
M: total number of organs considered.
N_d: decay number ($=\lambda_d L/V$).
n: porosity of porous medium.
P: Peclet number ($=VL/D$).
P': population of community.
p: intensity of pressure.
Q: rate of radionuclide transport with disperson effect considered.
Q': tate of radionuclide transport with dispersion effect neglected.
Q_0: rate of radionuclide transport at $x = 0$.
Q_w: rate of groundwater flow.
q': rate of fluid withdrawal from the transport system.
q_L: rate of heat loss of fluid.
R: retardation factor for porous medium.
R_f: risk conversion factor.
r: ratio of radioactivity being uptaken and that reached the wellpoint.
T: temperature of fluid.
T_s: number of years of life expectancy.
t: time.
t_d: duration of radionuclide leaching.
U: internal energy.
u: unit response function with dispersion effects being considered.
u': unit response function with dispersion effects being neglected.
V: interstitial velocity of groundwater.
$\underline{v}$: velocity vector.
x: space coordinate in horizontal direction.
z: height above reference plane.
δ: unit impulse function.
η: long-term health effects conversion factor.
θ: dimensionless time ($=\tau V/L$).
μ: viscosity of fluid.
λ_d: radioactive decay constant.
λ_L: radionuclide leaching factor.
ξ: instantaneous radionuclide transport correction factor.
ϱ: mass density of fluid.
τ: dummy time variable.

APPENDIX C

RADE MODEL: RADIOACTIVE ATMOSPHERIC DISPERSION AND EXPOSURE CODE

INTRODUCTION

The PRESTO-EPA code[1] for calculating the environmental transport of nuclides from low-level waste (LLW) disposal facilities, includes a Gaussian Plume model for atmospheric transport. However, the code only allows for the calculation of one wind sector, speed, stability, distance and population to be specified for each PRESTO-EPA calculation. In order to facilitate the determination of risk elements in the LLW disposal alternative risk-cost data matrix, a small program called RADE (Radioactive Atmospheric Dispersion and Exposure) has been written. The RADE code performs standard Gaussian Plume atmospheric dispersion calculations using subroutines from PRESTO-EPA. It then performs the Air-Pathway Unit Response analysis[2] and prepares the output in the form suitable for input to PRESTO-EPA. This memorandum documents the RADE code and its use to prepare input to PRESTO-EPA for the following settings:

1) Barnwell, South Carolina

2) West Valley, New York

3) Beatty, Nevada

METHOD OF ANALYSIS

One main objective in developing RADE is to utilize applicable subroutines in PRESTO-EPA without altering them. The MAIN program collects and prints out the necessary input data. It then calls the AIRTRM subroutine for the analysis. AIRTRM in turn calls several support subroutines. The MAIN program then performs the unit response analysis and outputs the parameters UEFF, XG, FTWIND, CHIQ and POP parameters for input into PRESTO.

After the sector-independent data are read in, RADE proceeds through a DO loop in which the sector-distance data are input, AIRTRM is called and the key data are accumulated.

The RADE code is organized so that several wind speed distributions, stability classes or populations within a sector can be included in the analysis, for this situation, the data for each distribution element are read in as a new sector. Thus, for example, the sector designation parameter, ISEC, can be larger than 16 even though only 16 compass sectors are used. If the type of stability formation or stability class varies with the sector-distance designations, then the original source height, read in Card 2 must be a negative number. The source height for each sector-distance designation is read from input card set 5B.

The parameters for the community designated as the PRESTO-EPA input community must be read in last. A complete listing of the RADE code is given on pages 7-12.

AIR CONCENTRATION AS A FUNCTION OF DISTANCE

In order to obtain some insight into the impacts from the atmospheric dispersion pathway for large populations at large distances from the facility, RADE2 was used to calculate CHIQ values as a function of distance for the West Valley meteorological conditions. The results are shown in Figure 1 for both four and three atmospheric stability conditions. At large distances both curves in the figure coincide because of the lid height condition of 300 m used in the calculations.

RESULTS

The air pathway unit response calculations for the three reference sites are summarized below. The complete RADE output, including the input data for these sites is given in Appendix B.

Site	FTWIND	POP	CHIQ (sec/m^3)
Barnwell, SC	0.080	111	1.20E-04
West Valley, NY	0.288	4,285	3.25E-07
Beatty, NV	6.025	50	5.23E-08

When performing the PRESTO calculations the CHIQ is read in so that subroutine AIRTRM is not called.

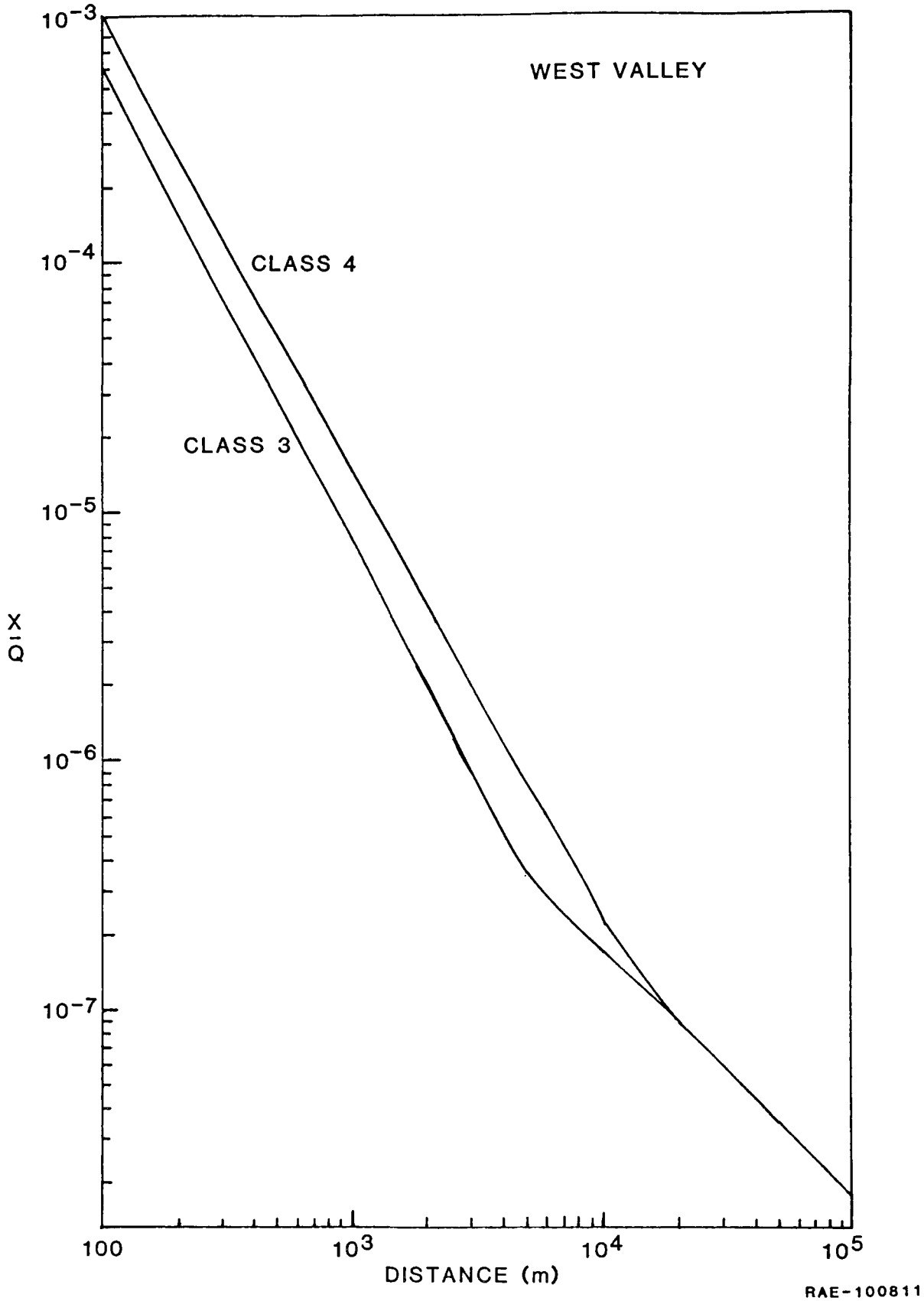
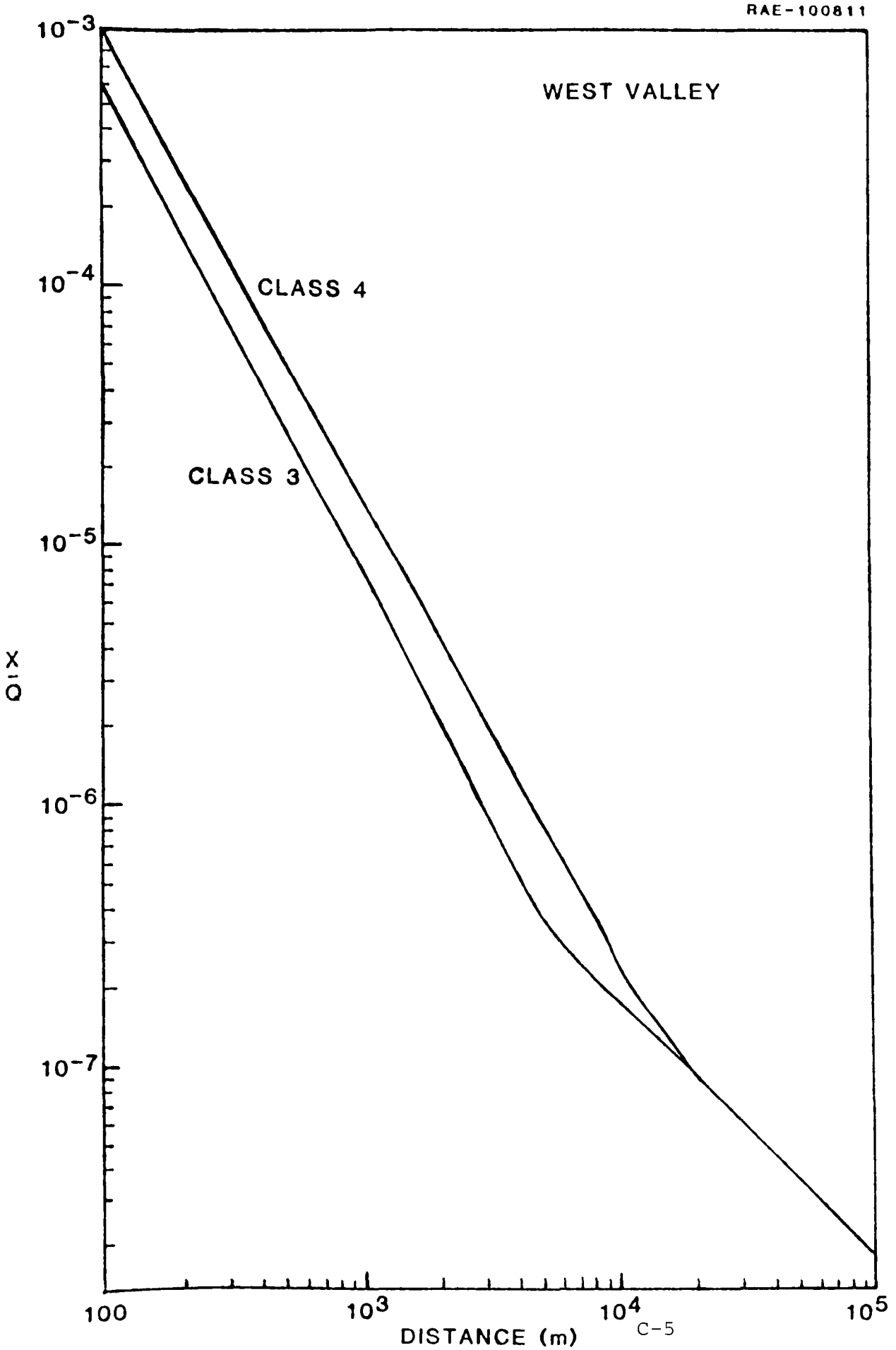


FIGURE 1. CHIQ AS A FUNCTION OF DISTANCE

FIGURE 1. CHIQ AS A FUNCTION OF DISTANCE



INPUT SPECIFICATIONS

The input format for RADE is given below:

Card No.	Format	Parameters
1	20A4	Title Card
2	8F10.0	H, VG, VD, HLID, ROUGH
3	8F10.0	CHIQ, RE1, RE2, RE3, RR, FTMECH
4	815	IT, IS, ISEC
5A	4A4	XNAME(J), J = 1,4
5B	8F10.0	H, XG, U, FTWIND, POP

Cards 5A and 5B are repeated for every sector-distance combination being analyzed.

A description of the input variables is given below.

Variable	Description
H	Source height (m)
VG	Velocity, gravitational fall (m/s)
HLID	Lid height (m)
ROUGH	Roughness parameter
CHIQ	1.0
RE1	Beginning coefficient in resuspension equation (not used)
RE2	Decay factor in resuspension equation (not used)
RE3	Final coefficient in resuspension equation (not used)
RR	(not used)
FTMECH	Fraction of year mechanical disturbance occurs (not used)
IT	Type of stability formulation 1 for PG 2 for Briggs-Smith
IS	Stability class
ISEC	Number of sector-distance combinations
XNAME	Community name
XG	Distance to the population in the sector of interest (m)
U	Windspeed in the sector of interest m/s)
FTWIND	Fraction of time the wind blows toward the sector of interest
POP	Population at the sector-distance of interest

RADE CODES

```
C     R A D E
C     RADIOACTIVE ATMOSPHERIC DISPERSION AND EXPOSURE.
C     INCORPORATES AIRTRM AND RELATED SUBROUTINES FROM PRESTO-EPA.
      PROGRAM RADE
      COMMON/AIR/H,VG,U,IT,IS,VD,XG,HLID,ROUGH,FTWIND,CHIQ,RE1,RE2,RE3
      DIMENSION A(20),XNAME(4)
      CALL ASSIGN(6,'LA:')
      WRITE(6,1000)
      READ(5,6000) (A(I),I=1,20)
      WRITE(6,6010) (A(I),I=1,20)
      T1=0.
      T2=0.
      READ(5,3005) HH,VG,VD,HLID,ROUGH
      READ(5,3005) CHIQ,RE1,RE2,RE3,RR,FTMECH
      READ(5,3010) IT,IS,ISEC
      WRITE(6,6400)
      WRITE(6,6416) ISEC
      IF (HH.GE.0.) WRITE(6,6405) HH
      WRITE(6,6410) VG
      WRITE(6,6420) VD
      WRITE(6,6435) HLID
      WRITE(6,6440) ROUGH
      WRITE(6,6445) IT,IS
      WRITE(6,6480) RE1,RE2,RE3
      WRITE(6,6500)
      DO 1 I=1,ISEC
      READ(5,6000) (XNAME(J),J=1,4)
      READ(5,3005) XG,U,FTWIND,POP
      IF(HH.LT.0.) READ(5,3010) IT,IS
      IF(HH.LT.0.) READ(5,3005) H,VG,VD,HLID,ROUGH
      IF(HH.GE.0.) H=HH
C
      CALL AIRTRM(EXPOS,DEPO)
C
      T0=EXPOS*POP*FTWIND
      T1=T1+T0
      T2=T2+POP
      WRITE(6,6510) (XNAME(J),J=1,4),XG,U,FTWIND,POP,EXPOS
    1 CONTINUE
      FTWIND=T1/(POP*EXPOS)
      PTOT=T2
      WRITE(6,6520)
      WRITE(6,6530) (XNAME(J),J=1,4),XG,U,FTWIND,POP,EXPOS
      WRITE(6,6540) PTOT
 1000 FORMAT(//,18X,55('*'),/,18X,'*',53X,'*',/,18X,'*',23X,'R A D E .
```

```
     &23X,'*',/,18X,'*',53X,'*',/,18X,'*',4X,'RADIOACTIVE ATMOSPHERIC DI
     &SPERSION & EXPOSURE',4X,'*',/,18X,'*',53X,'*',/,18X,55('*'))
 3005 FORMAT(8F12.6)
 3010 FORMAT(8I6)
 6000 FORMAT(20A4)
 6010 FORMAT(///,1X,20A4,///)
 6400 FORMAT(//,20X,'GENERAL INPUT DATA')
 6405 FORMAT(//,11X,'SOURCE HEIGHT IS ',F6.1,' METERS')
 6410 FORMAT(11X,'VELOCITY OF GRAVITATIONAL FALL IS',F6.2,
     & ' METERS/SECOND')
 6416 FORMAT(11X,'NO. OF SECTOR DISTANCE COMBINATIONS IS',I4,/,11X,
     & 'ENTER PRESTO REFERENCE POPULATION DATA LAST')
 6420 FORMAT(11X,'DEPOSITION VELOCITY IS ',F6.2,' METERS/SECOND')
 6435 FORMAT(11X,'LID HEIGHT IS ',F8.2,' METERS')
 6440 FORMAT(11X,'HOSKER ROUGHNESS FACTOR IS ',F6.2)
 6445 FORMAT(11X,'TYPE OF STABILITY FORMULATION IS ',I2,/,
     & 11X,'STABILITY CLASS IS ',I2)
 6480 FORMAT(11X,'RESUSPENSION FACTOR PARAMETERS ',3(4X,1PE12.4))
 6500 FORMAT(///,3X,'COMMUNITY NAME',7X,'XG',8X,'WIND VELOCITY',4X,
     & 'FTWIND',6X,'POP',11X,'CHIQ',/,24X,'(M)',10X,'(M/SEC)',26X,
     & 'CI/M**3 PER CI/SEC',/,90('-'),/)
 6510 FORMAT(2X,4A4,4X,F7.1,8X,F6.2,8X,F7.5,1X,F8.0,7X,1PE10.3)
 6520 FORMAT(/,90('-'),/,' ---- PRESTO INPUT ----',/)
 6530 FORMAT(2X,4A4,4X,F7.1,8X,F6.2,8X,F7.5,1X,F8.0,7X,1PE10.3,//,
     & 90('-'))
 6540 FORMAT(///,'   TOTAL AIR PATHWAY POPULATION IS ',F8.0)
      END
C
C
C
      BLOCK DATA
      COMMON/C/BY(6),BZ1(6),BZ2(6),BZ3(6),A1(6),A2(6),B1(6),B2(6),
     &B3(6),PY(6,5),PZ(6,5),QY(6,5),QZ(6,5),XM(50)
      DATA BY/.22,.16,.11,.08,.06,.04/,
     &    BZ1/.2,.12,.08,.06,.03,.016/,
     &    BZ2/0.,0.,.0002,.0015,.0003,.0003/,
     &    BZ3/1.,1.,-.5,-.5,-1.,-1./
      DATA A1/-.0234,-.0147,-.0117,-.0059,-.0059,-.0029/,
     &     A2/.35,.248,.175,.108,.088,.054/,
     &     B1/.88,-.985,-1.186,-1.35,-2.88,-3.8/,
     &     B2/-.152,.82,.85,.793,1.255,1.419/,
     &     B3/.1475,.0168,.0045,.0022,-.042,-.055/
      DATA PY/.469,.306,.23,.219,.237,.237,0.,.4,.36,.32,0.,.31,
     &        0.,1.7,1.44,.91,1.02,0.,.66,.66,.63,.53,.41,7.56,
     &        .34,.37,.40,.43,.46,7.56/
      DATA QY/.903,.885,.855,.764,.691,.594,0.,.91,.86,.78,0.,.71,
     &        0.,.717,.71,.729,.648,0.,.83,.83,.80,.80,.87,.52,
     &        1.00,.94,.88,.82,.76,.52/
      DATA PZ/.017,.072,.076,.14,.217,.262,0.,.411,.326,.223,0.,
     &        .062,0.,.079,.121,.91,1.93,0.,.14,.14,.21,.26,.13,.56,
     &        .037,.076,.16,.32,.66,1.37/
```

```
      DATA QZ/1.38,1.021,.879,.727,.61,.5,0.,.907,.859,.776,0.,.709,
     &        0.,1.2,1.046,.702,.465,0.,1.09,1.09,.98,.89,.83,.55,1.28,
     &        1.12,.96,.88,.63,.47/
      DATA XM/1.00,2.00,3.00,4.00,5.00,10.0,15.0,20.0,25.0,30.0,
     &        35.0,40.0,45.0,50.0,150.,200.,300.,400.,500.,600.,
     &        700.,800.,900.,1.E3,1.1E3,1.2E3,1.3E3,1.4E3,1.6E3,1.8E3,
     &        2.0E3,2.5E3,3.0E3,3.5E3,4.0E3,4.5E3,5.0E3,6.E3,7.E3,8.E3,
     &        1.0E4,1.5E4,2.E4,3.E4,4.E4,5.E4,6.E4,7.E4,8.E4,1.E5/
      END
C
C
C
      SUBROUTINE AIRTRM(EXPOS,DEPO)
      COMMON/AIR/H,VG,U,IT,IS,VD,XG,HLID,ROUGH,FTWIND,CHIQ,RE1,RE2,RE3
      IF(IS.LT.1.OR.IS.GT.6) WRITE(6,38) IT,IS
      IF(IT.EQ.5.AND.(IS.EQ.1.OR.IS.EQ.5)) WRITE(6,38) IT,IS
      IF(IT.EQ.6.AND.(IS.EQ.1.OR.IS.EQ.6)) WRITE(6,38) IT,IS
   38 FORMAT(2X,'INVALID COMBINATION OF IT=',I2,' AND IS=',I2)
      PI=3.141593
      IF(HLID.EQ.0.) HLID=12000.
      LID=HLID
      HH=H-VG*XG/U
      IF(HH.LT.0.) HH=0.
C
      CALL SIGMAZ(XG,IT,IS,ROUGH,SIGZ,IKPM,HLID,VG,U,HH)
C
      COR=1.
      IF(VG.EQ.0.0.AND.VD.EQ.0.) GO TO 45
C
      CALL DPLT(PI,HH,VD,U,IT,IS,XG,ROUGH,HLID,COR,VG,H)
C
   45 EXPO=EXP(-0.5*HH*HH/(SIGZ*SIGZ))
      EXPN=COR*2.032*EXPO/(SIGZ*XG)
      EXPOS=EXPN/U
      DEPO=EXPOS*VD
   80 CONTINUE
      RETURN
      END
C
C
C
      SUBROUTINE DPLT(PI,HH,VD,U,IT,IS,XG,ROUGH,HLID,COR,VG,H)
      DIMENSION XX(50),FX(50),AX(50)
      COMMON/C/BY(6),BZ1(6),BZ2(6),BZ3(6),A1(6),A2(6),B1(6),B2(6),
     &B3(6),PY(6,5),PZ(6,5),QY(6,5),QZ(6,5),XM(50)
      IF(IS.GT.6) IS=6
      IKPM=1
      IDUM=1
C
      CALL SIGMAZ(XG,IT,IS,ROUGH,SIGZ,IKPM,HLID,VG,U,HH)
C
```

```
      IKPM=IKPM+1
      SQRTPI=0.79785
      FX(IKP)=EXP(-0.5*HH*HH/(SIGZ*SIGZ))/SIGZ
      XX(IKP)=XG
      DO 20 I=1,IKPM
      XX(I)=XM(I)
C
      CALL SIGMAZ(XX(I),IT,IS,ROUGH,SIGZ,IDUM,HLID,VG,U,HH)
C
      FX(I)=EXP(-0.5*HH*HH/(SIGZ*SIGZ))/SIGZ
   20 CONTINUE
      L=1
C
      CALL SIMPUN(XX,FX,IKP,L,AX)
C
      COR=EXP(-SQRTPI*VD*AX(IKP)/U)
      RETURN
      END
C
C
C
      SUBROUTINE SIGMAZ(XG,IT,IS,ROUGH,SIGZ,IKPM,HLID,VG,U,HH)
      COMMON/C/BY(6),BZ1(6),BZ2(6),BZ3(6),A1(6),A2(6),B1(6),B2(6),
     &B3(6),PY(6,5),PZ(6,5),QY(6,5),QZ(6,5),XM(50)
      DIMENSION AONE(6),BONE(6),ATWO(6),BTWO(6),CONE(6),DONE(6),
     &CTWO(6),DTWO(6),RGH(6)
      DATA AONE/0.112,0.130,0.112,0.098,0.0609,0.0638/
      DATA BONE/1.06,0.950,0.920,0.889,0.895,0.783/
      DATA ATWO/5.38E-4,6.52E-4,9.05E-4,1.35E-3,1.96E-3,1.36E-3/
      DATA BTWO/0.815,0.750,0.718,0.688,0.684,0.672/
      DATA CONE/1.56,2.02,2.72,5.16,7.37,11.7/
      DATA DONE/0.0480,0.0269,0.,-0.060,-0.0957,-0.128/
      DATA CTWO/6.25E-4,7.76E-4,0.,186.,4290.,45900./
      DATA DTWO/0.45,0.37,0.,-0.225,-0.60,-0.78/
      DATA RGH/0.01,0.04,0.10,0.40,1.0,4.0/
      P=0.
      DO 8 I=1,50
      IKP=I
      IF(XM(I).GE.XG) GO TO 9
    8 CONTINUE
    9 IF(IKP.LT.2) IKP=2
      IKPM=IKP-1
      IPOINT=IT
      IF(IT.GT.3) IPOINT=4
      GO TO (10,20,30,40) IPOINT
   10 ALX=ALOG(XG)
      SIGZ=EXP(B1(IS)+(B2(IS)+B3(IS)*ALX)*ALX)/2.15
      GO TO 99
   20 ABS=ROUGH
      C1=YLAG(ABS,RGH,CONE,0,3,6,0)
      D1=YLAG(ABS,RGH,DONE,0,3,6,0)
```

```
      C2=YLAG(ABS,RGH,CTWO,0,3,6,0)
      D2=YLAG(ABS,RGH,DTWO,0,2,6,0)
      G=AONE(IS)*XG**BONE(IS)/(1.+ATWO(IS)*XG**BTWO(IS))
      F=C1*XG**D1
      F=ALOG(F)
      IF(ROUGH.LE.0.10) GO TO 26
      F=F+ALOG(1.+1./(C2*XG**D2))
      GO TO 27
   26 F=F-ALOG(1.+C2*XG**D2)
   27 SIGZ=G*F
      IF(SIGZ.LT.1.) SIGZ=1.
      GO TO 99
   30 SIGZ=BZ1(IS)*XG*(1.+BZ2(IS)*XG)**BZ3(IS)
      GO TO 99
   40 J=IT-3
      SIGZ=PZ(IS,J)*XG**QZ(IS,J)
   99 SIGCAL=2.*(HLID-0.5*HH)/2.15
      IF(SIGZ.LT.SIGCAL) GO TO 1000
      SIGZ=SIGCAL
 1000 RETURN
      END
C
C
C
      SUBROUTINE SIMPUN(XX,FX,NX,I,AX)
      DIMENSION XX(2),FX(2),AX(2)
      IF(I.LT.0) GO TO 30
      AX(1)=0.
      DO 10 IX=2,NX,2
      D1=XX(IX)-XX(IX-1)
      AX(IX)=AX(IX-1)+0.5*D1*(FX(IX)+FX(IX-1))
      IF(NX.EQ.IX) GO TO 20
      D2=XX(IX+1)-XX(IX-1)
      D3=D2/D1
      A2=D3/6.*D2*D2/(XX(IX+1)-XX(IX))
      A3=0.5*D2-A2/D3
   10 AX(IX+1)=AX(IX-1)+(D2-A2-A3)*FX(IX-1)+A2*FX(IX)+A3*FX(IX+1)
   20 RETURN
   30 AX(NX)=0.
      DO 40 IX=2,NX,2
      IC=NX+1-IX
      D1=XX(IC+1)-XX(IC)
      AX(IC)=AX(IC+1)+0.5*D1*(FX(IC+1)+FX(IC))
      IF(NX.EQ.IX) GO TO 20
      D2=XX(IC+1)-XX(IC-1)
      D3=D2/(XX(IC)-XX(IC-1))
      A2=D3*D2*D2/(6.*D1)
      A3=0.5*D2-A2/D3
   40 AX(IC-1)=AX(IC+1)+(D2-A2-A3)*FX(IC-1)+A2*FX(IC)+A3*FX(IC+1)
      RETURN
      END
```

```
C
C
C
      FUNCTION YLAG(XI,X,Y,IND1,N1,IMAX,IEX)
      DIMENSION X(1),Y(1)
      IND=IND1
      N=N1
      IEX=0
      IF(N.LE.IMAX) GO TO 10
      N=IMAX
      IEX=N
   10 IF(IND.GT.0) GO TO 40
      DO 20 J=1,IMAX
      IF(XI-X(J)) 30,130,20
   20 CONTINUE
      IEX=1
      GO TO 70
   30 IND=J
   40 IF(IND.GT.1) GO TO 50
      IEX=-1
   50 INL=IND-(N+1)/2
      IF(INL.GT.0) GO TO 60
      INL=1
   60 INU=INL+N-1
      IF(INU.LE.IMAX) GO TO 80
   70 INL=IMAX-N+1
      INU=IMAX
   80 S=0.
      P=1.
      DO 110 J=INL,INU
      P=P*(XI-X(J))
      D=1.
      DO 100 I=INL,INU
      IF(I.NE.J) GO TO 90
      XD=XI
      GO TO 100
   90 XD=X(J)
  100 D=D*(XD-X(I))
  110 S=S+Y(J)/D
      YLAG=S*P
  120 RETURN
  130 YLAG=Y(J)
      GO TO 120
      END
```

```
*****************************************************
*                                                   *
*                      R A D E                      *
*                                                   *
*    RADIOACTIVE ATMOSPHERIC DISPERSION & EXPOSURE  *
*                                                   *
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```

B E A T T Y

GENERAL INPUT DATA
NO. OF SECTOR DISTANCE COMBINATIONS IS 6
ENTER PRESTO REFERENCE POPULATION DATA LAST

SOURCE HEIGHT IS 1.0 METERS
VELOCITY OF GRAVITATIONAL FALL IS 0.03 METERS/SECOND
DEPOSITION VELOCITY IS 0.03 METERS/SECOND
LID HEIGHT IS 300.00 METERS
HOSKER ROUGHNESS FACTOR IS 0.01
TYPE OF STABILITY FORMULATION IS 1
STABILITY CLASS IS 4
RESUSPENSION FACTOR PARAMETERS 1.0000E-04 -1.5000E-01 1.0000E-09

COMMUNITY NAME	XG (M)	WIND VELOCITY (M/SEC)	FTWIND	POP	CHIQ CI/M**3 PER CI/SEC
BEATTY	17000.0	4.80	0.09000	900.	8.923E-08
DEATH VALLEY	40000.0	4.80	0.17000	25.	3.792E-08
DEATH VALLEY JN.	58000.0	4.80	0.11000	30.	2.615E-08
LATHROY WELLS	29000.0	4.80	0.25000	250.	5.231E-08
PAHRUMP	87000.0	4.80	0.25000	1000.	1.744E-08
FARM VILLAGE	29000.0	4.80	0.25000	50.	5.231E-08
---- PRESTO INPUT ----					
FARM VILLAGE	29000.0	4.80	6.02489	50.	5.231E-08

TOTAL AIR PATHWAY POPULATION IS 2255.

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*****************************************************
*                                                   *
*                      R A D E                      *
*                                                   *
*    RADIOACTIVE ATMOSPHERIC DISPERSION & EXPOSURE  *
*                                                   *
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```

B A R N W E L L

GENERAL INPUT DATA
NO. OF SECTOR DISTANCE COMBINATIONS IS 7
ENTER PRESTO REFERENCE POPULATION DATA LAST

SOURCE HEIGHT IS 1.0 METERS
VELOCITY OF GRAVITATIONAL FALL IS 0.10 METERS/SECOND
DEPOSITION VELOCITY IS 0.10 METERS/SECOND
LID HEIGHT IS 300.00 METERS
HOSKER ROUGHNESS FACTOR IS 0.01
TYPE OF STABILITY FORMULATION IS 1
STABILITY CLASS IS 4
RESUSPENSION FACTOR PARAMETERS 1.0000E-06 -1.5000E-01 1.0000E-11

COMMUNITY NAME	XG (M)	WIND VELOCITY (M/SEC)	FTWIND	POP	CHIQ CI/M**3 PER CI/SEC
BARNWELL CITY	8050.0	2.01	0.09000	5572.	6.968E-07
BLACKVILLE	21900.0	2.01	0.05000	2840.	1.654E-07
WILLISTON	18200.0	2.01	0.02000	3173.	1.990E-07
ELKO	16400.0	2.01	0.03000	329.	2.209E-07
HILDA	20300.0	2.01	0.06000	355.	1.785E-07
KLINE	16700.0	2.01	0.05000	315.	2.169E-07
SNELLING	480.0	2.01	0.05000	111.	1.201E-04
---- PRESTO INPUT ----					
SNELLING	480.0	2.01	0.07962	111.	1.201E-04

TOTAL AIR PATHWAY POPULATION IS 12695.

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*****************************************************
*                                                   *
*                      R A D E                      *
*                                                   *
*    RADIOACTIVE ATMOSPHERIC DISPERSION & EXPOSURE  *
*                                                   *
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```

W E S T V A L L E Y

GENERAL INPUT DATA
NO. OF SECTOR DISTANCE COMBINATIONS IS 33
ENTER PRESTO REFERENCE POPULATION DATA LAST

SOURCE HEIGHT IS 1.0 METERS
VELOCITY OF GRAVITATIONAL FALL IS 0.01 METERS/SECOND
DEPOSITION VELOCITY IS 0.01 METERS/SECOND
LID HEIGHT IS 300.00 METERS
HOSKER ROUGHNESS FACTOR IS 0.01
TYPE OF STABILITY FORMULATION IS 1
STABILITY CLASS IS 4
RESUSPENSION FACTOR PARAMETERS 1.0000E-06 -1.5000E-01 1.0000E-10

COMMUNITY NAME	XG (M)	WIND VELOCITY (M/SEC)	FTWIND	POP	CHIQ CI/M**3 PER CI/SEC
GLENWOOD	19300.0	5.00	0.07170	100.	7.545E-08
CHAFFEE	19300.0	5.00	0.08670	100.	7.545E-08
PROTECTION	21700.0	5.00	0.08670	100.	6.711E-08
SANDUSKY	22500.0	5.00	0.06380	500.	6.472E-08
ELTON	17700.0	5.00	0.10650	50.	8.227E-08
FARMERSVILLE	23300.0	5.00	0.09370	978.	6.250E-08
HUMPHREY C.	25700.0	5.00	0.04860	20.	5.666E-08
HUMPHREY	28200.0	5.00	0.04860	529.	5.164E-08

SUGARTOWN	23300.0	5.00	0.04270	150.	6.250E-08
GREAT VALLEY	25700.0	5.00	0.04270	2014.	5.666E-08
MARSHFIELD	19300.0	5.00	0.08890	100.	7.545E-08
LANGFORD	22500.0	5.00	0.08890	100.	6.472E-08
NEW OREGON	19300.0	5.00	0.08890	100.	7.545E-08
BOSTON	20900.0	5.00	0.06490	7687.	6.968E-08
COLDEN	21700.0	5.00	0.07170	3128.	6.711E-08
GOWANDA	23300.0	5.00	0.02500	2713.	6.250E-08
CATTARANGUS	21700.0	5.00	0.01320	1200.	6.711E-08
ELLICOTTVILLE	19300.0	5.00	0.04270	1677.	7.545E-08
ARCADE	20900.0	5.00	0.08670	3714.	6.968E-08
FRANKLINVILLE	20100.0	5.00	0.06530	3102.	7.245E-08
RICEVILLE	2400.0	5.00	0.09370	30.	2.565E-06
WEST VALLEY	5630.0	5.00	0.04860	600.	5.392E-07
ASHFORD VALLEY	5630.0	5.00	0.01320	100.	5.392E-07
EAST CONCORD	11300.0	5.00	0.04270	8171.	1.504E-07
SARDINIA	15300.0	5.00	0.08670	2792.	9.518E-08
YORKSHIRE	17200.0	5.00	0.08670	3620.	8.467E-08
LIME LAKE	14500.0	5.00	0.10650	1191.	1.004E-07
MACHIAS	12900.0	5.00	0.06380	2058.	1.180E-07
ASHFORD	14500.0	5.00	0.04270	1922.	1.004E-07
EAST OTTO	11300.0	5.00	0.01320	942.	1.504E-07
OTTO	18500.0	5.00	0.01320	828.	7.872E-08
COLLINS CENTER	17700.0	5.00	0.04990	5037.	8.227E-08
SPRINGVILLE	7420.0	5.00	0.07170	4285.	3.252E-07

---- PRESTO INPUT ----

SPRINGVILLE	7420.0	5.00	0.28829	4285.	3.252E-07

TOTAL AIR PATHWAY POPULATION IS 59638.

REFERENCES

1. C. Little, et al., "PRESTO-EPA: A Low-Level Radioactive Waste Environmental Transport and Risk Assessment Code--Methodology and User's Manual," EPA Report (in print).

2. C.Y. Hung, et al., "Use of PRESTO-EPA Model in Assessing Health Effects From Land Disposal of LLW to Support EPA's Environmental Standards," Presented at the 5th Annual DOE LLW participants Information Meeting, Denver, Colorado, August 30 - September 1, 1983.

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