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USER'S GUIDE FOR PRESTO-EPA-CPG/ POP OPERATION SYSTEM

Version 4.2

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**Developed by:
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**U.S. Environmental Protection Agency
Office of Radiation and Indoor Air
Washington, DC 20460**

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DISCLAIMER

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PREFACE

The mainframe versions of the PRESTO-EPA-POP and PRESTO-EPA-CPG models were developed for generating basic data to support EPA's rulemaking on the generally applicable environmental standards for the management and disposal of low-level radioactive waste (LLW).

Since the mainframe versions of the PRESTO-EPA-CPG and PRESTO-EPA-POP models were published in December 1987, the Office of Radiation and Indoor Air has received numerous requests from potential users urging the Office to convert the models to a form usable on a personal computer. This effort led to the development of a PC version of the PRESTO-EPA-CPG and PRESTO-EPA-POP models. The models were later expanded to include evaluation of contaminated soil sites as well as LLW sites, were designated as Version 4.0, and were intended strictly for within-Agency use.

The current model, Version 4.2, combines the previous PRESTO-EPA-CPG and PRESTO-EPA-POP operation systems into one operation system. It updates the user interface program to facilitate the editing of the input file and viewing of the resultant output file. The user interface is a menu-driven program that is supported by the Windows-95 operation system and performs several data-checking tests. In addition, the model is more flexible for evaluating sites with various forms of contamination.

The PRESTO-EPA-CPG/POP model predicts the maximum doses to a member of a critical population group and the cumulative numbers of health effects (cancers and genetic effects) among a general population residing on and downstream of a contaminated soil site, a LLW disposal site, or an agricultural application site. The individuals and populations may be exposed to radioactivity through atmospheric, groundwater, and surface water transport pathways; through drinking water, fish, vegetable, milk, and meat food-chain pathways; and through air immersion and direct external exposure pathways. Exposure through inhalation of dust and radon gas and inadvertent ingestion of soil is also included.

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1. INTRODUCTION

1.1 BACKGROUND

Under the Atomic Energy Act, as amended, the U.S. Environmental Protection Agency (EPA) has the authority to develop generally applicable standards for the disposal of low-level radioactive waste (LLW). Technical support for the standards includes an estimation of the health impacts from the disposal of LLW in a wide variety of facility types located in diverse hydrogeological settings.

As an aid in developing the standards, a family of PRESTO (Prediction of Radiation Effects from Shallow Trench Operation) codes, entitled PRESTO-EPA-POP, PRESTO-EPA-DEEP, PRESTO-EPA-CPG, PRESTO-EPA-BRC, and PATHRAE-EPA, has been developed under EPA direction (EPA87a through EPA87g.) The PRESTO-EPA-POP code was the first code developed and served as the basis for the other codes in the family. EPA uses the PRESTO-EPA code family to compare the potential health impacts (cumulative population health effects and maximum annual dose to a critical population group) to the general public and critical population group for a broad number of LLW disposal alternatives. Table 1-1 provides a brief description of the function of each member of the code family. The application of these codes has been described in detail elsewhere (Hu83a, Gal84, Ro84, Mey81, Mey84).

The PRESTO-EPA-CPG and PRESTO-EPA-POP codes were designed to estimate the maximum annual committed effective dose (CED) and the annual mortality and cancer incidence risks to a critical population group (PRESTO-EPA-CPG) and the number of fatal cancer deaths and serious genetic effects to the general population (PRESTO-EPA-POP) resulting from the disposal of LLW. The codes were later modified to include the modeling of contaminated surface soil sites and were designated Version 4.0 of the PRESTO-EPA-CPG/POP model.

The model calculates the health impacts exposure to the individuals and populations through atmospheric, groundwater, and surface water transport pathways; drinking water, fish, vegetable, milk, and meat food-chain pathways; and air immersion and direct external exposure pathways. Exposure through inhalation of dust and radon gas and inadvertent ingestion of soil are also included.

The model described in this report is an extension of the original PRESTO-EPA-CPG and PRESTO-EPA-POP models that makes them more flexible for evaluating sites with various forms of contamination. The most significant modification to the operation system is the combination of the previous PRESTO-EPA-CPG and PRESTO-EPA-POP operation systems into one operation system.

Table 1-1. Function of PRESTO-EPA Family of Codes

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 below regulatory concern (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.
PRESTO-EPA-CPG/POP	Estimates maximum doses, risks, and cumulative population health effects from LLW disposal sites, contaminated soil sites, land application sites, or emergency response sites.

The user interface is a menu-driven program that is supported by the Windows-95 operation system. The program is user-friendly and is designed to reduce human errors and to facilitate the preparation of the input file and the display of the output file.

1.2 CHANGES IN VERSION 2.1

Version 2.1 is the first version ever published since the model was converted to operate on a personal computer. An interface program, written in a DOS operation system, was also incorporated into the model to form an operation system. Four improvements were made to this version: (1) the addition of the daughter nuclide in-growth effects (DNIE) into the risk assessment, (2) the update of the dose and risk conversion factors to the 1994 level, (3) the addition of the annual mortality and risk incidence calculation, and (4) the adoption of the International System (SI) units.

The dose coefficients are extracted from the RADRISK data file (Du80) and the weighting factors are consistent with the definitions used in International Commission on Radiological Protection Publications 26 (ICRP77) and 30 (ICRP79). The effective dose equivalent is the weighted sum of the 50-year committed dose equivalent to the organs or tissues.

The cancer risk coefficients are calculated from radiation-risk models that are based on 1980 U.S. vital statistics. The genetic-risk coefficients for serious disorders to all subsequent generations are calculated from the product of the average absorbed dose to the ovaries and testes up to age 30 per unit intake before that age. Risk coefficients of 2.60×10^{-2} and $6.9 \times 10^{-2} \text{ Gy}^{-1}$ for low-LET and high-LET radiation, respectively, are used for the calculation of risk conversion factors (EPA89).

The Version 2.1 Operation System modifies the PRESTO-EPA-CPG model by integrating the daughter nuclide in-growth effects into the Version 2.0 model. The DNIE are calculated based on a crude assumption that the sorption characteristics of the parent and daughter nuclides are identical throughout the processes of leaching and groundwater transport. The DNIE are adjusted annually by using the correction factors derived from Bateman Equations (Ev55).

The adjustment for DNIE is performed only for those parent nuclides designated and built into the model. To simplify the modeling, the adjustment is carried up to four-member decay chains. The transport of daughter nuclides is not calculated in the model.

1.3 CHANGES IN VERSION 4.2

Five further improvements were made to this version of the model: (1) the expansion of the model applicability from the field of radioactive waste disposal to the field of soil cleanup, agricultural land application, land reclamation, and accidental spill, (2) the combination of PRESTO-EPA-CPG and PRESTO-EPA-POP into one operation system, (3) the addition of a radionuclide-specific database, (4) the improvement of user friendliness by converting the interface program to a Windows system, and (5) the addition of data collection guidelines for those input data not commonly known.

This version of the PRESTO-EPA-CPG/POP model not only evaluates exposures from near-surface low-level radioactive waste, as did the original PRESTO-EPA-CPG and PRESTO-EPA-POP models, but it has been expanded to include repeated agricultural application operations, long-term impacts from accidental spills, and contaminated soil sites. However, no change was made to the modeling of exposures from LLW disposal operations.

The radionuclide-specific data are collected for 825 radionuclides. The system will automatically pull out necessary data from the built-in databases based on the nuclides that the user selects. However, the maximum number of radionuclides that the user can select remains at 40.

In response to user requests, the model now includes input data guidelines for the input parameters, upward equivalent hydraulic conductivity and diffusivity used for the infiltration calculation, and the parameters, residual and saturated water contents used for the unsaturated zone radionuclide transport calculation. The guidelines are now included in the documentation for clay, loam, and sand.

2. PRESTO-EPA-CPG/POP OPERATION SYSTEM

The PRESTO-EPA-CPG/POP operation system is a menu-directed, user-friendly system. The system is designed to help the user of the PRESTO-EPA-CPG/POP model to easily prepare the input characterizing a scenario of interest files and to graphically view results. This chapter contains step-by-step instructions on installing the system, and creating, processing, and viewing the results of a scenario input file.

2.1 SYSTEM INSTALLATION

The PRESTO-EPA-CPG/POP operation system is designed to operate in the Microsoft Windows 95 environment. The computer should be equipped with at least 16 megabytes of RAM and have a minimum of 10 megabytes of available disk storage.

To install the PRESTO-EPA-CPG/POP operation system, either from a set of 3.5 inch diskettes or from a downloaded folder, proceed as follows:

1. A. 3.5 inch diskettes: Insert the PRESTO-EPA-CPG/POP diskette 1 into drive A.
 B. Downloaded folder: Stores all files in the folder.
2. Click the **Start** button and select **Run**.
3. Type **a:\setup.exe** (where a: is your floppy drive) or **c:\download\setup** (if option B is used), and click **OK**.
4. Follow the setup instructions on the screen.

2.2 START UP

To begin a guided tour of the PRESTO-EPA-CPG/POP operation system, access the **Programs** option under the **Start** button and then click on the PRESTO-EPA-CPG/POP globe icon.

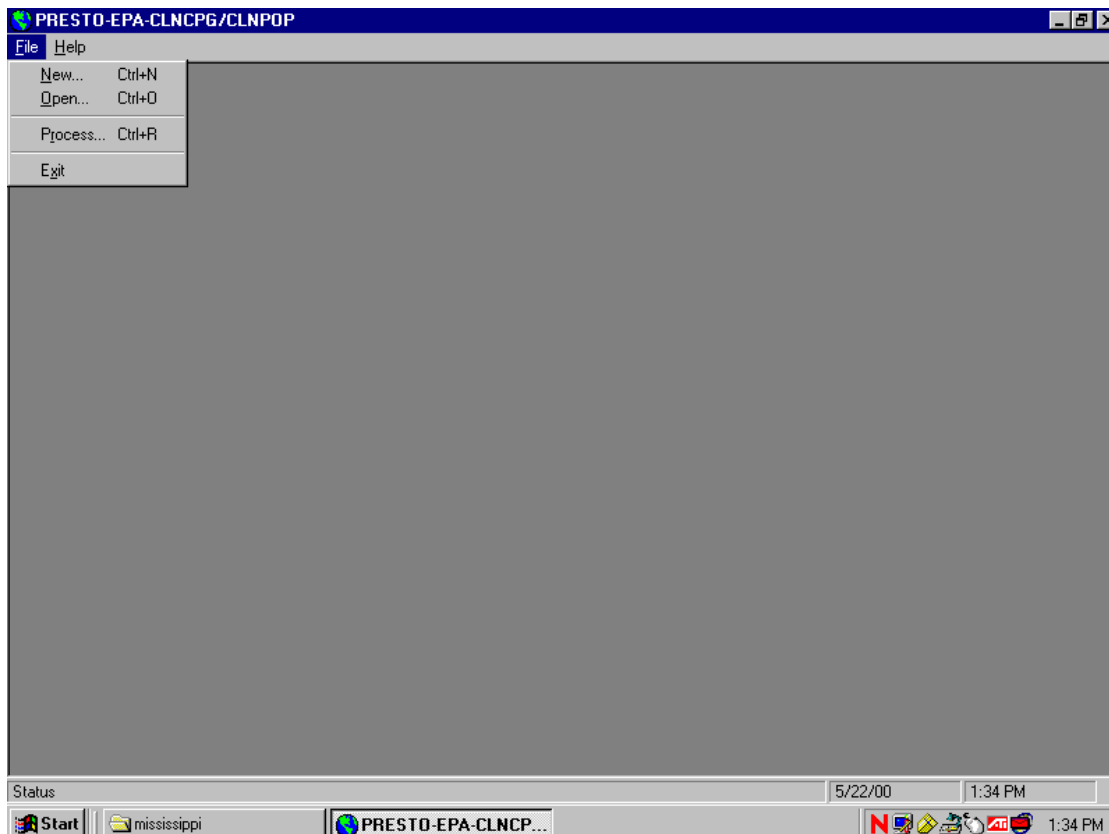
2.3 MOVING ABOUT IN THE SYSTEM

You can select an option in the PRESTO-EPA-CPG/POP operation system by either (1) clicking on the selection with the mouse or (2) moving to the selection with the keyboard using the **Tab**, **Shift-Tab**, or **arrow** keys. The menu options may also be selected by pressing and releasing the **Alt** key, followed by the appropriate highlighted letter.

2.4 SCENARIO INPUT FILE PREPARATION

The structure of the PRESTO-EPA-CPG/POP operation system consists of several options, each of which performs a designated function. The options shown under the **File** menu bar allow you to create a new input file, open an existing input file, and process one or more input files. A maximum of three input files may be opened at one time.

Figure 2-1. File Screen



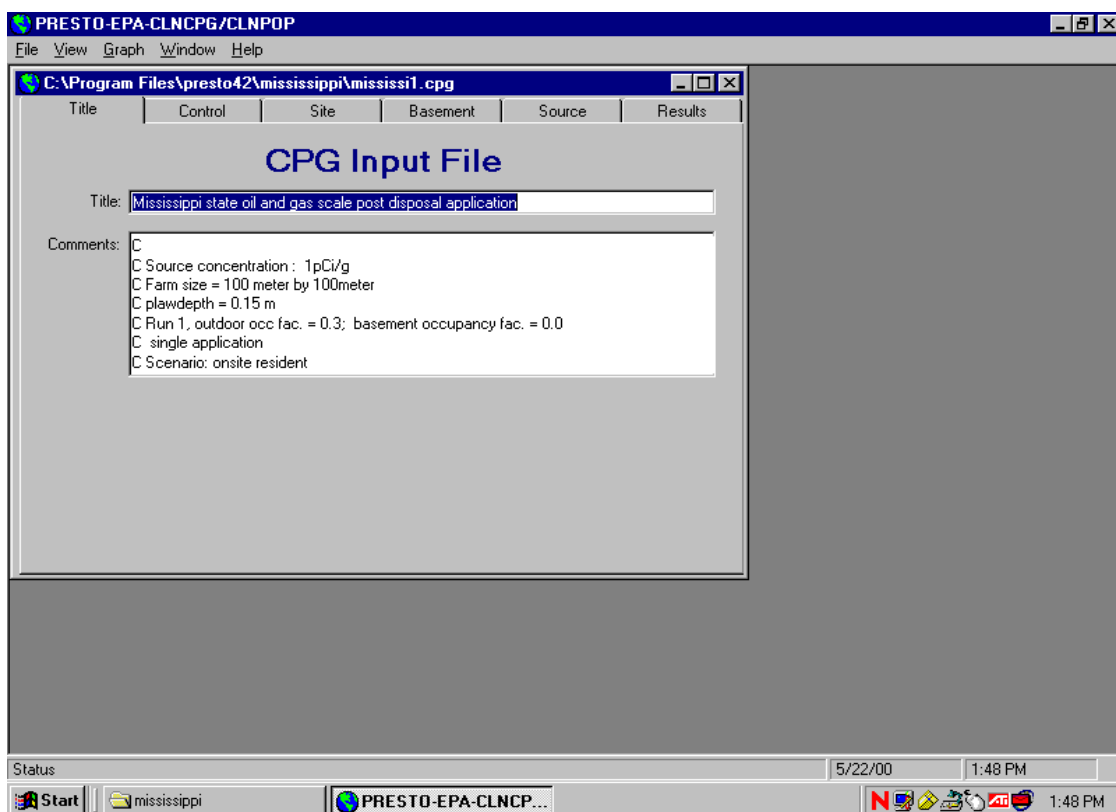
As shown in Figure 2-1 above, you can open an existing file or create an input file by activating the **File** menu option and then selecting **New** or **Open**. Alternatively, you can press and release the **Alt** key and then press **F** to activate the **File** menu option. To create a new input, select **New**. A window will appear prompting you to select the type of input file to create. The two available options/scenarios include either a CPG (Critical Population Group) input file or a POP (Population) input file. The CPG scenario estimates radiological doses to a critical population group and the POP scenario estimates the cumulative health effects to the general population. The type of file is indicated by the extension of the file name: “cpg” for critical population group and “pop” for general population. The file is named “Input File 1” until it is saved with a different name.

To access an existing file, select **Open** and specify the type of input file to open. The screen will now show a window with additional options that allow you to enter or modify input parameters.

2.4.1 Input File Parameter Categories

After you have opened an existing or new input file, the window will display six tabs that contain parameter descriptions and fields (**Title, Control, Site, Basement, Source, and Results**). These options allow you to modify or view parameters in a logical and organized manner. In addition to the options shown in the main menu, the **Site** and **Source** options contain a second level of options that will be discussed in more detail below. Appendix I contains a listing of the input parameters sorted by screen. The list also includes the units of each parameter and ranges of acceptable values.

Figure 2-2. Input Parameter Screen



As shown in Figure 2-2 above, from the main menu, select the **Title** option if it is not visible. The screen will show a window with several fields and field descriptions for each parameter. You can select input fields by either clicking on the field with the mouse or pressing the **Tab** key until the desired field is active. You can also see its specified unit by pointing and holding the mouse over the field. You can now input parameters by simply typing in the data and

then pressing the **Tab** key to select the next field. The other options are similar to the **Title** option in functionality.

Figure 2-3. Site Input Parameter Screen

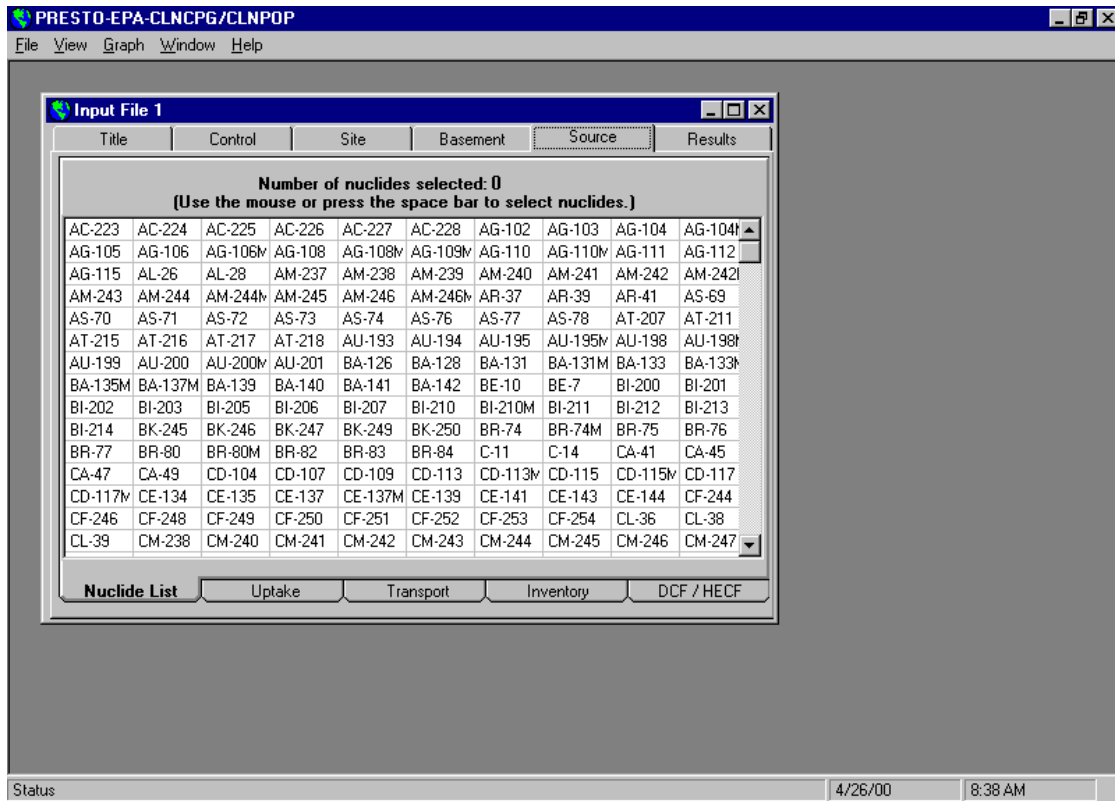
Parameter	Value
Total Annual Precipitation:	1.35
Annual Flow Rate of Stream:	3.57E+5
Distance to Onsite Well:	10.
Distance between site edge & ditch:	50.
Rainfall Factor:	50.0
Soil-Erodibility Factor:	0.13
Crop Management Factor:	0.30
Erosion Control Practices Factor:	0.1
Sediment Delivery Ratio:	1.0
Slope Steepness-Length Factor:	0.27
Fraction of Residual Saturation:	0.17
Fraction of Total Annual Precipitation for Infiltration Calculation:	0.00
Top Soil Layer Precipitation Run-off Fraction:	0.005
Bottom Soil Layer Precipitation Run-off Fraction:	0.1
Active Depth of Soil in Surface-Contaminated Region:	0.15
Width of the contaminated site measured perpendicular to groundwater flow:	1.0E+2
Length of contaminated site parallel to groundwater flow:	1.0e+2

Characteristics: Cover & Waste, Plant & Animal, Human Uptake, Vert. Zone & Aquifer, Atmosphere

Figure 2-3 above demonstrates that the **Site** tab contains a second level of tabs that are categorized by input parameters relating to plant, animal, and human uptake factors, cover and waste layers, vertical zone and aquifer, and the atmosphere. The definition of each input parameter is briefly presented on the screen. However, **Site Environment**, which is in the pull-down list located under the **Atmosphere** tab, requires additional explanation. This parameter indicates region-specific meteorological data to be selected for the analyses. Region-specific data are given for humid south, humid north, arid south, and arid north environments. You can also select the **User Defined** option when you want to create a user-defined meteorological data file (INFIL.INP). This file must be modified externally in a text editor.

The **Source** tab contains options relating to nuclide-specific data such as uptake factors, transport parameters, waste inventory, and dose and health effect conversion factors (see Figure 2-4 on the following page). Dose and health effect conversion factors are further grouped by pathway (i.e., ingestion, inhalation, immersion, and surface). The **Source** tab allows you to enter

Figure 2-4. Source Input Parameter Screen



nuclide-specific parameters into the input file. After you have selected this option, a new window appears with a list of 825 nuclides in alphabetical order. You can select a maximum of 40 nuclides to be included in the input file by double-clicking on the nuclide or by pressing the **Space bar** once the cell is active. The total number of nuclides selected is automatically updated and shown at the top of the window. Use the same method to deselect nuclides from the list.

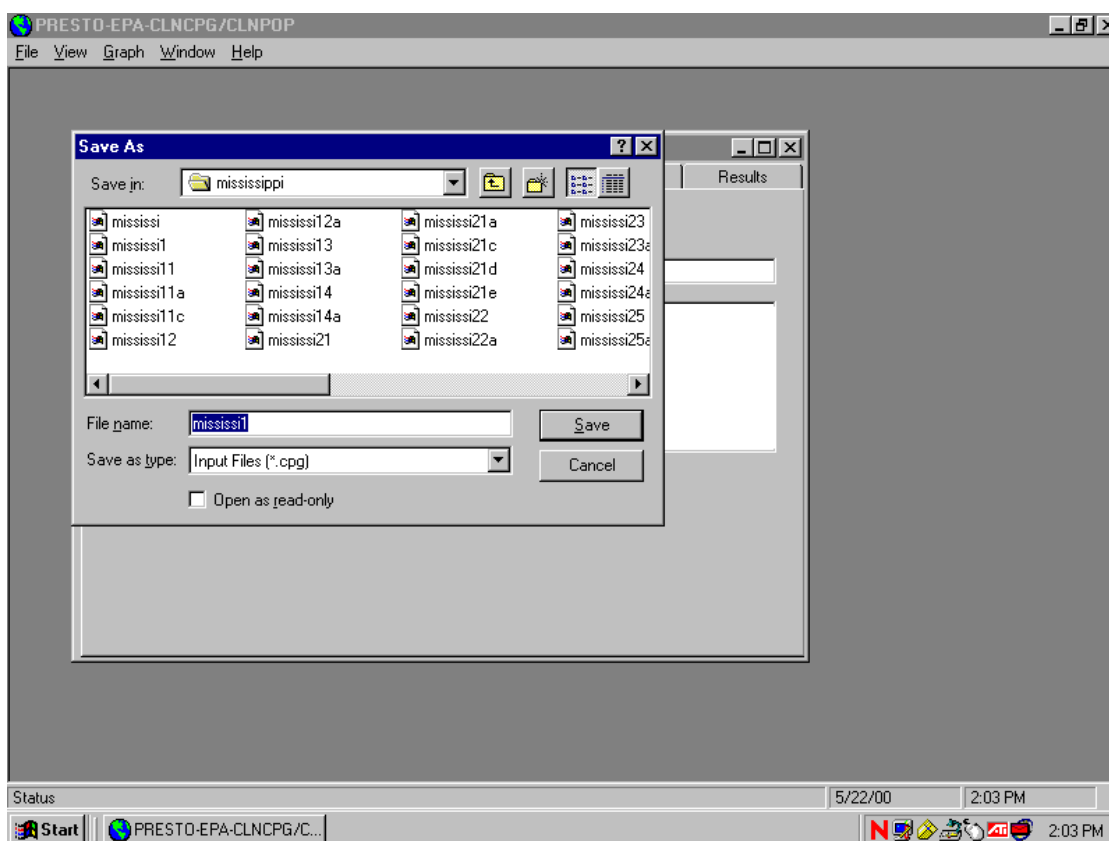
When a nuclide is added to the input file, the program automatically searches the dose and health effects conversion factor database for a default value (EPA88, EPA93, EPA99). If a value is found in the database, it is added to the input file. After you have selected the desired set of nuclides to include in the input file, you may re-enter new values for the dose and risk conversion factors. In addition, select the **Uptake**, **Transport**, and **Inventory** options to enter the appropriate nuclide-specific parameters. You also have the option of entering the radionuclide inventory in units of Bq or Ci by selecting the appropriate option at the top of the **Inventory** window.

2.4.2 Saving an Input File

To save the current input file, select the **Save As** option in the **File** menu bar, as shown in Figure 2-5 on the following page, and enter the desired filename. You can also change the path by choosing a different directory and drive. Once you have entered the filename, select the **Save**

option to save the input file or the **Cancel** option to return to the main menu without saving changes to the current file. When changes to an input file have not been saved, the system will warn you when you try to close the file. The same warning message appears if you try to exit the program without saving changes to the current file. The physical structures of the input files are documented in Appendix J-1 for PRESTO-CPG and Appendix J-2 for PRESTO-POP. They are provided for user reference only; you are **strongly encouraged** to modify these files only from within the operation system.

Figure 2-5. Save As Screen

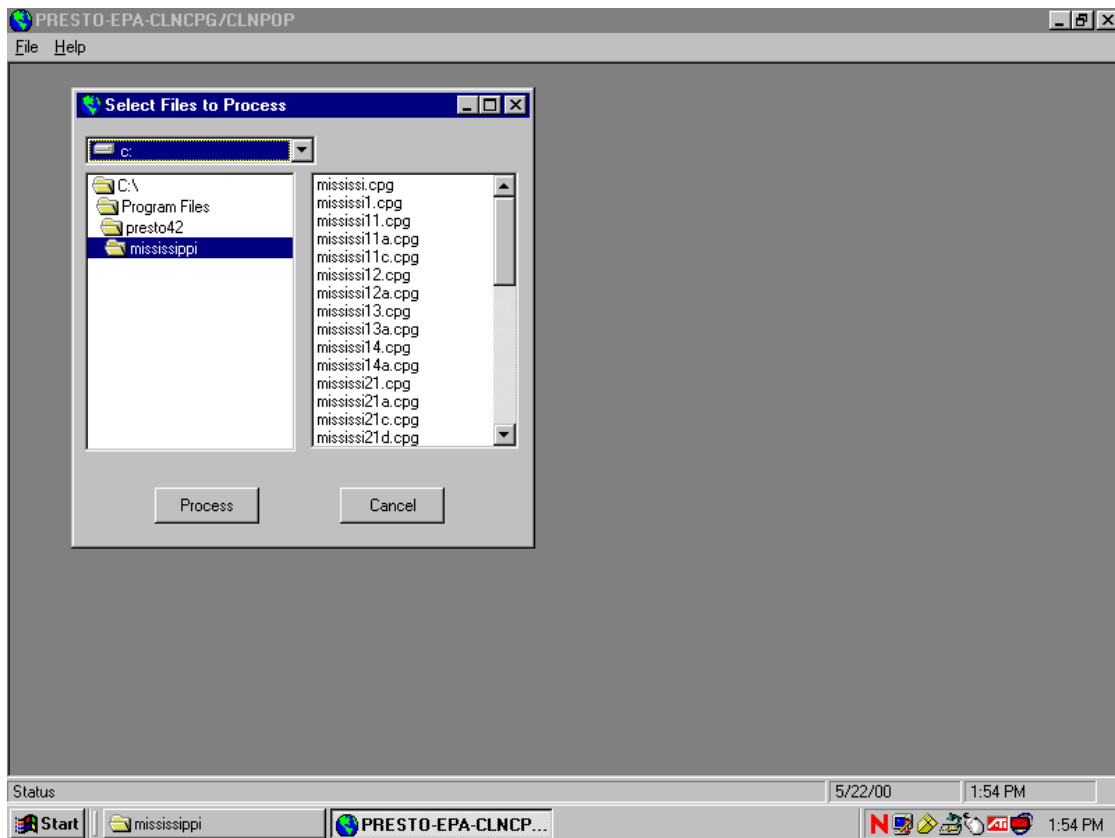


2.4.3 Processing an Input File

To process an input file, choose the **Process** option in the **File** menu bar (see Figure 2-6 on the following page). Locate the directory that contains the input files and then select at least one input file to process. A window will appear showing the file being processed and the number of input files that have yet to be processed.

After processing the input file, view the output by selecting the **Results** option. The system will prevent you from accessing this option until the input file has been processed. The output is organized into the following six categories:

Figure 2-6. Process Screen



1. Infiltration,
2. Initial Calculations,
3. Daughter Nuclide Effect Correction Factors,
4. Annual Summaries,
5. Maximum Individual Dose Summary or Total Risk Summary, and
6. Average Individual Dose or Health Effects.

The default units representing radionuclide activity and radiological doses are expressed as μSv and Bq. The units for activity and dose, however, can be changed to mrem and Ci by selecting the desired option shown on any one of the tabs.

2.4.4 Description of the Results

The results of PRESTO-EPA-CPG/POP are designed to be self-explanatory and contain descriptive comments, definitions, and intermediate and final tabulations. It is assumed that the results may be analyzed by users unfamiliar with PRESTO-EPA-CPG/POP structure. As illustrated by the sample input and output in Appendix K, the PRESTO-EPA-CPG/POP results are organized into the following sections.

2.4.4.1 Infiltration Calculations

The first section consists of the results for the infiltration calculations. The model calculates and outputs several data items. The most important of these are annual infiltration and annual precipitation. Annual evaporation, runoff, and infiltration are also calculated and output.

2.4.4.2 Initial Calculations

A set of tables under the heading “Initial Calculations” summarizes the radionuclide data used for the transport calculations. These tables include radionuclide distribution coefficients and nuclide inventories for both contaminated soil layers.

2.4.4.3 Daughter Nuclide In-Growth Effect Correction Factors

This section lists the daughter nuclide effect correction factors for each radionuclide. The derivation of these correction factors is presented in Appendix C.

2.4.4.4 Annual Summaries

Input control parameters determine the years for which intermediate results are printed. For these years, a number of hydrological and transport variables are output. Included are the status of the top layer (percentage intact), water depth in the contaminated zone, water loss by overflow and drainage from the contaminated zone, and radionuclide inventories. Radionuclide concentrations and flux values are also given for key pathways and regions of interest. Intermediate whole body doses to the critical population group or fatal cancer and genetic risks to the general population are other important results given in this section of the PRESTO-EPA-CPG/POP output.

2.4.4.5 Maximum Individual Dose Summary or Total Risk Summary

For the critical population group scenario, this section contains the data and results described in Section 2.4.4.4 for the year in which the maximum critical population dose occurs. This allows for specific identification of contributing pathways and radionuclides.

For the population scenario, this section contains the total risk summary for the entire simulation. These tables present a summary of the cumulative cancer incidence, cancer mortality, and genetic effects, the cumulative radioactivity pumped out of the well, and the cumulative release of radionuclides to the downstream basin for each radionuclide.

2.4.4.6 Average Individual Dose or Health Effects

For the critical population group scenario, the last section is similar to the results described in Section 2.4.4.5 except the results apply to the average exposed individual. For the general population scenario, this section provides the total number of fatal health effects and genetic health effects resulting from the contaminated soil site.

2.4.5 Graphing the Results of a Processed Input File

After processing either a CPG or a POP input file, you may graph the results by selecting the **Graph** option in the menu bar. For CPG results, the graphing options include: (1) individual dose, (2) mortality risk, (3) incidence risk, (4) well water concentration, and (5) total annual dose. For POP results, the graphing options include: (1) fatal cancer risks and (2) genetic risks. After you've selected a graph, a new window will appear allowing you to select a combination of nuclides and/or pathways to include in the graph. To view the graph by nuclide or pathway, select the appropriate tab.

You can modify the appearance of the graph by selecting the **Edit** option in the **File** menu bar. The chart style, text, and axis scale and type (i.e., linear or logarithmic) are a few of the formatting options that can be modified. Additionally, you may copy the graph and paste it into other applications by selecting the **Copy** option in the **Chart** menu bar.

The **Print** option in the **Chart** menu bar allows you to print the graph. The layout and formatting of the printed graph can be adjusted by selecting the **Layout** option on the **Print** dialogue box.

2.4.6 PRESTO-EPA-CPG/POP Help

At any time, you may access the **Help** option to view online help. PRESTO-EPA-CPG/POP Help contains instructions on using the system and detailed information on all of the input parameters. The help system also contains a table of contents, an index, and a search tab to help you locate the desired topic.

Additionally, parameter-specific help is provided whenever you place the mouse cursor over a parameter. The information displayed on the screen includes the parameter name, units of the parameter, and, if applicable, ranges of acceptable input values.

2.4.7 Exiting the PRESTO-EPA-CPG/POP Program

To exit the PRESTO-EPA-CPG/POP program, select the **Exit** option in the **File** menu bar. A warning message will appear on the screen if you have not saved changes to an opened input file. Select **Yes** to save and exit the program. Select **No** to exit without saving changes. To remain in the system, select the **Cancel** option.

3. THEORETICAL BACKGROUND OF THE PRESTO-EPA-CPG/POP MODEL

3.1 GENERAL DESCRIPTION OF THE MODEL

The model has been designed to calculate the maximum annual committed effective dose (CED) to a critical population group (CPG) and cancer incidence, fatal cancer effect, and serious genetic effect in the general population resulting from a near-surface disposal of low-level radioactive waste (LLW) site, from cleanup of a radiologically contaminated site, and from beneficial application/disposal of naturally occurring radioactive material (NORM) waste. The model simulates the transport of radionuclides from a site to the environmental receptors and human exposure through food chain pathways. The health effects for the general population are calculated from the radionuclide uptake rate and the health effect conversion factors.

The model assumes that the radionuclides may contaminate the near-ground surface in two soil layers – upper and lower layers – having different soil characteristics and contamination levels. Based on the nature of the exposure to the contaminants, three types of human receptors are considered: (1) onsite residents, residing on the contaminated site; (2) offsite residents, residing near the site; and (3) the general population, residing in the downstream basin, which extends all the way to the estuary.

The PRESTO-EPA-CPG model calculates the maximum individual dose to the onsite and the offsite residents depending on which is the critical population group, and the PRESTO-EPA-POP model calculates the cumulative number of fatal cancer effects and serious genetic effects to the onsite residents, offsite residents, and general population.

The onsite residents are assumed to be exposed to the contaminant through ingestion of (1) the drinking water pumped out from the groundwater and surface water contaminated, respectively, from subsurface leaching and overland flow; (2) foods (crops and cattle) grown on the farmland contaminated from initial contamination, subsequent chronic application (NORM waste disposal, contaminated irrigation water), and redeposit of radionuclides being transported from upstream through atmospheric process; and (3) fish caught from nearby contaminated surface waters, and through soil, inhalation of radon gas outdoors and indoors, direct exposure outdoors and in the basement, and immersion in the air that is resuspended from local soil or is transported upstream.

The offsite residents are assumed to be subject to the same exposure as the onsite residents except that there is no initial contamination of soil in the resident area and the exposure from radon gas is direct exposure from living in the basement. In addition, the offsite residents are assumed to be exposed to the radionuclides being transported from the contaminated site through atmospheric transport, which contaminates the foods grown in the area.

The general population is assumed to be exposed to all pathways that the offsite residents experience, except the groundwater and atmospheric transport pathways.

3.1.1 Description of a Near-Surface Low-Level Radioactive Waste Disposal Site

The life cycle of a LLW disposal site begins with site selection. Following site selection and regulatory approval, trenches are dug on the site. Waste materials, which may be a combination of trash, absorbing waste activate metal, and solidified wastes, are containerized and placed into each trench. Once a section of the trench is filled, the trench is backfilled to eliminate voids to decrease the potential for subsidence and cracking of the trench cap. Following backfilling, the trench is covered with a cap of soil, one to several meters thick, and mounded above grade to facilitate runoff and decrease infiltration. The cover soil may be clean or contaminated with spilled radionuclide.

The site is assumed to have a definite period of institutional control following the completion of the disposal operation. During the institutional control period, the cap is assumed to be maintained in its intact condition. However, the site is assumed to be a restricted land-use area and no residents are allowed to reside onsite.

The model can assess the health impacts to offsite residents and the general population downstream for various combinations of waste form and disposal operation alternatives.

3.1.2 Description of a Contaminated Soil Site

A contaminated soil site is a site with residual radionuclides resulting from the cleanup of an accidental spillage or the decommission of a nuclear facility. Unlike a radioactive waste disposal site, the contamination generally does not extend more than a few meters below the ground surface. However, the radionuclide release and transport processes at a contaminated soil site are similar to those at a radioactive waste disposal site. The contaminated soil site may be modeled as a single layer of contamination (with or without a clean soil cover layer) or as two distinct layers of contamination.

The model can assess the health impacts to onsite residents, offsite residents, and the general population.

3.1.3 Description of a Land Farming Site

A farmland application of NORM waste contaminated with a diminished level of radioactivity is normally used for the purpose of disposal and/or to benefit the land. Waste is applied to the surface of the selected site and then mixed, using a traditional plow, to a nominal depth. This operation can be either a one-time application or repeated annually for a given number of years. No cover or cap is present in this scenario. Additionally, no waste treatment, such as solidification or containerization, is modeled.

The model can evaluate long-term effects to onsite residents, offsite residents, and the downstream general population.

3.2 MATHEMATICAL FORMULATIONS

3.2.1 Model Assumptions and Structure

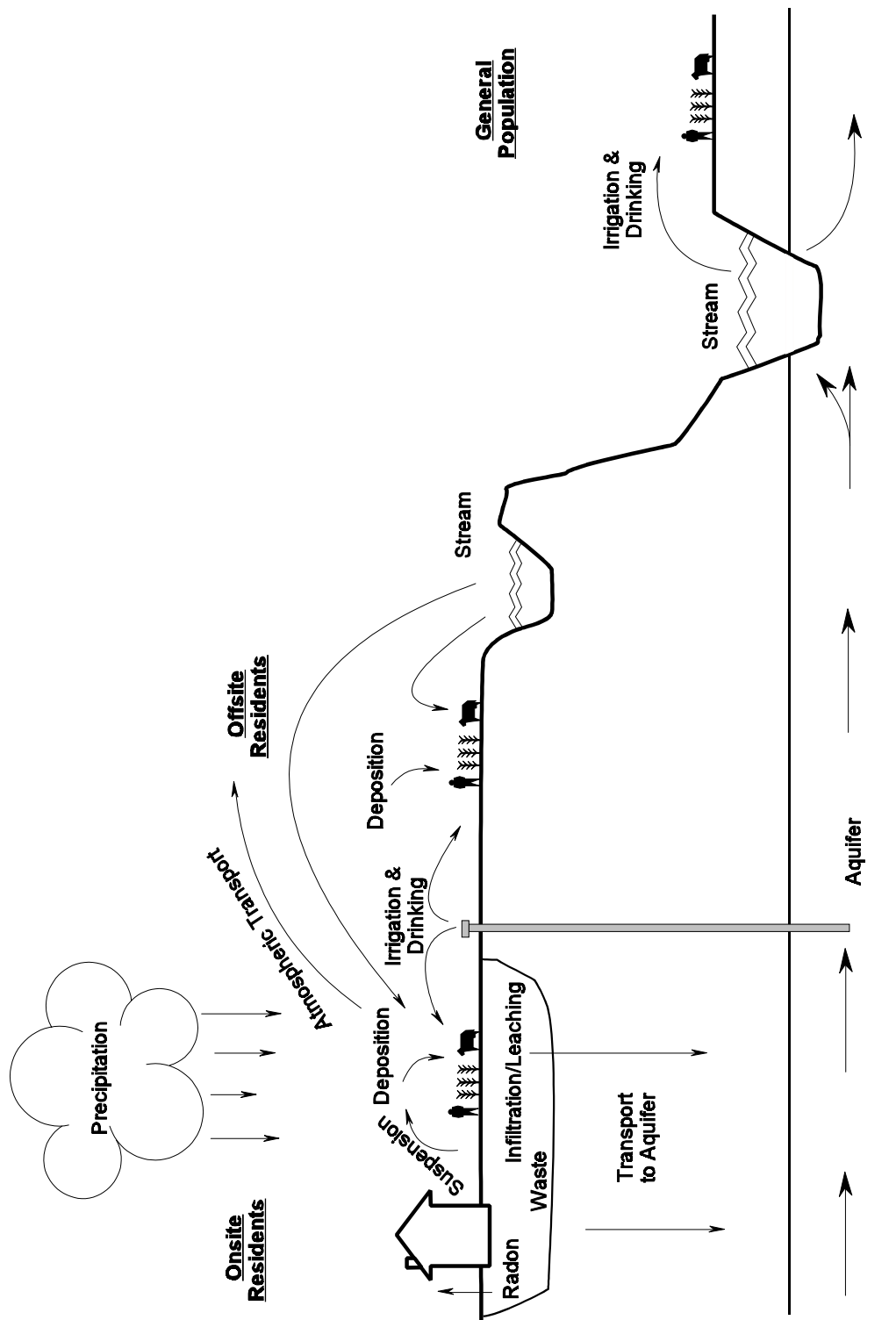
The PRESTO-EPA-CPG/POP operation system, which includes the PRESTO-EPA-CPG and PRESTO-EPA-POP models and their interface programs, is designed to accommodate a wide range of hydrogeologic and climatic conditions. It is also designed to handle the leaching of radionuclides and their subsequent groundwater transport through unsaturated and saturated hydrogeologic conditions, while taking into account nuclide retardation due to geochemical processes. The system's features account for the dynamic leaching process, the farming scenario, which simulates farming over the waste with root uptake of radionuclides from the waste matrix, and the reduction in the source inventory due to radionuclide decay during the simulation.

In general, hydrologic transport is the principal pathway by which the human receptors may become exposed to the radioactivity contaminating a disposal/contaminated soil site. Figure 3-1 shows a schematic of the pathways through which the water may transport radionuclides from a site to human receptors. The major source of water, which is the primary driving force for leaching the radionuclides out of the contaminated matrix, is from precipitation. The precipitated water at a site will either infiltrate into the soil layers, run off the site by overland flow, or evaporate into the atmosphere.

The transport of radionuclides from a site may occur by the infiltrated water or by the overland flow. A dynamic model, which calculates the evaporation loss and water transport based on their dynamic equations, is used to calculate the rate of infiltration (Appendix A). The infiltrated water entering the contaminated zone leaches out radionuclides from the contaminated zone. This contaminated water may either overflow from the top of the site or percolate downward to the subsoil and ultimately enter an aquifer.

Radionuclides that finally reach the aquifer will generally be transported at velocities less than or equal to the flow velocity of the water in the aquifer. This "retardation" is due to the interaction of radionuclides with solid media in the aquifer, known as the sorption effect. When the radionuclides being transported in the aquifer reach a well, they will be consumed by onsite and/or offsite residents through drinking, irrigation, and cattle feed pathways. Residual radionuclides in the aquifer are assumed to be transported farther downstream and impose additional health impacts on downstream general populations.

Figure 3-1. Environmental Transport Pathways



The contaminated water in the site will accumulate if the rate of infiltration exceeds the rate of exfiltration out the bottom of the contaminated zone. When the volume of water accumulated in the waste exceeds the total void space, contaminated water overflows onto the ground surface. The radionuclides in the contaminated water will then mix with the overland flow and be further transported into nearby streams.

This contaminated water will potentially be consumed by the local residents and by the population downstream via drinking, irrigation, cattle feed, and fishing pathways.

The model incorporates the effect of daughter nuclide in-growth in its final results by multiplying the parent-nuclide-caused health effects with its daughter nuclide in-growth effect correction factor, which is defined in Section 3.2.9. Up to the fourth member of the decay chain is included in this adjustment.

The complex physical and chemical interactions between the radionuclides and the solid geologic media have been grouped into a single factor, the distribution coefficient (K_d). Different K_d values can be used for the top contaminated soil layer, the bottom contaminated soil layer, the subsoil, and the aquifer.

The model assumes that the leaching process takes place in both the upper and lower soil layers. The top layer takes the infiltrated rainwater as its driving force to leach out radionuclides from the soil, while the bottom layer receives the leachate from the top soil layer and uses it as the driving force to leach the radionuclide in the bottom soil layer. Negative leaching, or sorption, may occur when the concentration of the radionuclides in the leachate exceeds the equilibrium concentration of radionuclides in the bottom soil layer.

The subsurface transport path of radionuclides is assumed to be vertical from the waste to the aquifer and then horizontal through the aquifer. The flow in the vertical flow regime is calculated either as saturated or unsaturated flow, depending on the relationship among the rate of exfiltration, the degree of saturation, and the properties of the geologic media. The transport of radionuclides in the aquifer is calculated by employing Hung's "optimum groundwater transport model" (Hu81), in which Hung's correction factor is used to compensate for the effects of longitudinal dispersion (given in Appendix B).

Three types of submodels are used in the PRESTO-EPA-CPG/POP model: unit response, bookkeeping, and scheduled event. The unit response submodels calculate the annual response of a given process. For example, the submodel INFIL calculates the annual infiltration through an intact top soil layer. 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 top layer.

Bookkeeping submodels keep track of the results of unit response submodels and user-supplied control options. For example, the TRENCH submodel calculates the level of standing water in the source-contaminated layers and the volume of water leaving the contaminated zone.

Annual concentrations of each radionuclide in environmental receptors, such as well water or the atmosphere, are used to calculate radionuclide concentrations in foodstuffs. Foodstuff concentrations and average ingestion and breathing rates are utilized to calculate the annual average radionuclide intake per individual in the onsite and local populations. These intake data are then used to estimate doses and health effects.

The atmospheric transport submodel assumes a constant dust loading and mixing height for estimating onsite concentrations. For offsite atmospheric transport, the system assumes that the entire local population resides within the same 22.5-degree sector. User-specified parameters give the fraction of year that the plume blows in that sector. The transport of the radionuclides from the source area to the local population is calculated by employing the Gaussian plume diffusion model. Therefore, each member of the local population will inhale the same quantity of each radionuclide.

Each person in a specific local community is assumed to consume the same quantities and varieties of food, all grown in the same fields, and to obtain his or her drinking and crop irrigation water from the same source. However, the user may specify the distribution of the sources of drinking and irrigation water supplies between well and stream. Table 3-1 lists the environmental pathways by which members of the onsite, offsite, and general populations are exposed to radioactivity.

Table 3-1. Exposure Pathways

Onsite Residents	Offsite Residents	General Population
Drinking water	Drinking water	Drinking water
Crop irrigation	Crop irrigation	Crop irrigation
Cattle feed	Cattle feed	Cattle feed
Fish ingestion	Fish ingestion	Fish ingestion
Soil ingestion	Soil ingestion	Soil ingestion
Dust inhalation	Dust inhalation	
Outdoor direct exposure (ground surface and air)	Outdoor direct exposure (ground surface and air)	
Basement gamma exposure		
Radon inhalation (indoor/outdoor)	Radon inhalation (outdoor)	

The mathematical formulation adopted by the model in calculating radionuclide concentrations in the pertinent environmental receptors is described in the following two sections.

3.2.2 Transport Pathways Involving Water

Figure 3-2 shows the surface water and groundwater transport pathways and their associated exposure pathways (see Section 3.2.1).

3.2.2.1 Infiltration Through Top Layer

The basic model for simulating the annual infiltration through the contaminated soil is based on the cap failure function in an interim version of the model. It assumes that a portion of the top layer material will degrade and allow the precipitated water to flow through the lower contaminated layer. The fraction of the top layer that 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 top layer, the annual infiltration through the top layer is divided into two components.

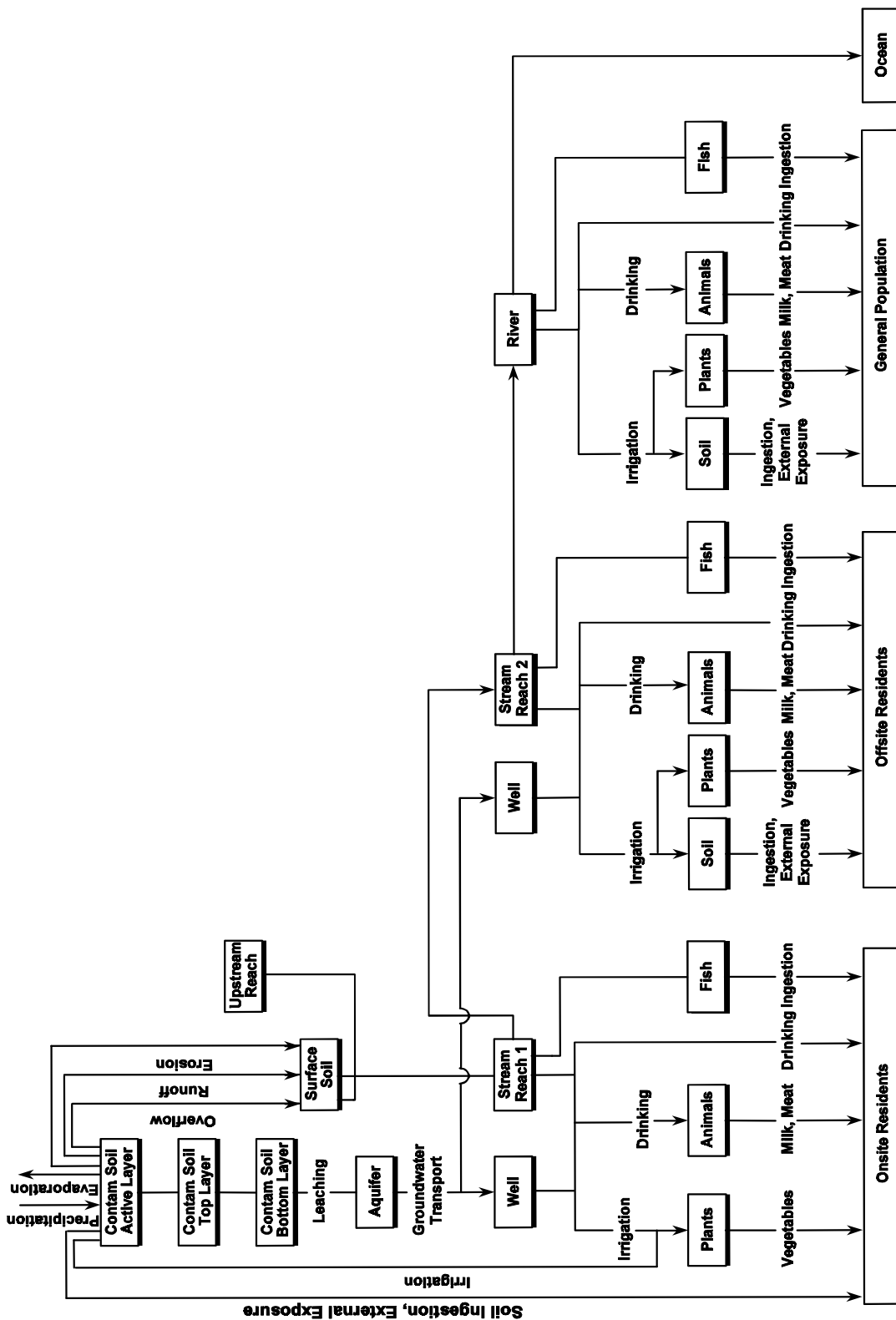
On the intact portions of the layer, the normal infiltration rate is calculated by the method developed by Hung (Hu83b), which is described in Appendix A. For the failed portion of the top layer, the infiltration equals the rainfall. Therefore, the volume of water entering the trench annually is calculated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

$$W_t = A_t[f_c \cdot W_s + (1 - f_c)W_a] \quad (3-1)$$

where

- W_t = volume of water entering trench in current year (m^3),
- A_t = area of trench (m^2),
- f_c = fraction of top layer that has failed (unitless),
- W_a = annual infiltration through intact cap or top layer (m),
- W_s = annual infiltration through bottom layer where cap has failed (m)
= $(P_a + I_r)f_b$
- P_a = annual precipitation (m),
- I_r = annual irrigation rate (m), and
- f_b = fraction of infiltration through bottom layer (unitless).

Figure 3-2. Water Transport Exposure Pathways



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

The component of annual infiltration through the intact portion of the top layer, 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 that describe the dynamics of overland flow, subsurface flow, and atmospheric dispersion systems. The basic equations employed in the model are:

$$Q_0 = \{(\text{Sin}\theta)^{1/2} H^{5/3}\}/n \quad (3-2)$$

$$dH/dt = P - E_o - q_o - Q_o/L \quad (3-3)$$

$$E_o = \begin{cases} E_p & \text{when } P + H/\Delta t > E_p \\ P + H/\Delta t & \text{when } P + H/\Delta t > 0 \\ 0. & \text{when } P + H/\Delta t = 0 \end{cases} \quad (3-4)$$

$$q_o = \begin{cases} K_s & \text{when } P - E_o + H/\Delta t > K_s \\ P - E_o + H/\Delta t & \text{when } K_s > P - E_o + H/\Delta t > 0 \\ 0. & \text{when } P - E_o + H/\Delta t = 0 \end{cases} \quad (3-5)$$

$$q_i = \begin{cases} K_s & \text{when } Z_g > Z_{\max} \\ 0. & \text{when } Z_g = Z_{\max} \end{cases} \quad (3-6)$$

$$dZ_g/dt = (q_i - q_o + q_t)/W_g \quad (3-7)$$

$$q_L = -D_e W_p/Z_p + K_e \quad (3-8)$$

$$q_L < E_p - E_o$$

$$q_v = -(E_p - E_o) \times \left[1 + \frac{0.5 \cdot Z_p}{0.66 \cdot (W_p + W_g)} \right]^{-1} \quad (3-9)$$

$$dZ_p/dt = -(q_p + q_t)/W_p \quad (3-10)$$

$$q_t = \begin{cases} q_0 & \text{when } Z_p > 0 \\ 0. & \text{when } Z_p = 0 \end{cases} \quad (3-11)$$

and

$$q_p = -\text{Max}(*q_L*, *q_v*) \quad (3-12)$$

where

- Q_o = rate of overland flow per unit width of trench cover ($\text{m}^3/\text{m}\cdot\text{hr}$),
- 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,
- θ = 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, it is numerically identical to the porosity for the gravity water (unitless),
- W_p = component of wetness for the pellicular water; under a fully saturated condition, it 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), and
- q_p = flux of pellicular water (m/hr).

The amount of annual infiltration through the top soil layer or trench cap is then calculated by integrating the hourly infiltration over the entire year.

The values of hydraulic conductivity and diffusivity to be used in the calculation of the flux for the pellicular water are theoretically functions of water content (see Equation 15, Appendix A). In order to simplify the calculation, the water content independent terms “upward equivalent hydraulic conductivity” and “upward equivalent hydraulic diffusivity” are introduced for the submodel. They are calculated from the soil characteristics obtained from soil tests.

For the purpose of modeling input data collection, three typical soils with appropriate soil characteristics are selected and their equivalent values are calculated. Details of the derivation/calculation are attached in Appendix D. The results of the calculation are presented in Table 3-2 below. These values are intended for use as guidelines in preparing the input parameters for equivalent conductivity and diffusivity.

Table 3-2. Soil Characteristics

Variables	Sand	Loam	Clay
upward diffusivity, m ² /hr	7.2E-4	3.6E-4	1.4E-4
upward hydraulic conductivity, m/hr	3.6E-6	1.4E-6	1.1E-6

3.2.2.2 Trench Cap Modification

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

The annual amount of erosion is subtracted from the layer 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:

$$I_i = f_r \cdot f_k \cdot f_l \cdot f_s \cdot f_c \cdot f_p \cdot f_d \tag{3-13}$$

where

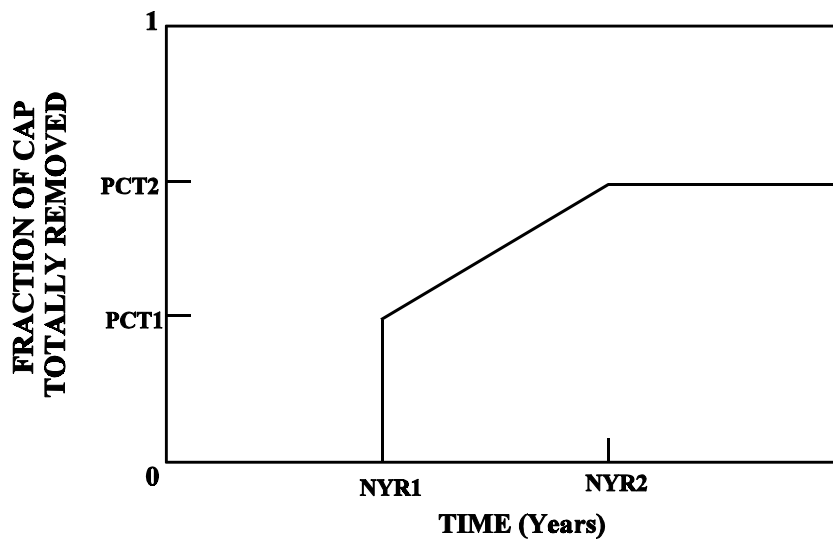
I_1	=	yearly sediment loss from surface erosion (tons/ha),
f_r	=	rainfall factor (f_r unit or 100 m-tons-cm/ha),
f_k	=	soil erodability factor (tons/ha/ f_r -unit),
f_l	=	slope-length factor (unitless),
f_s	=	slope-steepness factor (unitless),
f_c	=	cover factor (unitless),
f_p	=	erosion control practice factor (unitless), and
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 (3-13). The rainfall factor, f_r , expresses the erosion potential caused by average annual rainfall in the locality. The soil erodability 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 cover factor, f_c , represents the ratio of the amount 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 reduction in the erosion potential due to 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_1 are converted to (m/yr) within the code. See Appendix I for the description of input units.

The second method of top layer failure is mechanical disturbance due to human intrusion or some other means, such as channel erosion, which might completely destroy portions of the top layer. This phenomenon can be termed a partial failure, but in reality it is a total failure of some part of the top layer. The code user may specify some rate of failure as shown in Figure 3-3.

Figure 3-3. Trench Cap Failure Function



NYR1 is the year that the cap begins to fail,
 NYR2 is the year the cap ends additional failure,
 PCT1 is the fraction of cap failure at year NYR1, and
 PCT2 is the fraction of cap failure at year NYR2.

By specifying appropriate values for the time in Figure 3-3, the user may selectively simulate the failure of the cap or top layer from a portion of the site area. Mathematically, this function is represented by:

$$f_c = \begin{cases} 0, & t < \text{NYR1} \\ \frac{(\text{PCT2} - \text{PCT1})(t - \text{NYR1})}{(\text{NYR2} - \text{NYR1})} + \text{PCT1}, & \text{NYR1} \leq t \leq \text{NYR2} \\ \text{PCT2}, & t > \text{NYR2} \end{cases} \quad (3-14)$$

Even though PCT2 might be less than 1.0 in year NYR2, 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.

3.2.2.3 Rate of Infiltration Through Trench Cap

The amount of water infiltrating through and leaving the bottom layer annually is calculated by Darcy's law but cannot exceed the available volume of water remaining in the soil layer, which is expressed mathematically in:

$$V_B = \text{Min}(D_W \cdot A_T, I_T \cdot A_T) \quad (3-15)$$

in which D_W is calculated by:

$$D_W = V_W / (A_T \cdot W_T) \quad (3-16)$$

In above equations:

V_B = volume of water leaving contaminated zone annually (m^3/yr),

D_W = depth of water in contaminated zone during current year (m),

I_T = conductivity of material below the contaminated zone (m/yr),

A_T = trench area (m^2),

V_W = volume of water in contaminated zone (m^3), and

W_T = porosity of soil (unitless).

3.2.2.4 Rate of Overflow

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

$$V_O = (D_W - D_T)A_T \cdot W_T \quad (3-17)$$

where

V_O = volume of water overflowing site in a year (m^3),

D_W = depth of water in contaminated zone (m),

D_T = contaminated zone depth (m),

A_T = site area (m^2), and

W_T = porosity of soil material (unitless).

3.2.2.5 Radionuclide Leaching

Water in the contaminated zone may be contaminated by contact with the radioactivity. To calculate the concentration of radionuclides in the water exfiltrating out of the site, two model types are used, a dynamic model based on the chemical exchange and an empirical model based on the annual release fraction.

The user must choose one of the three options shown in Table 3-3 to calculate the concentration of radionuclides in the exfiltrating water.

Table 3-3. Leaching Options (LEAOPT) Specified

Option	Leach Calculation Method
1	Chemical exchange without solubility limit
2	Chemical exchange with solubility limit
3	Annual release fraction

Leaching options 1 and 2 utilize a dynamic model that estimates the radionuclide concentration in the water based on chemical exchange. When this model is selected, annual release rates are required for solidified wastes and activated metals. Contaminants released from these waste forms are then treated to chemical exchange, as is the absorbing waste.

The model is developed based on a multi-phase leaching concept (Hu86b), which simulates a leaching system under a field environment. The model assumes that the flow of infiltration is concentrated in preference paths and, thereby, forms a finger flow system. This flow system leads to the transport of radionuclides in two phases, the convective phase and the diffusive phase. These phases of transport are assumed to take place in the convective zone and the diffusive zone, respectively. The radionuclides in the diffusive zone must be transported to the convective zone before they can be transported downward through the convective process.

Due to the complexity in the modeling of the multi-phase leaching concept, a simplified and yet conservative model is used. The simplified model assumes an idealized steady, uniform leaching model to calculate the radionuclide concentration in the trench water based on the chemical exchange process. A correction factor is then added to account for the leaching process under field conditions derived from the multi-phase leaching concept (Hu86b). The final formula is expressed by:

$$C_{TW} = \frac{I_T \cdot FAC}{A_T(D_T W_T + D_T K_{d2} \rho_w)} \quad (3-18)$$

(Chemical exchange option)

and

$$FAC = \text{Min} [TINFL/PERMT, 1] \quad (3-19)$$

where

- FAC = a correction factor to account for the multi-phase leaching phenomenon experienced in field conditions,
- TINFL = effective annual infiltration rate (m/yr),
- PERMT = soil hydraulic conductivity (m/yr),
- C_{TW} = concentration of radionuclides in contaminated water (Bq/cm³),
- I_T = amount of activity in layer (Bq),
- A_T = site area (m²),
- W_T = porosity within soil layer (unitless),
- D_T = layer depth (m),
- K_{d2} = distribution coefficient within waste for radionuclides (ml/g), and
- ρ_w = density of waste material (g/cm³).

Leaching option 2 uses a solubility factor to estimate the maximum concentrations of radionuclides in the leachate. The solubility option may be used when the radionuclide solubility is low or information concerning K_d values is not available. The radionuclide concentration is estimated by:

$$C_{TW} = \text{Min} \left[\frac{S \cdot N_c N_v}{M}, \frac{I_T \cdot FAC}{D_T A_T W_T + A_T D_T K_{d2} \rho_w} \right] \quad (3-20)$$

(Solubility Option)

where

S	=	elemental solubility (g/ml),
M	=	mass of radionuclide (g/mole),
N _c	=	ratio (Bq/mole), and
N _v	=	ratio (ml/m ³).

Leaching option 3 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 3 is normally used for a solidified waste form. The model calculates the primary release of radionuclides 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 (3-18) 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.

If the constant fraction release model is chosen, the radionuclide concentrations in the water leaving the waste are given by:

$$C_{TW} = \frac{Q_U \cdot f_L + Q_P}{W} \quad (3-21)$$

where

C _{TW}	=	nuclide concentration in water leaving the waste (Bq/m ³),
Q	=	total unleached inventory (Bq),
f _L	=	annual leach fraction (1/yr),
Q _P	=	leached activity in waste left from previous year (Bq), and
W	=	volume of water in contaminated zone (m ³).

3.2.2.6 Waste Container Effects

Waste containers can inhibit nuclide leaching until they lose their integrity. The length of time before containers lose their integrity – container life – depends on their design, structural strength, and material. In PRESTO-EPA-CPG, the net radionuclide release is calculated by multiplying the radionuclide concentration in the trench water by the fractional container fracture factor (CFF), which is time-dependent. The fraction CFF is set to zero while all of the containers are intact. Once the containers start to fracture, CFF is assumed to increase linearly to a maximum value of 1, which represents failure of all of the containers.

3.2.2.7 Transport Between Contaminated Soil Layers

The leaching of the contaminated soil layers is modeled in three layers: active, top, and bottom layers. The active layer is subdivided from the top layer and remains the same thickness throughout the entire period of analysis, while the thickness of the remaining top layer decreases with the progress of the sheet erosion. The three-layer leaching model is based on the above two equations with the added capability that radionuclides may transfer from one contaminated layer to the other. For groundwater pathway calculation, the three-layer leaching model is implemented by first applying the leaching equations to the active layer to get a leachate concentration. The amount of radioactivity transferred from the active layer to the top layer or from the top layer to the bottom layer is the product of the leachate concentration in the layer and the volume of deep infiltrating water. That is:

$$\begin{aligned} P_{A1} &= C_{WA} \cdot V_b \\ P_{12} &= C_{W1} \cdot V_b \end{aligned} \tag{3-22}$$

where

- P_{A1} = radioactivity transferred from active layer to top layer (Bq),
- P_{12} = radioactivity transferred from top layer to bottom layer (Bq),
- C_{WA} = leachate concentration in active layer (Bq/m³),
- C_{W1} = leachate concentration in top layer (Bq/m³), and
- V_b = volume of water leaving bottom of waste (m³).

In some cases, site conditions do not allow the downward movement of water through the waste. In these cases, the waste becomes saturated and water overflows from the site. This leads to upward movement of radioactivity from the bottom layer to the top layer and from the top layer to the active layer. The amount of radioactivity transferred upward to the top and active layers is the product of the leachate concentration in the layer and the volume of water that overflows from the site. Mathematically it can be represented by the following equation:

$$\begin{aligned} P_{1A} &= C_{W1} \cdot V_0 \\ P_{21} &= C_{W2} \cdot V_0 \end{aligned} \tag{3-23}$$

where

- P_{1A} = radioactivity transferred from top layer to active layer (Bq),
- P_{21} = radioactivity transferred from bottom layer to top layer (Bq),

C_{w1} = leachate concentration in top layer (Bq/m³),

C_{w2} = leachate concentration in bottom layer (Bq/m³), and

V_o = volume of water overflowing the site (m³).

These equations are implemented in the LEACH subroutine, which is called by the main program for every radionuclide for each year of the simulation. As in the original PRESTO codes, the leached radionuclides either travel downward to the groundwater pathway or flow across the ground surface to the surface water pathway.

3.2.2.8 Transport Below Contaminated Zone

Once radionuclides have been leached out of the waste in the contaminated zone, they are transported vertically downward 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.

Because of the distinct nature of radionuclide transport in various reaches, the model subdivides the transport field into three reaches: vertical reach, collection reach, and horizontal reach. The solute transport analyses for each reach are conducted as detailed in the following subsections.

Vertical Reach

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 calculated from the equation:

$$SSAT = RESAT + (1 - RESAT) \times \left[\frac{ATINFL}{PERMV} \right]^{0.25} \quad (3-24)$$

where

RESAT = fraction of residual moisture content relative to saturated moisture content (unitless),

ATINFL = average exfiltration rate (m/yr), and

PERMV = saturated hydraulic conductivity in the vertical reach (m/yr).

Equation (3-24) is an empirical formula for predicting the fraction of saturation in a soil under dynamic infiltrating conditions as developed and applied by Clap and McWhorter (Cla78,

McW79). The exponent, 0.25, should theoretically be a function of soil type, but a conservative fixed value was selected based on the experimental data for simplicity. In order to justify the reliability of this equation, a benchmark study is conducted to compare it with another commonly used empirical formula. The results of the comparison fit quite well with each other. Details are presented in Appendix E.

In Equation (3-24), the residual moisture content is the moisture content remaining in the soil after long-term drainage through gravity, and it is equivalent to the component of moisture content for the hygroscopic water. RESAT is the ratio of this water content and the saturated water content and is a user input parameter. As a guideline for selecting this value, typical values for sand, loam, and clay soils are calculated based on the soil test data obtained in the laboratory (Hil76). Details of this calculation is presented in Appendix E. The results are 0.25, 0.42, and 0.77 respectively for sand, loam, and clay soils.

The parameter ATINFL is the average trench exfiltration rate. When there is no overflow of trench water, the rate is calculated by the expression:

$$ATINFL = [PCT2 \cdot (PPN+XIRR)+(2 - PCT2) \cdot XINFL] \cdot 0.5 \quad (3-25)$$

where

- PCT2 = maximum fraction of trench cap failure (unitless),
- PPN = annual precipitation rate (m/yr),
- XIRR = annual irrigation rate (m/yr), and
- XINFL = infiltration rate through the intact top layer (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:

$$V_v = ATINFL/(PORV \cdot SSAT) \quad (3-26)$$

$$R_v = 1 + (BDENS \cdot XKD3)/(PORV \cdot SSAT) \quad (3-27)$$

where

- BDENS = host formation bulk density (g/cm^3),
- XKD3 = distribution coefficients for the host formation (ml/g), and

PORV = subsurface porosity (unitless).

Horizontal Reach

The transport analysis for the horizontal reach calculates the radionuclide transport in the aquifer without lateral or vertical supply of radionuclide flux. The transport analysis employs Hung's groundwater transport model (Hu81, Hu86a, Appendix B). The basic equations for the model, as adopted from Hung, are:

$$Q = \eta Q_o(t-RL/V+t_L) \text{Exp}(-\lambda_d RL/V+t_L) \tag{3-28}$$

and

$$\eta = \int_0^\infty \frac{0.5(RP/\pi\theta^3)^{1/2} \text{Exp}[-N_d\theta-(P\theta/4R)(R/\theta-1)^2]d\theta}{\text{Exp}(-RN_d)} \tag{3-29}$$

$$= \frac{\text{Exp}[P/2 - (P/2)(1 + 4RL\lambda_d/PV)^{1/2}]}{\text{Exp}(-RL\lambda_d/V)}$$

where

- η = Hung's correction factor, to compensate for the dispersion effect,
- R = retardation factor,
- P = Peclet number, $V_H D_H/d$,
- θ = dimensionless time, $\tau V/L$,
- N_d = decay number, $\lambda_d L/V$,
- L = flow length, D_v or D_H (m),
- V = water flow velocity, V_v or V_H (m/yr),
- t = time of simulation (yr),
- d = dispersion coefficient (m^2/yr),

- λ_d = radiological decay constant (yr^{-1}),
- τ = dummy time variable (yr),
- Q = rate of radionuclide transport at the point of interest, which is at well point in this case (Ci/yr),
- Q_o = rate of radionuclide released at the upstream reach, which is at the downstream edge of a disposal site (Ci/yr), and
- t_L = time a leachate control system is in use (yr).

In the above equation, the horizontal retardation factor, R_H , is calculated by

$$R_H = 1 + (\text{BDENS} \cdot \text{XKD4})/\text{PORA}$$

in which

- XKD4 = distribution coefficient of the aquifer (ml/g), and
- PORA = aquifer porosity (unitless).

Collection Reach

This analysis calculates the rate of radionuclide transport in the aquifer while receiving the radionuclide flux from the vertical reach. The primary purpose of the analysis is to calculate the rate of radionuclide transport at the downstream edge of the site boundary.

The basic equation used to calculate the rate of transport at the site boundary is expressed as:

$$Q(t) = B \int_0^L \eta(x) q \left(t - \frac{Rx}{V} \right) \cdot U \left(t - \frac{Rx}{V} \right) \cdot \text{Exp} \left[- \left(\frac{Rx}{V} \right) \lambda_d \right] \cdot dx \quad (3-30)$$

in which

- Q = rate of radionuclide transport at the downstream edge of the disposal site (Ci/yr),
- B = width of the disposal site measured in the direction perpendicular to the groundwater flow (m),
- L = length of the disposal site in the direction of groundwater flow (m),

- $\eta(x)$ = Hung's correction factor for the flow reach from the downstream edge of the disposal site to the point of integration,
- q = radionuclide flux entering the aquifer at the point of integration (Ci/yr/m²), and
- U = unit step function.

To simplify the calculation, Hung's correction factor, $\eta(x)$, is assumed to be equal to 1.0 in the actual model analysis. This approximation is acceptable because the length of the integration reach should not exceed the length of the disposal site, which is relatively small, and the $\eta(x)$ value is almost always 1.0 under normal application. Furthermore, the model assumes the segment of integration, dx or Δx , to be one-hundredth of the length of the disposal site in conducting the numerical integration.

3.2.2.9 Radionuclide Breakthrough Time

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. From a practical viewpoint, the breakthrough time is approximated in the model by assuming that the radionuclide leaching is from a point source and that the dispersion effect on the radionuclide transport can be neglected. The vertical and horizontal transit time, t_v (yr) and t_H (yr), are calculated according to:

$$t_v = \frac{D_v R_v}{V_v}, \quad t_H = \frac{D_H R_H}{V_H} \quad (3-31)$$

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), and
- V_H = water velocity in aquifer (m/yr),
- and retardation factors R_v and R_H are as previously defined.

3.2.2.10 Concentration in the Well Water

Since the well point receptor for the calculation of maximum annual committed effective dose is, in general, fairly close to the edge of the disposal site, the concentration of radionuclides in the well water may vary considerably with the depth of the well screen installed. In addition,

the well may pump in dilution water from the stream layer below the screen level. The thickness of this layer is a complicated function of the aquifer characteristics and the rate of pumping.

Theoretical concepts used in the development of the basic equation for the calculation of well water concentration is detailed by Hung (Hu99) and is attached as Appendix H.

The PRESTO-EPA model assumes that the well screen is installed near the bottom of the aquifer, which is the most reasonable assumption based on current well drilling practice and State well water regulation in the United States. Furthermore, the model also assumes that the well will withdraw water uniformly from the layer of stream between the water surface to the bottom of the well screen.

The depth of the screen is the user-assigned depth and the potential dilution water that may be withdrawn from the layer below the screen is considered negligible. The overall assumptions tend to overestimate the concentration of radionuclide and is considered to be a conservative approach.

To calculate the radionuclide concentration at the well point, the rate of groundwater flow at the well point is calculated first. By considering the lateral dispersion of the flow, the total rate of flow available for dilution is calculated by:

$$W_A = V_A P_A D_A [B + 2 \cdot \tan(\alpha/2) D_H] \quad (3-32)$$

where

- W_A = the rate of contaminated water flow in the plume at the well point (m^3/yr),
- V_A = groundwater velocity (m/yr),
- P_A = porosity of aquifer material (unitless),
- D_A = depth of well penetrating into the aquifer (m),
- α = constant angle of spread of the contaminant plume in the aquifer (radian),
- B = site width (m), and
- D_H = distance from the center of site to the well (m).

The angle " α " 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/m³), is calculated by:

$$C_w = Q/W_A \quad (3-33)$$

where Q is the rate of radionuclide transport at the well point.

3.2.2.11 Rate of Water Consumption

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

$$V_u = [3.9E-7 \cdot W_I f_I L_I + U_w L_H + 1.5E4 \cdot L_A] N_p \quad (3-34)$$

where

- V_u = annual well water demand in liters (l/person·yr),
- 3.9E7 = 4492 m² irrigated per person X 8760 hr/yr,
- W_I = irrigation rate per unit area (l/m²·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,
- 1.5E4 = 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), and
- 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 noncontaminated 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.

3.2.2.12 Surface Stream Contamination

As previously mentioned, water will overflow the site onto the soil surface when the maximum depth of standing water is greater than the contaminated zone depth. If this occurs, radionuclides will be added to the surface inventory. The surface soil will then have a component adsorbed by the soil with concentration C_{SS} (Ci/kg) and a component of contaminated water in the surface soil of C_{SW} (Ci/m³). The material adsorbed by the soil remains in the soil and becomes a source term for resuspension and atmospheric transport (discussed in Section 3.2.3). The contaminated water in the surface soil is available to enter nearby surface water bodies via overland flow, or to percolate down to the aquifer.

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

$$C_{SW} = \frac{1000 \cdot I_s}{K_{d1}M_s + M_{w2}/\rho_w} \quad (3-35)$$

where

C_{SW} = radionuclide concentration in surface soil water (Bq/m³),

I_s = amount of radionuclide on surface (Bq),

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_w = density of water (g/cm³), and

1000 = conversion factors used for K_d (1 ml/g = 1 m³/1000 kg) and for ρ_w (1 g/cm³ = 1000 kg/m³).

Equation (3-35) is used to compute the concentration of radionuclides in the surface soil interstitial water.

The radionuclide concentration in the contaminated surface soil region, C_{SS} , is calculated using:

$$C_{SS} = C_{SW}K_{d1}/1000 \quad (3-36)$$

The contaminated region of surface soil is defined by the area of the site and has a user-specified depth or thickness. These parameters allow the calculation of soil mass (M_S) and water mass (M_W) in the contaminated soil region by:

$$M_S = 1000 \cdot \rho_S S_w S_L S_D, \quad M_W = 1000 \cdot W_S S_w S_L S_D \quad (3-37)$$

where

- ρ_S = soil bulk density (g/cm^3),
- W_S = soil porosity (unitless),
- S_L = length of the site (m),
- S_W = width of the site (m),
- S_D = depth of the active layer (m), and
- 1000 = conversion factor for the mass of soil and water.

Water falling on the contaminated soil region may either evaporate, run off, or infiltrate. 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:

$$W_S = f_r P S_w S_L \quad (3-38)$$

where P is the annual precipitation.

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

$$W_D = W_a S_w S_L \quad (3-39)$$

where W_a is the yearly infiltration rate for the farmland.

The annual amount of radionuclides moving 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_{SW} (Equation (3-35)). The amount of each radionuclide annually entering the deeper soil layers from the contaminated surface soil region is the product of W_D and C_{SW} . 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 calculated by using:

$$I_r = [3.9E7 \cdot W_1 f_1 S_I + U_w S_H + 1.5E4 \cdot S_A] \cdot N_P \cdot C_{RW} \quad (3-40)$$

where

I_r = annual amount of nuclide removed from stream (Bq),

C_{RW} = radionuclide concentration in stream (Bq/m³),

S_I = fraction of irrigation water obtained from stream,

S_H = fraction of drinking water obtained from stream, and

S_A = fraction of cattle feed water obtained from stream.

Other parameters are the same as defined for Equation (3-34).

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, V_U , rather than the stream flow by:

$$C_{RW} = R_S / V_U \quad (3-41)$$

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.

3.2.3 Atmospheric Transport Sources and Pathways

For some sites, atmospheric transport of radionuclides may be a major transport mechanism. Therefore, careful consideration has been given to obtaining an accurate atmospheric transport model. On the other hand, one of the goals in developing the PRESTO-EPA-CPG/POP model was to minimize the complexity of input data. A compromise solution to achieve these conflicting goals was to employ a simplified and compact algorithm for the model. The model assumes that the population is concentrated in a single, small community, and allows the user to enter an externally computed population average value, the air concentration, X , to source strength, Q , ratio. An example of a code that could be used for determining this ratio, X/Q , is AIRDOS-EPA (Moo79).

In most cases, the uncertainties in the computed atmospheric source strength for contaminated areas are greater 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, 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 judgment in determining population centroid.

The atmospheric transport pathways and exposure pathways used in the model are illustrated in Figure 3-4. Contaminants are transported/deposited from the exposed source to the downstream population and, as a result, a member of the onsite or local population may be exposed to the suspended radionuclides through inhalation/immersion pathways, through ingestion of crops contaminated following deposition on soil or crops, and through direct irradiation from ground surfaces.

3.2.3.1 Internal Model Capability and Formulation

The atmospheric transport portion of the system is discussed in two parts: (1) a description of source strength computation and (2) a discussion of the calculation of atmospheric concentration at the residence site of the specified at-risk population. For most applications, the model 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 contaminated site. 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 parameter values for a particular, identified site.

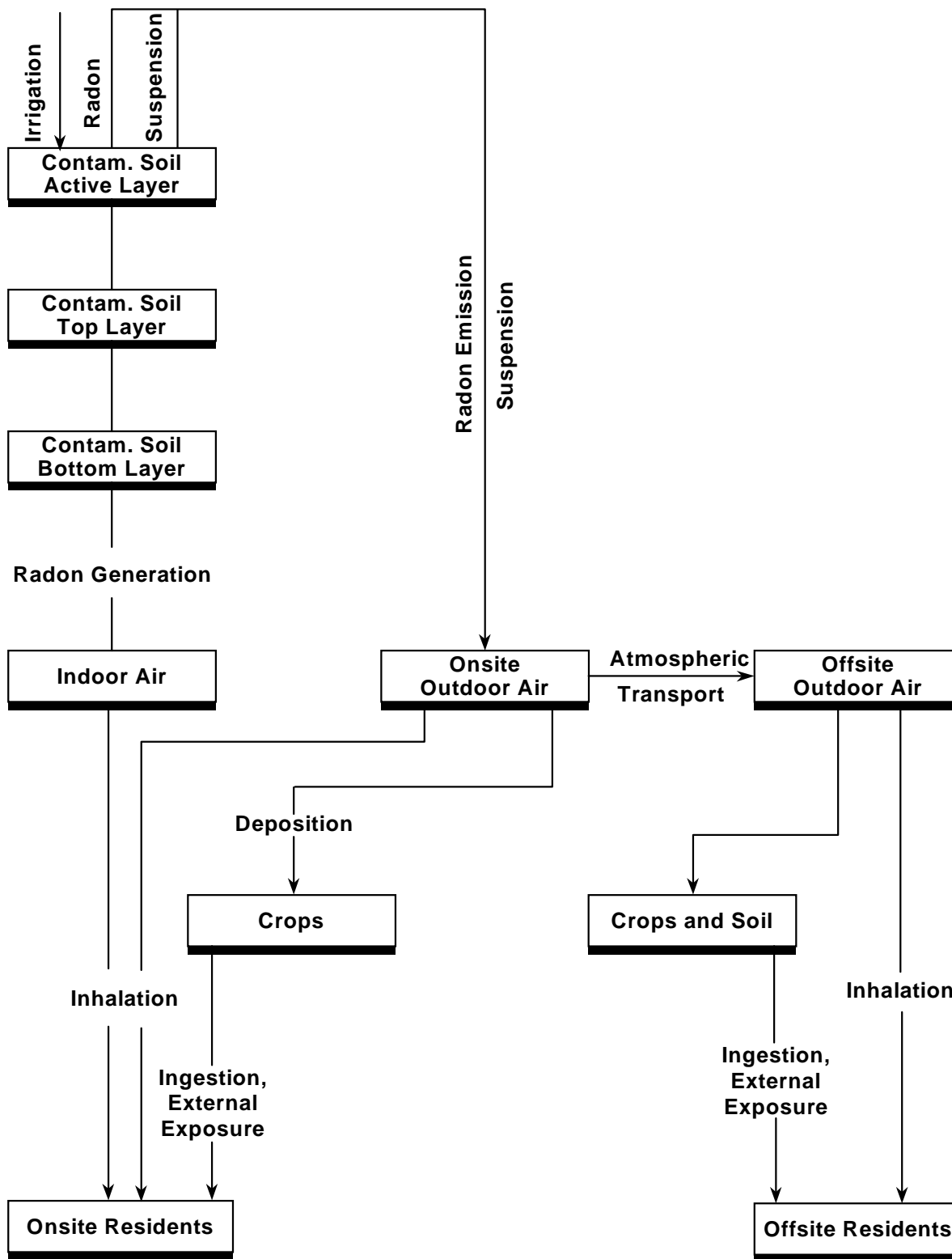
Where offsite population health effects are to be determined, the geometric population centroid specified by the user is the point for which a 22.5-degree sector average ground-level air concentration is determined. Where the population distribution subtends from the waste disposal area at an angle significantly greater than 22.5 degrees, the user should run the code separately for each subpopulation. A mean yearly value for the sector-averaged atmospheric concentration is computed by the system and is input to HESTAB for use in computing population health effects.

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 (S168). This approach is versatile and well documented. The model incorporates a Gaussian plume transport code called DWNWND (Fi80a) as a module, in subroutine form.

User inputs for the downwind atmospheric transport simulation allow specification of a surface radionuclide concentration at the waste site. Parameters used here include the initial surface radionuclide inventory and the chemical exchange coefficient for surface soils. The portion of radionuclides 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 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),
- 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), and
- x = distance from source to receptor (m).
- S = stability class,

Figure 3-4. Airborne Exposure Pathways in PRESTO Codes



3.2.3.2 Source Term Characterization

The release rate for atmospheric transport is termed the source strength. In the operation system, the source strength is directly dependent on the surface soil radionuclide concentrations from deposition and site overflow, C_G (Bq/m²). The source strength is the arithmetic sum of two parts: a time-dependent resuspension factor, R_e , (An75) and a resuspension rate, R_r , (He80).

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

$$R_e = R_{e1} \text{Exp}(R_{e2} T^{1/2}) + R_{e3} \quad (3-42)$$

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:

$$C_A = R_e C_G \quad (3-43)$$

and

$$C_G = 1000 \cdot C_s \rho_s S_D \quad (3-44)$$

where C_s is the radionuclide concentration of the soil and ρ_s and S_D are defined in Equation (3-37).

Using Anspaugh's values of 1E-4, -0.15, and 1E-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 for computing R_e .

The value of elapsed time appearing in Equation (3-42) 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 particulate. Therefore, it is assumed that a steady-state asymptotic value of R_e is appropriate for most sites appropriate for later additions to the surface inventory.

Users wishing to specify a time-independent windblown resuspension factor may do so by setting the values of R_{e1} and R_{e2} to 0. 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 Equation (3-43), C_A is the atmospheric concentration of radionuclides immediately above the site at a height of about one meter (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 model is the square root of the site area, A, and a tentative correction factor, F, which is computed using the equation:

$$F = [V_g(S_L S_W)^{1/2}/uH_D] \quad (3-45)$$

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

$$Q_r = C_A H_D u F A^{1/2} \quad (3-46)$$

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 the resuspension rate, R_r , having units of inverse seconds, as:

$$Q_{\text{mech}} = C_G A R_r F_{\text{mech}} \quad (3-47)$$

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

$$Q = Q_r + Q_{\text{mech}} \quad (3-48)$$

The airborne concentration is calculated as:

$$C_A = Q u \frac{A_T}{L} h \quad (3-49)$$

where

- C_A = airborne concentration (Bq/m³),
- Q = air pathway source term (Bq/s),
- u = wind speed (m/s),
- A_T = site area (m²),
- L = site length (m), and
- h = box model mixing height, set equal to 2 m.

3.2.3.3 Transport Formulation

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

$$X = \frac{Q}{2\pi u \sigma_y \sigma_z} \text{Exp}\left(-\frac{y^2}{2\sigma_y^2}\right) \left\{ \text{Exp}\left[-\frac{1}{2}\left(\frac{z+H}{\sigma_z}\right)^2\right] + \text{Exp}\left[-\frac{1}{2}\left(\frac{z-H}{\sigma_z}\right)^2\right] \right\} \quad (3-50)$$

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 ensured 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 (3-50) may be obtained from any of several reasonable conceptual transport and dispersion models.

Atmospheric transport at four regions in the United States of possible interest to individuals evaluating consequences of radionuclide transport has also been considered elsewhere. These sites include arid north (Fi81, Mi81), humid south (Fi80b), and humid north (Si66).

Implicit in Equation (3-50) 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 particulate in the effluent has 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:

$$H = H_{\text{eff}} - xV_g/u \quad (3-51)$$

for $H \geq 0$

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

3.2.3.4 Effects of a Stable Air Layer on Transport

The Gaussian plume formulation has been modified for use in the operation system 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:

$$\sigma_z(\text{limit}) = 2(H_L - H/2)/2.15 \quad (3-52)$$

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 $\sigma_z(\text{limit})$, the Pasquill-Gifford value of σ_z is used. For greater downwind distances, where σ_z is greater than or equal to $\sigma_z(\text{limit})$, the value of $\sigma_z(\text{limit})$ given in Equation (3-52) is used instead. The lid height is a user-specified value in the operation system. 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.

3.2.3.5 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 in 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:

$$W = V_d X, \quad (3-53)$$

where

$$X = \text{radionuclide concentration in air (Bq/m}^3\text{)}.$$

The magnitude of the plume depletion within the downwind sector may be found by integrating the deposition across the entire plume. Using Equation (3-50) and setting $z = 0$, it is found that:

$$\begin{aligned} \frac{dQ}{dx} &= \int_{-\infty}^{\infty} V_d x dy \\ &= \int_{-\infty}^{\infty} \left(\frac{V_d Q}{u\pi\sigma_y\sigma_z} \right) \text{Exp} \left[-\left(\frac{y^2}{2\sigma_y^2} \right) - \left(\frac{H^2}{2\sigma_z^2} \right) \right] dx \end{aligned} \quad (3-54)$$

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:

$$F_d = \frac{X_d}{X} = \text{Exp} \left\{ - \left(\frac{2}{\pi} \right)^{1/2} \frac{V_d}{u} \int_0^x \left(\frac{1}{\sigma_z} \right) \text{Exp} \left[\frac{-H^2}{2\sigma_z^2} \right] dx \right\} \quad (3-55)$$

Since σ_z is a complicated empirical function of x , Equation (3-55) must be evaluated numerically.

In the PRESTO-EPA model applications, the average value of radionuclide concentration X across a 22.5-degree downwind sector is the desired quantity. In this case, the transsector integration leads to the value 2.032 in the air concentration equation (Cu76). This value includes the $1/2\pi$ factor in Equation (3-50).

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:

$$X = (2.032 F_d F_w Q / u x \sigma_z) \text{Exp}[-H^2/2\sigma_z^2] \quad (3-56)$$

The value of H in Equation (3-56) must be an effective source height. This value is corrected in the model for plume tilt as in Equation (3-51) and the accompanying discussion. In the code, H is on the order of 1 m for reasonably flat sites but, in many other cases, different values should be used to account for local site characteristics, e.g., the presence of updrafts.

It has been noted that the choice of plume dispersion parameter σ_z is a user option in the system. Choice of appropriate parameterization depends on site meteorology, topography, and release conditions. The DWNWIND code (Fi80a), which has been included as part of the model, 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 the assessment of long-term performance of LLW disposal sites. 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 distances to population (<1 km). The so-called Pasquill-Gifford form of this parameterization (Hi62) has been tabulated by Culkowski and Patterson (Cu76), and is used in this model.

3.2.4 Food Chain Calculations

Mean concentrations of radionuclides in stream water, well water, and air are calculated by using the equations listed in Sections 3.2.2 and 3.2.3. This section describes how radionuclides in those 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 or watering cattle – or from fish from a contaminated stream. 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 DOSTAB and HESTAB submodels, which are modified from the DARTAB program (Be81). Radionuclide-related input consists of the constant concentrations in air (person-Bq/m³), constant concentrations on ground surface (person-Bq/m³), constant collective ingestion rate (person-Bq/yr), and constant collective inhalation rate (person-Bq/yr). Calculation of these variables follows.

Concentrations of radionuclides in air that affect the population or an individual are calculated as described in Section 3.2.3. It is assumed that the mean nuclide concentrations in air are constant during the total period of the simulation, as required, for input to HESTAB. Similarly, it is assumed that the maximum nuclide concentration in air is input to DOSTAB.

Concentration of each radionuclide on the ground surface, Q_s (Bq/m²), is calculated using:

$$Q_s = \text{CSPA} + \text{CSPI} \quad (3-57)$$

where

- Q_s = radionuclide concentration on the ground surface at the populated area of interest (pBq/m²),
- CSPA = radionuclide concentration in the soil used for farming due to atmospheric deposition (pBq/m²), and
- CSPI = radionuclide concentration in the soil used for farming due to irrigation (pBq/m²).

Appropriate unit conversions are made within the code.

The inhalation rate of radionuclides is calculated by multiplying the generic individual inhalation rate by the concentration of radionuclides:

$$Q_{inh} = U_a C_A \quad (3-58)$$

where

- Q_{inh} = rate of inhalation exposure (BQ/yr),
- U_a = inhalation rate (m³/yr), and
- C_A = mean ground-level radionuclide concentration at a point of interest (Bq/m³).

The units of Q_{inh} are converted to person·Bq/yr by the population size for input to the HESTAB subroutine.

The ingestion rate is the input to DOSTAB and HESTAB that requires the most calculation. Ingestion includes intake of drinking water, beef, milk, crops, fish, and soil. Except for drinking water and stream water, all of these media may be contaminated by either atmospheric processes or irrigation.

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

$$d = 3.6E15 \cdot C_A V_d \quad (3-59)$$

where

- d = mean rate of radionuclide deposition onto ground or plant surfaces (pBq/m²·hr),
- C_A = mean ground-level radionuclide concentration at the point of interest (Bq/m³),
- 3.6E15 = conversion factor, sec-pBq/hr-Bq, and
- V_d = deposition velocity (m/sec).

The following equation estimates the concentration, C_v , of a given nuclide in and on vegetation at the deposited location (except for H-3 and C-14):

$$C_v = [d \cdot R[1 - \text{Exp}(-\lambda_e t_e)] / (Y_v \lambda_e) + (B \cdot CS/P) \text{Exp}(-\lambda_d t_h)] \cdot RT \quad (3-60)$$

where

- C_v = radionuclide concentration in pBq/kg,

d	=	mean rate of radionuclide deposition onto ground or plant surfaces (pBq/m ² ·hr),
R	=	fraction of deposited activity retained on crops (unitless),
λ_e	=	effective removal rate constant for the radionuclide from crops (hr ⁻¹), which is the sum of the radioactive decay constant and the removal rate constant from weathering, λ_w ,
t_e	=	time period that crops are exposed to contamination during the growing season (hr),
Y_v	=	agricultural productivity or yield [kg (wet weight)/m ²],
B	=	radionuclide concentration factor for uptake from soil by edible parts of crops [pCi/kg (dry weight) per pCi/kg dry soil],
CS	=	soil radionuclide concentration updated yearly (Bq/kg),
P	=	effective surface density for topsoil [kg(dry soil)/m ²],
t_h	=	time interval between harvest and consumption of the food (hr), and
RT	=	root density correction factor.

One of the exposure pathways in the model is the absorption of radionuclides by plant roots. Radioactivity is absorbed by the plants, which are later consumed by humans and animals. Previously, the root density in the PRESTO models was assumed to be constant with depth. A more realistic model has now been implemented that accounts for varying root density with depth. The plant root density distribution is given by:

$$d = \alpha^2 \beta y \exp(-\alpha y) \quad (3-61)$$

where

d	=	root density distribution (dimensionless),
α	=	root density constant (m ⁻¹),
β	=	scaling factor (m), and
y	=	depth (m).

The variable root density is implemented in PRESTO as a correction to the previous constant root density model. The correction factor is the following.

$$RT = \frac{c_1 e_1 t_1 + c_2 e_2 t_2 + c_3 e_3 t_3}{c_1 (e_1 t_1 + e_2 t_2 + e_3 t_3)} \quad (3-62)$$

where

c = nuclide concentration in soil layer,

t = thickness of layer (m), and

subscripts 1, 2, and 3 denote active layer, top layer, and bottom layer, respectively.

Using the same convention, variables e_1 , e_2 , and e_3 denote:

$$e_1 = \int_{t_1}^{t_1} y \exp(-\alpha y) dy,$$

$$e_2 = \int_{t_1}^{t_1+t_2} y \exp(-\alpha y) dy, \text{ and}$$

$$e_3 = \int_{t_1+t_2}^{t_1+t_2+t_3} y \exp(-\alpha y) dy$$

All vegetative concentrations (vegetables, produce, pasture grass, and animal feed) are multiplied by the above correction factor to account for the nonuniform root distribution. In Equation (3-60), the value of CS is calculated by:

$$CS = (CSL + d \cdot \Delta t S_D) \text{Exp}[-(\lambda_d + \lambda_s) \Delta t] \quad (3-63)$$

where

CS = soil radionuclide concentration for this year (Bq/kg),

CSL = soil radionuclide concentration for last year (Bq/kg),

d = mean rate of radionuclide deposition ($\text{pBq/m}^2\text{-yr}$),

λ_d = radioactive decay constant (yr^{-1}),

λ_s = rate constant for contaminant removal (yr^{-1}),

Δt = time increment, equal to one year in PRESTO model, and

S_D = depth of active soil layer (suggested value is 0.1 meter).

The total soil concentration on the site where farming occurs is calculated as:

$$\text{SOCON} = 1\text{E}12 \cdot S_D(C_{\text{SW}}W_s + 1000 \cdot C_{\text{SS}}\rho_s) \quad (3-64)$$

where

- SOCON = soil radionuclide concentration (pBq/m²),
- S_D = depth of contaminated surface region (m),
- C_{SW} = radionuclide concentration in interstitial water of contaminated surface region (Bq/m³),
- W_s = porosity of surface soil (unitless),
- C_{SS} = radionuclide concentration in soil of contaminated surface region (Bq/kg),
- ρ_s = bulk density of surface soil (g/cm³),
- 1E12 = pBq/Bq, and
- 1000 = (kg/g)·(cm³/m³).

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

$$\lambda_s = \frac{r_s}{S_D(8760)\{1 + \rho_s K_d / W_s\}} \quad (3-65)$$

where

- λ_s = removal rate coefficient (hr⁻¹),
- r_s = watershed infiltration (m/yr),
- ρ_s = soil bulk density (g/cm³),
- K_d = distribution coefficient (ml/g),
- W_s = porosity (unitless), and
- 8760 = hr/yr.

Equation (3-60) 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:

$$C_f = f_p f_s C_p + (1 - f_p f_s) C_s \quad (3-66)$$

where

- C_f = the radionuclide concentration in the animals' feed (pBq/kg),
- C_p = the radionuclide concentration on pasture grass (pBq/kg) calculated using Equation (3-60) with $t_h = 0$,
- C_s = the radionuclide concentration in stored feeds in pBq/kg, calculated using Equation (3-60) with $t_h = 2160$ hr or 90 days,
- f_p = the fraction of the year that animals graze on pasture (unitless), and
- 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:

$$C_m = F_m C_f Q_f \cdot \text{Exp}(-\lambda_d t_f) \quad (3-67)$$

where

- C_m = the radionuclide concentration per liter in milk (pBq/l),
- C_f = the radionuclide concentration in the animal's feed (pBq/kg),
- F_m = the average fraction of the animal's daily intake of a given radionuclide that 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), and
- λ_d = 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:

$$C_f = F_f C_f Q_f \cdot \text{Exp}(-\lambda_d t_s) \quad (3-68)$$

where

- C_f = the nuclide concentration in animal flesh (pBq/kg),
 F_f = the fraction of the animal's daily intake of a given radionuclide that appears in each kilogram of flesh (d/kg),
 C_f = the concentration of radionuclides in the animal's feed (pBq/kg),
 Q_f = the amount of feed consumed by the animal per day (kg/d), and
 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, C_v , is estimated using Equation (3-60), but a different value of the retention fraction, R , is used. For irrigation, the second term of Equation (3-60) is modified by a factor of f_i , the 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 (3-60) becomes the irrigation rate, I_r , expressed as:

$$I_r = C_w W_I \quad (3-69)$$

where

- I_r = radionuclide application rate (pCi/m² hr),
 C_w = radionuclide concentration in irrigation water (pBq/l), and
 W_I = irrigation rate (l/m²-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 the animal's intake of water was added to Equations (3-67) and (3-68), respectively. This becomes:

$$C_m = F_m(C_f Q_f + C_w Q_w) \text{Exp}(-\lambda_d t_f) \quad (3-70)$$

$$C_F = F_f(C_f Q_f + C_w Q_w) \text{Exp}(-\lambda_d t_s) \quad (3-71)$$

where

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

Radionuclides ingested through the direct inadvertent ingestion of soil and the consumption of fish are also included in the model.

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

$$Q_{\text{ing}} = Q_v + Q_{\text{milk}} + Q_{\text{meat}} + Q_{\text{wat}} + Q_{\text{soil}} + Q_{\text{fish}} \quad (3-72)$$

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

$$Q_v = (C_{vI} + C_{vA})U_v \quad (3-73)$$

where

Q_v = annual radionuclide intake from vegetation (pBq/yr),

C_{vI} = radionuclide concentration in vegetation from irrigation (pBq/kg),

C_{vA} = radionuclide concentration in vegetation from atmospheric deposition (pBq/kg), and

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

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

As mentioned earlier, Equations (3-59) through (3-71) 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 transfer factor for the concentration in all vegetation, C_v , from the tritium concentration in the irrigation water is 1 when the units of C_v and C_w are in pBq/kg and pBq/l, respectively. Thus:

$$C_v = 1 \cdot C_w \quad (3-74)$$

In the same manner, the concentration of H-3 in an animal's feed, C_f , is also equal to C_w . Then, from Equations (3-70) and (3-71), the concentration of tritium in an animal's milk and meat can be written as:

$$C_m = F_m C_w (Q_f + Q_w) \quad (3-75)$$

$$C_F = F_f C_w (Q_f + Q_w) \quad (3-76)$$

where

- C_m = concentration of tritium in milk (pBq/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 (pBq/l),
- Q_f = animal's daily intake of forage (kg/d),
- Q_w = animal's daily intake of water (l/d),
- C_F = concentration of tritium in animal meat (pBq/kg), and
- 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:

$$C_v = (C_A/h)(0.75)(0.5)(1E15) \quad (3-77)$$

where

- C_v = tritium concentration in vegetation (pBq/kg),
- C_A = concentration of H-3 in air (Bq/m³),
- h = absolute humidity of the atmosphere (g/m³),
- 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, and
- 1E15 = (1E12 pBq/Bq)x(1000 g/kg).

The mean ground-level air concentration of H-3, C_A , is calculated using the equations in Section 3.2.3.

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

$$C_v = (C_A/0.16)(0.11)(1E15) \quad (3-78)$$

where

- C_v = C-14 concentration in vegetation (pBq/kg),
- C_A = mean ground-level concentration of C-14 in air (Bq/m³), also calculated from equations given in Section 3.1.3,
- r = ratio of the total release time of C-14 to the total annual time during which photosynthesis occurs, $r \leq 1$,
- .11 = fraction of the plant mass that is natural carbon,
- 0.16 = concentration of natural carbon in the atmosphere (g/m³), and
- 1E15 = (1E12 pBq/Bq)x(1000 g/kg).

3.2.5 External Radiation Risk to Residents

The methodology used in the calculation of external radiation risk to residents, including outdoor exposure and basement exposure, is detailed in Appendix F. The external exposures calculated for the outdoors and for the basement floor are adjusted for the finite area size. The methodology for calculating the area adjustment factor (designated as area factor for simplicity) is detailed in Appendix G.

3.2.5.1 Area Factor

For the purpose of calculating the area factor, it is assumed that (1) the multi-layer soil contamination considered in the PRESTO model can be converted to an equivalent surface contamination, (2) no radioactive decay takes place for the contaminants, (3) the contamination is homogeneous, and (4) the energy absorption buildup effect can be ignored. Based on these assumptions, the following basic equation is derived (see Appendix G for derivation) and used in the model:

$$F_a = \frac{\int_0^{L/2} \int_0^{B/2} (1/r^2) \text{Exp}(-\mu_a r) dx dy}{\int_0^\infty \int_0^\infty (1/r^2) \text{Exp}(-\mu_a r) dx dy} \quad (3-79)$$

where

F_a	=	area factor,
μ_a	=	linear attenuation coefficient of air,
r	=	distance from the receptor to the area segment of interest and is calculated by the square root of $x^2 + y^2 + 1$,
L	=	length of the contaminated site, and
B	=	width of the contaminated site.

Since the linear attenuation coefficient is a function of the photon energy level and the level of photon energy is radionuclide-dependent, the area factor should also be radionuclide-dependent. In order to simplify the procedures of calculation and yet to maintain accuracy, a preliminary study is conducted to identify the variation of area factor with radionuclide. The results of the analysis indicate that the range of variation for a series of square sites is relatively small within the range of practical application (see Appendix G).

The model uses U-234, which gives the upper-bound value of area factor as the representative radionuclide and uses the same area factor value for all other radionuclides.

3.2.5.2 Radiation Risk from Outdoor Ground Surface

The model calculates the gamma exposure from contamination outdoors. The sources of contamination include direct application, irrigation, atmospheric deposition, water overflow from the waste, and erosion of the waste in addition to the preexisting contaminants.

The outdoor gamma exposure is the sum of exposure imposed from the three layers of contamination considered in the model: an active surface layer, an upper layer immediately below the active layer, and a lower layer of contamination. Radiation from each layer is considered to be shielded by its own layer of soil and by any layers above it. The equations used in the calculations are given below.

The calculation of the gamma exposure takes the short-cut methodology recommended in the Federal Guidance Report No. 12 (EPA93). The dose coefficient for any depth of contamination is calculated based on the approximation method derived in Appendix F, from which the dose from the active layer is calculated by:

$$DS_a = CS_a \cdot BDENS \cdot 1000 \cdot S_{oc} \cdot F_a \cdot DCF_a \quad (3-80)$$

where

DS_a	=	dose from active layer (Sv/s),
--------	---	--------------------------------

- CS_a = nuclide concentration in active layer (Bq/kg),
 $BDENS$ = soil density in active layer (g/cm³),
 1000 = unit conversion (cm³/m³ and kg/g),
 S_{oc} = outdoor site occupancy factor (unitless),
 S_{ar} = area factor calculated based on the site area (Appendix G) (unitless), and
 DCF_a = dose conversion factor for the active layer (Sv/s per Bq/m³).

The dose conversion factor for the active layer, which has user-selected thickness, is calculated by:

$$DCF_a = DCF_{inf}(1.-Exp(-K_A \cdot D_a)) \quad (3-81)$$

and

$$K_A = -\left(\frac{1}{0.15}\right) \text{Ln}\left(1.-\frac{DCF_{15}}{DCF_{inf}}\right) \quad (3-82)$$

In above equations:

- DCF_{inf} = dose conversion factor for an infinitely thick contaminated layer (Sv/s per Bq/m³),
 μ = gamma attenuation coefficient for soil (m⁻¹),
 D_a = thickness of active layer, and
 DCF_{15} = dose conversion factor for a 15-cm contaminated layer (Sv/s per Bq/m³).

The dose from the top layer immediately beneath the active layer, the nominal top layer, is slightly more complicated because it has additional shielding from the active layer. The gamma exposure for this layer is calculated from the net difference between the exposure from the combined soil layer (active layer and the nominal top layer) and from the active layer. Mathematically, it is expressed by:

$$DS_{nu} = CS_{nu} \cdot BDENS \cdot 1000 \cdot S_{oc} \cdot F_a(DCF_{ul} - DCF_a) \quad (3-83)$$

where

- DS_{nu} = dose from nominal top layer (Sv/s),

- CS_{nu} = nuclide concentration in nominal top layer (Bq/kg),
 S_{oc} = outdoor site occupancy factor, and
 DCF_{ul} = dose conversion factor for the combined layer, or the active plus the nominal top layers (Sv/s per Bq/m³).

In Equation (3-83), the dose conversion factor for the combined soil layer is calculated from Equation (3-81), except the thickness of the soil layer is replaced by the thickness of the combined soil layer, that is, the entire top layer.

In the same manner, the dose due to the bottom layer is calculated by applying Equations (3-81) and (3-82) to calculate the dose conversion factor for the combined soil layer and the bottom layer. Finally, the dose due to the bottom layer is calculated by the dose equation expressed in:

$$DS_{Bll} = CS_{bl} \cdot BDENS \cdot 1000 \cdot S_{oc} \cdot F_a(DCF_{cl} - DCF_{nu}) \quad (3-84)$$

where

- DS_{bl} = dose from bottom layer (Sv/s),
 CS_{bl} = nuclide concentration in the bottom layer (Bq/kg), and
 DCF_{cl} = dose conversion factor for the thickness of the combined layer, including the entire upper layer and the lower layer (Sv/s per Bq/m³).

Details of the theoretical background and their derivation of equations are stated in Appendix F.

3.2.5.3 Basement Risk to Residents

The calculation of the basement gamma exposures to an onsite resident is a complex process. Since it is not, normally, a dominated pathway, some crude assumptions are needed to simplify the calculation. These assumptions include the use of a fixed basement geometry and construction materials so that the complex volume integral can be replaced with a simple coefficient.

The calculations are conducted for two components, the exposure from the side-wall layer and from the floor layer. The theoretical background and its derivation of the basic equations are described in Appendix F.

Side-Wall Layer

In order to simplify the processes of modeling, it is assumed that the basement has a fixed geometry having 2-meter high and 3-meter radius and is constructed from a concrete with wall thickness of 0.1 meter. In addition, if it is also assumed that the side-wall soil layer is contaminated with unit concentration of a specific radionuclide over the entire depth of side-wall, then the dose coefficient for an individual living in the basement for 24 hours a day can be calculated for this specific basic model as:

$$DC_{SWL} = 2\pi \int_0^{R+d} \int_0^H \frac{B(\mu_{SWL}, \rho_{SWL})}{\rho^2} \text{EXP}[-(\mu_{SWL}\rho_{SWL} + \mu_{CON}\rho_{CON} + \mu_{AIR}\rho_{AIR})] dzdr \quad (3-85)$$

where

- DC_{SWL} = dose coefficient for the side-wall layer,
- B = radionuclide buildup factor,
- μ = linear attenuation coefficient,
- ρ = length of the path in a medium measured in the direction connecting the source, dv, and the receptor, and the subscripts,
- SWL = side-wall layer,
- CON = concrete material, and
- AIR = air.

Comparing the ratio of the side-wall dose coefficient to the dose coefficient for the contamination of the same radionuclide to 0.15 meter indicates that the ratios vary with the radionuclide or its photon energy level. However, since the range of variation is small, a fixed conservative value, designated as the side-wall factor, can be chosen for the modeling.

The dose to an individual living in the basement is calculated by:

$$DS_{SWL} = \eta \cdot DC_{15} \cdot CS_{SWL} \cdot BDENS \cdot 1000 \cdot S_{oc} \cdot (D/H) \quad (3-86)$$

where

- DS_{SWL} = dose due to the side-wall layer (Sv/S),
- η = side-wall factor defined as the ratio of the dose to the individual living in the basic model basement to the dose coefficient for a surface soil contaminated to depth of 0.15 meter,

- CS_{SWL} = nuclide concentration in the equivalent side-wall layer, weighted average concentration within active layer, nominal top layer, and bottom layer (Bq/kg),
- D = total depth of the equivalent side-wall layer, including the active layer, nominal top layer, and the bottom layer (m), and
- H = depth of the basement, fixed to 2 meters.

Floor Layer

The calculation of the component from the floor layer is conducted using a similar approach as that employed in the calculation of the outdoor external dose rate. The dose coefficient for the contaminated soil layer beneath the basement floor (designated as the floor layer for simplicity) is calculated based on the assumption that the floor extends to the entire area of contamination and has a fixed concrete floor thickness of 0.1 meter. The dose to the individual living in the basement is then calculated by:

$$DS_{BFL} = CS_{BFL} \cdot DC_{BFL} \cdot BDENS \cdot 1000 \cdot S_{oc} \cdot F_A \cdot EXP(-\mu_{CNF} \cdot \rho_{CNF}) \quad (3-87)$$

where

- DS_{BFL} = dose to the individual due to the basement floor layer (Sv/S),
- ρ_{CNF} = overall equivalent thickness of the concrete floor measured along the path of gamma ray, which is conservatively assumed to be 0.1 meter,

DC_{BFL} = dose coefficient for the basement floor layer, which is calculated by Equations (3-81) and (3-82) for the actual contaminated soil layer thickness, and

μ_{CNF} = radionuclide-specific concrete attenuation coefficient.

3.2.6 Radiation Risk From Radon Inhalation

The model simulates the radon gas pathway to calculate doses from indoor and outdoor radon exposure. The calculations are activated automatically when radon decay-chain parents are included in the source term. A radon emanation fraction of zero can be used to ignore radon effects.

Indoor and outdoor radon exposures are calculated for a hypothetical house that is constructed on a contaminated soil site. Indoor radon enters the house through the basement floor. Outdoor radon is generated in the contaminated soil and released to the atmosphere.

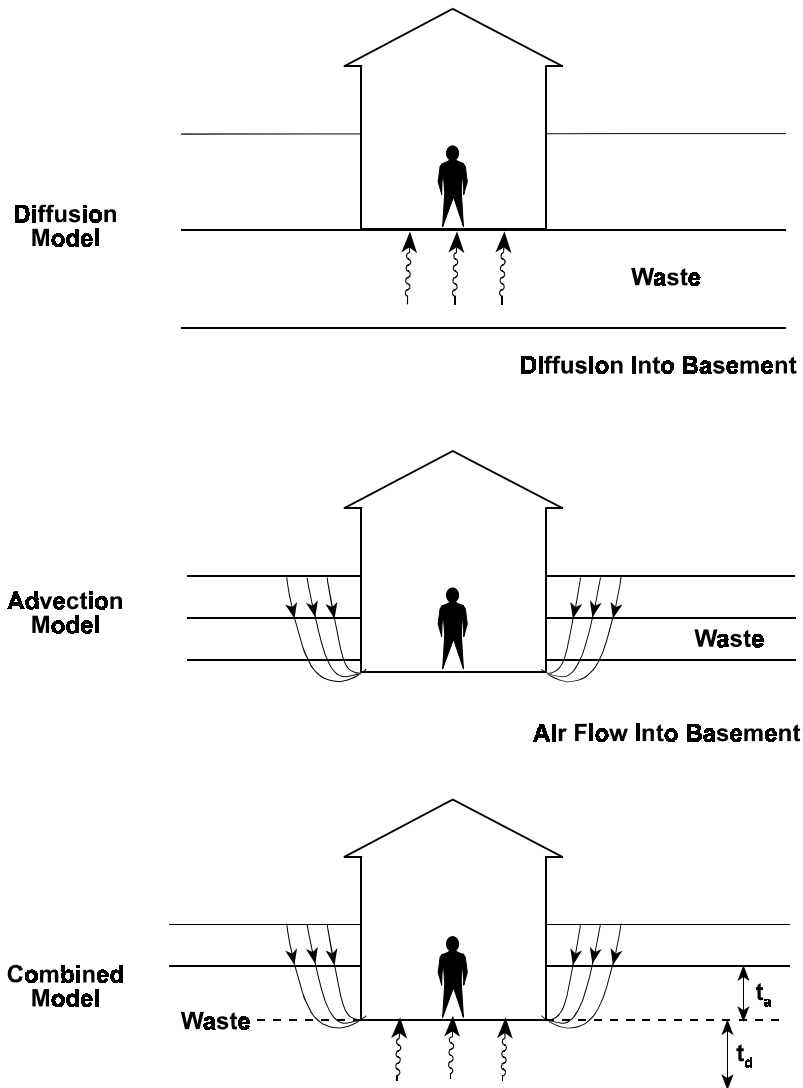
There are two main processes by which radon can be transported in soil: diffusion and advection. Depending on the conditions, either of these processes may dominate the dose.

For conditions where all of the contaminated soil is lower than the basement floor elevation, only the diffusion model is used. For cases where all of the waste is higher than the basement floor, the advection model is used (see Figure 3-5). In some cases, where the basement extends only partway through the waste, both models are used. The diffusion model applies to the portion of the waste below the basement floor and the advection model applies to the rest.

In the operation system, the advection model is supplemented with the diffusion model to account for radon that is generated directly below the house. Together, the advection and diffusion models calculate the total radon concentration in the basement.

The radon calculations take into account up to three layers of contamination: the bottom layer, the nominal top layer (the top layer less the active layer), and the active layer (the uppermost portion of the top layer). The three contaminated layers may have different radium concentrations and generate different amounts of radon gas. The model accounts for the radon gas contribution from all three contaminated layers.

Figure 3-5. Diffusive and Advective Radon Models



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3.2.6.1 Diffusion Model

Diffusive radon gas transport is driven by the radon concentration gradient and is the dominant transport process when there is little or no movement of the soil gas. The radon concentration gradient is maintained by the continuous generation of radon gas in the soil.

The diffusion transport model is appropriate for calculating the radon flux into a basement excavated in soil containing radium. The diffusion model also calculates the radon

flux from contaminated soil directly to the outdoor atmosphere. The equation used in the model is cited from NRC84 and shown below:

$$J_d = \frac{2R\rho E\sqrt{\lambda D_w} \tanh\left(\sqrt{\frac{\lambda}{D_w}} t_d\right) \text{Exp}\left(-\sqrt{\frac{\lambda}{D_c}} t_c\right)}{\left[1 + \frac{P_w}{P_c} \sqrt{\frac{D_w}{D_c}} \tanh\left(\sqrt{\frac{\lambda}{D_w}} t_d\right)\right] + \left[1 - \frac{P_w}{P_c} \sqrt{\frac{D_w}{D_c}} \tanh\left(\sqrt{\frac{\lambda}{D_w}} t_d\right) \text{Exp}\left(-2\sqrt{\frac{\lambda}{D_c}} t_c\right)\right]} \quad (3-88)$$

where

- J_d = diffusive radon flux (Bq/m²-s),
- t_d = thickness of waste below basement floor (m),
- R = radium content of waste (Bq/kg),
- ρ = waste density (kg/m³),
- E = radon emanation coefficient (dimensionless),
- λ = radon decay constant (s⁻¹),
- D_w = radon diffusion coefficient of waste (m²/s),
- D_c = radon diffusion coefficient of concrete basement floor (m²/s),
- P_w = porosity of waste (dimensionless), and
- P_c = porosity of concrete (dimensionless).

3.2.6.2 Advection Model

The diffusion model applies anywhere there is no advective movement of soil gas. However, for sandy soils and high negative pressure differentials, radon transport may be dominated by advective flow through joints and cracks in a house's foundation.

Typically, a significant portion of the radon that enters a house is due to a negative pressurization of the house's interior. This causes a flow of soil gas into the house through joints and cracks in the foundation and basement floor. An analytical expression for the advective flow is discussed and illustrated in Mo88. The negative pressure in the house causes a flow of soil gas from the ground surface near the house, downward and into the basement through shrinkage cracks, usually around the perimeter of the basement floor.

The air that is drawn into the basement comes from the atmosphere outside the house. It is assumed that the radon concentration in the outdoor atmosphere is zero, or at least very small. However, as the atmospheric air is drawn downward through the soil, it picks up and accumulates radon that is always being produced in the soil. The longer it takes the air to travel from the surface to the basement crack, the more radon it accumulates. For very long air travel times, the radon concentration approaches a limiting value determined by the radium content of the soil. Alternatively, for very rapid air flow, the radon entry rate into the house is limited by the available time for the radon gas to mix into the airflow.

The pressure-induced advective flow occurs only in a region near the house. As indicated in Mo88, the advective flow region extends outward from the foundation about the same distance as the depth of the basement floor. Mo88 also indicates that the advectively transported radon comes from the soil at the sides of the house. Thus, the advective model is only concerned with radon that is generated at the sides of the house within a few meters of the foundation wall.

The equations for the advective radon transport model are shown below. The expression for the volumetric air flow rate into the basement is (Mo88, equation 20):

$$Q = \frac{L P_f}{\mu} \left[\frac{C_f L_s}{12 w^3} + \frac{\cosh^{-1}\left(\frac{2Z}{w}\right)}{\pi k} \right]^{-1} \quad (3-89)$$

where

- Q = soil gas flow rate into basement (m³/s),
- L = perimeter crack length (m),
- P_f = negative pressure inside house (Pa),
- μ = soil gas viscosity (Pa-s),
- C_f = 1, as per Mo88,
- L_s = thickness of concrete basement floor (m),
- w = shrinkage crack width (m),
- z = depth of basement floor below ground (m), and
- k = soil intrinsic permeability (m²).

The soil intrinsic permeability in Equation (3-89) is related to its hydraulic conductivity as follows:

$$k = \frac{(K/3.15E+7) \mu_w}{\rho_w g} \quad (3-90)$$

where

- ρ_w = density of water (1000 kg/m³),
- μ_w = viscosity of water (0.001 Pa-s),
- g = acceleration of gravity (9.8 m/s²),
- K = hydraulic conductivity (m/yr), and
- 3.15E+7 = seconds per year.

Once the volume of soil gas flowing into the basement has been determined from Equation (3-89), the radon concentration in the gas can be calculated. The radon concentration in the pore space of contaminated soil builds up to an equilibrium value according to:

$$C_r = \frac{R\rho E}{p} [1 - e^{-\lambda t}] \quad (3-91)$$

where

- C_r = radon concentration in soil gas (Bq/m³),
- R = radium content (Bq/kg),
- ρ = soil dry bulk density (kg/m³),
- E = radon emanation fraction (dimensionless),
- λ = radon-222 decay constant (s⁻¹),
- p = waste porosity (dimensionless), and
- t = residence time of air in contaminated soil(s).

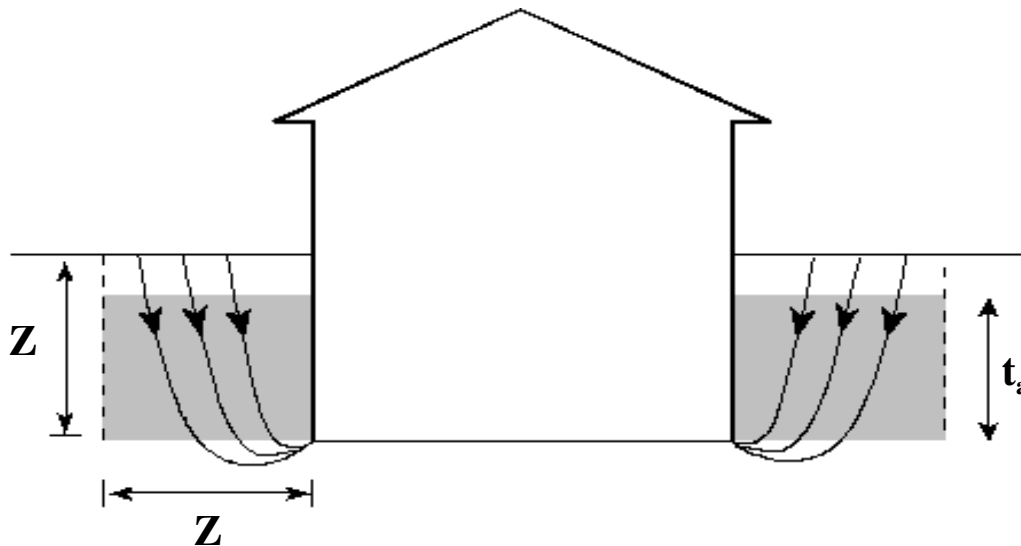
The residence time can be estimated by dividing the advective air flow rate into the pore space volume through which the air flows.

$$t = \frac{V}{Q} = \frac{Lz^2 p t_a}{Qz} \quad (3-92)$$

where

- V = volume of advective flow region in waste (m³),
 Q = soil gas flow rate into basement (m³/s),
 L = basement perimeter (m),
 z = depth of basement floor below ground (m), and
 p = porosity of soil around foundation.

Figure 3-6. Advective Flow Region Around Basement



The volume flow rate, Q , is calculated from Equation (3-89). The volume of the advective flow region, V , is based on a volume of soil adjacent to the house foundation, as shown in Figure 3-6. The air volume in the soil is approximately Lz^2p , where L is the basement perimeter, z is the depth of the basement floor, and p is the porosity of the soil around the foundation. This volume is then modified by the ratio of the waste thickness above the basement floor divided by the depth of the basement floor below the ground surface (t_a/z). This accounts for the fraction of the total depth that is contaminated.

The advective flow region is assumed to extend vertically from the ground surface to the depth of the basement floor. Also, it is assumed to extend an equal distance horizontally from the foundation wall and to extend around the entire perimeter of the house. While it is not possible to precisely define the advective flow region, these assumptions are sufficient to capture the main parameter dependencies and permit a reasonable estimation of the radon entry rate into the house.

The advective radon entry rate into the house is given by:

$$J_a = QC_r \quad (3-93)$$

where

- J_a = radon entry rate due to advection (Bq/s),
- Q = defined by Equations (3-89) and (3-90) (m^3/s),
- C_r = defined by Equations (3-91) and (3-92) (Bq/m^3).

3.2.6.3 Indoor Radon Concentration

The total radon concentration in the basement can now be evaluated using the diffusive and advective radon entry rates and an assumed ventilation rate of the house. The indoor radon concentration is given by:

$$C = \frac{J_d A_b + J_a}{\lambda_a A_b h} \quad (3-94)$$

where

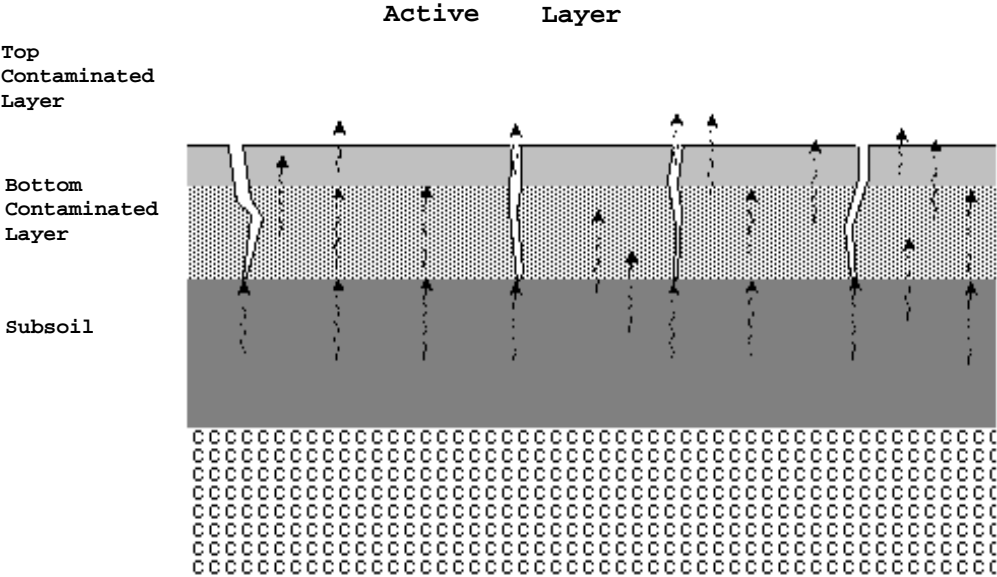
- C = radon concentration in house (Bq/m^3),
- J_d = diffusive radon flux into basement ($Bq/m^2 \cdot s$),
- A_b = area of basement floor (m^2),
- J_a = advective radon entry rate into the basement (Bq/s),
- λ_a = basement ventilation rate (air changes/s), and
- h = height of basement rooms (m).

3.2.6.4 Outdoor Radon Concentration

In all cases, the outdoor radon flux is calculated from the diffusion equation (Equation (3-88)). The outdoor radon flux may consist of up to three components. As illustrated in Figure 3-7, radon may originate in (1) the bottom contaminated layer, (2) the nominal top contaminated layer, or (3) the portion of the top layer known as the active layer. All three of these regions may have different radon-parent concentrations and, therefore, different radon generation rates.

For radon gas originating in the active layer, Equation (3-89) is applied with no clean soil over the radon-generating layer. The flux of radon from the rest of the top layer is also calculated with Equation (3-88), but a clean cover layer (with the thickness of the active layer) is present.

Figure 3-7. Radon Diffusion through Soil Layers and Cover Defects



The radon flux from the bottom layer is calculated in two parts, because some of this radon must diffuse all the way through the top layer, while some is released to the surface through cracks in the top layer. The fraction of the top layer that has failed is calculated by the model cap failure function. However, in both the intact and failed portions of the top layer, the radon flux is calculated from Equation (3-88). The failed portions are simply modeled by not having a soil cover above them.

After the total diffusive radon flux to the atmosphere is calculated, the outdoor radon concentration is calculated as:

$$C_o = \frac{(J_1 + J_2 + J_3 + J_4) L_x}{H u} \quad (3-95)$$

where

- C_o = outdoor radon concentration (Bq/m³),
- J_1 = radon flux from active layer (Bq/m²-s),
- J_2 = radon flux from top layer (Bq/m²-s),
- J_3 = radon flux from bottom layer through top layer (Bq/m²-s),
- J_4 = radon flux from bottom layer through cracks (Bq/m²-s),
- L_x = length of contaminated site (m),
- H = outdoor mixing height (m), and
- u = average wind speed (m/s).

3.2.6.5 Dose From Radon Inhalation

The dose from inhaling indoor and outdoor radon is calculated from the following equation:

$$D = (C B_{oc} + C_o S_{oc}) DF \quad (3-96)$$

where

- D = radon dose (uSv/yr),
- C = indoor radon concentration (Bq/m³),
- B_{oc} = fraction of year spent in basement (dimensionless),
- C_o = outdoor radon concentration (Bq/m³)

S_{oc} = fraction of year spent onsite outdoors (dimensionless), and

DF = radon inhalation dose factor (uSv/yr per Bq/m³).

3.2.7 DOSTAB Calculations

In the preceding sections it was shown how calculations are made of radionuclide concentrations in air, ground, vegetation, and water, and of annual ingestion and inhalation rates. These concentrations and rates of exposure are utilized by the DOSTAB portion of the model to generate tables of radiological dose and resulting health effects. This section describes the mathematical calculations made within DOSTAB. For the most part, the equations and text have been taken from the DARTAB documentation report (Be81), Section 3.3 titled, "General Equations."

3.2.7.1 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:

$$D_{ijl}(k) = (K_j E_{ij}(k) \cdot DF_{ijl})/P(k) \quad (3-97)$$

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.

DOSTAB 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 3-4 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.

Table 3-4. Unit of Exposure and Dose Rate Factors Used in DOSTAB

Pathway	Unit of Factor	
	E_{ij}	DF_{ijl}
Ingestion	(Person-pBq)/yr	(μ Sv/yr)/(pBq/yr)
Inhalation	(Person-pBq)/yr	(μ Sv/yr)/(pBq/yr)
Air immersion	(Person-pBq)/yr	(μ Sv/yr)/(pBq/yr)
Ground exposure	(Person-pBq)/yr	(μ Sv/yr)

Mean individual dose rates are calculated using:

$$D_{ijl} = [\sum_k P(k) \cdot D_{ijl}(k) / \sum_k P(k)] \quad (3-98)$$

Note that the critical population group 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 μ Sv/yr.

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

$$D_{il}(k) = \sum_j D_{ijl}(k) \quad (3-99)$$

or over all nuclides:

$$D_{jl}(k) = \sum_i D_{ijl}(k) \quad (3-100)$$

The total dose to the l th organ at location k , $D_l(k)$, is then:

$$D_l(k) = \sum_j \sum_i D_{ijl}(k) \quad (3-101)$$

The dose equivalent (mrem), H , for the l th organ is given as:

$$H_l(k) = QF_{(low-let)} D_{l(low-LET)} + QF_{(high-LET)} D_{l(k,high-LET)} \quad (3-102)$$

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:

$$D_{ij}(k) = \sum_1 W_1 D_{ijl}(k) \quad (3-103)$$

where W_1 are weighting factors for the various organ doses, which maintains:

$$\sum_1 W_1 = 1 \quad (3-104)$$

Weighting factors for various organs as recommended by the International Commission on Radiological Protection (ICRP79) are used for the dose calculation.

3.2.8 Health Effects Estimates

The health effects or individual risk of premature death for an individual residing at location k for the l th cancer, i th radionuclide, and j th exposure pathway are given by:

$$R_{ijl}(k) = 10^{-5} K_j \cdot E_{ij}(k) \cdot RF_{ijl}/P(k) \quad (3-105)$$

where

K_j is a numerical factor used to reconcile the units of $E_{ij}(k)$ and RF_{ijl} ,

$E_{ij}(k)$ is the exposure rate for the i th radionuclide by j th pathway,

RF_{ijl} is the risk factor for the l th cancer due to an unit exposure of i th radionuclide through the j th pathway, and

$P(k)$ is the exposed population at k th location.

The total individual risk from the exposure of all nuclides through all pathways can, therefore, be written as:

$$R(k) = 10^{-5} \cdot \sum_j K_j \cdot \sum_i E_{ij}(k) \sum_l RF_{ijl}/P(k) \quad (3-106)$$

The collective health effects are expressed in the health effects rate. Therefore, the total equivalent fatal cancer rate in an exposed population is calculated by:

$$HE = (10^{-5}/T_e) \cdot \sum_j K_j \cdot \sum_k \cdot \sum_i E_{ij}(k) \cdot \sum_l RF_{ijl}/P(k) \quad (3-107)$$

where T_e is mean individual life expectancy (70.7 years).

Users desiring a complete discussion of the development of the health effects should consult the DARTAB documentation report (Be81, pp. 5-10) and compare the risk conversion factors used in the HESTAB submodel to those in the cancer risks estimation report (EPA94).

3.2.9 Daughter Nuclide In-Growth Effect Correction

The previous version of the PRESTO model calculated the health impacts resulting from only the parent nuclide, while ignoring the health impacts contributed by its progeny. This simplification may, in some cases, cause significant error in the results of the risk assessment.

Since the original model was designed to evaluate health impacts without progeny effects, one of the simplest approaches in integrating the progeny effects into the current model (Version 4.2) is to introduce a progeny effect correction factor. This correction factor can then be used to adjust the results obtained from the original model to account for the progeny effects.

In order to simplify the analysis, a crude assumption is imposed upon the progeny effect correction factor analysis. The model assumes that the sorption characteristics of the parent nuclide and its progeny are identical throughout the processes of leaching and transport through the geosphere. This assumption seems to be unrealistic, but the error incurred from it is not excessive and, in most cases, is on the conservative side. Furthermore, this assumption is widely adopted in most other screening risk assessment models.

When the above assumption is imposed, the ratio of the activities between progeny and parent nuclides in a designated moving control volume at any given time can be calculated from the Bateman equation. This ratio is then used to calculate the correction factor to account for progeny in-growth effects.

3.2.9.1 Decay Chains

For the purpose of assessing the health impacts from contaminated soil sites, the following simplified decay chains are selected for incorporation into the PRESTO-EPA-CPG/POP model:

1. Am-243 \longrightarrow Pu-239 \longrightarrow U-235
2. Cm-244 \longrightarrow Pu-240 \longrightarrow U-236 \longrightarrow Th-232
3. Pu-238 \longrightarrow U-234 \longrightarrow Th-230 \longrightarrow Ra-226 \longrightarrow Pb-210

4. Pu-241 \longrightarrow Am-241 \longrightarrow Np-237
5. Pu-242 \longrightarrow U-238 \longrightarrow U-234 \longrightarrow Th-230 \longrightarrow Ra-226 \longrightarrow Pb-210

These decay chains assume that those progeny not shown in the chains can be ignored for the analysis. The error introduced from this simplification is considered to be insignificant for an application to a screening model.

Altogether, 13 parent nuclides are considered and built into the model for calculating their progeny effects: Am-243, Pu-239, Cm-244, Pu-240, U-236, Pu-238, U-234, Th-230, Ra-226, Pu-241, Am-241, Pu-242, and U-238. The model evaluates the progeny in-growth effects up to the fourth member of the chains shown above; the effects from the fifth and higher members are neglected. Using Pu-242 as an example, the analysis calculates the potential health effects induced from U-238, U-234, and Th-230 only. All other progenies are ignored.

3.2.9.2 Dose Equivalent Calculation

The derivation of the correction factors representing the progeny in-growth effects for the second, third, and fourth decay chain members are derived and included in Appendix C. The results are summarized as follows:

$$\eta_2 = \lambda_2 \frac{CF_2}{CF_1} \left(\frac{1}{\lambda_2 - \lambda_1} + \frac{\text{EXP}(-(\lambda_2 - \lambda_1)t)}{\lambda_1 - \lambda_2} \right) \quad (3-108)$$

$$\eta_3 = \lambda_2 \lambda_3 \left(\frac{CF_3}{CF_1} \right) \left(\frac{1}{(\lambda_3 - \lambda_1)(\lambda_2 - \lambda_1)} + \frac{\text{EXP}(-(\lambda_2 - \lambda_1)t)}{(\lambda_3 - \lambda_2)(\lambda_1 - \lambda_2)} \right. \\ \left. + \frac{\text{EXP}(-(\lambda_3 - \lambda_1)t)}{(\lambda_2 - \lambda_3)(\lambda_1 - \lambda_3)} \right) \quad (3-109)$$

$$\eta_4 = \lambda_2 \lambda_3 \lambda_4 \left(\frac{CF_4}{CF_1} \right) \left(\frac{1}{(\lambda_4 - \lambda_1)(\lambda_3 - \lambda_1)(\lambda_2 - \lambda_1)} + \frac{\text{EXP}(-(\lambda_2 - \lambda_1)t)}{(\lambda_4 - \lambda_2)(\lambda_3 - \lambda_2)(\lambda_1 - \lambda_2)} \right. \\ \left. + \frac{\text{EXP}(-(\lambda_3 - \lambda_1)t)}{(\lambda_4 - \lambda_3)(\lambda_2 - \lambda_3)(\lambda_1 - \lambda_3)} + \frac{\text{EXP}(-(\lambda_4 - \lambda_1)t)}{(\lambda_3 - \lambda_4)(\lambda_2 - \lambda_4)(\lambda_1 - \lambda_4)} \right) \quad (3-110)$$

where

- η = correction factor relative to the parent nuclide,
- λ = radionuclide decay constant,
- CF = dose conversion factor, and
- 2,3,4 = subscripts respectively for the second, third, and fourth decay chain members.

The combined progeny effects correction factor used in the model is the sum of all correction factors, that is:

$$\eta = 1 + \eta_2 + \eta_3 + \eta_4 \quad (3-111)$$

The equivalent dose for the combined parent and daughter nuclides is calculated by:

$$D_C = D_p \times \eta \quad (3-112)$$

where

- D_C = combined equivalent dose for parent and daughter nuclides, and
- D_p = calculated equivalent dose for the parent nuclide.

3.2.9.3 Cumulative Health Effects Calculation

A similar approach is also taken to calculate the cumulative health effects to the general population. The same equations as used in the calculation of the correction factors for the dose are used to calculate the same correction factors for the cumulative health effects, except health effects conversion factors are used instead of dose conversion factors.

Since the cumulative health effects calculation has to deal with the time-dependent correction factors over the time period of analysis, it is necessary to take a few more steps to correct the health effects, as follows.

The net increment in the cumulative health effects due to each decay member during a time increment of Δt can be expressed as:

$$\Delta HE_c = Q_1 \cdot I \cdot HECF_1 \cdot \Delta t (1 + \eta_{h2} + \eta_{h3} + \eta_{h4})$$

or

$$\Delta HE_c = I \cdot HECF_1 \cdot Q_1(t) \cdot \eta_h(t) \cdot \Delta t$$
(3-113)

where

- HE_c = combined parent and progeny health effects,
- I = intake factor,
- $HECF_1$ = cumulative health effect conversion factor for the parent nuclide, and
- η_h = combined daughter nuclides correction factor.

Therefore, the cumulative health effects from the decay chain for the period of analysis can be written as:

$$HE_c = I \cdot CF_1 \int_0^T Q_1 \cdot \eta_h \cdot dt$$
(3-114)

where T denotes the duration of the health effects analysis.

Equation (3-114) implies that the combined health effects for a four-member decay chain can be calculated from the integration of the product of parent-nuclide activity in the control volume and its combined correction factor over the duration of the health-effect analysis.

The model involves the evaluation of the cumulative health effects resulting from the consumption of the radionuclide-contaminated-water in the environmental receptors. The radionuclide concentration in an environmental receptor is calculated annually and its average value is calculated at the end of the yearly loop analysis. The average human exposure rate and its subsequent health effects are calculated from this average value. The integration of the progeny in-growth effects simply involves the modification of the average concentration as:

$$C_{ave} = \left(\frac{1}{T} \right) \int_0^T C(t) \cdot \eta(t) \cdot dt$$
(3-115)

After the average radionuclide concentration in the environmental receptor is calculated, the health effects for the local population are calculated through the existing biological and human exposure pathways model.

The health effects in the downstream basin population are calculated from the product of the cumulative release of radioactivity and the health effects conversion factor for the regional basin population (Section 3.3). In integrating the daughter nuclide in-growth effect into the health effect assessment, the released parent nuclide activity should be corrected annually. The correction of the cumulative radionuclide release is calculated by:

$$Q_T = \int_0^T Q_1(t) \cdot \eta(t) \cdot dt \quad (3-116)$$

where Q_T denotes the corrected cumulative radionuclide release.

After the activity of the parent nuclide released to the downstream basin is corrected for daughter nuclide in-growth effects, the total health effects for the downstream basin population are calculated by multiplying corrected cumulative radionuclide release with the precalculated conversion factor for the parent nuclide. The calculation of the conversion factors for the regional basin population is discussed in next section.

3.3 HEALTH EFFECTS INDUCED IN THE REGIONAL BASIN POPULATION

The PRESTO-EPA-POP model estimates the potential cumulative health effects induced in the regional basin population downstream of the contaminated site for a period of 10,000 years. Health effects are calculated for the onsite, offsite, and regional basin populations for up to 10,000 years. Any radioactivity not withdrawn from the groundwater or surface water by the onsite or offsite populations is released as residual radioactivity to the basin population. The cumulative health effects in the basin population are obtained by multiplying the cumulative residual nuclides with their conversion factors.

The residual radionuclides include those radionuclides not consumed by the onsite or offsite community. The total health effects are then calculated from the sum of the health effects obtained from the onsite, offsite, and regional basin analyses.

The regional basin analysis assumes that all of the communities located in a regional water basin downstream from the disposal site can be combined into a single composite community. The transport of radionuclides from the disposal site through the hydrologic pathway continues. 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. The radionuclides considered are those that enter the aquifer from the contaminated soil site and those that enter the regional stream by way of runoff.

In order to determine the potential health effects induced in the regional basin population, the residual radionuclide activity released into the regional basin is multiplied by a conversion

factor precalculated for each radionuclide. The conversion factors, which are nuclide-dependent, are based on local water use characteristics and the hydrologic pathway.

3.3.1 Calculations of Regional Basin Health Effects

The model uses health effects conversion factors (see Section 3.3.2) to calculate the potential health effects induced in the regional basin population from the cumulative activity of residual radionuclides released to the regional basin. 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 induce no health effects.

Releases of radionuclides into the regional basin can be simulated for up to 10,000 years. The annual nuclide releases into the regional basin are collected in the model in 10 periods of 1,000 years each. The annual releases of each nuclide from the surface runoff and from the aquifer to the regional basin during the first millennium are collected in the array variable QDWSB.

In simulating the cumulative release of radionuclides through the surface water pathway, the radionuclide concentrations in surface soil and surface water are calculated first according to Equations (3-35) and (3-36). These concentrations change from year to year as a result of wind resuspension, surface water runoff, trench water overflow, and the seepage of water from surface soil to the aquifer.

The amount of radionuclides released from the surface soil to the surrounding surface streams is calculated using the current year surface water nuclide concentration, C_w (Ci/m³); the area of the contaminated site, A_T (m²); the annual precipitation rate, P_a (m/yr); the current year's amount of water overflow, V_o (m³/yr); and a transfer factor, f_r , as follows:

$$W_s = f_r(P_a \cdot S_w \cdot S_l + V_o) \quad (3-117)$$

$$SSTREM = W_s \cdot C_w \quad (3-118)$$

where W_s (m³/yr) is the amount of water that enters the surrounding streams from runoff and site overflow during the current year of simulation, and SSTREM (Bq/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 net amount released into the regional basin, SSTREM(N), is calculated by subtracting the amount of nuclides being removed from the nearby stream(s) for consumption by the onsite or offsite populations. The calculation is accomplished by 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 into the basin each year for the first millennium is:

$$\begin{aligned} & (\text{STFLOW} - \text{VOLUSS}) \cdot \text{STCON}(N) \quad \text{if } \text{VOLUSS} < \text{STFLOW} \\ & 0 \quad \text{if } \text{VOLUSS} \geq \text{STFLOW} \end{aligned} \tag{3-119}$$

In calculating the cumulative release of residual radionuclides from the aquifer to the basin, the amount being pumped out for local population consumption is first subtracted from the annual transport of radionuclides. In calculating the cumulative release of radionuclides into the regional 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 total release of radionuclides, QLBTTH, into the basin is the sum of the releases from surface water and groundwater pathways over 10,000 years; i.e.,

$$\text{QLBTTH} = \text{QDWSB} + \sum_{j=2}^{10} \text{QLB}(j) \tag{3-120}$$

where QDWSB and QLB(j) are the total radionuclides released into the regional basin in the first and jth millennium, respectively.

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 radionuclides into the regional basin by a health effect conversion factor, i.e.,

$$\text{HE} = \text{QLBTTH} \cdot \text{BHECF} \tag{3-121}$$

The basin health effects conversion factors, which are used to determine the health effects to the regional basin population from residual radionuclides, are model input values and are precalculated by using PRESTO-EPA-CPG/POP as a tool. The methodology for calculating these factors is discussed in Section 3.3.2.

The regional basin health effects and the genetic effects for ith nuclide over the entire 10,000 years of analysis can now be calculated by summing up the effects for over 10,000 years, respectively as follows:

$$\begin{aligned} (\text{Basin Health Effects})_i &= [(\text{Residual nuclide, 1st 1,000 years})_i \\ &+ (\text{Residual nuclide, last 9,000 years})_i] \cdot \text{BHECF}_i \end{aligned} \tag{3-122}$$

and

$$(\text{Basin Genetic Effects})_i = [(\text{Residual nuclide, 1st 1000 years})_i$$

$$+ (\text{Residual nuclide, last 9,000 years})_i] \cdot \text{BGECECF}_i \quad (3-123)$$

The total regional basin health effects and genetic effects resulting from all basin residual nuclides are calculated by summing the effects over all nuclides as follows:

$$\text{Total Basin Health Effects} = \sum_{i=1}^N (\text{Basin Genetic Effects})_i \quad (3-124)$$

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

3.3.2 Conversion Factors for Regional Basin Health Effects

The basin health effect and genetic effect conversion factors are used to calculate the impacts of residual radioactivity entering a regional water basin. The conversion factors, which are nuclide-specific, are made up of two components, one calculated from the terrestrial pathway (BHECF_{ti}) and the other from the aquatic pathway (BHECF_{ai}). Separate conversion factor values are calculated for health effects and for genetic effects. The methodology discussed below is applicable to the calculation of both cancer effect and genetic effect conversion factors.

3.3.2.1 Terrestrial Pathway

The nuclide-specific health effects conversion factors for the terrestrial pathway (BHECF_{ti}) are calculated in two steps using the results from analyses for the local population. The first step determines the health effects to the local population resulting from the withdrawal of a unit activity of a specific nuclide from the local well or stream.

The second step involves the calculation of the fraction of activity discharged into the basin being withdrawn for consumption by the regional basin population. The BHECF_{ti} is then determined by multiplying the fraction of activity withdrawn with its health effect per bequerel conversion factor, that is:

$$\text{BHECF}_{ti} = \text{HE}/\text{Ci}_i \times f \quad (3-125)$$

where

BHECF_{ti} = terrestrial HECF for radionuclide i ,

HE/Ci_i = health effects to the local population per unit activity of i th radionuclide withdrawn from the local well or stream, and

f = fraction of activity withdrawn from the basin river per unit activity released to the regional basin.

The fraction of activity withdrawn by the regional basin population is calculated based on the local population water usage and a standard ratio of river flow to population. The per-capita water use (including the water for drinking, cattle feed, and irrigation) for the local population is calculated based on appropriate model inputs. Assuming that the per-capita regional basin water use and the ratio of population to river flow in the regional basin are constants, then the fraction of activity that will be withdrawn by the regional basin communities can be calculated by:

$$f = \frac{U/P}{Q/P} \quad (3-126)$$

where

f = fraction of activity withdrawn from the basin river per unit activity released to the regional basin,

(U/P) = per-capita water usage (m³/person-yr), and

(Q/P) = ratio of river flow to population, 3,000 m³/person-yr (EPA85).

The per-capita water consumption is calculated from the individual water consumption, local irrigation requirement, and cattle feed demand. For regions where multiple water sources are 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 arrogation pathway switch will equal 0.5. The per-capita water consumption as built in the model is expressed by:

$$U = [3.9 \times 10^7 W_i f_i L_i + U_w L_h + 1.5 \times 10^4 L_a] \quad (3-127)$$

where

U = per-capita water consumption (l/person-yr),

3.9x10⁷ = 4492 m² irrigated land per person x 8760 hr/yr,

W_i = irrigation rate (l/m²-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,
1.5×10^4	=	annual water fed to cattle consumed by humans (l/person-yr),
L_a	=	animal pathway switch, and
N_p	=	size of population.

In calculating the health effects resulting from pumping the contaminated water from groundwater or surface water, the air pathway sources should be shut off by setting surface contamination equal to zero. In addition, no onsite farming or basement exposures are included. Therefore, the health effects in the exposed population are due to the consumption of the contaminated water only.

When the local per-capita water consumption is divided by a standard ratio of river flow to population, the fraction of the regional basin radioactivity that will be withdrawn by the regional basin community is determined. This calculation is based on two assumptions: that the local water usage is comparable to regional basin water consumption, and that the national average ratio of river flow to population is applicable to the regional basin. Studies show that while regional basin population and river flow vary widely, the ratio of the river flow to the population remains relatively constant (EPA85).

As noted previously, the fraction of nuclides that are not pumped out by the regional basin community are assumed to enter the ocean, which acts as a nuclide sink. The health effects resulting from these activities collected in the ocean are assumed to be negligible.

3.3.2.2 Fish Pathway

One pathway that was viewed as negligible for the local population but is considered in calculating regional basin health effects is that of contaminated fish ingestion. A separate $HECF_{fi}$ for fish is determined based on the following equation:

$$BHECF_{fi} = (P/Q) \cdot B_{fi} \cdot U_f \cdot (D/C)_i \quad (3-128)$$

where

$BHECF_{fi}$	=	health effects conversion factor from consumption of fish, per curie of radionuclide i released to the regional basin,
(P/Q)	=	population-to-river flow ratio (person-yr/3,000 m ³) (EPA85),
B_{fi}	=	fish bio-accumulation factor (Bq/kg-fish per Bq/l of radionuclide i in water) (NAS71),

- U_f = annual fish consumption rate (6.9 kg/person-yr) (Ru80), and
- $(D/C)_i$ = conversion factor for health effects per curie of nuclide i ingested, obtained from HESTAB data file.

3.3.2.3 Basin Health Effects Conversion Factor

The conversion factors for regional basin health effects can now be calculated by summing the conversion factor from the terrestrial pathway and the conversion factor from the fish pathway. Thus, the nuclide-specific regional basin $HECF_i$ can be expressed as:

$$BHECF_i = BHECF_{ti} + BHECF_{fi} \quad (3-129)$$

The same methodology discussed above is also used for the calculation of the genetic effects conversion factors. The only differences are that the genetic effects per unit activity are used in place of health effects per bequerel in Equation (3-125), and that the genetic effects conversion factor is used in place of the health effects conversion factor in Equation (3-128).

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