APPENDIX H

TRANSPORT AND DOSE MODELS

This appendix describes the analytical models used to determine the transport of waste constituents through the environment. It also discusses potential exposure of individuals to such constituents resulting from the alternative actions evaluated in this environmental impact statement (EIS). The primary transport is via the groundwater pathway; Section H.1 describes the hydrogeologic models used to evaluate that pathway. Atmospheric pathways provide more routes for exposure via deposition and uptake in foods and by inhalation; Section H.2 describes models used for these evaluations.

H.1 HYDROGEOLOGIC MODELS

This section describes the hydrogeologic models used to support this EIS. The assessments in the EIS are based on data and study results presented in Environmental Information Documents (EIDs). The computer models are identified in several documents (Colven et al., 1985; Stephenson et al., 1987; Merrell, Rogers, and Bollenbacher, 1986; Rogers, Merrell, and Bollenbacher, 1986; Merrell and Rogers, 1986). The hydrogeologic models discussed in this appendix are PATHRAE, MOD3D, and SWIFT II. TC

H.1.1 PATHRAE

PATHRAE is an analytical model used to provide a basis for quantitative estimates of the human health risks associated with land disposal of wastes. This code was developed originally for the U.S. Environmental Protection Agency (EPA) for low-level radioactive waste disposal. It was modified to estimate health risks and environmental effects of removal and closure options for low-level radioactive, mixed, and hazardous waste disposal sites on the Savannah River Plant. PATHRAE has also been used in performance assessments of new disposal facilities for hazardous wastes, mixed wastes, and low-level radioactive wastes. The value of the PATHRAE model is its simplicity of operation and its presentation of analysis results for a set of waste constituents and pathways.

PATHRAE was the primary model used to provide a basis for the relative environmental consequences of the various approaches considered for existing waste sites and new disposal facilities. The following paragraphs evaluate the ability of PATHRAE to perform this task as a basis for comparative evaluation of alternative strategies, as opposed to site-specific decisions that would be based on more precise determinations of environmental consequences. Such determinations require site-specific groundwater flow data such as input, in more complex cases, to three-dimensional models (as well as site-specific information on waste inventories and soil-waste interactions), and would be prepared as part of the regulatory agency interactions required for specific project proposals. TC

The PATHRAE evaluation was performed by the following methods:

- Comparison with other analytical models

- Comparison with measured concentrations

- Comparison with three-dimensional numerical solutions

- Evaluation of the selection of model input values and their effect (i.e., sensitivity on model results)

TE A comparison with other analytical results indicates good agreement between PATHRAE and a slightly more complex analytical model (Looney, King, and Stephenson, 1987). This indicates that two simplifying assumptions in PATHRAE (i.e., plug flow in the unsaturated zone and uncoupled longitudinal and transverse dispersion in the saturated zone) do not have a significant effect on transport predictions. PATHRAE predicts higher concentrations than a three-dimensional dispersion model. This indicates that neglecting the vertical dispersion causes PATHRAE to be more conservative than the more sophisticated three-dimensional model. PATHRAE also predicts concentrations that are higher than those predicted by the EPA VHS model, which was developed specifically to develop conservative models of land disposal scenarios. In the concentrations presented in Chapter 4 and Appendix F for the 1- and 100-meter wells, this conservatism was increased by neglecting the transverse dispersion component of the PATHRAE model.

Figure H-1 presents the results of a comparison of PATHRAE 1-meter well predictions for SRP waste site assessments to average 1985 downgradient concentrations, which suggests that the methods used for prediction produced generally reasonable results. Based on the data, approximately 73 percent of the predictions are within a factor of 10 of the measured values, with considerable scatter both above and below the "1x" line, particularly at low concentrations (i.e., less than 100 micrograms per liter or picocuries per liter). However, at concentrations above several hundred micrograms per liter or picocuries per liter, the PATHRAE predictions improve considerably with only a few underpredictions of measured values.

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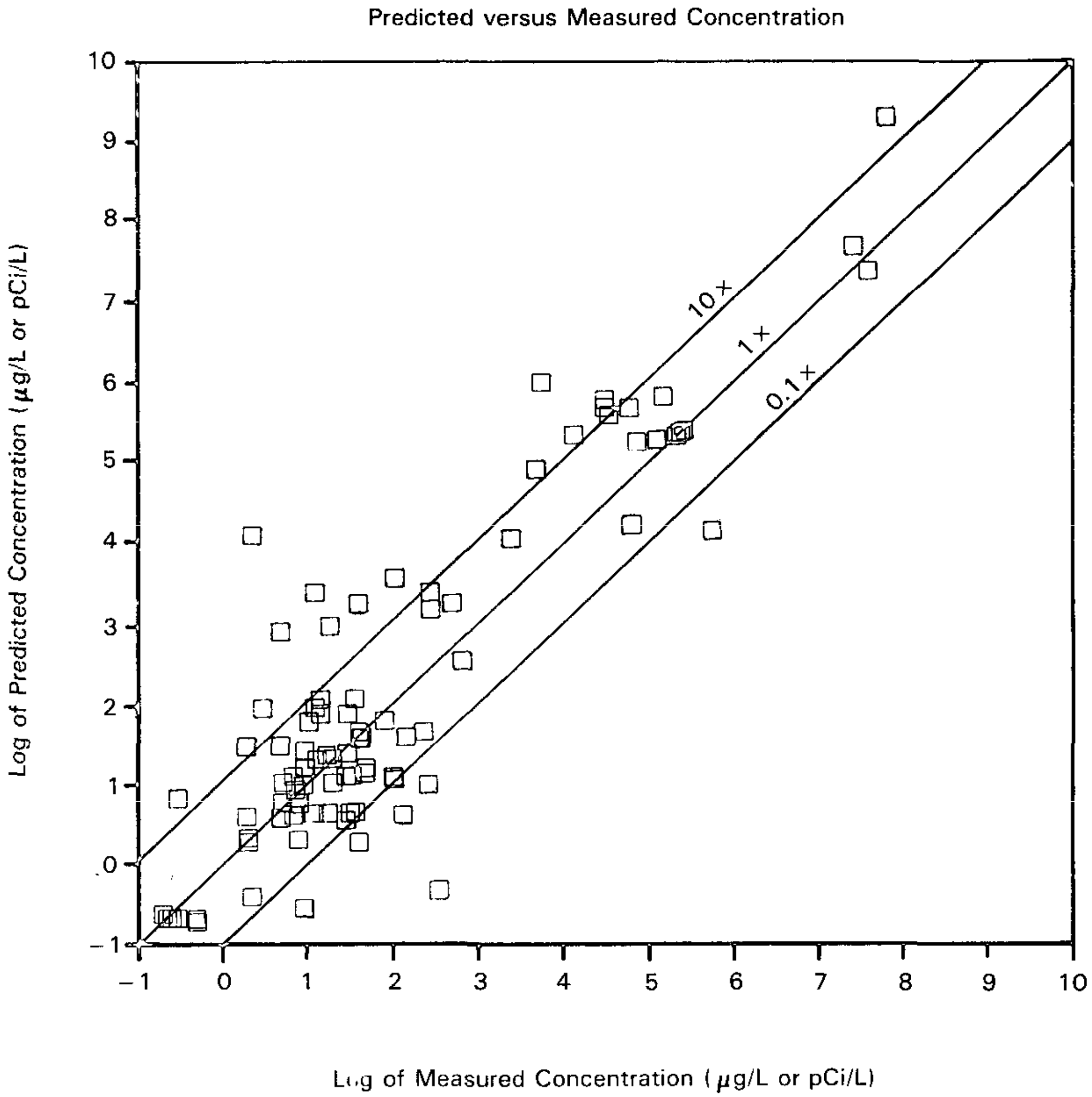
Thus, in a comparison of PATHRAE success in predictions of exceedances of groundwater protection guidance, PATHRAE predicted 36 exceedances while 33 exceedances were measured (of which PATHRAE predicted 28). With respect to waste sites, PATHRAE predicted at least one exceedance at each of 14 sites, compared to 13 sites with at least one measured exceedance; all 13 sites were identified by the PATHRAE predictions.

Researchers also compared PATHRAE results to those generated by "more sophisticated" three-dimensional flow and transport models (Looney, King, and Stephenson, 1987). The three-dimensional models were used in the A- and M-Areas and the F- and H-Areas, where detailed geohydrologic data were available. Generally, the peak concentrations predicted by PATHRAE are higher by factors of 10 or greater than those predicted by the three-dimensional models. The model comparisons suggest that PATHRAE is conservative but sufficiently accurate to compare relative differences in various waste management approaches.

Researchers applied sensitivity analyses to bound the range of predicted concentrations that would result from the uncertainty in estimating the input parameters. The input parameters that have the most significant effects on results are assumed inventory, groundwater flow rate, and leach rate. These studies indicate that the variations due to uncertainties in input parameters



Source: Looney, King, and Stephenson, 1987.

Figure H-1. Verification of PATHRAE Model Results

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are less than the inherent uncertainties of the model. The worst-case deviation for a single parameter was less than a factor of 10 (Looney, King, and Stephenson, 1987).

In summary, these four studies indicate that the PATHRAE model is sufficiently accurate to make relative comparisons between generic waste management approaches for the purposes of this EIS. However, specific conceptual design-level and/or permitting decisions would require more detailed site-specific modeling.

The PATHRAE model has some limitations:

- It is one-dimensional.
- It was not used to incorporate results of groundwater remedial actions in the overall analysis.
- It was not used to predict spatial distribution of plumes.
- It was not used to determine concentration distribution.
- It is not suitable for determining effects of remedial actions (e.g., groundwater pumping in M-Area).

Researchers can investigate site performance for radioactive/hazardous waste disposal with relatively few parameters to define the site condition. This characteristic makes the model useful for the evaluation of a wide range of radioactive and hazardous waste disposal problems. The modified version of PATHRAE can evaluate the environmental and health risk due to nonradioactive contaminants by the input of equivalent model parameters.

General inputs to the model include the following:

- Dimension and size of the source
- Flow rate of the receiving surface stream
- Distance to the receiving surface stream
- Depth to the aquifer
- Aquifer distance to accessible location
- Bulk density of aquifer materials
- Groundwater flow velocities
- Longitudinal and lateral dispersivities
- Total waste volume
- Density of waste

- Parameters associated with vegetation and air deposition
- Atmospheric parameters such as atmospheric stability, wind speed, diffusion coefficient, precipitation, etc.
- Soil retardation characteristics
- Porosity of aquifer
- Cover thickness and impermeability
- Mixing thickness of aquifer
- Surface erosion rate

The contaminant transport through the aquifer is determined by the solution of either the one-dimensional advection equation or the one-dimensional advection-dispersion equation with decoupled longitudinal and transverse dispersion. In association with this methodology, the model includes the following assumptions:

1. The aquifer is one-dimensional, consisting of an infinitely long homogeneous, isotropic porous medium.
2. The releases of contaminants from the source are constant or are an exponentially decaying function of time.
3. Only adsorption-desorption equilibrium of contaminant between water and aquifer materials is considered in calculating the effect of retardation. Effects of pH, redox potential, and thermodynamically competing species are neglected.
4. The movement of contaminants in the unsaturated zone is described in terms of plug flows.

The code contains algorithms for analyzing 10 different pathways. The pathways that were modeled include groundwater movement to hypothetical water wells nearby, groundwater movement to the Savannah River, waste erosion and movement to the Savannah River, food consumption on reclaimed farm, and consumption of crops grown through natural biointrusion.

For groundwater movement to nearby water wells, the pathway consists of downward migration of the modeled waste components through advection and diffusion or as a result of dissolution in percolating precipitation. The waste components move downward through the unsaturated zone to the aquifer and move horizontally to nearby wells downstream (in the sense of aquifer flow). Two hypothetical well scenarios were analyzed: one immediately adjacent to the waste disposal facility (i.e., the 1-meter well) and one 100 meters downstream from the edge of the facility. The models for both vertical and horizontal movement of waste materials account for chemical retardation by the soils. Once withdrawn from the well, the water is assumed to be consumed directly by individuals and used to irrigate crops that are then consumed by these same individuals.

For groundwater movement to surface streams, the pathway is similar to that described above, but the modeled waste components are assumed to continue to move through the aquifer until released to surface waters. For the purpose of analyzing the potential impacts of releases through this pathway, the release was assumed to be into the Savannah River, with its downstream consumer populations. The waste components are assumed to be mixed completely with water in the Savannah River.

The following subsections present equations describing the transport and dose via groundwater to surface waters and to wells.

H.1.1.1 Groundwater Pathway to a Surface Stream

The dose from groundwater migration to a river is calculated from:

$$D = \frac{Q \lambda_L f_0 U_t}{q_w} (DF) \tag{H-1}$$

where:

- Q = inventory of the radionuclides (picocuries) or toxic chemicals (kilograms)
- q_w = flow rate of the river (cubic meters per year)
- f_o = fraction of the inventory arriving at the river from transport through the aquifer
- λ_L = fraction of each nuclide/chemical leached from the inventory in a year
- U_t = annual equivalent surface-water uptake by an individual (cubic meters per year)
- DF = dose conversion factor for radionuclides (millirem per picocurie) = 1 for chemicals
- D = (units) dose in millirem for 1 year or kilograms per year for chemicals

In Equation H-1, the product of Q and λ_L represents the release rate of radionuclide/chemical from the source. Parameter f_o determines the fraction of radionuclide/chemical released from the source that can reach the river. U_t is the amount of river water consumed by an individual. DF defines the dose to an individual for each unit of radionuclide or chemical uptake.

Transport of contaminants through the aquifer can be described by the advection-dispersion equation:

$$\frac{\partial C}{\partial t} = -\frac{V}{R} \bullet \frac{\partial C}{\partial X} + \frac{D_L}{R}\frac{\partial^2 C}{\partial X^2} - \lambda C \tag{H-2}$$

where:

C = concentration of contaminant (picocuries per liter or milligrams per liter)
V = seepage velocity of the groundwater flow (meters per year)
X = distance along the mean groundwater flow direction (meters)
D_L = longitudinal dispersion coefficient along the direction of flow (square meters per year)

R = retardation factor $= 1 + \frac{\rho \text{ rho}}{p} k_d$ (Freeze and Cherry, 1979)

λ = first-order decay constant
ρ = aquifer density
p = aquifer porosity
k_d = sorption coefficient in the aquifer (cubic meters per kilogram)

If the dispersion term is neglected, Equation H-2 reduces to the one-dimensional advection equation with radioactive decay

$$\frac{\partial C}{\partial t} = -\frac{V}{R} \bullet \frac{\partial C}{\partial X} - \lambda C \qquad \text{(H-3)}$$

Parameter f_o of Equation H-1 can be calculated for either dispersive or nondispersive groundwater transport. For the nondispersive case, the line source is assumed to decrease in inventory with time at a constant fraction due to both the release of contaminant and radioactive decay. The solution of the one-dimensional advection equation (Equation H-3) for this boundary condition, parameter f_o, is as follows:

$$f_0 = 0 \text{ for } t \leqslant t_1 - t_0$$

$$f_0 = \frac{V_a}{LR\lambda_L} \bullet [1 - \exp[-\lambda_L (t - (t_1 + t_0))]] \quad \text{for} \quad t_1 - t_0 < t < t_1 \qquad \text{(H-4)}$$

$$f_0 = \frac{V_a}{LR\lambda_L} \bullet \exp[-\lambda_L (t - t_1)] \quad [1 - \exp(-\lambda_L t_0)] \quad \text{for} \quad t_1 \leqslant t$$

where:

t = time (years)
t_0 = RL/V_a
t_1 = $R(L+X_w)/V_a$

R = retardation factor $= 1 + \frac{\rho}{p} k_d$

k_d = sorption coefficient in the aquifer (cubic meters per kilogram)
ρ = aquifer density (kilograms per cubic meter)
L = length of waste site in direction parallel to aquifer flow (meters)
V_a = interstitial horizontal aquifer velocity (meters per year)
X_w = distance of groundwater flow from nearest edge of burial pits to the river (meters)
p = aquifer porosity

For dispersive groundwater transport, the source is considered to be a line of point sources that release contaminants, with the exception of radioactive decay, at a constant rate. The solution of the one-dimensional advection-dispersion equation (Equation H-2) for this boundary condition, the parameter f_o, can be expressed as:

$$f_0 = \frac{1}{N} \sum_{j=1}^{N} [Fj(t) - Fj(t-1/\lambda_L)] \tag{H-5}$$

where:

$Fj(t)$ = 0.5 U (t) [erfc (z-) + exp(d_j) erfc (z+)]
$U(t)$ = unit step function
$z\pm$ = $\dfrac{d_j^{1/2}[1 \pm t/Rt_{wj}]}{2[t/Rt_{wj}]^{1/2}}$
d_j = distance from sector center to access location divided by the longitudinal dispersivity
t_{wj} = water travel time for distance d_j (years)
N = number of mesh points in numerical integration

The disposal area of length L is divided into N sectors of equal length. A point source of the appropriate magnitude is placed at the center of each sector. The distance d_j is measured from the center of sector j to the access location. The summation shown in Equation H-5 represents the integration of the point source analytical solutions to approximate an area source.

H.1.1.2 Groundwater Pathway to a Well

The dose from groundwater migration with discharge to a well is calculated from:

$$D = \frac{Q\lambda_L f_0 U_2(DF)}{q_w} \tag{H-6}$$

The aquifer flow rate q_w is given, in this case, by:

$$q_w = \begin{cases} WLP & \text{for } H_w > L_p \\ WL_pV_ap & \text{for } H_w < L_p \end{cases}$$

where

- W = width of waste pit perpendicular to aquifer flow (meters)
- P = water percolation rate (meters per year)
- L_p = length of well casing in aquifer (meters)
- H_w = vertical dimension of contaminated zone in aquifer (meters)
- v_a = horizontal velocity of aquifer (meters per year)
- U_2 = *annual equivalent total uptake of well water by an* individual (cubic meters per year)

Continuity of mass for the contaminated water in the unsaturated and saturated zone requires that

$$H_w = \frac{P \bullet L}{p \bullet v_a} \tag{H-7}$$

In addition to modeling the effects of longitudinal dispersion in the aquifer, the well pathway can account for any transverse dispersion that might occur. This reduces the conservatism when calculating contaminant doses for the well pathway. In modeling of transverse dispersion, the term f_o in Equations H-5 and H-6 is modified by an additional multiplicative term, f_t, given by:

$$f_t = \frac{1}{2} \operatorname{erf} \left[\frac{(y_w + W/2)R}{2(D_y t)^{1/2}} \right] - \frac{1}{2} \operatorname{erf} \left[\frac{(y_w - W/2)R}{2(D_y t)^{1/2}} \right] \tag{H-8}$$

where:

- y_w = distance to well from center of waste area in the direction perpendicular to the aquifer flow (meters)
- D_y = transverse dispersion coefficient (square meters per year)

For the limiting case in which D_y goes to zero, f_t becomes equal to 1. Therefore, the effects of transverse dispersion can be ignored by choosing D_y equal to zero.

Although a portion of the model's algorithms associated with subsurface transport have been verified analytically by comparison with the simplified analytical solutions and other independent calculations using different programs, the overall model has not been verified with field measurements. A report prepared by Clemson University discusses PATHRAE code sensitivity and verification (Fjeld et al., 1986).

H.1.2 MOD3D

The MOD3D model, which was developed by the U.S. Geological Survey, simulates three-dimensional groundwater flow in a porous, heterogeneous, and anisotropic medium with irregular boundaries. The uppermost hydrologic unit can have a free water-table surface. Stress can be applied to the system in the form of well discharge/recharge, and as recharge from precipitation. A modified version of this model extends its application to simulations involving head-dependent sources and sinks such as river, springs, or drains, and evapotranspiration. These modifications also enhance the effectiveness of the iterative solution process used by the original version.

This model can simulate groundwater flow in both a fully three-dimensional and a quasi-three-dimensional manner, depending on the availability of data and the requirements of computer memory. It can simulate each hydrologic unit with one or more layers and permits the use of variable grid spacing. If the analysis can neglect the storage in a confining bed and the associated horizontal component of flow, the model can incorporate the effects of vertical leakage through a confining bed into the vertical component of the anisotropic hydraulic conductivity of adjacent aquifers.

The iterative numerical technique used to solve the set of simultaneous block-centered, finite-difference, approximated, algebraic equations is the strongly implicit procedure. This method converges faster and has fewer rounds of errors than the iterative alternating direction implicit method.

Groundwater flow in a three-dimensional, heterogeneous, and anisotropic porous medium can be expressed as

$$\nabla \bullet (K_{ij}\frac{\partial h}{\partial x_j}) = S_s\frac{\partial h}{\partial t} + W(x,y,z,t) \qquad \text{(H-9)}$$

where:

∇	=	vector differential operator
h	=	hydraulic head (L)
S_s	=	specific storage (L^{-1})
K_{ij}	=	tensor of hydraulic conductivity (LT^{-1})
X_j	=	distance in the space direction j (L)
$W(x,y,z,t)$	=	volumetric flux per unit volume of aquifer (T^{-1}) representing source/sink of the porous medium

Assuming that the coordinate axes x, y, and z are aligned with the principal directions of the hydraulic conductivity tensor, the crossproduct terms drop from Equation H-9. It reduces into the following form:

$$\frac{\partial}{\partial x}(K_{xx}\frac{\partial h}{\partial x}) + \frac{\partial}{\partial y}(K_{yy}\frac{\partial h}{\partial y}) + \frac{\partial}{\partial z}(K_{zz}\frac{\partial h}{\partial y}) = S_s\frac{\partial h}{\partial t} + W(x,y,z,t) \qquad \text{(H-10)}$$

in which K_{xx}, K_{yy}, and K_{zz} are the components of the hydraulic conductivity in the three principal directions x, y, and z. In the finite-difference approach, it is often convenient to represent a hydrologic unit by one layer of nodes. Thus, if Equation H-9 is multiplied by the thickness (b) of the hydraulic unit, Equation H-10 can be written as:

$$\frac{\partial}{\partial x}\left(T_{xx}\frac{\partial h}{\partial x}\right) + \frac{\partial}{\partial y}\left(T_{yy}\frac{\partial h}{\partial y}\right) + \frac{\partial}{\partial z}\left(b\, K_{zz}\frac{\partial h}{\partial z}\right) = S^{1}\frac{\partial h}{\partial t} + bW(x,y,z,t) \qquad \text{(H-11)}$$

in which T_{xx} and T_{yy} are the principal components of the transmissivity tensor, and S^{1} is the storage coefficient. Although the model is designed to solve Equation H-10, it will solve Equation H-9 by substituting hydraulic conductivity, specific storage, and W(x,y,z,t) for transmissivity, storage coefficient, and bW(x,y,z,t), respectively. If the upper hydrologic unit is under water-table conditions, the specific yield is used to replace the storage coefficient in Equation H-10. The transmissivity in Equation H-10 is defined as a function of the head obtained from the previous iteration. That is,

$$T^{\,n}_{xx\ (i,j,k)} = K_{xx\ (i,j,k)} \bullet b^{n-1}_{i,j,k} \qquad \text{(H-12)}$$

where:

$b^{n-1}_{i,j,k}$ = the saturated thickness of the upper hydrologic unit at iteration n-1

n = iteration index

The required input data to simulate an aquifer under a stress of pumping are the transmissivity or hydraulic conductivity, storage coefficient or specific storage, initial head distribution, geometry of the hydrologic unit, dimension and layout of the finite-difference grid, length of pumping periods, number of pumping wells, pumping rates, and other simulation control parameters.

This model incorporates the following assumptions and limitations:

1. Aquifer properties can be heterogeneous and anisotropic.

2. Aquifer properties and hydrologic characters are uniform within each block of the model grid.

3. The perimeter of the aquifer should be described by a no-flow boundary.

4. Grid axes are parallel to the principal directions of the transmissivity tensor if the aquifer is anisotropic.

5. Head-dependent sources/sinks can also be simulated.

6. Darcy's Law can be applied in the porous media of the aquifers.

7. A simulated aquifer can be represented by such boundary conditions as constant head, constant flux, and head-dependent flux.

8. Only one horizontal anisotropy factor is allowed for each layer.

9. Overpumping can create an irreversible dry cell.

10. If the same aquifer is simulated by several layers and the water table is expected to traverse more than one layer, the cells can be converted incorrectly to no-flow cells.

11. Because the conversion to no-flow is irreversible, only declines in the water table can be simulated.

12. A confining layer with a given vertical hydraulic conductance is assumed to be below the water-table layer because vertical hydraulic conductance is left as a non-zero constant until the cell is converted to a no-flow cell.

McDonald and Harbaugh (1984) developed this modular, three-dimensional, finite-difference model to simulate groundwater flow in the porous medium. Their main objectives were to produce a program that can be modified readily, is simple to use and maintain, can be executed on a variety of computers with minimum changes, and is relatively efficient with respect to computer memory and execution time.

This model has been applied to a number of studies, including various aquifer and flow conditions in the A/M-Area and the Separations (F and H) Areas. In addition to the field application, this model also has been compared successfully with simplified analytical solutions. This model has better convergence than the quasi-three-dimensional model. In general, it has been appropriately validated, modified, and documented. Reliable results can be obtained, especially for aquifers in which the properties are ideally stratified and the groundwater flow in the porous medium can be modeled for the condition of confining beds.

MOD3D has been used in conjunction with SWIFT II for a number of groundwater flow and transport investigations. MOD3D provided the flow results; SWIFT II provided the contaminant transport results. The waste sites or locations studied were the A/M-Area, the F- and H-Area seepage basins, and the low-level radioactive waste burial ground. Published results of these field problems are not available at present. In addition, the published results of code verification of MOD3D are not available, even though the code has gone through the USGS review process. However, the model includes detailed mass balance algorithms to provide confidence in convergence and apportioning of sources and sinks.

H.1.3 SWIFT II

SWIFT II (Sandia Waste Isolation Flow and Transport for Fractured Media) (Reeves and Cranwell, 1981; NRC, 1986) is a general nuclide transport code to describe migration from the repository through the groundwater system. It is based on the finite-difference method, and solves not only for flow and solute transport but also for heat and brine transport. TE

The code simulates the flow and transport of energy, solute, and radionuclides in a geologic medium. SWIFT II is a three dimensional finite-difference groundwater flow and nuclide transport code. The model takes into account saturated flow in an isothermal or heated porous medium as well as sorption *and desportion mechanisms. In addition, the code takes into explicit account* nuclide decay and the creation of daughter products. For the nuclide decays, the code considers conservation of dissolved contaminants, energy, and total liquid mass. The fluid density can be a function of pressure, temperature, and concentration. Viscosity can also be a function of temperature and concentration. Aquifer properties can vary spatially. Hydrodynamic dispersion is described as a function of velocity. Boundary conditions allow natural water movement in the aquifer, heat losses to the adjacent formation, and location of injection, production, and observation points anywhere in the system.

SWIFT II solves four differential equations, together with a number of submodels describing the nonlinearities, in a sequential manner. Options include:

- Steady-state or transient flow
- Steady-state or transient density-dependent brine transport
- Solute transport
- Heat transport
- Dual porosity or discrete fracture-matrix
- *Salt dissolutioning*
- Well bore
- Radioactive waste-leach source
- Heterogeneous and/or anisotropic media
- Confined and/or water table conditions and recharge
- Recharge and/or wells

The code is fairly general and can be used to examine most farfield problems. It contains many options in terms of geometry, processes, and boundary conditions. Because it contains heat flow, it can also be used to examine some near-field problems.

SWIFT II is a general-purpose code and is applicable to most geologic media, including fractured rock. The main limitation would be due to the availability of data. It can be valid in many cases to perform a horizontal or vertical averaging. SWIFT II can still be used to perform a one- or two-dimensional simulation for this purpose.

Sensitivity analysis has been performed on both physical and numerical parameters.

Verification of numerical decay processes appear in the SWIFT II documentation. Verification of flow, heat, and solute transport also appear in which eight problems are documented.

Three field comparison problems for flow, heat, and solute transport have been performed (Colven et al., 1985).

H.2 ATMOSPHERIC TRANSPORT PATHWAY

Modeling calculations to determine potential risk to human populations due to atmospheric transport of waste materials have been made using a variety of computer codes. The pathway scenarios were inhalation of polluted air and ingestion of contaminated food by individuals and the offsite population. The occupational risk to personnel from airborne contaminants generated during actual waste site closure operations was included.

H.2.1 SOURCE TERMS

Atmospheric source terms for the site were estimated from soil inventories. Contaminants selected for atmospheric transport modeling were the same as
TC those analyzed for the subsurface transport exposure scenario (Looney et al., 1987). Atmospheric source terms account for volatilization of select contaminants (i.e., organics), dust generated by suspension of contaminated soil due to wind erosion, and dust generated as a consequence of excavation of contaminated soil from the site. The time-dependent nature of atmospheric source
TC terms was estimated to account for the time period of interest in this analysis. SESOIL (Bonazountas and Wagner, 1984), an EPA soil layer model, was used to estimate the soil contaminant concentration profiles as a function of time. SESOIL accounts for potential upward transport (volatilization) and downward movement (infiltration) of each contaminant for each remedial action. Airborne contaminant loadings are estimated using SESOIL and MARIAH (a National Oceanographic and Atmospheric Administration box model) (Holton et al., 1986). SESOIL estimates the amount of contamination entering the atmosphere over time from the site via volatilization. MARIAH estimates suspended dust loading to the atmosphere and excavation-generated dust loading due to digging, vehicular movement, and dumping. The source term for potential atmospheric transport away from the site - the contaminant loading due to dust - is the product of the dust loading and the contaminant concentration in the top soil layer.

H.2.2 TRANSPORT AND DOSE MODELS

The transport of waste constituents from a waste disposal facility to potential receptor sites through atmospheric dispersion was modeled using the XOQDOQ computer code (Sagendorf, Goll, and Sandusky, 1982). The XOQDOQ code is an NRC model that is used for routine release of atmospheric dispersion calculations to the SRP. The code was modified to handle area source terms. The XOQDOQ transport code uses a modified Gaussian plume model to estimate constituent concentration as a function of distance and direction from a waste site. Time-dependent source strength and meteorological conditions were input parameters.

The calculation of the transport of materials from the SRP by the atmosphere is based on meteorological conditions that are measured continuously at seven onsite meteorological towers and at a 365-meter television transmitting tower 30 kilometers northwest of the geometric center of the Plant. These meteorological measurements were to calculate the dispersive characteristics of the atmosphere by methods used in the nuclear industry (NRC, 1977a).

H.2.2.1 Nonradiological Exposures

After waste contaminant concentrations at potential receptor locations were determined, the results were translated into individual and population exposures. The maximally exposed individual at the site boundary and general population exposures to airborne substances via inhalation and ingestion pathways were determined. The CONEX computer code (Holton et al., 1986) uses XOQDOQ transport results and local population demographics to estimate time-dependent population exposures to nonradioactive airborne substances. The TERREX computer code (Holton et al., 1986) also uses XOQDOQ transport results along with local crop production data and local population demographics to estimate population data and local foodstuff uptake. The population demographics used in the CONEX and TERREX codes are estimated using a population growth model. Using census data from 1980 as the initial basis, the population growth model estimates the surrounding population from 1980 to 2050. After 2050, the population is assumed to be constant. After the end of the assumed 100-year period of institutional control (2085), the SRP site is assumed to be inhabited by the public. Hence, the air receptor is closer to the waste site at the end of the institutional control period.

Risk posed to the public population was calculated using a computer code called MILENIUM (Holton et al., 1986). For each potential airborne contaminant, the MILENIUM code translates time-dependent exposure results into a population dose and into a maximally exposed individual dose. The code uses the dose results and appropriate unit cancer risk (UCR) values and acceptable daily intake (ADI) factors (explained in Appendix I) to estimate excess risks for the population and a maximally exposed individual at the SRP boundary.

Risk posed to the worker involved in waste excavation activities was estimated using the MARIAH and MILENIUM computer codes. MILENIUM uses the source term results generated by MARIAH and appropriate UCRs and ADIs to estimate excess worker risk. A conservative assumption built into these models is that the occupational workforce would not use special protective clothing during waste excavation operations.

H.2.2.2 Radiological Exposures

To calculate the doses and corresponding human health risks associated with the atmospheric transport of radioactive waste materials, DOE used transport and dosimetry models developed for the nuclear industry. These models were developed by the U.S. Nuclear Regulatory Commission (NRC) and others for assessing the effects of operations of licensed commercial nuclear facilities (NRC, 1977b; ICRP, 1978). The radioactive transport and dose models have been implemented in the following computer programs:

- MAXIGASP: Calculates maximum and average doses to offsite individuals from atmospheric releases

- POPGASP: Calculates population doses from atmospheric releases

MAXIGASP and POPGASP are Savannah River Laboratory (SRL) modified versions of the NRC program GASPAR (Eckerman et al., 1980). The modifications enable the input of specific SRP physical and biological data. SRL did not modify the basic calculational methods used in the GASPAR program (Marter, 1984).

The pathway scenarios considered for the calculation of doses received by individuals and the offsite population are inhalation, ingestion, and exposure to direct radiation from material deposited on the ground.

DOE used the annual average concentration and deposition factors calculated with the XOQDOQ program in the MAXIGASP and POPGASP programs, along with data on population distribution, vegetable crop production, milk production, and meat production, to calculate offsite radiation exposure.

The direct gamma exposure pathway calculates the external radiation dose to an individual standing directly over a waste site. This scenario allows the cover material over the waste to erode at a specified rate so the degree of shielding provided by the cover can decrease in time. This pathway also assumes that no loss of contaminants occurs by leaching to the groundwater pathways. The time dependence of the source term is defined solely by radioactive decay.

H.2.2.3 Cumulative Radiological Effects

In evaluating the radiological impacts for the no-action alternative and during the first year after the implementation of the other three options, the cumulative effects of the operation of all nuclear facilities in the affected region also were considered. This region includes the Savannah River Plant and the area within 80 kilometers of the Plant.

The impacts from the following nuclear facilities, which represent existing and planned operations, were considered in calculating cumulative effects:

- The SRP, which includes four production reactors (L, P, K, and C) with associated support facilities, in addition to the low-level radioactive waste and mixed waste sites

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- The F- and H-Area Effluent Treatment Facility (ETF), to be constructed at H-Area on the Plant

- The Defense Waste Processing Facility (DWPF), under construction at S-Area on the Plant

- The Fuel Materials Facility (FMF), under construction at F-Area on the Plant

- The Fuel Production Facility (FPF), to be constructed at H-Area on the Plant

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- The Vogtle Electric Generating Station (Unit 1 is operating), Unit 2 is under construction across the Savannah River from the southwestern boundary of the Plant

- The Barnwell Nuclear Fuel Plant (BNFP; not operating) adjacent to and east of the Plant

- The Chem-Nuclear Services, Inc., low-level radioactive disposal site adjacent to BNFP (no releases expected)

Table H-1 lists the maximum individual and population doses associated with each of these facilities as base-case doses derived from documentation that summarizes doses for releases from each facility (DOE, 1986).

To estimate the cumulative impact of the operation of all nuclear facilities in the region, including each of the four waste management strategies, DOE combined the base-case doses in turn with the doses from the Dedication strategy, the Elimination strategy, and the Combination strategy. Because the dose from the No-Action strategy is included in the total base-case dose, the cumulative impact associated with that strategy would be the same as for the base case. TE

H.3 ENVIRONMENTAL DOSE COMMITMENT

Man can receive doses externally from radioactive materials outside the body or internally from the intake of radioactive material by inhalation or ingestion. Radionuclides that enter the body are distributed to various organs and are removed by normal biological processes and radioactive decay. The rate at which each radionuclide is removed from the body depends on its chemical, physical, and radiological properties. Historically, dose calculations have included an accounting of doses resulting from the fraction of radionuclides retained in the body for 50 years following the year of intake. The dose commitment factors used in these dose calculations include this 50-year integrating period.

Similarly, radioactive materials released in a given year remain in the environment for varying lengths of time, depending on many environmental factors and on the decay rate of each radionuclide. The environmental dose commitment (EDC) concept has been used to account for this activity.

EPA developed the EDC concept, defining the environmental dose commitment as "...the sum of all doses to individuals over the entire time period the material persists in the environment in a state available for interaction with humans." The EPA report presenting this concept (EPA, 1974) describes its implementation and presents some sample calculations. These calculations integrate doses for 100 years following radionuclide release rather than "the entire time period." This 100-year integrating period is distinct from the 50-year integrating period discussed above because it deals with the accumulation of doses from residual radioactivity in the environment rather than in the body.

This analysis uses the 100-year integrating period; in other words, all collective (population) dose calculations include an accounting of collective doses caused by environmental radioactivity levels for 100 years following each year's release. The 100-year period provides meaningful results by accounting for impacts over a period of time that is about equal to the maximum lifetime of an individual; thus, it provides a measure of risk to an individual. Longer integrating periods or an infinite time integral would require extremely speculative predictions about the human environment for thousands of years into the future.

Table H-1. Annual Cumulative Maximum Individual and Collective Doses from Atmospheric and Liquid Releases from Indicated Facilities

Dose	Release	Facilities						
		SRP[a]	ETF[a]	DWPF	FMF	FPF[b]	Vogtle[c]	Total
Annual maximum individual (millirem per year)	Atmospheric	1.5×10^{1}	-8.0×10^{-3}	7.0×10^{-4}	5.6×10^{-3}	4.0×10^{-5}	5.4×10^{-1}	1.6×10^{1}
	Liquid	1.1	7.2×10^{-2}	3.7×10^{-3}	6.5×10^{-4}	–	9.9×10^{-1}	2.2
	Combined	1.6×10^{1}	6.4×10^{-2}	4.4×10^{-3}	6.3×10^{-3}	4.0×10^{-5}	1.5	1.8×10^{1}
Annual collective (person-rem per year)	Atmospheric	1.1×10^{2}	-9.3×10^{-1}	9.4×10^{-2}	7.4×10^{-1}	4.1×10^{-3}	4.8×10^{-1}	1.1×10^{2}
	Liquid	3.2×10^{1}	1.1×10^{1}	5.4×10^{-1}	9.2×10^{-2}	–	–	4.4×10^{1}
	Combined	1.4×10^{2}	1.0×10^{1}	6.3×10^{-1}	8.3×10^{-1}	4.1×10^{-3}	4.8×10^{-1}	1.5×10^{2}

[a]The values in the SRP column include continued use of the F- and H- Area seepage basins. The values in the ETF column represent <u>changes</u> in doses resulting from operating the ETF rather than using the seepage basins. The sums of the dose values in the two columns represent SRP doses with the ETF in operation.
[b]There will be no radioactive liquid releases during normal FPF operations.
[c]Georgia Power Company, 1985.

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For the EDC calculations, changes in environmental characteristics were not predicted. Population size and distribution were based on the latest estimates. The analysis assumed that the historic meteorology would continue into the future and that food production and consumption patterns would be static.

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