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Development of a Multimedia Radionuclide Exposure Assessment Methodology for Low-Level Waste Management

Y. Onishi
G. Whelan
R. L. Skaggs

March 1982

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MANAGEMENT

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Richland, Washington 99352

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EXECUTIVE SUMMARY

Pacific Northwest Laboratory (PNL) and Los Alamos National Laboratory (LANL) are jointly developing a methodology for assessing exposures of the air, water, and plants to low-level waste (LLW) as a part of an overall development effort of a LLW site evaluation methodology. The assessment methodology will predict LLW exposure levels in the environment by simulating dominant mechanisms of LLW migration and fate. The methodology we describe here is more sophisticated than those for a basic worst-case scenario analysis, but less sophisticated than a full-scale, extremely detailed, and expensive and time-consuming onsite survey. The methodology consists of a series of physics-based models with proven histories of success; the models interact with each other to simulate LLW transport in the ecosystem. A scaled-down version of the methodology was developed first by combining the terrestrial ecological model, BIOTRAN; the overland transport model, ARM; the instream hydrodynamic model, DKWAV; and the instream sediment-contaminant transport model, TODAM (a one-dimensional version of SERATRA).

The methodology was used to simulate the migration of ^{239}Pu from a shallow-land disposal site (known as Area C) located near the head of South Mortandad Canyon on the LANL site in New Mexico. The scenario studied here assumed that ^{239}Pu would be deposited on the land surface through the natural processes of plant growth, LLW uptake, dryfall, and litter decomposition. Run-off events would then transport ^{239}Pu to and in the canyon. The model provided sets of simulated LLW levels in soil, water and terrestrial plants in the region surrounding the site under a specified land-use and a waste management option. Over a 100-yr simulation period, only an extremely small quantity (6×10^{-9} times the original concentration) of buried ^{239}Pu was taken up by plants and deposited on the land surface. Only a small fraction (approximately 1%) of that contamination was further removed by soil erosion from the site and carried to the canyon, where it remained. Hence, the study reveals that the environment around Area C has integrity high enough to curtail LLW migration under recreational land use.

The methodology demonstrated that it is capable of providing a reasonably accurate assessment of the ability of specific LLW sites to confine the release of LLW. The methodology is currently being expanded to include atmospheric and groundwater compartments.

1.0 INTRODUCTION

Legislators and private citizens are very concerned about the release of radionuclides from low-level waste (LLW) disposal sites into the environment and the danger that these releases present to humans and the environment. Currently, the U.S. Nuclear Regulatory Commission (NRC) is preparing regulations that will define the necessary conditions for adequately disposing of LLW (NRC 1979). The objectives of the LLW management program are to locate, operate, close, and maintain LLW disposal facilities in such a way that LLW is contained within the facilities, and to ensure that potential releases do not violate the environmental standards and criteria (emphasizing long-term protection) set by the NRC and the U.S. Environmental Protection Agency (EPA).

The regulatory standards can be met if the disposal sites have natural features that impede waste migration, or if artificial barriers are created to have the same effect. To evaluate the ability of a disposal site to contain LLW over a long period of time (say several hundreds of years) under present and future land use conditions, a LLW site-evaluation methodology must predict the long-term integrity of environment in curtailing LLW migration in the biosphere. LLW released from the disposal site undergoes complex interactions in the biosphere. Since mathematical models can integrate these complex, interactive mechanisms into a single system, they must be an integral part of a LLW site-evaluation methodology.

Currently, Los Alamos National Laboratory (LANL) and Pacific Northwest Laboratory (PNL) are jointly developing a LLW site-evaluation methodology. The overall effort encompasses the development of: 1) a usable and credible LLW site-evaluation methodology for present and future land use around specific LLW sites; 2) long-term scenarios; and 3) a multimedia LLW exposure-assessment methodology. This report describes work performed for the effort to develop a multimedia LLW exposure assessment methodology.

Obtaining an accurate evaluation of a LLW site is very important, because overestimation of the environmental exposure of LLW may result in unnecessary expenses and unreasonable restrictions on LLW management. On the other hand,

underestimation may result in irreversible damage to human health and the environment. In principal, the more accurate the evaluation, the more extensive the evaluation effort. The current joint effort by PNL and LANL to develop a multimedia LLW exposure assessment methodology is designed to provide helpful and moderately accurate prediction at reasonable cost. Elementary worst-case scenario analyses (e.g., Science Application Inc. 1981 and many man-dose models such as Fletcher et al. 1973 and Soldat et al. 1979) are too simple for the purpose of a reasonably accurate assessment, but the extremely detailed site survey programs are too expensive and time consuming. The goal is to develop an assessment methodology that permits: 1) the consideration of site-specific conditions and characteristics, 2) the simulation of realistic environmental transport mechanisms, and 3) comparative evaluation of LLW disposal consequences over many hundreds of years. To arrive at a suitable system for simulating the dynamics of LLW transport in the ecosystem, we assembled several models at appropriate interfaces (Figure 1). Model selection was guided by the following four principal considerations:

1. the models must be capable of simulating dynamic environmental processes with a moderate degree of resolution in space and time
2. the models must be generic but, at the same time, capable of being calibrated on a site-by-site basis. They should not require excessive data or require data that are difficult, expensive, or time consuming to acquire
3. the models must be capable of being coupled to each other so that the simulation includes LLW migration in multimedia and the ensuing aspects of exposing humans to the waste
4. all of the models must have a proven history of success.

Models representing terrestrial plants, soil, surface water, and aquatic sediment (Figure 1) were selected as integral parts of the assessment methodology. The models are BIOTRAN, a terrestrial ecology model (Wheeler et al. 1977 and Gallegos et al. 1980); ARM, an overland transport model, (Donigian

FIGURE 1. Multimedia Radionuclide Exposure Assessment Methodology

and Crawford 1976); DKWAV, an instream hydrodynamic model (Whelan 1980); and TODAM, an instream sediment-contaminant transport model, which is a simplified version of SERATRA (Onishi et al. 1978).

Prior to the full development of the proposed assessment methodology, this scaled-down version of the multimedia methodology was used to simulate the migration of ^{239}Pu from Area C, a shallow-land disposal site located near the head of South Mortandad Canyon on the LANL site in New Mexico. The scenario studied herein assumed that through natural processes of plant growth and radionuclide uptake, dryfall, and litter decomposition, ^{239}Pu would be deposited on the land surface. Runoff events would then transport ^{239}Pu to and in the canyon. Hence, the terrestrial-plant, soil, surface-water, and aquatic-sediment compartments of the methodology were employed. South Mortandad Canyon lies within the drainage field of Mortandad Canyon, which is one of many canyons dissecting Pajarito Plateau in north-central New Mexico. The intermittent streams draining Mortandad and South Mortandad Canyons are narrow and steep, with thin layers of alluvial sediment. During major summer thunderstorms, surface flows may traverse a few thousand meters in the canyon channels before disappearing in the alluvium (Hakonson et al. 1976).

The purpose of this application effort was to demonstrate and evaluate the attributes of the methodology. Specifically, we:

- examined the availability of data required by the methodology and the practicability of the data requirements of the methodology
- determined the adequacy of the methodology's level of sophistication in light of current understanding of how LLW migrates and persists in biotic and abiotic media, and in light of the uncertainty over data and predicted results
- examined the coupling of several components of the methodology and the ability of the integrated methodology to simulate LLW migration and fate in several media
- evaluated the relative significance of LLW pathways and various transport and decay mechanisms
- examined the applicability and limitations of the methodology.

The terrestrial ecology modeling was performed by LANL, and the overland and instream modeling was performed by PNL. This report mainly describes the work performed by PNL. Chapter 2 presents the conclusions and recommendations of the study. Chapter 3, briefly discusses the major mechanisms of LLW transport and fate occurring on land surfaces, in surface water and in the atmosphere. Transport occurring in groundwater and terrestrial plants is also briefly discussed. Chapter 3 also gives a brief evaluation of available models applicable to these environments, the rationale behind model selection, and a description of selected models. Chapter 4 describes the results of the scaled-down methodology as applied to Mortandad and South Mortandad Canyons.

2.0 CONCLUSIONS AND RECOMMENDATIONS

Because the migration and fate of LLW in a given environment are controlled by a complex set of mechanisms, the model used to simulate natural events must itself be complex enough to simulate transport mechanisms, decay, intermedia transfer and biological uptake to predict LLW exposure levels. The model also provides a standardized approach for examining the complex and interdependent processes of LLW transport.

A scaled-down version of the methodology consisting of BIOTRAN, ARM, DKWAV and TODAM models simulated the migration of ^{239}Pu (with the assumed source strength of 800 pCi/g) from Area C. After 100 yr of simulation with BIOTRAN, radionuclide concentrations on the land surface of the waste-disposal site were predicted to be 5×10^{-6} pCi/g or 6×10^{-9} times the original ^{239}Pu concentration disposed of in the ground under recreational land use. Over the 2-yr simulation period following the 100-yr BIOTRAN simulation, 1% of ^{239}Pu on the land surface had been removed by soil erosion events and transported to the canyon. The simulation also revealed that, once in the canyon, up to 99% of the ^{239}Pu was carried by sediments during occasional runoff events. Apparently, ^{239}Pu has a high affinity for being adsorbed by sediment. Simulation results revealed that all of the ^{239}Pu eroded from Area C remained in the canyon. Depositions of sediment in the channel cause some concern, because radioactive contaminants might concentrate in the bed sediments and serve as a wide-spread, long-term source of contamination.

The following conclusions and recommendations were derived from this study:

1. The multimedia LLW exposure assessment methodology discussed here has demonstrated that it is capable of evaluating how specific sites curtail LLW migration.
2. The methodology could be used as:
  - a scientific decision-making tool for regulating the management of LLW. The methodology might also be used to estimate the probable levels of LLW exposure to humans and the environment.

- a research tool for evaluating the relative effectiveness of natural features at the site in curtailing LLL migration to the biosphere, or of purposeful strategies, e.g., specific engineered barriers, special disposal methods, or other controls.
  - a research tool for estimating the relative significance of environmental pathways and LLW transport and fate mechanisms
  - a means of providing a rational base for analyzing field sampling data.
3. Since the methodology is physics-based and simulates dominant dynamic processes in the environment, its data requirements are rather extensive, and it requires moderate effort to apply. The data requirements were judged reasonable, however, because the methodology is intended to provide a reasonably accurate LLW exposure assessment at selected specific-sites and is not intended as a preliminary screening tool for a great number of potential LLW sites. Most of the required data are available, but, as is the case in applying the methodology at the LANL site, the quantity of available data is rather limited. Some required data must be synthesized from other available data. Field data collection and laboratory experiments should be coordinated more extensively with mathematical models, and the current methodology should optimize the data generating efforts for LLW site assessment.
  4. The models BIOTRAN, ARM, DKWAV, and TODAM are (roughly) equally sophisticated, but their time frames and land-surface areas under consideration differ. Since the methodology is currently being expanded to include atmospheric and groundwater compartments, the framework for coupling the models must be improved. For example, BIOTRAN, ARM, and the unsaturated groundwater model must share a common computational scheme for precipitation, infiltration, and excess runoff to ensure the water balance. We also recommend that TODAM will be combined with an appropriate hydrodynamic model to

form one model representing the surface-water and aquatic sediment compartments. These compartments are needed to update changes in flow conditions, cross-sectional shapes, and bed slopes due to bed and bank erosion. This feature is especially important for a long-term predictions.

5. Limited data were used to partially test the methodology's validity. Additional tests are strongly recommended.

3.0 MULTIMEDIA RADIONUCLIDE EXPOSURE ASSESSMENT METHODOLOGY

The proper management of LLW sites relies on information about release mechanisms and pathways and the volume of radioactive waste at a site. Mathematical models used to evaluate risks to man and biota must integrate all of the complex mechanisms that control radionuclide fate--i.e., contaminant transport; decay processes and the production of daughter products; transfers between or among environmental media such as air, water, and soil; and biological uptake. This chapter discusses the major mechanisms of radionuclide transport and fate on land, in surface waters, and in air, and evaluates the applicability and limitations of the models. The interface between radionuclide transport and groundwater and terrestrial plants is also briefly discussed. The chapter ends with a brief description of the models selected for this study.

3.1 RADIONUCLIDE TRANSPORT ON LAND SURFACES

3.1.1 Physical Mechanisms of Radionuclide Transport on Land Surfaces

Radionuclides released to a land surface may, as a result of subsequent rainfall or snowfall, migrate into surface water systems. Migration can occur by several pathways, including overland runoff, subsurface runoff or flow through shallow soils, groundwater flow due to infiltration and percolation, and erosion and transport of soil particles. The importance of each pathway depends largely on the individual radionuclides. While highly adsorptive radionuclides tend to move mainly with the sediment, highly soluble radionuclides follow the water pathways.

Hydrology

During a storm, precipitation separates into precipitation excess and loss. Precipitation excess is the portion of the total precipitation that forms surface runoff, traveling toward a receiving surface-water body over the ground surface (overland flow). Precipitation loss is the remaining portion of total precipitation that does not become surface runoff. Examples of

precipitation loss are infiltration, interception, evaporation, transpiration, and depression storage. Infiltrating precipitation can make its way to a stream through subsurface flow, moving laterally through the upper soils, as groundwater flow, or as percolation to the groundwater table (Chow 1964).

The distribution of precipitation into each of the components described is dependent on many factors, most of which can be classified into two major categories: climatic factors and physiographic factors. Climatic factors include the amount and distribution (spatial and temporal) of precipitation; the presence of snow and ice; and the effects of temperature, wind, solar radiation, and humidity on evapotranspiration and snowmelt. Examples of physiographic factors are the size, shape, slope, orientation, elevation, and drainage density of a watershed, plus such factors as land use, type of cover, soil type and the topography of the watershed (Chow 1964).

Erosion

Soil erosion basically involves the detachment and movement of soil particles into a channel. Aspects of the process are sheet/rill erosion, gully erosion, and channel erosion. Sheet/rill erosion is the critical mechanism governing the removal of radionuclides from the top few inches of a land surface.

Sheet erosion is the gradual and relatively uniform removal of topsoil from land surfaces. As raindrops strike the land, fine soil particles become detached from soil aggregates and are dispersed in every direction. On a slope, raindrop splashes tend to move the soil particles a short distance downhill. The splashes' main effect, however, is to produce soil fines that are available for transport downslope by overland flow. The soil fines are either entrained by the overland flow or added to the flow by raindrop splashes. The soil fines are transported downslope until the flow forces are diminished to the extent that transport is no longer possible and deposition occurs. Subsequent overland flow will pick up the deposited fines and transport them further downslope. This process is repeated until the soil particles are discharged into a stream channel where they are added to the sediment load of the stream (Crawford and Donigian 1973).

Radionuclide Transport

The interactions between radionuclides and the soil greatly influence the migration of radionuclides from land surfaces. Radionuclide adsorption to soil particles provides the means of transport to stream channels for many radionuclides. Furthermore, the strength of the adsorption determines the importance of overland, subsurface, and groundwater flow as mechanisms of transport for soluble radionuclides. The partitioning of the radionuclide between soil and water can also affect the rate at which radionuclides are transferred to the atmosphere through volatilization.

Among the many factors that influence the adsorption-desorption of radionuclides are the type of radionuclide, its chemical form and oxidation state, the sediment's particle size, the concentration and mineralogic content of the suspended sediment, and various environmental conditions such as temperature, dissolved oxygen content, concentration, pH, oxidation-reduction potential, salinity, presence of competing ions, mixing characteristics, and pollutional stresses. The effects of these various factors have been illuminated to a certain extent by the work of Gloyna et al. (1963-1972). Sayre et al. (1963) have determined that sorption capacity of sediments is roughly proportional to the available surface area. Generally, the sediments eroded from the land surface are minute particles the size of silt and clay fines. As pointed out by Sayre et al. (1963), the total surface area of 1- μ diameter clay is 1,000 times that for 1-mm-diameter sand for a given weight. Nelson et al. (1966) showed that small silt- and clay-sized particles contain much greater concentrations of radionuclides than do coarser particles.

3.1.2 Mathematical Models for Overland Radionuclide Transport

A number of mathematical models encompassing a broad range of complexity can be used to estimate the migration of radionuclides from land surfaces. These models have been developed and applied mainly to the movement of pesticides from agricultural lands. Because radionuclides behave much like other contaminants in the environment, these models can generally be modified to handle radionuclides as well.

The available models can be separated into three groups based on their complexity and the extent to which they represent the physical processes involved. The simplest models involve the use of an empirical equation to estimate the average soil loss from a catchment. Since many radionuclides are adsorbed to eroded soil materials, transport of these radionuclides can be estimated by multiplying sediment yields by factors denoting the radionuclide concentrations or activities in the soil. The models require only limited data on such things as watershed characteristics and loading factors and can be applied with ease and expedience but, in some cases, with questionable accuracy.

A more complex group of models simulates the hydrologic processes that separate precipitation into overland, subsurface, and groundwater flow, and lead to the erosion of soils. Similar to models of the first group discussed above, the resulting water or sediment runoff (or both) are multiplied by a loading factor to determine the number of radionuclides transported from the watershed. These models require considerable quantities of detailed hydrologic and meteorologic data and information on watershed characteristics, land use, and loading factors. These complex models are generally more accurate, at least conceptually, than the previously described group of models.

A final group of models, in addition to simulating rainfall-runoff and soil erosion, simulates adsorption-desorption of radionuclides, and their loss due to radionuclide decay, and the migration of both dissolved and sorbed radionuclides to a stream's edge. In addition to requiring the same data as the second group of models, these models also require detailed information about the chemical characteristics of radionuclides on the land surface and their antecedent distribution. Proper use of these models should provide the highest degree of accuracy.

An exhaustive review of the many models in each of the above groups is beyond the scope of this report. However, an example for each group is briefly described below to guide the selection of a overland model for this study.

GROUP I: McElroy et al. (1976)

McElroy et al. (1976) developed a simple loading function to compute the loadings of heavy metals attached to sediment. The loading function, which assumes that the amount of heavy metal delivered to the stream is directly proportional to the sediment load, is as follows:

$$Y(HM)_S = a C_S(HM) Y(S)_E \quad (1)$$

where

$Y(HM)_S$ = yield of heavy metals in sediment, kg/day (lb/day)

$C_S(HM)$ = concentration of heavy metals in eroded soil, ppm

$Y(S)_E$ = sediment yield in metric tons/yr (tons/yr)

a = conversion factor to obtain proper units of load. If $Y(S)_E$ is expressed in metric tons/yr, $a = 2.74 \times 10^{-6}$; if $Y(S)_E$ is expressed as English tons/yr, $a = 5.48 \times 10^{-6}$.

To predict the discharge of radionuclides, $C_S(HM)$ is substituted for the yield of radionuclides in sediment, $C_S(RAD)$, in the equation above.

The information needed to estimate radionuclide discharges includes the amount of sediment produced, $Y(S)_E$, and the concentrations of radionuclides attached to the eroded soils, $C_S(RAD)$. $C_S(RAD)$ is determined from field measurements whereby radionuclide concentrations divided by sediment concentrations for a storm or single sample basis provide estimates of the factor. $Y(S)_E$ is computed using the modified version of the Universal Soil Loss Equation originally developed by Wischmeier and Smith (1966), as described below:

$$Y(S)_E = \sum_{i=1}^n A_i (R K L S C P S_d)_i \quad (2)$$

where

$Y(S)_E$ = sediment loading from surface erosion, tons/yr
n = number of sub-areas in the area

Source areal factor:

A_i = acreage of sub-area, i, in acres

Source characteristic factors:

R = rainfall factor, expressing the erosion potential of average annual rainfall in the locality, is a summation of the individual storm products of the kinetic energy of rainfall in hundreds of ft-tons/acre and the maximum 30-min rainfall intensity in in./hr for all significant storms on an average annual basis

K = soil erodibility factor, commonly expressed in tons/acre/R unit

L = slope length factor, dimensionless ratio

S = slope gradient factor, dimensionless ratio

C = cropping management factor, dimensionless ratio

P = erosion control practice factor, dimensionless ratio

S_d = dimensionless sediment delivery ratio.

The reliability of estimating radionuclide discharges through the procedure discussed above is a function of three factors: 1) the accuracy of the sediment loads delivered to the stream, 2) the accuracy of the soil radionuclide concentration measurements or estimates, and 3) the variability of radionuclide concentrations in the eroded soils. Of these three factors, the one concerning variability of radionuclide content will be the most uncertain. Additionally, since Equation (1) can compute only a sorbed radionuclide, it can be used only for radionuclides with a high affinity to sediment.

GROUP II: Donigian and Crawford (1979)

The nonpoint source code (NPS) is a continuous hydrologic simulation model that predicts runoff and sediment and nonpoint pollutant contributions to stream channels (Donigian and Crawford 1979). The two major components of the model--LANDS and QUAL--simulate the hydrologic response of the watershed; and sediment production, sediment transport, and pollutant transport; respectively. Basically, LANDS is an accounting procedure for surface and subsurface moisture. It simulates all flow components (overland, subsurface, and groundwater flow) and moisture storage by representing the processes of interception, infiltration, percolation, evapotranspiration, and snow accumulation and melt.

QUAL simulates nonpoint source pollution from up to five different land uses. Processes include accumulation and removal of sediment through dustfall and human activities, erosion of soil particles by rainfall detachment and transport by overland flow, and production of soil fines available for transport by tillage operations. The pollutant loadings to a receiving stream are represented as follows:

$$\text{POLP}(t) = \text{ERSM}(t) \cdot \text{PMP} \quad (3)$$

where $\text{POLP}(t)$ is the mass of pollutant transported during time interval, t ; PMP is the potency factor; and $\text{ERSN}(t)$ is the sediment loss to the stream during time interval t .

To determine radionuclide discharge, the model uses potency factors applicable to radionuclides. Field measurements are the best means of evaluating the potency factors.

Proper application of this or similar models involves several major steps: 1) data collection and analysis, 2) preparation of meteorological data, 3) evaluation of model parameters, and 4) model calibration and verification. Data requirements include detailed meteorological information, physical watershed characteristics, and observed runoff and water quality.

While requiring considerably more effort, the level of accuracy obtained through the use of models in this group is much greater than for those of the first group. Group II models integrate the significant physical mechanisms to provide a continuous accounting of rainfall-runoff and soil erosion-deposition. This provides a better foundation on which to estimate radionuclide discharge than the average sediment yield values used by Group I models.

GROUP III: Crawford and Donigan (1973)

The Agricultural Runoff Management Model (ARM) developed by Crawford and Donigan (1973) to simulate pesticide migration and fate is essentially the same as the NPS model except in the way it handles the transport of contaminants. Developed for agricultural watersheds, ARM uses a modified Freundlich adsorption/desorption equation to determine the amount of pesticide dissolved in the overland flow and the amount of sorbed pesticide attached to soil fines being transported by the flow. Both single-valued and non-single-valued formulations of this equilibrium-type algorithm have been included. The relationship used in ARM to model pesticide adsorption and desorption in soil is as follows:

$$X/M = K C^{(1/N)} + F/M \quad (4)$$

where

X/M = contaminant adsorbed per unit soil, mg/gm

F/M = contaminant adsorbed in permanent fixed state per unit soil;

F/M is less than or equal to FP/M , where FP/M is the permanent fixed capacity of soil in mg/gm for contaminants. This can be approximated by the cation or anion exchange capacity for that particular soil type.

C = equilibrium contaminant concentration in solution, mg/ml

N = exponent

K = coefficient.

The contaminant degradation component of ARM determines the quantity of contaminant available for transport, either by overland flow or by soil erosion, at any time after its application to a watershed. The major processes that act to attenuate contaminants in the soil are volatilization and microbial, chemical, and photochemical degradation. Because of the way the adsorption-desorption and pesticide degradation elements are represented, the model can study radionuclides simply by using appropriate coefficients for the radionuclides of interest.

Application of ARM or other Group III models (Patterson et al. 1974, Frere et al. 1975, Roesner et al. 1975) requires additional effort beyond that of Group II models to determine appropriate coefficients for the contaminant transport and degradation (or decay) and to perform the calibration for the contaminant transport as well as for the hydrology and sedimentation aspects. This additional effort is offset by a better representation of the contaminant transport mechanisms, which potentially enhances the accuracy of predictions produced by the model.

3.2 RADIONUCLIDE TRANSPORT IN NON-TIDAL RIVERS

The models reviewed in this section were designed to simulate radionuclide transport in particulate or dissolved forms (or both) in nontidal rivers as reported by Onishi et al. (1980).

3.2.1 Physical Mechanisms of Radionuclide Transport in Non-Tidal Rivers

Radionuclides discharged to rivers are dispersed by currents and turbulent mixing. These radionuclides may then be absorbed from solution onto sediments and biota, producing sorbed radionuclide concentrations much higher than dissolved concentrations. This process may create a significant pathway to man. Most of the available methods for evaluating radionuclide transport in surface waters do not account for radionuclide uptake or desorption into water. Furthermore, these methods do not account for transport, deposition and resuspension of contaminated sediments. Models that do not include sorption effects on radionuclide transport predict that radionuclides will be flushed from surface waters at the same rate at which water is exchanged (US NRC 1977). In

actuality, sorption effects would cause some radionuclides to flush much more slowly, at approximately the rate at which the stream components are exchanged in the water system (US NRC 1978a,b; US EPA 1978). Hence, in some cases, neglecting the detention effects precludes making accurate predictions about radionuclide distributions in time and space. This section reports the results of field and laboratory studies which document the importance of the above-mentioned physical and chemical mechanisms affecting radionuclide transport in rivers. The importance of sediment-radionuclide interaction on radionuclide migration is illustrated by Churchill et al. (1965).

Hydrodynamic Transport

Hydrodynamic transport is the principal transport mechanism for radionuclides; its primary aspects are advection and diffusion/dispersion. In the Clinch River system, Parker et al. (1966) found from field studies and mass-balance analyses that most of the radionuclides being discharged from operations at the Oak Ridge National Laboratory passed through the water phase as dissolved radionuclides, with the exception of ^{137}Cs , which has a large distribution coefficient.

Suspended Sediment

Churchill et al. (1965) noted that radionuclide field measurements obtained from the Clinch River near Oak Ridge National Laboratory in Tennessee in the early 1960s indicated that approximately 90% of ^{137}Cs released was adsorbed by the river's suspended sediment within a 10-mile reach downstream of the effluent discharge. As indicated above, sediment plays an important role in radionuclide transport in rivers. One of the earliest investigations to determine the importance of uptake of radionuclides by sediment was conducted by Carritt and Goodgal, who studied the uptake of various radionuclides in solutions containing suspensions of Roanoke River and Chesapeake Bay sediments, bentonite, fuller's earth, and powdered Pyrex® glass (Sayre et al. 1963). The percentage of radionuclides sorbed by sediments ranged between 10 and 90%. In addition, one of their tests showed that sorption by the sediment particles resulted in particulate radionuclide concentrations of >10,000 times those in the surrounding solution.

Tritremmel et al. (1966) conducted tests to examine the uptake of various fission products by typical suspended sediment loads. They found that ^{90}Sr , ^{131}I , ^{140}La , and other fission products tested were adsorbed fairly rapidly on suspended sediment but were not permanently fixed. Results indicated that 50% of the radionuclides may have redissolved prior to equilibrium conditions. From their data they estimated that sorption by suspended sediment may reduce the quantity of radionuclides in reactor effluents discharged into the Leitha-Muhl Canal in Austria by 25 to 30 percent.

In field studies conducted on the Clinch and Tennessee Rivers, Pickering et al. (1966) found that bottom sediment in the Clinch River contained 21% of the ^{137}Cs , 9% of the ^{60}Co , 0.4% of the ^{106}Ru and 0.2% of the ^{90}Sr released to the river during nearly 20 years of operation at the Oak Ridge National Laboratory. Moreover, based on vertical distributions of radionuclide concentrations in the bed sediment, they concluded that the ^{137}Cs and ^{60}Co were incorporated in the bottom sediment by adsorption and by deposition of suspended sediment.

Existing data indicate that some radionuclides are adsorbed to river sediment rather quickly and then migrate with sediment movements. Deposition of suspended sediment with adsorbed radionuclides can result in a build-up of radioactivity if the rate of accumulation through deposition exceeds the rate of natural radioactive decay. Moreover, some of the contaminated sediment could be resuspended, particularly during periods of high flow, and then be redeposited further downstream. Thus, an understanding of sediment transport mechanisms is essential to the prediction of radionuclide concentrations.

Bottom Sediment

Direct sorption by bed sediment appears to have a much smaller effect on radionuclide transport than does sorption by suspended sediment. In hydraulic flume experiments, Bhagat and Gloyna (1965) found that suspended matter subsequently deposited to the bed significantly increased the concentration of ^{103}Ru in the bottom sediment. Only less than 5% of the total ^{103}Ru in the form of $^{103}\text{RuNO}$ was adsorbed directly by bottom sediment. When kaolinite clay suspensions were added, as much as 50% of the $^{103}\text{RuNO}$ in the water was

adsorbed by the suspended matter. As a result of deposition, the amount of $^{103}\text{RuNO}$ in the bottom sediment increased from 1% to 15% of the total ^{103}Ru discharged into the flume.

Gloyna and his associates (Reynolds and Gloyna, 1963; Rowe and Gloyna, 1964; Clanton and Gloyna, 1964, Yousef and Gloyna, 1964) determined from additional studies that sorption by bed sediment is dependent on contact time, the mineralogical and chemical composition of the sediment, and on physical characteristics. They also found that uptake by bed sediment is slow and only partially reversible.

Yousef et al. (1970) pointed out that radionuclide migration within the bed is limited to near the bed's surface, while activity in deeper layers results from physical processes such as sediment deposition and erosion. Gloyna and others (Rowe and Gloyna 1964; Yousef and Gloyna 1964; Bhagat and Gloyna 1974; Shih and Gloyna 1966) investigated the migration of radionuclides to deeper layers of bed sediments and found the specific activity of the bed sediments decreased exponentially with depth. After 15 to 30 days of contact time, the depth at which the specific activity of the sediment was 0.1% of that near the surface was about 2 cm for ^{65}Zn and 14 cm for ^{137}Cs (Gromiec and Gloyna 1973). Data on radionuclide concentrations in the Columbia River also indicate that radionuclides penetrate into the bed's sediments (Gloyna and Ledbetter 1969).

Results of analyses performed by Brisbin et al. (1974) showed that over 68% of the total ^{137}Cs was located in the upper 20 cm (7.9 in.) of bed sediment in Steel Creek near Savannah River Laboratory during nine years of plant operation (1961 to 1970). Experimental data collected by Mahmood et al. (1973) using bottom sediment from the Ismailia Canal in Egypt indicated that for both flowing and nonflowing ecosystems, the ^{60}Co incorporated in the bottom sediment was detected mainly in the upper layer, while ^{134}Cs and ^{89}Sr migrated to deeper layers.

Biota

Mahmood et al. (1973) found through laboratory investigations that the maximum uptake of ^{89}Sr , ^{60}Co , and ^{134}Cs by various aquatic plants was reached

after 1 to 4 days. A major portion, ranging from 45% to 85% of the sorbed radionuclides, was released after 2 to 4 days.

Armstrong and Gloyna (1967), Bhagat and Gloyna (1964), and Rowe and Gloyna (1964) found that sorption by plant surfaces is rapid and almost completely reversible. They concluded from their experimental results that plants play an insignificant role in radionuclide transport, except perhaps in slow-moving and enriched streams and impoundments. Gloyna and Ledbetter (1969) noted that radionuclide concentration data from the Clinch River indicate that the effects of plant uptake may be neglected in a mass-balance analysis of radionuclides.

The studies described above reveal that the transport mechanisms of advection and dispersion are the most important mechanisms controlling the migration and fate of radionuclides. The studies above also indicate that sediment effects on radionuclide transport become important 1) if sediment concentration is high, 2) if the radionuclides have a high affinity to sediment, or 3) if long-term transport and accumulation of the radionuclides are of concern. Under these conditions, if sediment effects are not considered, there is significant potential for erroneous evaluations of radionuclide distributions in surface water. The decay of radionuclides becomes an important factor if the radionuclide has a short half-life or if its long-term transport and fate are of concern.

3.2.2 Mathematical Models for Rivers

The greatest number of radionuclide-transport models are those developed for rivers. The models range from those specifically developed to predict radionuclide concentrations in a river to those developed as a part of a man-dose model.

The radionuclide-transport models for rivers cover a wide range of complexity and use. These models may be divided into three groups based on their model sophistication and applicability. (A detailed discussion of the models can be found in Onishi et al. 1980.) Group I contains models that are a part of a larger man-dose model. These models generally incorporate dilution of the material by the flow volume of the river and calculate decay of the radionuclide according to the travel time to a specified downstream point. A

few of the models predict sediment concentrations but do not account for reductions in dissolved radionuclide concentration by adsorption. Models in this group require the least amount of data and are very easy to apply. However, models in this group are the least accurate in representing the real phenomena of radionuclide transport and fate as compared to models in other groups.

Most of the other models (Groups II and III) predict a river's dissolved radionuclide concentration by solving the advection-dispersion equation. Models in Group II consider many transport and fate mechanisms but do not consider such complete sediment-radionuclide interactions as adsorption/desorption, nor the transport, deposition and resuspension of particulate radionuclides sorbed by sediment. Governing equations with associated initial and boundary conditions are solved either analytically or numerically. These models require considerable quantities of data on channel and flow characteristics and may require information on sediments if sediment-radionuclide interactions are part of the model. These models can predict radionuclide distributions more accurately than models in Group I. Group II models are more suited to calculate distributions of radionuclides when dissolved radionuclides are dominant.

Only a few models belong in Group III. These models simulate both dissolved and particulate radionuclide transport by fully considering adsorption/desorption mechanisms and dynamic transport, deposition, and scouring of sediment. These models use numerical schemes (e.g., finite-difference and finite-element methods) to solve their governing equations with associated initial and boundary conditions. These models are the most versatile tools for evaluating the migration and fate of radionuclides under a wide range of conditions. They are expected to provide the most accurate predictions because, conceptually, these models are more sound than those in Groups I and II. However, their input data requirements and running costs and time are high.

A summary of these models is given in Table 1 (reprinted from Onishi et al. 1980) and representative models from each group will be discussed below.

TABLE 1. Summary of Radionuclide Transport Models (Onishi et al., 1981)

Author and/or Radionuclide Transport Models	Modeling Substances			Mechanisms				Hydrodynamics Simulation	Hydrodynamics Simulation	No Spatial Dimensions	Dimensionality			Time Dependence		Solution Technique*	Applicable Water Body**	Field Application
	Dissolved Radionuclide Transport	Particulate Radionuclide Transport	Sediment Transport	Advection	Diffusion/ Dispersion	Absorption of Radionuclides	Radioactive Decay				One- Dimensional	Two- Dimensional	Three- Dimensional	Steady- State	Dynamic			
Fletcher and Dotson, 1971	X	X	X	X			X				X				X	FD	R,L	
Bramati et al., 1973	X			X		X					X			X		A	R,L	X
Soldat et al., 1974	X			X		X					X			X		FD	C,E,R,L	X
Watts, 1976	X			X			X				X			X		A	R	X
Martin et al., 1976	X			X			X				X				X	A	R,L	X
Buckner and Hayes, 1976	X	X		X	X	X	X				X				X	FD	R	X
Shih and Gloyna 1976	X	X		X	X	X					X				X	A	R	
Armstrong and Gloyna, 1968	X	X		X	X	X					X				X	FD	R	
White and Gloyna, 1969	X	X		X	X	X					X				X	FD	R	
Shull and Gloyna, 1968	X	X	X	X		X					X			X		A	R	
Onishi et al., 1976, 1977, 1978, 1979																		
FETRA	X	X	X	X	X	X	X					X		X	X	FE	C,E,R	X
SERATRA	X	X	X	X	X	X	X					X			X	FE	R,L	X
TODAM	X	X	X	X	X	X	X	X	X		X			X	X	FE,FD	R,E	X
Onishi and Whelan (Onishi et al., 1980)	X	X	X	X		X	X				X				X	FD	R,L	X
Falco, Onishi and Arnold (Onishi 1980)	X	X		X		X	X				X				X	A	R	
Fields, 1976 (CHNSD)	X	X	X	X		X					X			X		FD	R	
Churchill, 1976	X	X		X		X	X				X	X			X	FD	E	X
Vanderploeg et al., 1976	X	X				X	X			X					X	A	L	X
Booth, 1976	X	X				X	X				X				X	A	L	
Eraslan et al., 1977																		
RADONE	X			X	X						X				X	I	E,R	
RADTMO	X			X	X							X			X	I	C,E,L	
HOTSED		X	X			X		X			X				X	I	E,R	
Chapman, 1977		X		X	X	X	X					X			X	FD	R	
Smith et al., 1977	X	X		X	X	X	X				X	X	X		X	FD	R,L	
Code11 (USNRC, 1978)																		
NRC River Model	X	X		X		X	X				X			X		A	R	X
NRC Lake Model	X	X		X		X	X			X					X	A	L	X
NRC Estuarine Model	X	X		X	X	X	X				X				X	A	E	X
O'Connor and Farley, 1978	X	X		X		X	X	X	X			X		X		FD	E	X

*A - Analytical Solution; FD - Finite Difference; FE - Finite Element; I - Integration; P - Probabilistic
 **C - Coastal Systems and the Great Lakes; E - Estuarine Systems; L - Lake and Impoundment Systems; R - River Systems

GROUP I: Fletcher and Dotson (1971), Fletcher, Dotson, Peterson,
and Betson (1973)

The model developed by Fletcher et al. (1973) is one of the earliest models to compute the passage of radionuclide doses to man through liquid and gaseous pathways (Figure 2a). The liquid pathway submodel, one of the components of this model (Figure 2b) considers dissolved and particulate radionuclide transport and sediment transport. Various options include radionuclide contributions from numerous point sources, such as contribution from air deposition and overland flow, flood deposition, and groundwater transport. The liquid-pathway submodel is an unsteady, one-dimensional code and calculates the temporal and spatial (longitudinal) distributions of dissolved radionuclide concentrations as well as concentrations of radionuclides attached to suspended and bottom sediments of various sizes. The transport model can also account for the effects of flow stratification and sediment trapping in reservoirs. The dissolved radionuclide concentration at a given location is found by applying the mass conservation equation with radioactive decay as follows:

$$C_{x,t} = \frac{1}{Q_{x,t}} \left\{ Q_{(x-\Delta x, t-\Delta t)} C_{(x-\Delta x, t-\Delta t)} e^{-\lambda \Delta t} + \sum_1^n Q_i C_i \right\} \quad (5)$$

where

$C_{x,t}$ = dissolved radionuclide concentration at location x and time t

C_i = dissolved radionuclide concentration of tributary

$Q_{x,t}$ = flow rate at location x and time t

Q_i = tributary flow rate

λ = radionuclide decay

The sediment transport rate, S , is found analytically from the following equation:

$$S = \alpha Q^b \quad (6)$$

FIGURE 2a. Overall Structure of Model Developed by Fletcher et al. (1973)

(b) WATER TRANSPORT MODEL

FIGURE 2b. Water Transport Submodel Developed by Fletcher et al. (1973)

where Q is the flow rate and a and b are constants that must be estimated for each sediment size range. The concentration of radionuclides attached to sediment is calculated from the known dissolved radionuclide concentration and the distribution coefficient value, K_d . Since the quantity of radionuclides adsorbed to the sediment is not subtracted from the dissolved concentration, strictly speaking, the mass conservation in a stream reach is not satisfied.

This model and other man-dose models (Bramati et al. 1973, Soldat et al. 1974, Watts 1976, Martin et al. 1976) express the very complicated aspects of liquid-pathway transport using extremely simplified formulations. Consequently, it is very easy to apply this model and other man-dose models, but their predictions are usually either overestimated or underestimated.

GROUP II

Gloyna et al. (1963-1972). Several unsteady, one-dimensional mathematical models resulted from a series of studies on radioactivity transport in water undertaken by E. F. Gloyna and his associates at the University of Texas at Austin. Investigations were made of the transport and dispersion of strontium, cesium, ruthenium, cobalt, and chromium with adsorption-desorption mechanisms included. In addition to model development, a series of experiments were conducted in a model river consisting of a dual-channel flume and various other components. Each channel of the metal flume was 200 ft long, 1.25 ft wide, and 2 ft deep with a slope that could be varied between zero and 0.006.

Three of the resulting models discussed below are based on the unsteady, one-dimensional, advection-diffusion equation with a source/sink term added to include the effect of sorption and desorption by aquatic plants or nonmovable bottom sediment, or both. The governing equations for dissolved radionuclides and particulate radionuclides adsorbed by plants and bed sediment are as follows:

$$\frac{\partial C}{\partial t} + U \frac{\partial C}{\partial x} = \epsilon_x \frac{\partial^2 C}{\partial x^2} + m_j K_j [C_j - KdC] \quad (7)$$

$$\frac{\partial C_j}{\partial t} = K_j [KdC - C_j] \quad (8)$$

where

C = dissolved radionuclide concentration

C_j = concentration of particulate radionuclide attached to j^{th} sorbent such as plants and bed sediment

Kd = distribution coefficient

K_j = mass transfer coefficient for j^{th} sorbent

m_j = mass of j^{th} sorbent

U = mean velocity in x direction.

As shown above, the last term in Equation (7) expresses the adsorption-desorption mechanisms. The three models differ in their inclusion of particulate radionuclide sorbents (e.g., plants and bed sediment) as discussed below.

- Shih and Gloyna (1967, 1969, 1970). The model developed by Shih and Gloyna in 1967 is capable of describing the transport of radionuclides injected instantaneously into a stream. The model takes into account the effects of nonmovable bed sediment in adsorbing a dissolved radionuclide.

Assumptions inherent in the model include a constant average velocity and a zero initial radionuclide concentration in both water column and bed sediment. Even though the solutions are analytical, they contain complicated integrals that must be solved numerically. Shull and Gloyna (1968) note that the solutions developed by Shih and Gloyna work well for a case in their low-velocity hydraulic flume but produce errors when applied to natural rivers.

- Armstrong and Gloyna (1968). Armstrong and Gloyna obtained a more general model by solving the above described equations numerically by an explicit finite-difference method. Their model includes the effects of adsorption-desorption by plants. To verify the model, they conducted radionuclide dispersion tests in the river model flume for both instantaneous and continuous releases of ^{85}Sr . Agreements of the computer results to the measured data for instantaneous and continuous releases were very good and reasonable. Since this model is a numerical simulation model, its applicability is wider than that of the analytical model developed by Shih and Gloyna. However, Armstrong and Gloyna's numerical solution requires further improvement because of the instability caused by an initial zero radionuclide concentration on the bed.

- White and Gloyna (1969). The one-dimensional mathematical model developed by White and Gloyna includes the mechanisms of sorption and desorption by both plants and bottom sediment and uses an implicit finite difference solution method. It is capable of treating instantaneous, continuous, or time-varying releases of radionuclide activity. The model showed good agreement with data measured in their hydraulic flume concerning both continuous and instantaneous releases of ^{85}Sr .

All three aforementioned models and a model developed by Buckner and Hays (1975) and Buckner et al. (1975) simulate unsteady, one-dimensional radionuclide transport by accounting for radionuclide adsorption-desorption with bed sediment and aquatic plants. The foundations and formulations of these models are sound. The analytical solution obtained by Shih and Gloyna (1967) is especially useful because it is relatively simple to apply. However, a shortcoming of these models is that the sediment and plants are treated statically. The models do not consider transport, sorption-desorption with suspended sediment; or deposition and resuspension mechanisms of contaminated sediment. Because direct adsorption by bed sediment has in many cases a much smaller effect on radionuclide transport than does sorption by suspended sediment (Bhagat and Gloyna 1965), these models are rather limited to rivers and certain lakes where almost no suspended sediment exists.

Eraslan et al. (1977).

- RADONE, RADTWO, SEDONE. Researchers at Oak Ridge National Laboratory have developed several transient, one- and two-dimensional, far-field mathematical models for assessing the environmental impact of power plant operations on estuaries and rivers, lakes and cooling ponds, and coastal regions. The project, known as the "Unified Transport Approach (UTA) for the Assessment of Power Plant Impacts," includes models for thermal, chemical, radiological, and biological impacts (Eraslan et al. 1977). Formulations of the UTA models will be fundamentally similar in theory, enabling the application of virtually identical numerical solution methods and computational algorithms to all UTA models. All are discrete-element formulations with Runge-Kutta-Gill numerical solutions.

As part of the UTA program, three radionuclide transport models have been developed for application to estuaries. According to Eraslan et al. (1977), RADONE is a transient, one-dimensional transport model for dissolved radionuclides applicable to controlled rivers and estuaries; RADTWO was purportedly developed directly from the thermal code, FARTMP, and is a transient, two-dimensional, dissolved-radionuclide transport model for lakes, estuaries, and coastal regions. Neither RADONE nor RADTWO has yet been verified (Eraslan et al. 1977). A third radionuclide model applicable to estuaries, HOTSET, is a one-dimensional model for the transport of radionuclides adsorbed on sediment (Fields 1976).

Chapman (1977). In the dissolved and particulate contaminant transport model developed by Chapman (1977), the basic equation for dissolved contaminants is a two-dimensional, unsteady, advection-diffusion equation with a sink-source term containing the Freundlich formulation of adsorption-desorption mechanisms. In this equation, the longitudinal velocity component was deleted by assuming that the flow is uniform and that the coordinate system moves with the flow. However, longitudinal velocity varies vertically according to the logarithmic velocity distribution and has a Von Karman's constant equal to 0.4. The suspended sediment profile was assumed to be independent of time and longitudinal direction. A particulate contaminant equation was formulated as the

rate of change of particulate concentration being equal to the amount adsorbed by or desorbed from sediment. This type of expression is the same as those obtained by Shih and Gloyna (1967) and Buckner and Hayes (1975). The dissolved and particulate contaminants were then solved numerically by an explicit finite difference scheme.

In general, the model is applicable to cases with steady flows in uniform channels. However, it has several inherent limitations which preclude its wider application. Most of all, the assumption that flow and sediment concentration are independent of longitudinal distance is not generally valid. Since sediment transport rate is, in general, proportional to the third to fifth power of velocity, the model's assumption of a constant sediment concentration with longitudinal distance and time greatly limits its applicability. The model must also assume that there is no initial concentration of contaminants in a study region.

GROUP III: Onishi et al. (1976, 1979a,b), Onishi (1977)

- Model SERATRA and TODAM. Onishi et al. (1976) developed an unsteady, two-dimensional (longitudinal and vertical) model known as SERATRA (Sediment-Radionuclide Transport,) as a means of more realistically predicting radionuclide migration by coupling radionuclide transport with sediment transport. SERATRA is a finite-element model that predicts movements of sediment and radionuclides and other toxic contaminants. It uses general advection-diffusion equations with decay and sink/source terms and appropriate boundary conditions. The model consists of the following three submodels coupled to describe sediment-contaminant (radionuclide) interaction and migration: 1) a sediment transport submodel, 2) a dissolved contaminant transport submodel, and 3) a particulate contaminant (contaminant adsorbed by sediment) transport submodel.

Since the movements and adsorption capacities of sediment vary significantly with sediment sizes or sediment types, the sediment transport submodel simulates transport, deposition, and erosion of three sediment size fractions

(or sediment types) of cohesive and noncohesive sediments. The dissolved-contaminant transport submodel includes mechanisms of contaminant adsorption/desorption, as well as radionuclide decay and contaminant degradation resulting from hydrolysis, oxidation, photolysis, and chemical volatilization and biological activity where applicable. The particulate contaminant transport submodel simulates transport, deposition and erosion of contaminants associated with each sediment size fraction.

Adsorption/desorption mechanisms were expressed by a distribution coefficient and a transfer rate which describes how quickly dissolved and particulate contaminant concentrations reach their equilibrium condition. A formulation of these sorption mechanisms is similar to the right hand side of Equation (8). Unlike most other models (e.g., Armstrong and Gloyna 1968), SERATRA uses different distribution coefficients for adsorption and desorption and treats adsorption/desorption mechanisms as not being fully reversible.

SERATRA also predicts changes in bed conditions for sediments and particulate contaminants, e.g., riverbed elevation changes due to sediment deposition and scour; and longitudinal and vertical distributions of bed sediment and particulate contaminants in the river bed. One of the shortcomings of the model is that it has a vertical dispersion coefficient but does not have a longitudinal dispersion term. The model uses a nonuniform vertical distribution of longitudinal velocity to convect sediment and contaminant longitudinally. However, the model does not consider a transverse variation in the longitudinal velocity or its affect on sediment and contaminant transport.

The model has been applied to a 100-km reach of the Columbia River near the Hanford Nuclear Reservation in Washington (Onishi 1977) and to a 37-km reach of the Clinch River near Oak Ridge National Laboratory, Tennessee (Onishi et al., 1979a). SERATRA has also been applied to 68-km reaches of Four Mile and Wolf Creeks in central Iowa (Onishi et al. 1979b). The Columbia River, Clinch River, and Four Mile and Wolf Creeks are large, intermediate, and small rivers or streams, respectively. In these cases, SERATRA simulated

sediment migration and radionuclide and pesticide transport. For the Columbia River, simulation was conducted for the movement of ^{65}Zn with and without the adsorption mechanism to identify the effects of ^{65}Zn adsorption by sediment. The cases included the continuous release or instantaneous release of ^{65}Zn , and the absence of new releases of ^{65}Zn to a river bed already contaminated by ^{65}Zn .

For the Clinch River, simulation was also conducted under conditions of continuous or instantaneous releases of ^{137}Cs and ^{90}Sr . These radionuclides were selected because field data on them are available, and because they have extreme adsorption characteristics (most of the ^{137}Cs is transported with sediment, while the majority of the ^{90}Sr is transported in a dissolved form). Comparison of computed results with measured data for both the Columbia and Clinch Rivers indicates good agreement for both sediment and radionuclide distributions (Figure 3). However, because limited data were compared, the model's validity cannot be considered fully tested. Currently, SERATRA is being applied to the Cattaraugus and Buttermilk Creeks near the West Valley radioactive-waste burial site in New York; this study constitutes a comprehensive verification test. Examination of simulation results for the Columbia and Clinch Rivers clearly indicates that suspended sediment has a significant effect on transport of some radionuclides.

Several limitations of SERATRA are as follows: the model requires extensive input data that may limit its applicability; it requires rather extensive computer time, in which long-term, continuous simulations can be expensive; the model cannot handle a reversible flow, such as a tidal flow; and the model treats adsorption/desorption mechanisms as complete reversible mechanisms.

The two-dimensional model, SERATRA was simplified to an unsteady one-dimensional (longitudinal) code named TODAM (Transient One-Dimensional Degradation and Migration model). TODAM includes the longitudinal dispersion term and can handle a reversible flow and dry bed conditions. The model is suitable to many rivers where vertical distributions of sediment and radionuclides are not a major concern. Along with the hydrodynamic model DKWAV (Whelan

FIGURE 3. Longitudinal Distributions of Dissolved, Particulate, and Total ^{137}Cs in the Clinch River, Tennessee (Onishi et al. 1979a)

1980), TODAM was applied to Mortandad and South Mortandad Canyons in Los Alamos, New Mexico to estimate movements of flow and sediment and radionuclide distributions in the canyons (Chapter 4). Both canyons represent intermittent streams.

Onishi et al. (1976), Onishi and Wise (1978).

- Model FETRA. Onishi et al. (1976) also developed an unsteady sediment-contaminant transport model called FETRA. This model solves longitudinal and lateral distribution of sediment and contaminant transport by including sediment-contaminant interaction and wave-sediment interaction mechanisms. FETRA

is applicable to relatively wide and shallow water bodies such as wide estuaries, coastal waters, the Great Lakes, as well as wide rivers.

The model has general advection-diffusion equations with decay and sink-source terms. In the implementation of the finite element method, triangular elements are used with six nodes associated with each element. Quadratic approximations are made with each element. Similar to SERATRA (Onishi 1977), as discussed above, FETRA consists of the following three submodels joined to include the mechanisms of sediment-contaminant interaction: 1) a sediment transport model, 2) a dissolved contaminant transport model, and 3) a particulate contaminant transport model. Transport of sediment and particulate contaminants is simulated for each sediment size fraction or sediment type. Because of the importance of wave mechanisms in suspending sediment in coastal waters and large lakes, FETRA also includes mechanisms of sediment suspension by waves in both offshore and surf zones.

The model was applied to the James River estuary to determine the longitudinal movement of sediment and the migration of the pesticide, Kepone, in both dissolved and particulate forms under unsteady estuary flow conditions (Onishi and Wise 1978). Sediment transport modeling was conducted for cohesive and noncohesive sediments and for organic matter. Comparison of computed sediments and contaminants (mainly particulates) with measured data indicates good agreement. The model also calculated the scour and deposition of sediments and Kepone in the James River estuary. FETRA was also applied to Pacific coastal waters to simulate the migration and accumulation of ^{239}Pu and ^{90}Sr .

The limitations of FETRA, like those of SERATRA, are as follows: 1) the model requires extensive input data that may limit its applicability, and 2) it requires rather extensive computer time such that a long-term, continuous simulation may be very expensive.

Fields (1976). The model developed by Fields, CHNSED, is capable of simulating the transport of trace contaminants in the dissolved form and the particulate form (adsorbed to sediment) as runoff from the land surface or as

flow in a channel. CHNSED is a composite model that combines the Wisconsin Hydrologic Transport Model (WHTM) with a sediment transport model for a rectangular stream channel (SEDTRN). Both the WHTM and the SEDTRN models are components of the Unified Transport Model, which simulates the transport of a trace contaminant through a watershed ecosystem. In the channel, the dissolved contaminant transport is governed by the hydrologic flow routing of the WHTM, while the adsorbed contaminants are routed according to the sediment transport dynamics of SEDTRN.

Fields uses the distribution coefficient, K_d , to allow for simulation of first-order chemical exchange between the water and the sediment in the channel system. The K_d value for the concerned contaminant is assumed to be the same for each of the 12 sediment size classes included in SEDTRN and for each river reach. In reality, however, K_d values do change significantly with sediment sizes and types due to large variations in exchange capacities and surface areas of sediments (Kennedy 1965, Jenne and Wahlberg 1968). Commonly, clay-size sediments have a larger K_d value than silts, which in turn have a greater K_d value than sand (Pruter and Alverson 1972). Although CHNSED does not allow for a variable K_d value, it does include a parameter (specified by the user) for each sediment size class to indicate whether the size class is to be included in sediment-contaminant interactions.

As discussed above, most radionuclide transport models are based on advection/diffusion equations. The models range from simple analytical solutions to sophisticated numerical models. Except in very simple cases, numerical models are needed to accommodate the wide ranges of channel geometry, flow and sediment characteristics, and radionuclide characteristics found in the real world.

In summary, the important mechanisms of radionuclide transport and fate in surface waters are 1) advection and dispersion of radionuclides, 2) radionuclide decay, and 3) radionuclide contributions to the surface water system from outside sources, and 4) radionuclide-sediment interactions. As shown in Table 1, most available models include the first three mechanisms. These models are applicable to short-term radionuclide migration and to cases

where the radionuclides have small a low to being adsorbed and the sediment concentrations in the receiving water body are very low. Unfortunately, many waste management situations do not fall into these categories. Rather they involve radionuclides having high affinity to sediments (as is the case for plutonium, americium, and cesium); they involve high concentrations of sediment in the receiving water body; and they involve long-term migration and accumulation of the contaminants. Under the latter circumstances, radionuclide-sediment interactions must be included in the analysis. Hence, any generic model used to simulate the instream migration of radionuclides originating from low-level waste burial sites must be a numerical model designed to incorporate all of the major mechanisms of contaminant migration and fate. The models that satisfy these requirements are SERATRA (Onishi 1977), FETRA (Onishi and Wise 1978), and CHNSED (Field 1976). Because of their success in past applications and their flexibility, SERATRA/TODAM was selected for use in this study with some modification.

3.3 RADIONUCLIDE TRANSPORT IN THE ATMOSPHERE

3.3.1 Physical Mechanisms of Radionuclide Transport

The storage, handling, and disposal of nuclear wastes may involve both accidental and routine releases of these materials into the atmosphere. Routine releases during waste processing are usually made directly to the atmosphere. Accidental releases can occur either directly to the atmosphere or through the suspension of contaminated soil by the wind. This latter process is of primary interest in this study. Airborne contamination can be transported and diffused by the wind, deposited on the surface by both dry and wet removal processes, and subsequently resuspended by the wind for further downwind transport. Each of these processes is briefly discussed below.

Transport and Diffusion

Material released to the atmosphere is transported in the direction of the mean wind and diffused in the vertical and crosswind directions by fluctuations in the wind. For surface releases, the effective wind speed

increases as the effective transport height increases with distance from the source. For continuous releases, diffusion in the downwind direction may usually be ignored compared to dilution by the mean wind.

Crosswind and vertical diffusion is controlled by atmospheric turbulence and is a function of atmospheric stability and downwind distance. Diffusion is high during the day when convective or unstable conditions exist and low during the strong inversion or stable conditions that occur on clear nights. Overcast skies or strong winds, or both, cause neutral stability, an intermediate diffusion condition.

Deposition

Airborne contaminants are deposited on land surfaces by both dry and wet processes. Contaminants are transported to the surface for dry deposition by gravitational settling and turbulent diffusion, the dominant process depending on particle size. Thus, the dry deposition of small particles is more strongly dependent on meteorological conditions than the deposition of large particles. For near-surface sources, wet deposition is primarily due to below-cloud collection of the contaminants by precipitation. The dominant mechanisms are Brownian diffusion and inertial impaction, and nucleation processes may be ignored except for long-range transport.

Resuspension

A primary source of airborne contamination is the suspension of contaminated soil at the waste storage site. Similarly, material transported by the wind and deposited downwind of the site may be subsequently resuspended for further atmospheric transport. These suspension processes are functions both of the soil conditions (e.g., soil type, moisture, ground cover) and of the meteorology (e.g., wind speed). Present knowledge of resuspension rates and how they depend on these variables is very limited.

3.3.2 Mathematical Models for Atmospheric Transport

The transport, diffusion, deposition, and resuspension of airborne contamination is affected by a variety of physical processes. A model which

includes all of these mechanisms becomes quite complex and expensive to use. Simpler models can be developed for restricted situations by including only those mechanisms that are important to the situation of interest. Often, this simplification is achieved by investigating only a limited domain in space and time. Atmospheric transport models are discussed below with these simplifications in mind.

Transport and Diffusion

The most commonly used transport and diffusion model is the Gaussian plume model. The model assumes that contaminants are transported at the speed and in the direction of the wind at the source and has a bivariate normal distribution in the crosswind horizontal and vertical directions. The standard deviations of the horizontal and vertical crosswinds, σ_y and σ_z , are usually empirical functions of downwind distance and atmospheric stability. The Gaussian plume model is best applied at downwind distances less than 10 km because of spatial variability in the mean wind field and limited knowledge of σ_y and σ_z beyond that distance. The model is typically applied for each hour of the period of interest over which temporal changes occur in wind speed, wind direction, and atmospheric stability. Adding up the results for these cases reveals mean contaminant levels.

The Gaussian plume model has been extended to greater distances by following contaminant puffs or plume segments as they are advected in a spatially and temporally varying wind field. At distances greater than 10 to 100 km, the turbulent crosswind dispersion described by σ_y is small compared to the dispersion due to the variability of the wind field. The vertical dispersion, σ_z becomes less important after the contaminant diffuses to the top of the mixed layer and becomes uniformly distributed in the vertical direction.

Gradient transfer or K-theory models may be used to look in more detail at the vertical dispersion process. Gradient transfer models may be particularly useful when studying situations for which direct observations of σ_z are not available, such as at large downwind distances or for contaminants subject to chemical transformation or deposition. Unfortunately, the eddy diffusivity

is also well known only for a limited number of situations. Furthermore, K-theory models usually require a numerical solution, and some compromise must inevitably be made on resolution or the size of the computational domain.

Deposition

The flux of contaminants to the ground caused by dry deposition is commonly expressed as a deposition velocity, v_d , multiplied by the ground-level airborne contamination. The deposition velocity becomes minimal for particles roughly $0.1 \mu\text{m}$ in diameter and increases for larger particles due to gravitational settling and for smaller particles due to Brownian diffusion. The deposition velocity also increases with increasing friction velocity, u_* , and increasing surface roughness, z_0 , due to the role of turbulent diffusion in carrying contaminants to the surface.

The deposition boundary condition is easily incorporated in gradient-transfer models because the models explicitly compute vertical fluxes. More care is required for Gaussian models and also for gradient-transfer models with coarse vertical resolution near the surface, because deposition alters the vertical profile of contamination near the surface from that assumed by the nondepositing transport and diffusion model. Horst (1979) discusses this problem and compares the predictions of several simple Gaussian diffusion-deposition models with an exact solution.

In contrast to dry deposition, precipitation removes airborne contamination at all heights. The loss per unit volume is estimated as a scavenging rate, ξ , multiplied by the local airborne concentration of the contaminant. This loss is an additional term in the advection-diffusion equation, which is solved numerically in K-theory models. Since ξ can often be assumed to be independent of height, Gaussian plume models are easily modified to account for wet deposition by multiplying the plume formula by the factor $\exp(-\int \xi dt)$ where t is travel time.

The scavenging rate for particles is determined from the collection efficiency, $E(a,R)$, for particles of radius R . The collection efficiency depends on a particle size similar to that of the deposition velocity. It has a

minimum value near 0.1 μm , increasing for larger particles due to inertial impact, and increasing for smaller particles due to Brownian diffusion. The efficiency is also a complicated function of drop size, decreasing with drop size for small particles and small drops. The scavenging rate is found by integrating the product of the drop's cross sectional area, the collection efficiency, the particle size distribution, and the drop size distribution over all particle and drop sizes. This may be usefully approximated by the ratio of the rainfall rate to the mean drop size, multiplied by an effective collection efficiency.

Resuspension

Information about the suspension and resuspension of surface contaminants is limited. The most extensive literature on soil movement by the wind deals with the formation and movement of sand dunes (Bagnold 1941) and the erosion of agricultural lands (Chepil and Woodruff 1963). The latter research has identified many factors that influence soil movement: wind speed, source geometry relative to the wind direction, surface roughness and vegetative cover, and soil structure and moisture content. Equations have been developed for the horizontal flux of saltating sand grains 0.1 mm to 0.5 mm in diameter, and an empirical wind erosion equation (Woodruff and Siddoway 1965) has been developed to estimate the annual average soil loss from a cultivated field.

Even though soil erosion is an important mechanism governing the production of suspended aerosols, vertical fluxes of aerosols have not been quantitatively related to horizontal fluxes, nor have air concentrations been related to either of these. The nuclear industry has historically summarized its experience in terms of a resuspension factor (Stewart 1967),

$$K_r = \frac{\text{Airborne Contamination (radioactivity/m}^3\text{)}}{\text{Surface Contamination (radioactivity/m}^2\text{)}}, \quad (9)$$

which is found to vary from 10^{-10} m^{-1} to 10^{-5} m^{-1} . Unfortunately, this quantity is determined not only by the suspension process, but also by the

spatial distribution of surface contamination and by the atmospheric dispersion/deposition process. Hence, K_r is likely to be unique for each situation. The resuspension rate,

$$\Lambda = \frac{\text{Vertical Flux of Contaminant (radioactivity/m}^2\text{-sec)}}{\text{Surface Contamination (radioactivity/m}^2\text{)}} \quad (10)$$

is a local parameter determined solely by the suspension process. Hence, it is more likely to have general application, although it still depends on many factors of weather and soil. Sehmel and Lloyd (1976) find values of $10^{-10} \text{ sec}^{-1}$ to 10^{-8} sec^{-1} for suspension of particles less than $10 \text{ }\mu\text{m}$ in diameter from a lightly vegetated, arid location.

When the surface contaminant concentration and the suspension or resuspension rate are known, it is a straightforward process to compute the resulting airborne contamination using either a plume or gradient transfer model to describe the atmospheric dispersion. The surface source is divided into a number of elements, each of which is represented as a ground-level source of contamination. A small area may be represented as a point source beyond a downwind distance of two to five times its dimensions. Horst (1979b) has developed a Gaussian diffusion/deposition/resuspension model of this type and used it to show that the airborne contamination without deposition and resuspension is an upper limit for the annual-average, ground-level, airborne contamination.

3.4 RADIONUCLIDE TRANSPORT IN GROUNDWATER

3.4.1 Physical Mechanisms of Radionuclide Transport in Groundwater

Radionuclide transport in groundwater is almost entirely dependent on the flow of the transporting media, namely the water, whether it be in saturated aquifer flow or unsaturated flow through the vadose zone. Without flow the transport can only be by diffusion, which is an extremely slow process.

Both the amount of water stored in an aquifer and the rate of flow through the aquifer are related directly to the geology of the aquifer. The geologic materials range from bedrock to unconsolidated sediments and include limestone, sandstone, sand, and gravel. Aquifers, therefore, exhibit a variety of characteristics and conditions which renders describing them difficult and time consuming.

Flow is primarily dependent upon pressure head differences, porosity, and transmissivity. The processes of recharge and discharge link the groundwater system to the rest of the hydrologic cycle. Thus, the system is dynamic. In general, groundwater moves under the influence of gravity and pressure in the direction of lesser head. The rate and direction of flow can be described by a set of well-known equations which have given rise to models. Typically, groundwater flow is on the order of a centimeter per day; in contrast, surface stream flow which is usually several meters per second.

Radionuclides in groundwater move by the general mechanisms of advection with the fluid movement and then dispersion. Radionuclides are also influenced by sorption, radioactive decay, the geochemical characteristics of the aquifer, and the concentrations of competing macro-ions in the transporting fluid.

Much of the flow of water beneath the earth's surface takes place above a saturated aquifer in the unsaturated zone or vadose zone. The unsaturated flow zone constitutes the transition region between the atmosphere and a (saturated) groundwater system. Passage of water through this zone is very dynamic and dependent on detailed variations in soil-water hydraulic properties. Rain falling directly on the soils' surface is usually viewed as the primary water input to this zone. But water derived from agricultural irrigation can also be a substantial input. Seepage beneath rivers and lakes can also be included, if the underlying geological region is composed essentially of granular, soil-type media.

Water in the unsaturated medium moves downward under the influence of gravity as long as there is a sufficient quantity present to overcome the restraining forces of capillary hydraulic potential (matric potential). Water

is extracted from the unsaturated zone as surface evaporation and as transpiration by growing plant roots. The rates of these extraction processes depend directly on available solar energy (radiation) and surface winds.

Water in the soil moves by two physical mechanisms: capillary Darcian flow (liquid phase) and vapor diffusion. Darcian flow is described by hydraulic conductivity, matric potential gradients, and hydraulic properties, which manifest an extreme variation with even small changes in water content. Vapor diffusion is mainly important near the soil surface as a result of thermal gradients, and this process is primarily in control of actual surface evaporation. Maximal potential evaporation rates estimated by energy balance methods such as Penman's can be realized only when sufficient water is near the soil surface; otherwise, a dry soil layer limits evaporation.

Growing plants are powerful and efficient extractors of stored water in soil. The relationship between potential transpiration and climatic conditions must be quantified in an unsaturated zone simulation. Active plant roots have a water withdrawal distribution that must also be taken into account. Indeed, an accurate estimation of water balance over an areal region is impossible without detailed quantification and modeling of transpiration and root growth.

Water storage by a soil profile is characterized by a water content distribution, which ultimately depends on the detailed spatial variability of hydraulic properties. Infiltrating water that exceeds soil-water holding capacity at a particular profile location will contribute to deep drainage, which may possibly enter the ground water, depending on the depth of the water table. Therefore, a single set of measured hydraulic properties cannot correctly represent an areal region. An estimate of the statistical distribution is required to determine areal drainage flux.

Unfortunately, the physical mechanisms responsible for contaminant transport by unsaturated fluid flow are still not completely understood and have not been fully described mathematically. Most transport modeling assumes that wetting fronts of high water content associated with intense rainfall are mainly responsible for the downward convective movement of soluble

contaminants. The migration of contaminants toward the soil surface has also been observed and modeled under extreme drying conditions. More research and model development is needed to fully assess the long-term impacts of unsaturated flow coming in contact with shallow buried radioactive contaminants.

3.4.2 Mathematical Models for Groundwater

Almost any modeling of transport related to groundwater starts with a flow model. Models are divided into categories according to dimensionality, i.e., one-, two-, and three-dimensional models, and by solution techniques, i.e., analytic, finite difference, or finite element. Most common are the two-dimensional models which consider saturated flow in an aquifer as essentially horizontal flow. These aquifer flow models can handle flow in phreatic, confined, leaky, non-leaky or multiple aquifer systems. Some contain submodels for vertical flow through the unsaturated zone.

Three-dimensional models tend to be larger and more costly to run but are powerful tools in determining flow patterns as well as the configuration of the free water-table surface. One-dimensional flow models are generally not used in aquifer simulations but find more use in the unsaturated zone modeling, where flow generally takes place downward from the ground surface.

Several models are available under various categories. Holcomb Research Institute, Butler University, under a grant from EPA undertook a model survey and compiled and published a document (Bachmat et al. 1978) evaluating the present status of numerical models for use in groundwater resource management.

A set of groundwater, transport and man-dose codes were adopted by Pacific Northwest Laboratory for use in radioactive waste isolation assessments (Silviera et al. 1980). Some of these codes would be applicable to low-level waste management. They have the advantage of rigorous application to radionuclide transport problems and are being well documented (Nelson and Schur 1980).

One of the currently available unsaturated models is the UNSAT computer model developed by Gupta et al. (1978) to describe unsaturated water movement under general agricultural conditions. The model, however, is flexible and

has been applied to various experimental investigations (Gee and Simmons 1979). Because unsaturated zones are hydrologically so complex, a considerable variety of site-specific input data is required in order to perform a proper simulation.

3.5 RADIONUCLIDE TRANSPORT IN TERRESTRIAL BIOTA

3.5.1 Physical Mechanisms of Radionuclide Transport in Terrestrial Biota

When radionuclides are placed underground, plants can take them up through their root system. Or, if radionuclides are exposed to the ground surface, they can infiltrate the soil and eventually enter the ecological system through the plant uptake process. Transpiration by growing plant roots and radionuclide uptake by plants are directly related to the availability of water and the concentration of dissolved radionuclides in an unsaturated groundwater zone. Hence, moisture movement and radionuclide transport in an unsaturated zone, as discussed in Section 3.4, are critical elements in determining plant growth and the source strength of contaminants in a terrestrial ecological system. Once radionuclides are taken up by plants, the radionuclides can accumulate within the plant community and later transfer to other parts of the environment via plant decay or the food chain.

3.5.2 Mathematical Models for Terrestrial Biota

Biotic processes and their relation to the environment can be represented by terrestrial ecological models. Examples of these models are the Terrestrial Ecosystem Hydrology Model (TEHM) (Huff et al. 1977) and a biotic model, BIOTRAN, developed by the Los Alamos National Laboratory (Wheeler et al. 1977). TEHM simulates processes that occur in atmosphere-soil-plant-water flow.

BIOTRAN is a compartmentalized model that simulates the biotic and abiotic transport of transuranics. The abiotic module is primarily responsible for generating the driving forces for biomass production, in addition to accounting for the transfer of transuranics. A key component of the abiotic module is a meteorological input data generator. Precipitation, daily

temperature, and insolation are generated stochastically using random selection from a normal distribution based on each parameter's long-term average and variance. These inputs, along with data on mean elevation, field and holding capacities, wilting point, control soil water, and evapotranspiration drive the production of biomass. A simple bookkeeping system is used to keep track of transuranic contributions from the air and soil, as well as transuranic levels in the soil, in plant biomass, and on the land surface.

The biotic module simulates biomass accumulation as it is affected by photosynthetic and respiratory rates at a given time in the development of the plant structure. Both of these processes are treated independently in response to temperature and moisture variations. Two distinct classes of plants considered are cool- and warm-season perennials. Biomass is divided into four components: roots; above-ground, living vegetation; above-ground, standing dead matter; and litter. The cycling of transuranics through each compartment is simulated continuously. Uptake of transuranics from the soil is a function of root biomass and its distribution. The conversion of dead standing vegetation to litter, the primary mechanisms by which transuranics are deposited on the soil surface, is governed by temperature and soil and water conditions.

3.6 MODEL SELECTION

Criteria for the selection of models used for this study included the following:

1. Models must consider major mechanisms of radionuclide transfer and fate such as:
  - transport mechanisms
  - radionuclide decay and production of daughter products
  - intermedia transfer
  - positive and negative contributions from outside sources.
2. Model assumptions must be documented as to their validity.

3. Models for various media must be compatible or easily modified to include intermedia transfer of radionuclides.
4. Models must have proven histories of success under past applications.
5. Models must be nonproprietary.
6. Requirements for data, modeling time, and modeling cost for selected models must be reasonable.

Based on these criteria, models for overland, surface water and terrestrial plants were selected.

3.6.1 Overland Model

Processes controlling runoff, soil erosion, and radionuclide transport on the land surface are continuous in nature even though radionuclide migration may take place only during runoff-producing storm events. Hence, a continuous simulation model must be selected for this study not only to simulate radionuclide transport during a storm event but also to simulate the processes that change a soil's moisture content, and the events that influence soil conditions and radionuclides between storms. Furthermore, a model must handle both dissolved and particulate (those adsorbed by soil) radionuclides. Clearly, the comprehensive nature of the Group III models (see Section 3.1) best meets these requirements. Besides the ARM model described in Section 3.1, several other models fit in Group III, such as ACTMO (Frere et al. 1975), AGRUN (Roesner et al. 1975), and WHTM (Patterson et al. 1974). However, of all these models, ARM is the most extensively tested. Its hydrologic components have been successfully tested on numerous watersheds across the country (Donigian et al. 1977). Although the sediment and pesticide components have been tested only on a few watersheds, testing of other models such as ACTMO and AGRUN is even more limited. Hence, the ARM model was selected for this study to predict runoff and sediment and contaminant loadings at the edge of a stream.

The ARM model is composed of the five major components which continuously simulate the following:

- hydrodynamic response of watersheds
- soil erosion
- contaminant adsorption and removal
- contaminant decay and degradation
- nutrient transformation and removal.

The last component was not used for this study.

The hydrologic component of ARM simulates the dominant physical processes that act to convert precipitation into runoff (i.e., interception, infiltration, interflow, percolation and evapotranspiration). ARM also has the capability to simulate the mechanisms of snow accumulation on the land surface and runoff due to melting of the snow pack. The transfer of water between major components of the hydrologic cycle simulated by ARM is not only used to calculate runoff, but is also used in the calculation of soil erosion and the vertical movement of contaminants in the soil profile.

The soil erosion component of ARM simulates sediment loss from the land surface due to both sheet and rill erosion. Soil fines are scoured from the land surface and transported by overland flow. The availability of these materials for transport is a function of rainfall impact. The availability of soil fines is also a function of the tillage operations or conservation practices that may have occurred on the watershed.

Contaminants placed in a water-sediment mixture will eventually come to equilibrium, with a certain fraction of the contaminant adsorbed to the sediment materials and the remainder dissolved. The adsorption and removal component of ARM uses a modified Freundlich adsorption/desorption isotherm algorithm to partition the available contaminant between that dissolved in the overland flow and that adsorbed to soil fines being transported by the flow.

The contaminant degradation component of ARM determines the quantity of contaminant available for transport, either by overland flow or by soil erosion, at any time after its application to a watershed. The major processes that act to attenuate contaminants in the soil are degradation and decay, which are lumped together in a simple, first-order decay algorithm.

One of the major limitations of the model is the quantity of historical data required to calibrate model coefficients and parameters. This limitation has been reduced somewhat by the regionalization of many hydrologic parameters. A second limitation is that the ARM model is not formulated to handle the effects of channel processes on runoff and sediment transport. As a result, the model is applicable only to watersheds that are small enough that channel processes can be assumed negligible. Modeling various catchment sizes by ARM is limited by climatic and topographic characteristics. Generally, it was found that ARM was applicable on areas from 2 to 5 km² (or about 1 to 2 mi²) (Donigian and Crawford 1976). Because of this limitation, the application of ARM to a large watershed requires the subdivision of the watershed into smaller subcatchments, and a number of individual model applications to the smaller subcatchments.

3.6.2 In-stream Models

To obtain accurate temporal and spatial distributions of radionuclides in streams, a mathematical model must include all the important mechanisms of both dissolved and particulate radionuclide transport phenomena, such as advection and dispersion of contaminants; contaminant decay and degradation; and the interaction of contaminants with sediments. Furthermore, an instream model must be compatible with models for other media, including the continuous simulation model, ARM. As discussed in Section 3.2, SERATRA, FETRA, and CHNSED satisfy these requirements. SERATRA is applicable to nontidal rivers and impoundment, CHNSED is applicable to rivers; and FETRA is best suited for estuaries and coastal waters.

The proceedings of a workshop on models for environmentally assessing radionuclide releases (Oak Ridge National Laboratory 1978) identify SERATRA and FETRA as the most advanced models capable of simulating the transport of both dissolved and particulate contaminants with sediment interaction. A recent review of mathematical transport models by Onishi et al. (1980) supports this conclusion.

SERATRA was selected for this study, because of its completeness in the model formulation, flexibility, and past application experiences. One of SERATRA's shortcomings is that it requires a hydrodynamic model to provide hydrodynamic data (depth and velocity distributions over the entire simulation period). SERATRA also requires lengthy computer time, partially because it computes vertical as well as longitudinal distributions of sediment and radionuclides.

To eliminate or reduce these shortcomings, SERATRA was simplified to a one-dimensional model. TODAM, the modified one-dimensional version of SERATRA, will be combined with the hydrodynamic model, DWOPER (Fread 1973). It should be noted that in the special case of Mortandad and South Mortandad Canyons, the hydrodynamic model DKWAV (Whelan 1980) was employed. TODAM is composed of the following three submodels joined to describe sediment-contaminant interaction and migration:

- sediment transport
- dissolved-contaminant transport
- sorbed-contaminant (contaminants adsorbed by sediment) transport.

These submodels solve an advection-diffusion equation with decay and sink/source terms with appropriate initial and boundary conditions. These submodels use the finite element computation method with the Galerkin weighted residual technique.

Since the movement and adsorption capacities of sediments vary significantly with grain size, the sediment transport submodel solves the migration (transport, deposition and scouring) for three size fractions of cohesive and noncohesive sediments. The model includes the mechanisms of advection and diffusion/dispersion, fall velocity and cohesiveness, deposition on the river bed, resuspension from the river bed (bed erosion and armoring), and sediment contributions from tributaries and point and nonpoint sources. Sediment mineralogy and water quality effects are implicitly included through the second, third and fourth mechanisms.

The dissolved contaminant transport submodel includes the mechanisms of advection and diffusion/dispersion of dissolved contaminants; adsorption (uptake) of dissolved contaminants by sediments or desorption from sediments;

radionuclide decay; degradation of dissolved contaminants due to hydrolysis, oxidation, photolysis and biological activities, if applicable; contaminant volatilization, if applicable; and contributions of dissolved contaminants from point and nonpoint sources.

Partitioning a radionuclide into dissolved and particulate phases was performed in the model by using distribution coefficients and transfer rates. The distribution coefficient is a function of many parameters, including contaminant chemical state, water quality, sediment sizes, mineralogy, and organic content. Unfortunately, the available information about distribution coefficients is not advanced enough to determine these functional relationships. Hence, in this model, values of the distribution coefficient are assigned to the model as input data reflecting the effects of these influencing variables. The adsorption of contaminants by sediments or desorption from the sediments is assumed to occur toward equilibrium with a transfer rate (with the unit of the reciprocal of time). For TODAM, it was assumed that a contaminant has the same distribution coefficient for moving (suspended and bed load sediments) and stationary (motionless) bed sediment. These two types of sediments have different transfer rates reaching equilibrium conditions determined by distribution coefficients. Degradation is one mechanism of nonradioactive-contaminant transport and fate in the aquatic environment. TODAM considers degradation, volatilization, and radionuclide decay, which gives it wide applicability. Contaminant degradation includes both chemical and biological reactions resulting from hydrolysis, oxidation, photolysis, and biological activities. Due to a lack of data on the degradation of contaminants, the degradation mechanisms just described were considered only for the dissolved contaminants. The transport of sorbed contaminants is simulated for those associated with each sediment size fraction.

TODAM includes the mechanisms of advection and diffusion/dispersion of sorbed contaminants; adsorption (uptake) of dissolved contaminants by sediments or desorption from sediments; radionuclide decay; deposition of sorbed contaminants to the river bed or resuspension from the river bed; and

contributions of sorbed contaminants from point and nonpoint sources into the system. TODAM also computes changes in river bed conditions, including bed elevation, sediment size distribution, and sorbed contaminant distribution within the bed.

As discussed previously, contaminants in surface water will interact with contaminants from the land surface, the atmosphere, and groundwater. The interactions may involve radionuclides carried by runoff and eroded soil from the land surface, seepage from or to the ground water, dry and wet deposition of contaminants from the atmosphere, and volatilization of contaminants into the atmosphere (where applicable). Positive or negative contributions of contaminants and their interactions from outside sources are included in the one-dimensional model as a source or sink, except for volatilization, which is directly formulated in the model. Since each medium has its inherent time frame during which the contaminant migration and fate occurs, the time-frame difference between surface water and other media requires careful attention. A detailed description of TODAM is presented in a separate report under preparation.

The following items are input-data requirements of TODAM:

- channel geometry
  - flow characteristics
    - depth and velocity distributions
- sediment characteristics
  - sediment size distribution
  - density
  - critical shear stresses and erodibility coefficient for cohesive sediment as expressed in Partheniades (1962) and Krane's (1962) formulations
- contaminant characteristics
  - distribution coefficients
  - transfer rates
  - decay and degradation rates or associated parameters
  - initial conditions
  - boundary conditions

1. Sediment simulation for any given time

- longitudinal distributions of total sediment (sum of suspended and bed load) concentration for each sediment size fraction
- longitudinal distribution of sediment size fractions in the river bed
- change in bed elevation (elevation changes due to sediment deposition or scour or both)

2. Contaminant simulation for any given time

- longitudinal distribution of dissolved contaminant concentration
- longitudinal distribution of contaminant concentration adsorbed by sediment for each sediment size fraction
- longitudinal and vertical distributions of contaminant concentrations in the bottom sediment within the bed for each sediment size fraction.

The model is applicable to nontidal rivers and estuaries. However, because of its one-dimensionality, the model can be applied only to water bodies whose variations of depth, velocity, sediment concentration and contaminant concentrations in the vertical and lateral directions are small.

Like SERATRA, TODAM has several shortcomings. It requires rather extensive input data, thus placing a limit on its practical use. The model simulates contaminant migration and fate by including all important mechanisms. This requires fairly long computer time. Because we do not fully understand cohesive sediment transport (especially deposition and scouring), the model may not adequately predict silt and clay migration.

As we discussed earlier in this chapter, except in very simple cases, solutions to complex, partial-differential equations (PDE) are needed, and numerical techniques are usually employed instead of analytical techniques. Numerical discretization of the PDE's represents an approximation. Due to this approximation, the numerical solution and the analytical solution may not be exactly the same. Two common conditions usually discussed when numerical techniques are employed are stability and convergence. Stability refers to the ability of a numerical scheme to provide solutions which are bounded at a given time as the time increment tends toward zero. It alone does not necessarily mean that the deviation between the true solution and its numerical

approximation will be in any sense small. Convergence is the condition in which the solution of the numerical scheme approaches the true solution of the PDE for a finite grid size. Convergence many times focuses on defining a grid size small enough to approximate the partial differentials in the PDE. Numerical solutions computed by TODAM for two different grid sizes are presented in Figure 4 together with an analytical solution to the following time-dependent advection-diffusion equation:

$$\frac{\partial C}{\partial t} + U \frac{\partial C}{\partial x} = \epsilon_x \frac{\partial^2 C}{\partial x^2} \quad (11)$$

in which C = concentration, t = time, x = distance, U = flow velocity, and ϵ_x = diffusion coefficient. The analytical solution being:

$$C(x,t) = \exp [-t (\pi^2 + 1) - x] \sin(\pi x) \quad (12)$$

with the assumptions $U = -2$, $\epsilon_x = 1$, and concentrations at the boundaries equaling zero. As Figure 4 reveals, the smaller the grid size, the closer the numerical approximation is to the analytical solution. Δx was reduced from 0.1 to 0.025, while Δt was reduced from 0.05 to 0.0125.

If the "curves" are integrated with respect to x , the total mass per unit area can be calculated. The areas under the numerical solutions are larger than the area under the analytical solution. This discrepancy is due to unspecified gradients ($\partial C / \partial x$) at the boundaries (possibly having a different mass flux at the boundaries). As the grid gets larger, the numerical approximation for the gradients gets courser, resulting in a discrepancy of the "curves". It should be noted that, numerically, mass is conserved. A trade-off is usually required between grid size and degree of convergence. If the grid size is too small, the cost prohibits use of the model and if the grid size is too large, the results may converge to an unacceptable level.

FIGURE 4. Comparison of an Analytical Solution with Numerical Solutions

The ^{239}Pu transport simulation (calibration) results as presented in Section 4.3.2 indicate that the numerical discretization of South Mortandad Canyon under this study was probably too coarse to obtain accurate solutions. Even though mass was numerically conserved, the quantity of ^{239}Pu tended to increase as it progressed downstream. By reducing the grid size, the solution using the numerical approximation of the PDE for computing the ^{239}Pu concentrations should converge to the true solution. The solution of the grid size for the Mortandad Canyon ($\Delta x \approx 200$ m) was twice that of South Mortandad Canyon ($\Delta x = 405$ m) with the sediment mass (for cohesive sediment) being conserved with $\pm 8.4\%$. The 8.4% represents a 92.1% decrease over that of South Mortandad Canyon. In a study reported by Parkhurst et al. (1981) in which TODAM was used to simulate the transport and fate of the pesticide diazinon in an experimentally controlled channel, an increase in the resolution of the grid size significantly improved the accuracy of results. For this study, the channel was composed of alternating narrow-shallow riffles and wide-deep pools, which necessitated computational nodes 10.16 meters apart ($\Delta x = 10.16$ m). The discrepancy between observed and simulated diazinon loadings at five locations in the channel was within measurement error with the total diazinon mass being conserved within $\pm 1.5\%$.

Distributed Kinematic Wave Model for Channel Flows

Originally, the dynamic wave model DWOPER (Fread 1973) was to be used to route flows in Mortandad and South Mortandad Canyons, Los Alamos, New Mexico. However, as discussed in Section 4.13, DWOPER was found not to be applicable to these small "ditch like" systems. Hence, a kinematic wave model, DKWAV, was substituted to calculate depth and velocity distributions in these canyons.

The following paragraph briefly describes the model, its application, and limitations as they relate to this study. Further justification of this substitution is presented in Section 4.13. DKWAV (Distributed Kinematic Wave Model for Channel Flows) is an unsteady, one-dimensional, second-order, explicit, finite-difference model which simulates the hydrodynamics in dendritic river systems to obtain time-varying distributions of depth and velocity in a

channel. [The original model was applied to Ralston Creek watershed in Iowa (Whelan 1980).] The model, which can be easily combined with overland flow models, routes flows through arbitrarily shaped channels in which the channel reach is divided into sections bounded by points called "nodes." Flow routing is performed from node to node by a marching solution.

The equations of motion with the kinematic wave approximation are numerically analyzed via a modified version of the Lax-Wendroff, second-order, finite-difference scheme (Kibler and Woolhiser 1970; Rovey et al. 1977). Numerical stability is based upon the Courant conditions (Linsley et al. 1975). Point inflow or continuous (or both) lateral inflow is included in which point inflow occurs at nodes or continuous lateral inflow occurs between nodes. Associated with each channel section is its own seepage velocity. A cross-sectional area versus discharge relationship exists for each segment between nodes. Based on this relationship, other characteristic parameters (flow depth, wetted perimeter, and so forth) of each section can also be obtained (Croley 1977). Each channel section retains its own individual characteristics, which include a roughness parameter, lateral or point inflow rates, slope, seepage velocity, and natural cross-sectional shape. A more detailed model description, together with model formulations, are presented in a separate report under preparation.

Advantages of DKWAV are as follows:

1. The model incorporates a fast and efficient routing scheme which uses little computer storage.
2. Because there are few restrictions on the model and few parameters that must be estimated, a solution is easy to obtain.
3. The model allows for channel sections of zero length, which allows for an accurate description of dendritic river systems.
4. Any natural or geometric cross-sectional shape may be employed. Each section also retains its own roughness and slope parameters.

5. The initial conditions can be represented by a dry channel, or a base flow can be imposed.
6. The model allows for seepage from each channel section.
7. The model handles both point inflow and continuous lateral inflow or any combination of the two.
8. The model allows for insertion of channel or overland flood hydrographs anywhere within the watershed configuration.
9. The model handles culverts.

DKWAV has the following limitations:

1. A kinematic wave approximation is employed instead of a complete momentum equation of unsteady open channel flows. Hence,
  - a. The model is restricted to areas where the bed slope dominates other terms (pressure, inertia, and the like) in the momentum equation.
  - b. Backwater effects cannot be modeled.
  - c. Kinematic shocks cannot be modeled. These may arise when there is a considerable change in the flow over a relatively short distance.
2. A modified Lax-Wendroff, second-order, explicit, finite-difference scheme is used to analyze the equations of motion. Hence, a rather strict stability criterion is required based upon the Courant condition. This criterion imposes relatively small-time increments. A criterion has been developed establishing the conditions under which the kinematic wave approximation may be used.
3. Since the model cannot be calibrated a priori, several calibrating runs are necessary.

3.6.3 Terrestrial Ecological Model

As stated previously, the primary criterion of model selection is the soundness of the model concept in including all major mechanisms affecting

radionuclide migration and fate. The biological transport model for transuranics and LLW, BIOTRAN, is one of few currently available models and was selected by LANL for this study (see Section 3.5). Although BIOTRAN was originally developed for the LANL environs, it is general enough to be applied to other sites with minor modification.

As discussed in Section 3.5, BIOTRAN stochastically generates precipitation, temperature, and insolation on a daily basis stochastically by using random selection from a normal distribution based on each parameter's long-term average and variance. These outputs, together with soil moisture parameters at the site are used to drive biomass production and root uptake of transuranics of LLW. A bookkeeping system is used to track inputs of contaminants from air and soil, as well as build-ups in contaminant levels in the soil, plant, and herbivore biomasses of the site. Cycling of radionuclide contaminants through the compartments of living and dead plant parts, humus, and upper soil layers is simulated continuously. As plant communities mature through successive stages to climax, a state of quasi-equilibrium is typically reached, which is characteristic of the site soil type, temperature, and moisture conditions, and of the plant community being simulated. Contaminant levels at this stage, along with other physical and biological parameters, are used as part of the inputs to the abiotic models.

3.6.4 Atmospheric and Groundwater Models

Development of the atmospheric model, AIRTRAN, is currently underway to simulate transport, deposition and resuspension of LLW. AIRTRAN is being applied to the LANL site. PNL and LANL are also running the unsaturated ground-water model, UNSAT (Gupta et al. 1978, Gee and Simmons 1979) and BIOTRAN to identify the importance of unsaturated groundwater movement on plant transpiration and to upgrade the plant transpiration mechanisms formulated in BIOTRAN. In the next fiscal year, AIRTRAN and saturated and unsaturated groundwater models along with an accompanying transport model, are expected to be incorporated to the multimedia LLW exposure assessment methodology.

4.0 APPLICATION OF METHODOLOGY

Prior to the full development of the proposed assessment methodology, LANL and PNL decided first to apply a scaled-down version of the methodology (involving terrestrial plants and overland and instream compartments) to an actual waste disposal site. The purpose of this effort was to evaluate and, at the same time, demonstrate the attributes of the methodology. We evaluated 1) the capability of each component of the methodology to simulate radionuclide transport within its media--inclusive of the major mechanisms of radionuclide migration and fate, 2) the soundness of coupling several components of the methodology and the ability of the integrated methodology to simulate radionuclide migration in several media, and 3) the applicability and limitations of the integrated methodology and its required input data.

Although state-of-the-art models with proven histories of success were selected for each component of the integrated methodology, the coupling of these models required careful attention as to the mass conservation between components, the differences in temporal and spatial resolution among selected models, and the selection of appropriate interfaces between models, based on the physical, chemical, and biological processes involved. In particular, we evaluated the adequacy of the methodology in light of the accuracy of LLW source characteristics at the waste disposal sites, our current understanding of the radionuclide migration and risk involved, our efforts to perform the analyses, and the availability of required input data.

4.1 LOGISTICS OF APPLYING THE MODEL

The following sections describe the test site, the selection process of the numerical codes, the methodology employed for coupling the codes, and the waste disposal scenario under consideration.

4.1.1 The Site

The LANL reservation was selected as the application site. Environmental data collected at the site are rather typical of those available at many DOE waste disposal sites. However, the reservation's unique hydrologic,

watershed, and meteorological conditions (e.g., large spatial and temporal variations in hydrology) strongly challenge the current modeling capabilities.

The laboratory is located on the eastern slope of the Jemez Mountains, a portion of the Pajarito Plateau in north-central New Mexico (Figure 5). The plateau is dissected by numerous canyons, all of which eventually drain into the Rio Grande River. Waste burial sites are located on the tops of finger-like mesas with a relief of 25 to 60 m above the floors of adjacent canyons. The area has a semi-arid, continental mountain climate. Annual precipitation averages about 46 cm at the higher elevations (2200 m), and about 18 cm at the lower elevations (1650 m). Summer rainfall accounts for about 75% of the annual precipitation (Hakonson et al. 1976).

A review of available data showed that among all the canyons comprising the LANL reservation, Mortandad and South Mortandad Canyons are the best documented. Since 1943, liquid radioactive wastes have been treated and released to Mortandad Canyon. The stream channel near the contaminant discharge point is narrow (generally less than 1.5 m in width), rocky, and contains thin layers of alluvial sediment. Effluent discharges to the canyon rapidly percolate into the alluvium within about 800 to 1200 m downstream of the outfall. During major storms, large surface water flows traverse about 3000 m below the outfall before disappearing into the alluvium (Hakonson et al. 1976). Area C, a small shallow, land disposal site, is located in a side-canyon (South Mortandad Canyon) within the Mortandad Canyon drainage. Figure 6 shows Mortandad and South Mortandad Canyons, as well as the location of Area C and the effluent discharge point (outfall).

Figure 6 also shows the location of the meteorological stations and stream-flow gauges in the Mortandad Canyon catchment. Three rain gauges are within close proximity to the basin, and a number of other daily total rainfall gauging stations are located on or near the LANL reservation. Storm runoff is measured at two locations in the canyon: GS-1 (Gauging Station 1) and GS-2 (Gauging Station 2). Stream-flow records at these two stations are highly discontinuous. Few data were collected on sediment or radionuclide transport during storm events.

FIGURE 5. Topography of the Los Alamos Area in New Mexico

4.1.2 Pathway Selection

A preliminary analysis of various pathways of radionuclide transport, which could result in the release of plutonium from LANL burial grounds to the environment, showed that plant, overland and instream pathways must be evaluated in detail (Wheeler et al. 1977). Wheeler et al. (1977) showed that over

FIGURE 6. Mortandad and South Mortandad Canyons Upstream of the "Garden" Location

relatively long time periods, deep rooted plants could take up plutonium and deposit it on the land surface through dry-fall and litter decomposition. Subsequent precipitation can transport contaminated soils overland to channel systems. Once in a channel, radionuclides can migrate considerable distances through processes of instream sediment transport, deposition, and resuspension.

These findings, along with the ready availability of biotic and abiotic codes to simulate plant, overland, and surface-water pathways, provided the basis for selecting the limits of the methodology demonstration effort.

4.1.3 Model Selection

The types of models required to simulate radionuclide migration through these three pathways include: a terrestrial ecology model, an overland runoff-soil erosion model, and in-stream sediment-radionuclide transport model. The primary criteria for the initial selection of models were the availability of state-of-the-art modeling capability in each of the above areas, and the adequacy of the models in describing the unique characteristics of the sites. Mortandad and South Mortandad Canyons represent a unique region, which calls for a model with certain unique features. This region's channels are generally dry between storms. The channels may be described as ditch-like, with high representative roughness values as opposed to the more traditional river bed. Low flow discharges and depths are typical in the region. In most instances, the seepage rate from the channel exceeds the inflow rate to the channel. Consequently, the channel goes dry by the time the flow reaches the Garden location (Figure 6). The channel slopes are also quite high, ranging from 1% to 20%.

PNL has been active for a number of years on the subject of overland and instream contaminant transport. They have developed and are now refining models to predict radionuclide migration in rivers, lakes, estuaries, and oceans. The river/river-run reservoir model, SERATRA (Onishi 1977; Onishi et al. 1979a), has been most extensively tested on the Columbia River in Washington, on the Clinch River in Tennessee, and on Cattaraugus and Buttermilk Creeks

in New York. Simulations of pesticide migration have also been performed on Four Mile and Wolf Creeks in Iowa using SERATRA (Onishi et al. 1979b). For this study, SERATRA was simplified to a one-dimensional model called TODAM (Transient One-Dimensional Degradation and Migration Model) as discussed in Chapter 3.

To supply depth and discharge distributions to TODAM, The dynamic wave model, DWOPER (Fread 1973), was originally employed as the hydrodynamic code for routing flows in Mortandad and South Mortandad Canyons. Several problems were encountered when DWOPER was applied to this region with its unique features. The main problem was supplying appropriate non-zero initial conditions to DWOPER. Such conditions are not an issue in large river systems, but they become increasingly important in smaller systems. The steep slopes, low flows, and small depths made it increasingly hard for DWOPER to handle these flows. The problem arose in the iterative scheme of DWOPER for the solution of the water surface elevations in the river reach. When the depth change during the iteration process became larger than the depth of flow, then the water surface elevation dropped below the channel bottom. It must be noted that DWOPER was developed and tested on many larger river systems with excellent results. By employing the kinematic criterion, as described by Yevjevich and Mahmood (1975), the adequacy of the kinematic flow approximation for this study can be analyzed as follows. When Manning's roughness values of 0.03 and 0.06 were used in the analysis of the channel section with the smallest grade (slope equalling 0.0116), the respective Froude numbers (Fr) were 0.63 and 0.33, respectively. The criterion for establishing kinematic flow is $Fr = 1.5$. When the Froude number is below 1.5, the dynamic waves are rapidly attenuated (Yevjevich and Mahmood 1975) and the flow is well within the criterion for establishing kinematic flow. Due to these flow characteristics in Mortandad and South Mortandad Canyons, the Distributed Kinematic Wave Model for Channel Flows, DKWAV (Whelan 1980) was selected as the hydrodynamic code for routing flows through the channel system.

The overland, runoff-soil erosion model selected for this study was ARM, the Agricultural Runoff Management model (Donigian and Crawford 1976). While

ARM was developed specifically to model pesticide and nutrient losses from agricultural lands, the model is highly flexible and applicable to radionuclide transport as well. Over the last few years, LANL has been actively developing a biological transport model for plutonium and other transuranics. Although the model BIOTRAN was originally developed for the LANL environs, it is general enough to be applied to other sites with minor modifications. The specific capabilities of BIOTRAN, TODAM, DKWAV, and ARM are presented in Chapter 3.

4.1.4 Model Coupling

To adequately simulate the migration of radionuclides through the plant-uptake, litter dryfall, and water/sediment transport pathways, the selected models had to be coupled at appropriate interfaces. Figure 7 presents the coupling methodology for the models and the information transferred at each interface. BIOTRAN is the first model in the system. It is coupled to ARM and supplies it with two types of simulation results, namely, predicted radionuclide concentrations varying temporally on the land surface and monthly estimates of the percent of area covered by vegetation. The latter is required to estimate the ARM parameters controlling evapotranspiration, interception, snowmelt, and soil erosion as it is affected by forest density. The second model in the system, ARM, generates two types of information, which provide boundary conditions to the models DKWAV and TODAM. ARM produces information on overland runoff and edge-of-stream sediment and radionuclide loadings. The third model in the system, DKWAV, performs the hydrodynamic channel simulation and provides the in-stream flow discharge, cross-sectional area, flow depth, shear stress, net inflow, and wetted perimeter for use by TODAM. TODAM, the fourth model in the system, simulates the in-stream migration and fate of sediment and radionuclides.

4.1.5 Waste Disposal Scenario

A waste management scenario was developed for the Area C disposal site in South Montandad and Montandad Canyons to provide a basis for demonstrating the methodology's applicability. It was assumed that management of Area C had

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FIGURE 7. Model Coupling and Information Transfer

ceased, and that natural vegetation had become reestablished over a period of 100 years. Ponderosa pine, deciduous shrubs (e.g., oak), and cool season grasses, in succession, were the three plant types simulated by BIOTRAN.

The disposal site conditions were assumed to be as follows:

- Annual precipitation = 7.1 cm.
  - Soil properties are approximately 28% field capacity, 35% holding capacity, and have a wilting point of 10%.
3. Plants grow over the LLW burial trenches, which are characterized as having a 1.5 m cap containing no radioactivity and a layer of fill 6.5 m deep that is uniformly contaminated with ^{239}Pu to a concentration of 800 pCi/gm.

- All plants are assumed to have essentially the same depth and distribution of roots; roots are present in the first 2 cm, 75% of the roots appear in the next 25 cm, 18% appear in the following 25 cm, 4% in the following, and so on until only a small fraction ($1.5 \times 10^{-2}\%$) of the roots actually makes contact with the first layer (25 cm) of waste. We assume that none of the trench cover is lost to wind or water erosion.

Through the natural process of plant uptake, litter dryfall, and litter decomposition, ^{239}Pu is deposited on the land surface. Runoff events produced by successive rainfall and snowmelt transport ^{239}Pu , in both dissolved and particulate forms, to South Mortandad Canyon (which joins Montandad Canyon). Transport, deposition, and resuspension of sediments in the channel within the canyon spread the contaminants downstream in the canyons.

The modeling approach for the above scenario involves simulating 100 years of plant growth at Area C with BIOTRAN. Then, using the data that BIOTRAN has produced with regard to plutonium deposition, the other models - ARM, DKWAV, and TODAM - simulate the overland and in-stream transport of ^{239}Pu for two years. Coupling the models includes predicted plutonium levels on the land surface input to ARM. The runoff events produced by ARM are provided to DKWAV for the hydrodynamic simulation. Outputs of runoff, soil erosion, and plutonium produced by ARM and DKWAV are then input to TODAM. Since the BIOTRAN simulation was performed by LANL, this report mainly discusses the results of using ARM, DKWAV and TODAM.

4.2 OVERLAND MODELING

This section describes the model calibration effort followed by the simulation results of ARM for runoff and sediment and ^{239}Pu migration on the land surface.

4.2.1 Model Calibration

Prior to applying ARM to Mortandad and South Mortandad Canyons, several model parameters had to be estimated through a calibration process. Model calibration involves establishing initial input values for certain hydrology

and sediment parameters using guidelines presented in the ARM users manual (Donigian and Davis 1978). These values are then adjusted until simulation results match the measured data.

In all, over thirty hydrology and sediment parameters had to be evaluated. Sensitivity studies have shown, that ARM simulation results are more sensitive to some parameters than others (Donigian and Crawford 1976), so only a few parameters were adjusted during the calibration process. The most important calibration parameters and the methods used in their evaluation are described below.

LZSN, the nominal, lower-zone, soil-moisture storage parameter, is related to the annual cycle of rainfall and evapotranspiration. An initial value of 5.0 was determined from a map of LZSN values of the United States based on numerous past calibrations of the hydrologic component of ARM and the Stanford Watershed Model. During the calibration process, LZSN was reduced to 3.0 to increase the peak rate of runoff and runoff volume over those of the initial calibration run.

INFIL is an index to the mean infiltration rate on the watershed and is generally related to soil permeability. Values for INFIL have been estimated using the Soil Conservation Service's hydrologic soil groups (1974). Based on these values, the soils in Mortandad Canyon have an INFIL value of 0.05. During the calibration process, this value was reduced to 0.02 to increase runoff volume.

INTER affects the interflow component of runoff and is used to adjust runoff timing. A map of INTER values for the United States shows a value of 3.0 for the Mortandad Canyon region. Based on the calibration procedure, this value was reduced to 0.02 so we could simulate hydrographs with higher peaks and shorter durations.

NN is the roughness coefficient for overland flow. The guidelines suggest a value of 0.35 for both the heavy turf and forest-litter conditions found in Mortandad Canyon. This value was not changed during the calibration process.

The main calibration parameter for the sedimentation is KSER. This single parameter combines the effects of slope, overland flow length, sediment particle size, and surface roughness on sediment transport capacity of overland flow. Suggested values range from 0.01 to 5.0. However, during the calibration process, a value of 8.0 was established. A comparison of initial and final values for each of the above parameters is summarized in Table 2.

TABLE 2. Comparison of Initial and Final Values of ARM Calibration Parameters

<u>Calibration Parameter</u>	<u>Initial Value</u>	<u>Final Value</u>
LZSN	5.0	3.0
INFIL	0.05	0.02
INTER	3.0	0.02
NN	0.35	0.35
KSER	0.01-5.0	8.0

The radionuclide transport simulation in ARM is largely dependent on the value of three input parameters, the distribution coefficient K_d as K_d , the equilibrium radionuclide concentration in solution C , and the initial radionuclide level, SSTR. The methods for estimating these parameters are described below.

1. Data describing the distribution coefficient, K_d , for the land surface sediment distribution, were not available for Area C. Fortunately, the instream distribution of radionuclides for Mortandad Canyon can be found in a report by Hakonson et al. (1976). Using these data, we estimated a K_d value for each of the sediment size classes (sand, silt, and clay). Then, using the land-surface, sediment size-distribution of nearby DP Canyon reported by Purtyman (1974), a bulk K_d value of 21,600 ml/mg was determined for the Mortandad Canyon watershed. These computations are summarized in Table 3.
2. The initial activity level for Area C was determined from the 100-year BIOTRAN simulation. Assuming that only the top 2 cm of the

TABLE 3. Summary of Area C Kd Estimates

Composite Sediment	Kd Value Associated With Instream Sediment (Hakonson et al. 1976) (m ³ /kg)	DP Canyon Sediment Size Distribution (Purtyman 1974)	Area C Kd Value (m ³ /kg)
Sand	3	20%	21.6
Silt	15	20%	
Clay	30	60%	

soil surface is subject to erosion, and assuming that the erodable material consists of 99% soil and 1% litter-humus, there were 520 pCi of ²³⁹Pu uniformly distributed over the surface of Area C, available for transport at the beginning of the ARM simulation period.

3. Finally, the equilibrium radionuclide concentration in the dissolved form was estimated to be 3.6 pCi/l of water based on instream measurements reported by Hakonson (1976).

The area selected for the model calibration of ARM was the portion of Mortandad Canyon above the GS-1 gauging station (see Figure 6). This catchment is approximately 1.24 sq. km in size, 10% of which is impervious (i.e., parking lots, streets, and roof tops). The time period selected for model calibration was June 1, 1975 to August 11, 1976. Meteorological input data for this period were obtained from the stations listed in Table 4. The first 12 months of the calibration time period were used to stabilize the simulated water balance and dampen the effect of assumed initial antecedant moisture conditions. The last two months were used for model calibration. During this period, six major storm runoff events and several smaller events were measured at GS-1. Discharge data are also available at GS-2 for the same time period. No sediment discharge or radionuclide data were available at either station.

Usually, the acceptability of an ARM calibration effort has been based on the accuracy of simulated annual, monthly, and individual storm runoff volumes. The next most important criterion is the temporal distribution of runoff during individual storm events. While these criteria are important in

TABLE 4. Sources of Meteorological Input Data

<u>Data Type</u>	<u>Frequency</u>	<u>Station Name and Location</u>
Precipitation	30 minutes	OHL Bldg, LASL, SM-43, LASL LAMF, LASL
Wind	Daily Total	SM-43, LASL
Solar Radiation	Daily Total	Albuquerque, NM
Maximal/Minimal Air Temperature	Daily	SM-43, LASL
Relative Humidity	Daily Average	SM-43, LASL
Dewpoint	Daily Average	(1)
Pan Evaporation	Daily Total	(2)

(1) Calculated from air temperature and relative humidity

(2) Calculated from wind, solar radiation, air temperature, and dewpoint

achieving a proper simulation of watershed hydrology, sediment transport may not be adequately simulated. Since radionuclide migration is largely controlled by sediment movement, a third criterion, transport energy for sediment, was also included in the selection of model parameters. This criterion is of particular importance because of the lack of measured sediment data needed to estimate erosion-related model parameters.

Figures 8 through 13 compare the simulated hydrographs for each storm event in July and August of 1976 with those measured at GS-1. These figures also show the precipitation distributions measured at the OHL, SM-43, and LAMF gauges, in addition to the weighted areal average distributions input to ARM. The simulated hydrographs for all but the 7/21/76 storm event were shifted in time to facilitate direct comparison of simulated and measured runoff. Table 5 gives a comparison between simulated and measured runoff volumes for each runoff event, as well as monthly totals for July and August.

FIGURE 8. Comparison of Simulated and Measured Runoff for the Storm of 7/14/76

Several factors contributed to the differences between simulated and measured runoff at GS-1. These factors were:

The contoured precipitation data provided by LANL for each event in July and August were the basis for developing representative areal average storm volumes for input to ARM. Because rainfall was highly variable over time for each event (see Figures 10 and 12), it was difficult to construct a

FIGURE 9. Comparison of Simulated and Measured Runoff for the Storm of 7/17/76

FIGURE 10. Comparison of Simulated and Measured Runoff for the Storm of 7/21/76

representative rainfall distribution for upper Mortandad Canyon. Since the OHL gauge is the closest recording gauge to the centroid of the catchment, the rainfall distribution measured at this location was used to distribute the areal volumes throughout five of the six events. The precipitation distribution measured by the SM-43 gauge was found to be more representative for the 7/21 storm event. The assumption of a uniform spatial distribution and the use of a single point measurement to typify the temporal distribution of

FIGURE 11. Comparison of Simulated and Measured Runoff for the Storm of 7/27/76

rainfall over upper Mortandad Canyon partially explain the differences in hydrograph shape. They also explain the 2 to 4 hr differences in timing. During certain events, timing differed by as much as 2 hr at the OHL, SM-43, and LAMF gauges. These differences may be due to the high temporal variability of summer precipitation or to errors in resetting the recorders when charts were replaced at each gauge. The timing of the measured hydrographs should be accurate since timing was established using effluent discharge records from the outfall (see Figure 9).

FIGURE 12. Comparison of Simulated and Measured Runoff for the Storm of 8/1/76

The PNL version of the ARM model does not account for the effects of channel routing on hydrograph shape and timing. The model should be applied to areas that are small enough so that channel processes are negligible. The limiting catchment size is approximately 2 sq. km. Although upper Mortandad Canyon is only 1.24 sq. km in size, it appears that the channel above GS-1 has an effect on hydrograph shape, as indicated by the relatively flat shape of the measured hydrographs. A long, narrow watershed like Mortandad Canyon

FIGURE 13. Comparison of Simulated and Measured Runoff for the Storm of 8/11/76

will generally produce flat hydrographs, while round- or pear-shaped watersheds tend to produce peaked hydrographs. Channel routing effects above GS-1 may also have contributed to differences in timing between simulated and measured runoff.

The ARM model assumes that all impervious areas in a catchment are located adjacent to the stream channel. This assumption is generally acceptable for agricultural watersheds, which have little or no impervious area. In

TABLE 5. Comparison of Simulated and Measured Runoff at GS-1, Mortandad Canyon

Date	Precipitation (in.)	Storm Runoff	
		Measured (in.)	Simulated (in.)
7/12/76	0.39	0.00006	0.003
7/14/76	0.72	0.016	0.011
7/16/76	0.72	0.019	0.045
7/20/76	0.35	0.0002	0.002
7/21/76	0.55	0.015	0.018
7/22/76	0.14	-	0.003
7/26/76	0.25	-	0.001
7/27/76	0.43	<u>0.014</u>	<u>0.008</u>
Monthly			
Total for			
JULY		0.066	0.091
8/1/76	0.53	0.006	0.007
8/11/76	0.80	<u>0.017</u>	<u>0.022</u>
Monthly			
Total for			
AUGUST		0.023	0.029

upper Mortandad Canyon, however, approximately 10% of the total area is impervious, and most of it is located at the upper end of the catchment. Runoff generated from this area has to travel over a large pervious area before entering a drainage channel. In order to represent this effect in the ARM model, the amount of impervious area had to be reduced to 1%.

Since there were no measured suspended sediment data for the GS-1 station for July and August of 1976, simulation results were evaluated on the basis of a limited number of observations at the GS-2 station. Instantaneous measurements of discharge and suspended sediment loadings for the September 15, 1974 runoff event, as obtained from Hakonson et al. (1976), are shown in Figure 14. The solid line on the figure represents a best-fit relationship through the measured values.

FIGURE 14. Simulated and Measure Suspended Sediment Loads in Mortandad Canyon

As discussed by Hakonson et al. (1976), a change in the suspended sediment load-discharge relationship occurs at $0.25 \text{ m}^3/\text{s}$. Figure 15, a plot of suspended sediment concentrations and discharge, shows that concentrations are relatively constant from 0.07 to $0.25 \text{ m}^3/\text{s}$, but that concentrations increase significantly with discharge from 0.25 to $0.30 \text{ m}^3/\text{s}$. A possible interpretation of this occurrence is that $0.25 \text{ m}^3/\text{s}$ corresponds to a threshold condition at which the bed shear stress is sufficiently high to begin resuspending the bed material. At greater discharges, increasing amounts of suspended material produce higher concentrations. At lower discharges, on the other hand, the relatively constant concentrations indicate that all available fine material from the land surface is being transported. This is a characteristic of washload (such as clay and silt materials), since its concentration in the

FIGURE 15. Measured Discharge and Suspended Sediment Concentrations at GS-2; Storm of 9/15/74. (Taken from Hakonson et al. 1976.)

stream is primarily determined by the amount eroded from the land, not by the stream's transport capacity. Hence, the lower portion of the solid line (Figure 15) represents the relationship between sediment load and discharge from the 452-acre drainage basin above GS-2 for the September 15 runoff event. Since no other data are available, it is assumed that this relationship is representative of long-term conditions.

Of the six runoff producing events, only four had enough runoff to generate appreciable soil erosion. Most of the simulated loadings at GS-1 are reasonably close to the sediment load-discharge relationship observed at GS-2 1475 meters downstream (see Figure 14), indicating that the ARM calibration for Mortandad Canyon is reasonable.

4.2.2 ARM Application Results

Based on the results of the hydrology and sediment calibration discussed in Section 4.2.1, the ARM model was applied to the 3.18-km² drainage area of

Mortandad Canyon above the Garden. For the simulation, this area was divided into thirteen subareas, including Area C, with drainage areas of 7 to 246 acres. Overland flow and sediment erosion were simulated for all thirteen subareas, while radionuclide transport was simulated for Area C only.

The simulation period for the application of ARM to Mortandad Canyon above the Garden was from June 1, 1975 to May 30, 1977. The overland flow and sediment and radionuclide transport simulation results for Area C are shown in Figures 16, 17, and 18. Over the simulation period, 1% of the initial 502 pCi of ^{239}Pu deposited on the land surface at Area C was transported from Area C into South Mortandad Canyon. However, as pointed out in Section 4.2.1, a full year of simulation is required by ARM to stabilize the water balance. Therefore, the results for the storms in 1975 are somewhat questionable. The remaining significant event is that of July 17, 1976. This event produced a peak discharge of $0.02 \text{ m}^3/\text{s}$ from Area C, a maximum sediment concentration of about 17 kg/m^3 , and radionuclide concentration of $1.7 \times 10^{-4} \text{ pCi/m}^3$.

4.3 IN-STREAM MODELING

This section presents the calibration and simulation results of DKWAV and TODAM for in-stream flow, sediment transport, and radionuclide transport in Mortandad and South Mortandad Canyons.

4.3.1 DKWAV Model Calibration

Prior to applying DKWAV to Mortandad and South Mortandad Canyons, seepage velocities and roughness coefficients had to be estimated for each channel cross section. Model calibration involved adjusting the roughness parameters and the seepage velocities until the patterns produced by the simulation results reflected those patterns produced by the measured data. The area employed for calibration was the channel section on Mortandad Canyon between gauging stations GS-1 and GS-2. Since ARM provides the upstream boundary conditions at GS-1 and lateral inflow information, predicted hydrographs at GS-2 were compared with measured data.

In calibrating DKWAV, five basic parameters were investigated: flow volume (V), time of concentration (t_c) between GS-1 and GS-2, peak discharge (Q_p),

SOUTH MORANDAD CANYON - SOURCE - 7/17/75 - 8/11/76

FIGURE 16. Simulated Discharge, Area C

FIGURE 17. Simulated Total Sediment Concentrations, Area C

FIGURE 18. Simulated Total Radionuclide Concentrations, Area C

Manning's roughness coefficient (n), and seepage velocity (V_s). The storms of July 16, 1976 and August 11, 1976 were employed in the calibration procedure. These storms were chosen because they were previously used in the calibration process of ARM.

The calibration process was subdivided into four areas involving 1) running DKWAV for various values of n and V_s ; 2) analyzing the resulting 25 runs; 3) determining the seepage velocity and roughness coefficient; and 4) establishing the validity of these latter two coefficients. Approximately twenty-five runs for both storms were performed using DKWAV. " n " ranged from 0.02 to 0.20, while V_s ranged from 1.0×10^{-5} m/s to 5.4×10^{-5} m/s. Lateral inflow was supplied by ARM and all major mechanisms (seepage, lateral inflow, and so forth) occurring in the reach between GS-1 and GS-2 were included.

In analyzing the twenty-five runs (all pertaining to GS-2) peak discharges and seepage velocities were plotted on log-log paper for lines of constant n . This plot revealed that the seepage velocity had to be decreased as the roughness coefficient was increased to maintain a constant peak discharge at GS-2. Since the time of concentration increases with the bed roughness, the seepage velocity has to be increased to maintain Q_p . The reverse is true if the roughness coefficient is decreased. Hence, for a given Q_p and travel time (between hydrograph peaks), the seepage velocity and roughness coefficient may be uniquely determined.

Watershed response to the July and August storms was different from run to run, as measured at GS-1. Individually, each storm could be modeled. It was impossible to obtain a general relationship among peak discharge, flow volume, travel time, seepage velocity, and roughness value to reproduce the hydrographs resulting from both storms. This conclusion is substantiated by the measured results of the July and August storms. Table 5 presents the peak discharges, flood volumes, travel times, and percent decrease in volume for each storm at GS-1 and GS-2. Figures 19, 20, 21 and 22 present the measured results (designated by circles) for both storms at GS-1 and GS-2. At GS-1, the measured hydrographs are similar in shape, magnitude and flow volume (Figures 19 and 21 and Table 5) for the July and August storms. Less than a

FIGURE 19. Simulated Flow Rate by ARM with Measured Data at GS-1 for the July 1976 Storm

FIGURE 20. Simulated Flow Rate by DKWAV with Measured Data at GS-2 for the July 1976 Storm

FIGURE 21. Simulated Flow Rate by ARM with Measured Data at GS-1 for the August 1976 Storm Event

kilometer downstream at GS-2, however, the hydrographs are radically different between these two runoff events. The hydrograph peak and volume for August decreased by 90% and 85%, respectively, and the travel time was 45 minutes. On the other hand, the peak of the July hydrograph did not decrease; its travel time between the hydrograph peaks at GS-1 and GS-2 was zero; and its volume only decreased by 10.4%. Obviously, the rainfall distribution had an effect on the flood routing, but when considering all circumstances (length of routing, velocity, initial shape of the hydrograph, initial flood volume, and peak), certain trends should still be seen (e.g., hydrograph attenuation, translation, and so forth). Since the watershed responded differently to the July and August events, different conditions appear to have existed on the catchment for each storm.

FIGURE 22. Simulated Flow Rate by DKWAV with Measured Data at GS-2 for the August 1976 Storm Event

To determine the seepage velocity and the roughness coefficient, measured data from the July and August storm events were employed. It must be noted that the lateral inflow supplied by ARM had a pronounced effect on the computed hydrographs as they were routed from GS-1 to GS-2. The computed hydrographs for the July and August storm events were modeled based upon the changes in peak discharge and flow volume exhibited by the measured data. The peak discharge and flow volume for each computed hydrograph were chosen so that the same relationship between the seepage velocity and the roughness coefficient existed for each storm event in the channel between GS-1 and GS-2. By employing the actual concentration time caused by the August storm, the roughness coefficient and seepage velocity were determined as 0.045 and 2.35×10^{-5} m/s, respectively. Figures 19 and 21 present the flood hydrographs predicted by ARM at GS-1 for the July and August storms, respectively, while Figures 19 and 21 present the flood hydrographs predicted by DKWAV for the July and August storm events, respectively, at GS-2. These hydrographs

are designated by solid lines in Figures 19 through 22. Table 6 presents the computed results for peak discharge, flow volume, concentration time, and percent decrease in volume at GS-1 and GS-2 for the July and August storms.

The calibration results were independently checked with data provided by LANL and by Hakonson et al. (1976). LANL provided seepage information for two different levels of flood discharge. For average discharges of $0.0185 \text{ m}^3/\text{s}$, the seepage rate ranged from $7.8 \times 10^{-6} \text{ m}^3/\text{s}$ per meter of length to $1.1 \times 10^{-5} \text{ m}^3/\text{s}$ per meter of length. DKWAV predicted $2.3 \times 10^{-5} \text{ m}^3/\text{s}$ per meter of length. For average discharges of $0.10 \text{ m}^3/\text{s}$, the seepage rate reported by LANL ranged from $1.2 \times 10^{-5} \text{ m}^3/\text{s}$ per meter of length to $3.4 \times 10^{-5} \text{ m}^3/\text{s}$ per meter of length. DKWAV predicted $3.5 \times 10^{-5} \text{ m}^3/\text{s}$ per meter of length. Hence, the seepage rate selected by this study was judged reasonable.

Hakonson et al. (1976) provided data pertaining to the September 15, 1974 storm, which is similar to the July 16, 1976 event (considering the quantity of data available). The measured hydrographs pertaining to the September event at GS-1 and GS-2 have been plotted in Figures 22 and 23, respectively (triangles). Hakonson et al. (1976) also provided information about peak discharge, water depth, and velocity at GS-2 for the September event ($0.3 \text{ m}^3/\text{s}$, 0.31 m , 1.5 m/s , respectively). From the computed results provided by DKWAV for the July storm event, the peak discharge, water depth, and velocity were $0.35 \text{ m}^3/\text{s}$, 0.31 m , and 1.1 m/s , respectively, revealing good agreement with measured data.

The seepage values computed for the channel reach between GS-1 and GS-2 were used on South Mortandad Canyon. The roughness values which were used on South Mortandad Canyon were increased to 0.065 and 0.055 for the upper and lower half of South Mortandad Canyon, respectively. In general, tributaries typically have larger roughness values, than main streams, because when comparing tributaries to their main streams, their flow depths and discharges are smaller. Tributary channels are usually less developed, and more obstruction to the flow exists (e.g., trees, large stones, and other debris). Hence, although data were not provided, this assumption on the roughness values appears reasonable.

TABLE 6. Data for the Storms of July 16, 1976 and August 11, 1976

	<u>July 16, 1976</u>	<u>August 11, 1976</u>
MEASURED RESULTS		
<u>GS-1 (Measured Data)</u>		
Peak Discharge (Qp)	0.156 m ³ /s	0.125 m ³ /s
Flow Volume (V)	599.0 m ³	604.0 m ³
 <u>GS-2 (Measured Data)</u>		
Peak Discharge (Qp)	0.156 m ³ /s	0.012 m ³ /s
Flow Volume (V)	537.0 m ³	90.0 m ³
Concentration Time (tc)	0	45 minutes
% Decrease in Volume	10.4	85.0
 COMPUTED RESULTS		
<u>GS-1 (ARM Results)</u>		
Peak Discharge (Qp)	0.323 m ³ /s	0.106 m ³ /s
Flow Volume (V)	1161.0 m ³	468.0 m ³
 <u>GS-2 (DKWAV Results)</u>		
Peak Discharge (Qp)	0.356 m ³ /s	0.066 m ³ /s
Flow Volume (V)	779.3 m ³	141.7 m ³
Concentration Time (tc)	30 minutes	45 minutes
% Decrease in Volume	32.9	69.7

FIGURE 23. Simulated Flow Rate by ARM with Measured Data at GS-1 for the September 1974 and July 1976 Storm Events

GS-2 LOCATION

FIGURE 24. Simulated Flow Rate by DKWAV with Measured Data at GS-2 for the September 1974 and July 1976 Storm Events

4.3.2 TODAM Calibration

Sediment transport and adsorption/desorption capacities differ significantly by sediment size, so TODAM simulated the following seven substance: sand, silt, clay, particulate ^{239}Pu adsorbed by sand, particulate ^{239}Pu adsorbed by silt, particulate ^{239}Pu adsorbed by clay, and dissolved ^{239}Pu . Calibrating TODAM involved defining a number of parameters, performing model runs, inspecting the results, and repeating the procedure until satisfactory results were obtained. The parameters, most of which were defined by sediment size, were sediment diameter, fall velocity, specific gravity, erodibility coefficient and critical shear stress associated with the cohesive soils, distribution coefficients of ^{239}Pu with three sizes of sediments, and the transfer rates of ^{239}Pu to sediment. Selecting a longitudinal dispersion coefficient value was also a necessity. The method employed for assigning values to each parameter is described below:

Sediment Diameter

Because the sediment size distributions of silt and clay for Mortandad and South Mortandad Canyons were unavailable, geometrical mean values associated with these size fractions were used instead. The sediment diameters for silt and clay were assumed to be 1.6×10^{-5} m and 1.0×10^{-6} m, respectively.

The size distribution of sand, on the other hand, was available for DP Canyon (Purtymun 1974), which neighbors Mortandad and South Mortandad Canyons. We assumed that the sediment distribution for all three canyons would be similar. The median sand-sediment size of DP Canyon (i.e., the size for which 50% of the material is finer or coarser) of 2.5×10^{-4} m was assumed to mirror sand fractions in Mortandad and South Mortandad Canyons. Since 98% of the channel bed is composed of sand (Purtymun 1974 and Hakonson et al. 1976), this value was also assumed to be the bed sediment size.

Fall Velocities

The fall velocities associated with each sediment diameter size were calculated from Ruby's Formula (Vanoni 1975). The fall velocities for sand, silt, and clay were assigned as 3.1×10^{-2} m/s, 1.06×10^{-4} m/s, and 4.11×10^{-7} m/s, respectively.

Specific Gravity

We assumed that Mortandad and South Mortandad Canyons represent alluvial streams, since 98% of their river beds are composed of sand. The specific gravity was, therefore, selected as 2.65.

Erodibility Coefficient and the Critical Shear Stress Associated With Cohesive Sediments

The river bed is mainly composed of sand, which indicates that silt and clay tend to be carried through the system. The bed shear stresses as provided by DKWAV ranged from less than 1 kg/m^2 to about 20 kg/m^2 . The critical shear stresses associated with deposition for silt and clay were determined with the aid of Shield's diagram (Vanoni 1975). The critical shear stresses associated with the deposition of silt and clay were estimated to be 0.01 kg/m^2 and $3.3 \times 10^{-3} \text{ kg/m}^2$, respectively. The critical shear stress associated with scour, and the erodibility coefficient were determined through the calibration process to produce an appropriate amount of cohesive soil to be scoured from the bed. Through calibration, the silt and clay have a critical shear stress of erosion of 7 kg/m^2 and 10 kg/m^2 , respectively, and an erodibility coefficient was selected to be $0.2 \text{ kg/m}^2\text{-day}$. With these values, the bulk of the silt and clay will be carried through the system.

Distribution Coefficient (Kd) of ^{239}Pu With Sediment

Based upon the previous investigations of Ecker et al. (1979) and Onishi et al. (1979a, 1979c), we assumed that the ratio of the Kd values for sand, silt, and clay would be 1 : 5 : 10, or

$$(Kd)_{\text{sand}} : (Kd)_{\text{silt}} : (Kd)_{\text{clay}} = 1 : 5 : 10 \quad (11)$$

Hakonson et al. (1976) provided runoff sample concentrations of ^{239}Pu . From these data, the K_d value associated with the bulk sediment was estimated as $12 \text{ m}^3/\text{kg}$.

Data presented by Purtymun (1974) revealed that 35% of the suspended sediment in DP Canyon was silt and clay, while 65% was sand. We assumed that neighboring Mortandad and South Mortandad Canyons have similar sediment distributions. Purtymun also reported that "The silt and clay are largely montmorillonite and illite which are weathering products of tuff... These clay minerals are believed to be responsible for the effectiveness of the Los Alamos soil in removing plutonium from waste solutions." Based on these observations, we assumed that 75% of the soil composite was clay and that 25% was silt. The suspended sediment composition could, therefore, be expressed as

$$\text{sand} : \text{silt} : \text{clay} = 0.65 : 0.0875 : 0.2625 \quad (12)$$

The bulk K_d values may be expressed as

$$\begin{aligned} & (0.65)(K_d)_{\text{sand}} + (0.35)(0.25)(K_d)_{\text{silt}} + (0.35)(0.75)(K_d)_{\text{clay}} \\ & = 12 \text{ m}^3/\text{kg} \end{aligned} \quad (13)$$

Combining Equations (12) and (13) and solving for the K_d values associated with sand, silt, and clay gives the following information:

$$(K_d)_{\text{sand}} = 3.23 \text{ m}^3/\text{kg} \approx 3 \text{ m}^3/\text{kg} \quad (14)$$

$$(K_d)_{\text{silt}} = 15 \text{ m}^3/\text{kg} \quad (15)$$

$$(K_d)_{\text{clay}} = 30 \text{ m}^3/\text{kg} \quad (16)$$

Equations (14), (15), and (16) represented the K_d values associated with sand, silt, and clay for which the desorption mechanism was not the main phenomenon.

Since the K_d values for desorption are rarely known and the K_d values for adsorption are generally known, the later K_d values are usually used. In the case where radioactive effluent is discharged directly to the water body, this procedure appears valid. Since most of the ^{239}Pu is adsorbed onto the sediment before it reaches the stream, the K_d values should depict the desorption phenomenon as the main mechanism. Using K_d values for adsorption, on the other hand, is not very effective. The K_d values should reflect the ability of the sediment sizes to retain the radionuclides once these nuclides have been adsorbed. Equations (14), (15), and (16) were modified as follows to reflect sediment retention properties:

1. For a discharge of $0.3 \text{ m}^3/\text{s}$ the sediment discharge can be estimated as 300 kg/min (Figure 25). The overland composite concentration will be:

$$\frac{300 \text{ kg/min}}{(0.3 \text{ m}^3/\text{s}) (60 \text{ sec/min})} = 16.7 \text{ kg/m}^3 \approx 17 \text{ g/l} \quad (17)$$

This calculation has also been checked by ARM.

2. To estimate the channel composite concentration, we used data on the September 15, 1974 storm provided by Hakonson et al. (1976). For a discharge of $0.3 \text{ m}^3/\text{s}$, the instream sediment concentration was estimated as

$$\frac{(14 + 56) + (17 + 74)}{4} \text{ g/l} \approx 40 \text{ g/l} \quad (18)$$

3. Since silt and clay represent 35% of the instream sediment concentration, the concentration for silt and clay can be estimated as

$$(0.35)(40 \text{ g/l}) \approx 14 \text{ g/l} \quad (19)$$

FIGURE 25. Total Sediment Flow Rate Distributed in Time at GS-1 on Mortandad Canyon for the 16 July 1976 Storm

4. By assuming that most of silt and clay comes from overland erosion, silts and clays would thus comprise 80% of the overland erosion (14 g/l of 17 g/l $\approx$ 80%). This was further supported by data reported by Purtymun (1974). Purtymun (1974) reported the size distribution of weathered tuff which constitutes the majority of surface soil. By assuming that only silt, clay, and very fine sand are eroded from the overland, the percentage of silt and clay in eroded overland soil can be estimated

$$\frac{0.42}{0.42 + 0.09} \times 100\% = 82\% \approx 80\% \quad (20)$$

5. Since the ratio of the amount of clay to silt is three to one, the ratios of sand, silt, and clay at the stream's edge can be calculated as

$$\text{Sand: } (1.00 - 0.80) \times 100\% = 20\% \quad (21)$$

$$\text{Silt: } (0.80)(0.25) \times 100\% = 20\% \quad (22)$$

$$\text{Clay: } (0.80)(0.75) \times 100\% = 60\% \quad (23)$$

6. By combining Equations (14), (15), (16), (21), (22), and (23), the ratios of the radionuclides associated with the sand, silt, and clay can be estimated as

$$\text{Sand : Silt : Clay} = (0.20)(3) : (0.20)(15) : (0.60)(30) \quad (24)$$

The percentages of radionuclides associated with the sand, silt, and clay are 3%, 14%, and 83%, respectively.

7. Radionuclides have more of an affinity for smaller sediment sizes than for larger ones. As such, we assumed that ^{239}Pu already

adsorbed by silt could not be adsorbed by sand, and that ^{239}Pu already adsorbed by clay could not be adsorbed by sand or silt. The K_d values were then adjusted to

$$(K_d)_{\text{sand}} = (3 \text{ m}^3/\text{kg}) (0.03) = 0.09 \text{ m}^3/\text{kg} \quad (25)$$

$$(K_d)_{\text{silt}} = (15 \text{ m}^3/\text{kg}) (0.03 + 0.14) = 2.55 \text{ m}^3/\text{kg} \quad (26)$$

$$(K_d)_{\text{clay}} = (30 \text{ m}^3/\text{kg}) (0.03 + 0.14 + 0.83) = 30 \text{ m}^3/\text{kg} \quad (27)$$

Based on Equations (25), (26), and (27), the K_d values employed for the Mortandad and South Mortandad Canyon simulations were set at $0.09 \text{ m}^3/\text{kg}$ for sand, $2.55 \text{ m}^3/\text{kg}$ for silt, and $30 \text{ m}^3/\text{kg}$ for clay.

Longitudinal Dispersion Coefficient

Several formulations have been proposed and several experiments have been performed for selecting the longitudinal dispersion coefficient value, ϵ_x . One such formulation as presented by Fischer (1967) is:

$$\epsilon_x = \frac{0.011 U^2 b^2}{d U_*} \approx \frac{0.011 Q^2}{d^3 U_*} \quad (28)$$

in which ϵ_x = longitudinal dispersion coefficient (m^2/s), U = flow velocity (m/s), b = stream top width (m), d = flow depth (m), U_* = shear velocity (m/s), and Q = flow discharge (m^3/s). By assuming a discharge, velocity, depth, and shear velocity of $0.3 \text{ m}^3/\text{s}$, 1.5 m/s , 0.31 m , and 0.25 m/s , respectively, ϵ_x can be approximated as equaling $0.1 \text{ m}^2/\text{s}$. These values were presented earlier and reflect typical values for the area. A major restriction in the Mortandad Canyon case is that the longitudinal dispersion process should not dominate the in-stream advection process. In fact, the reverse should be true and the dispersion process should be of minor importance. By

assuming a value for ϵ_x of $0.1 \text{ m}^2/\text{s}$, this restriction is met. In addition, variations in this value produce little change in the results, thereby confirming the dominance of the advection phenomenon.

Transfer Rate (K_j) of Radionuclides

The transfer rate expresses the time required to reach an equilibrium condition as determined by K_d (K_j has a unit of reciprocal of time). By checking the mass balance of radionuclides and the time it takes to reach an equilibrium condition, an estimate of K_j can be obtained. To obtain values for K_j , a steady-state condition was imposed on Mortandad Canyon with typical hydrodynamic values provided by DKWAV. We obtained a K_j value of 0.1 day^{-1} by conserving the mass balance for a Δt and Δx at 60 seconds and ~ 200 meters, respectively. Because of insufficient data, the same K_j values were assumed to be valid for all sediment sizes.

Since no lateral inflow data were available for the September 15, 1974 flow, the July 16, 1976 flow was used in the calibration procedure. As mentioned previously in the calibration section of DKWAV, the flood hydrographs for the July 16, 1976 storm at GS-1 and GS-2 were similar to the measured hydrographs at GS-1 and GS-2 for the September 15, 1974 storm. The peak discharges, flow velocities, and flow depths were quite similar. In calibrating TODAM, we compared the sediment flux at GS-2 for the July 16, 1976 storm with the flux produced by the September 15, 1974 storm.

Figure 25 presents the sediment flux distributed temporally at GS-1 for the July 16, 1976 flow event. This figure represents the upstream boundary condition as provided by ARM. Figure 26 presents sediment transport results in which the curve represents the predicted sediment flux for the July event. In addition, the measured sediment fluxes 6 cm below the water surface and above the stream bed are also presented for the September 15, 1974 flow event. The timing and magnitude of the predicted and measured sediment hydrographs were quite similar. Hakonson et al. (1976) provided the September flow information at GS-2, while TODAM predicted the July results at GS-2. Because watershed and hydrologic conditions influence sediment loadings, one may

FIGURE 26. Total Sediment Flow Rates Distributed in Time at GS-2 on Mortandad Canyon for the 15 September 1974 and 16 July 1975 Storms

expect similar loadings for similar flow discharges as for the case of the flow events of September 15, 1974 and July 16, 1976. This relationship is not always true, though.

Figures 25 and 26 indicate that complex phenomena occurred between GS-1 and GS-2, namely lateral inflow, infiltration, lateral sediment loadings, deposition and resuspension of sediments, advection, and dispersion. It should be noted that the sediment hydrographs at GS-1 and GS-2 for both the July and September flow events are significantly different, indicating that the hydrographs are not just undergoing pure translation. From GS-1 to GS-2, the base times of the hydrographs increase by over 300%, while the peak discharges increase by over 540%. If the hydrographs underwent only pure translation, then it would be legitimate to question using TODAM to model complex mechanisms. Modeling a hydrograph that undergoes translation is very simple and would not indicate whether important mechanisms are being modeled correctly or at all. However, because the hydrographs at GS-1 and GS-2 are significantly different, and because TODAM includes many important complex transport phenomena, the reasonable agreement between the simulated and measured sediment loads indicates that the transport phenomena are being modeled fairly well and that the assumptions for defining the basic parameters of the model appear reasonable.

The calibration results for the July 16, 1976 flow event are presented in Figures 27 through 31. The results are presented for two locations on Mortandad Canyon--GS-1 and the confluence where Mortandad and South Mortandad Canyons join. In addition, results for the September 8, 1975 storm event are presented for two locations on South Mortandad Canyon (Figures 32 through 43). This storm event was included because Mortandad Canyon was not contaminated by ^{239}Pu . It should be noted that all results presented herein pertaining to GS-1, the head or the source of Mortandad and South Mortandad Canyons, represent upstream boundary conditions as provided by ARM.

We assumed that a no flow condition existed in each channel between storms and that when each flow event ended, the sediment and associated contaminants were immediately deposited on the channel bottom. The only activity

FIGURE 27. Discharges Distributed in Time at GS-1 on Mortandad Canyon for the 16 July 1976 Storm

MORTANDAD CANYON - CONFLUENCE - JULY 16, 1976

FIGURE 28. Discharges Distributed in Time at the Confluence
on Mortandad Canyon for 16 July Storm

FIGURE 29. Sediment Flow Rates Distributed in Time at GS-1 on Mortandad Canyon for the 16 July 1976 Storm

FIGURE 30. Sediment Flow Rates Distributed in Time at the Confluence on Mortandad Canyon for the 16 July 1976 Storm

FIGURE 31. The Longitudinal Variation of the Net Change in the Bed Elevation on Mortandad Canyon for the 16 July 1976 Storm

FIGURE 32. Discharges Distributed in Time at the Source on South Mortandad Canyon for the 8 September 1975 Storm

FIGURE 33. Discharges Distributed in Time at the Confluence on South Mortandad Canyon for the 8 September 1975 Storm

FIGURE 34. Sediment Flow Rates Distributed in Time at the Source on South Mortandad Canyon for the 8 September 1975 Storm

FIGURE 35. Sediment Flow Rates Distributed in Time at the Confluence on South Mortandad Canyon for the 8 September 1975 Storm

FIGURE 36. Longitudinal Variation of the Net Change in Bed Elevation on South Mortandad Canyon for the 8 September 1975 Storm

FIGURE 37. Plutonium-239 Flow Rates Distributed in Time at the Source on South Mortandad Canyon for the 8 September 1975 Storm

SOUTH MORTANDAD CANYON - CONFLUENCE - SEPTEMBER 8, 1975

FIGURE 38. Plutonium-239 Flow Rates Distributed in Time at the Confluence on South Mortandad Canyon for the 8 September 1975 Storm

FIGURE 39. Particulate ^{239}Pu Flow Rates Distributed in Time at the Source on South Mortandad Canyon for the 8 September 1975 Storm

FIGURE 40. Particulate ^{239}Pu Flow Rates Distributed in Time at the Confluence on South Mortandad Canyon for the 8 September 1975 Storm

FIGURE 41. Particulate ^{239}Pu Concentrations in PC/kg Distributed in Time at the Source on South Mortandad Canyon for the 8 September 1975 Storm

FIGURE 42. Particulate ^{239}Pu Concentrations in PC/kg Distributed in Time at the Confluence on South Mortandad Canyon for the 8 September 1975 Storm

FIGURE 43. Simulated ^{239}Pu Bed Contamination on South Mortandad Canyon for the 8 September 1975 Storm

accounted for between flow events was the radioactive decay of the radionuclides. Since the half-life of ^{239}Pu is so long, its impact was negligible. A minimum depth of flow of 1.0×10^{-3} m was also assumed for each flow event. A minimum flow depth was defined because, when the depth of flow becomes very small, assumptions about the channel become less important and assumptions about overland flow become more important. The roughness parameter could increase by a factor of ten (Whelan 1980). Only the portions of the stream containing flow were numerically routed. When a flow rate at a given channel section becomes less than a pre-assigned minimum discharge, the channel section was treated as dry. In other words, the computations followed the flood wave as it proceeded down the channel (if the need arose).

Figures 27 and 28 present the hydrodynamics for Mortandad Canyon. As the figures reveal, there is little attenuation of the discharge hydrograph from GS-1 to the confluence; while concurrently, there is hydrograph translation. The attenuation is lessened because lateral inflow and channel seepage tended to negate each other.

Sediment discharge hydrographs distributed by sediment size fraction for GS-1 and the confluence are presented in Figures 29 and 30, respectively. The composition of the flux at the upstream end is 60% clay, 20% silt, and 20% sand. The peak sediment flux increases from 5 kg/sec to approximately 31 kg/sec, which is the flux computed by $\{UC - \epsilon_x (\partial C / \partial x)\}$ at each cross section. The increase is mainly due to bed erosion and, to a lesser extent, to the lateral inflow of sediment. The erosion occurs because the capacity of the stream to transport sediment is greater than the amount of sediment supplied by the overland erosion process. Because of this erosion, the flux of sand increases dramatically (the bed is composed of 98% sand), while the flux of silt and clay does not change significantly. The silt and clay are being carried through the system with little deposition or erosion (they comprise only 2% of the bed).

Figure 31 presents the longitudinal variation of the net change in the bed elevation on Mortandad Canyon. Since the inflow of sediment at the upstream end is less than the stream's capacity to transport sediment, a net

resuspension of bed sediment occurs in the first two channel sections. The third section presents a reach under stable conditions, where the capacity of the flow to transport sediment appears to equal the amount of sediment supplied from upstream and from lateral inflow. The sediment transport capacity was computed using the Toffaletti formulation (Vanoni 1975), which is based on the assumed (power law) velocity profile. Since velocities and cross-sectional shapes of Sections 2 and 3 (assuming similar discharges) are very similar and since the amount of sediment from Section 2 nearly equals the transport capacity of Section 3, very little resuspension or deposition of sediment occurs in Section 3. The longitudinal variation of the velocity increases from Section 3 to Section 5, while the flow velocity in Section 6 is very near that of Section 5. This phenomenon explains the sediment resuspension process which occurs in Sections 4 and 5 and the minor deposition in Section 6. Further downstream, the longitudinal variation in velocity decreased from Sections 8 through 10. In addition, the slopes and bed widths in Sections 9 and 10 were smaller relative to other sections, thereby resulting in deposition in these sections. The velocity and slope of Section 11 increased significantly over that of Section 10, thereby increasing the transport capacity in this section. Resuspension of sediment was the resulting process. Overall, 98.4% of the total net sediment erosion consisted of sand and, for this particular storm, deposition and erosion occurred in only the topmost bed layer. The depth of the bed was arbitrarily assumed to be 0.75 meters, consisting initially of 8 layers. At the conclusion of the storm, all resuspension and deposition of sediment and radionuclides were confined to the eighth (the topmost) layer. When a multiple-layered bed technique like this one is used, the effects of armoring can be considered. It should be noted that the net change in bed elevation depends also upon the designated width of the bed (which is constant for each section). The smaller the width, the larger the change in elevation when a given amount of sediment is deposited or resuspended. This comment applies equally to South Mortandad Canyon.

As mentioned previously, the results of the storm event of September 8, 1975 on South Mortandad Canyon are presented in Figures 32 through 43.

Figures 32 and 33 present the discharge hydrographs for the head and confluence of the canyon, respectively. The peak discharge at the confluence is approximately 4.5 times the peak discharge at the head of the canyon. This increase is due to lateral inflow.

Figures 34 and 35 present the sediment discharge hydrographs distributed by size fraction at the head and confluence of the canyon, respectively. Figure 34 represents the upstream boundary condition as presented by ARM and has a sediment distribution of 60% clay, 20% silt, and 20% sand. The minor increase in silt and clay from the head to the confluence is due to lateral inflow and some erosion from the bed. The sand flux, on the other hand, increases significantly, largely because of erosion from the bed, which is composed of 98% sand.

Figure 36 represents the simulated net change in bed elevation, varying spatially for the September 8, 1975 storm event. The first two sections of South Mortandad Canyon underwent erosion, because the lateral sediment inflow and the upstream sediment flux could not satisfy the transport capacity of the flow. The midsection (Section 4) underwent deposition due to an increase in the section's width and a decrease in its sediment transport capacity. The erosion at the downstream end is due to steep slopes and high flow velocities, thereby increasing the transport capacity of the area. Overall, 98.7% of the total net sediment erosion consisted of sand, and deposition and resuspension of sediment occurred in only the topmost layer. Similar to the Mortandad Canyon case, the depth of the bed was assumed to be 0.75 meters and the bed was assumed to consist initially of eight layers. At the conclusion of the storm, all resuspension and deposition of sediment were confined to the eighth and topmost layer.

Figures 37 and 38 present the ^{239}Pu flux in the particulate and dissolved phases at the head and confluence of the canyon, respectively. As stated before, the flux was computed by $\{UC - \epsilon_x (\partial C / \partial x)\}$ at each cross section. Figure 37 represents the only location where ^{239}Pu enters the channel, but as the contaminant hydrograph passes the confluence location, its mass flux doubles, indicating that at least $\pm 100\%$ error occurred. This

numerical discrepancy is mainly due to the numerical discretization adopted for South Mortandad Canyon which is too coarse, and is explained in greater detail in Section 3.6.2. Another possibility causing this problem is a minor lack of mass conservation of flow in TODAM when flow data were supplied by DKWAV, due to the compatibility problem between TODAM and DKWAV. The figures reveal that the particulate phase dominates the dissolved phase, because ^{239}Pu had already adsorbed onto the sediment prior to in-stream migration. Because of ^{239}Pu 's strong affinity for the sediment, desorption is a more difficult process than adsorption, as reflected in the model by the high equilibrium coefficient (K_d) values for sand ($0.09 \text{ m}^3/\text{kg}$), silt ($2.55 \text{ m}^3/\text{kg}$), and clay ($30 \text{ m}^3/\text{kg}$).

Figures 39 and 40 present the particulate phase of the ^{239}Pu flux distributed by sediment size at the head and confluence of South Mortandad Canyon, respectively. The particulate radionuclide distribution as presented by Figure 39 has 3.0% associated with sand, 14.0% associated with silt, and 83.0% associated with clay. At the confluence, the particulate radionuclide distribution for sand, silt, and clay is 1.2%, 14.0%, and 84.1%, respectively. Due to the higher affinity that cohesive sediments have for contaminants, the plutonium associated with silt and clay increased slightly, while for sand it decreased.

The corresponding particulate phase of the ^{239}Pu concentration (pCi/kg) for each sediment size at the head and confluence of the canyon is presented in Figures 41 and 42, respectively. As expected, the concentration of ^{239}Pu with respect to each sediment size decreased by approximately 50% as the flow moved downstream. This is due to the increase of less contaminated sediment suspended by the flow conditions. The contaminant concentration per unit weight of sediment size decreases with an increase in sediment loads, because the contaminated sediment was mixed with less contaminated sediment eroded from the bed and sediment transported into the channel as lateral inflow.

Figure 43 presents the simulated bed contamination results varying spatially for the September 8, 1975 storm event. This figure indicates that the contaminant concentration is highest where the sediment deposition is heaviest. Several locations (Sections 1, 3, 5, and 6) exhibit bed contamination where

there is net erosion. Since a contaminant, when deposited, is distributed equally within the top layer for a given sediment size fraction, it is possible to resuspend part of the bed layer and still have contaminants present in the bed. If the flow conditions are such that the sediment transport capacity is exceeded by the amount of sediment instream, then deposition occurs along with the possible contamination of the bed layer. Once it is contaminated, the bed layer will always be contaminated until the bed sediment in that layer has been totally resuspended. At the conclusion of the storm, all resuspension and deposition of sediment and radionuclides were confined to the eighth (topmost) layer. The total amount of ^{239}Pu deposited in the bed as indicated by Figure 43 represents 0.17% of the total inflow of ^{239}Pu at the upstream boundary condition. The rest of the ^{239}Pu (99.83%) was transported beyond the confluence of Mortandad and South Mortandad Canyon and was deposited in the Mortandad Canyon channel bed upstream of the Garden. All of the ^{239}Pu contaminating the bed was associated with sand, as there was no deposition of silt and clay upstream of the confluence. This indicates that when multiple storms are considered, sand will play an important role in contaminating the bed above the confluence. The direct adsorption/desorption mechanism between the dissolved phase of ^{239}Pu and the sediment comprising the top layer of the bed was assumed to be zero, since no data existed for determining these transfer rate coefficients (K_j values for the bed). Since changes in LLW concentration in the beds are due to contaminated sediment deposition and resuspension, this assumption appears reasonable.

4.3.3 Application Results

The models DKWAV and TODAM were applied to Mortandad and South Mortandad Canyons to simulate movement of flow, sediment, and ^{239}Pu for two years from June 1, 1975 to May 30, 1977. During this period, six major runoff events (July 17, 1975; September 8, 1975; September 9, 1975; September 10 and 11, 1975; July 16, 1976; and August 11, 1976) were predicted by ARM. TODAM simulated the movement of sand, silt, clay, particulate ^{239}Pu adsorbed by sand, particulate ^{239}Pu adsorbed by silt, particulate ^{239}Pu adsorbed by clay, and dissolved ^{239}Pu .

Figures 44 through 56 present the simulation results for the six storm events on Mortandad Canyon, while Figures 57 through 93 present the simulation results for the six storms on South Mortandad Canyon. Figures 44, 45, and 46 present the simulated flow rates at GS-1, GS-2, and the confluence, respectively, on Mortandad Canyon. There is little change in the magnitude of the flow events from GS-1 to the confluence because lateral inflow and channel seepage negate each other.

The total sediment flux at GS-1 is presented in Figure 47, while the corresponding sand, silt, and clay sediment flow rates are presented in Figures 48, 49, and 50. These sediment flow rates represent the upstream boundary conditions as predicted by ARM. The composition of the sediment flux at GS-1 is 60% clay, 20% silt, and 20% sand for all storms. Figures 51 and 52 depict the simulated total sediment flow rates at GS-2 and the confluence, respectively, and Figures 53, 54, and 55 illustrate the simulated sand, silt, and clay sediment flow rates at the confluence, respectively. The sediment flux for all flow events increased significantly from GS-1 to GS-2, while for most flow events the sediment flux decreased about 30% from GS-2 to the confluence. (The July 16, 1976 flow event has been discussed in detail in Section 4.3.2.) The increase in sediment flux from GS-1 to GS-2 is due to the resuspension of mostly sand from the bed, and the decrease in sediment flux from GS-2 to the confluence is due to slower flow velocities and decreasing slopes in the downstream portion of the channel. The dramatic increase in the simulated sand rate can be seen in Figures 48 and 53. Predicted sediment discharge rates of silt and clay, as presented in Figures 49, 50, 54, and 55 from GS-1 to the confluence, do not change significantly from storm to storm, indicating that finer sediments are being swept through the system with little net deposition or resuspension.

The bed elevation changes directly reflect sediment resuspension and deposition during flow events. Sediment resuspension generally occurs between GS-1 and GS-2, while deposition usually occurs in the downstream portion of the channel, according to the simulation results. Figure 56 presents the simulated longitudinal variation of the net change in bed elevation for the

FIGURE 44. Discharges Distributed in Time at GS-1 on Mortandad Canyon

FIGURE 45. Discharge Distributed in Time at GS-2 Mortandad Canyon

FIGURE 46. Discharge Distributed in Time at the Confluence on Mortandad Canyon

FIGURE 47. Total Sediment Flow Rates Distributed in Time at GS-1 on Mortandad Canyon

FIGURE 48. Sand Flow Rates Distributed in Time at GS-1 on Mortandad Canyon

FIGURE 49. Silt Flow Rates Distributed in Time at GS-1 on Mortandad Canyon

FIGURE 50. Clay Flow Rates Distributed in Time at GS-1 on Mortandad Canyon

FIGURE 51. Total Sediment Flow Rates Distributed in Time at GS-2 on Mortandad Canyon

FIGURE 52. Total Sediment Flow Rates Distributed in Time at the Confluence on Mortandad Canyon

FIGURE 53. Sand Flow Rates Distributed in Time at the Confluence on Mortandad Canyon

FIGURE 54. Silt Flow Rates Distributed in Time at the Confluence on Mortandad Canyon

FIGURE 55. Clay Flow Rates Distributed in Time at the Confluence on Mortandad Canyon

FIGURE 56. Longitudinal Variation of the Net Change in the Bed Elevation on Mortandad Canyon (7/17/75-8/11/76)

FIGURE 57. Discharges Distributed in Time at the Source on South Mortandad Canyon

SOUTH MORTANDAD CANYON - MID POINT - 7/17/75 - 8/11/76

FIGURE 58. Discharges Distributed in Time at the Midpoint on South Mortandad Canyon

FIGURE 59. Discharges Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 60. Total Sediment Flow Rates Distributed in Time at the Source on South Mortandad Canyon

FIGURE 61. Sand Flow Rates Distributed in Time at the Source on South Mortandad Canyon

FIGURE 62. Silt Flow Rates Distributed in Time at the Source on South Mortandad Canyon

FIGURE 63. Clay Flow Rates Distributed in Time at the Source on South Mortandad Canyon

FIGURE 64. Total Sediment Flow Rates Distributed in Time at the Mid-point on South Mortandad Canyon

FIGURE 65. Total Sediment Flow Rates Distributed in Time
at the Confluence on South Mortandad Canyon

FIGURE 66. Sand Flow Rates Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 67. Silt Flow Rates Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 68. Clay Flow Rates Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 69. Longitudinal Variation of the Net Change in the Bed Elevation on South Mortandad Canyon (7/17/75 - 8/11/76)

FIGURE 70. Total Particulate Flow Rates Distributed in Time at the Source on South Mortandad Canyon

FIGURE 71. Particulate Flow Rates Associated with Sand Distributed in Time at the Source on South Mortandad Canyon

SOUTH MORTANDAD CANYON - SOURCE - 7/17/75 - 8/11/76

FIGURE 72. Particulate Flow Rates Associated with Silt Distributed in Time at the Source on South Mortandad Canyon

FIGURE 73. Particulate Flow Rates Associated with Clay Distributed in Time at the Source on South Mortandad Canyon

FIGURE 74. Total Particulate Flow Rates Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 75. Particulate Flow Rates Associated with Sand Distributed in Time at the Confluence on South Mortandad Canyon

SOUTH MORTANDAD CANYON - CONFLUENCE - 7/17/75 - 8/11/76

FIGURE 76. Particulate Flow Rates Associated with Silt Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 77. Particulate Flow Rates Associated with Clay Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 78. Dissolved ^{239}Pu Flow Rates Distributed in Time at the Source on South Mortandad Canyon

FIGURE 79. Dissolved ^{239}Pu Flow Rates Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 80. Total Particulate and Dissolved Flow Rates Distributed in Time at the Source on South Mortandad Canyon

FIGURE 81. Total Particulate and Dissolved Flow Rates Distributed in Time at the Mid Point on South Mortandad Canyon

FIGURE 82. Total Particulate and Dissolved Flow Rates Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 83. Total Particulate Concentrations in PC/kg Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 84. Particulate Concentrations in Pc/Kg Associated with Sand Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 85. Particulate Concentrations in Pc/Kg Associated with Silt Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 86. Particulate Concentrations in Pc/Kg Associated with Clay Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 87. Total Particulate Concentrations in Pc/m^3 Distributed in Time at the Confluence on South Mortandad Canyon

SOUTH MORTANDAD CANYON - CONFLUENCE - 7/17/75 - 8/11/76

FIGURE 88. Particulate Concentrations in Pc/m^3 Associated with Sand Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 89. Particulate Concentrations in Pc/m^3 Associated with Silt Distributed in Time at the Confluence on South Mortandad Canyon

SOUTH MORTANDAD CANYON - CONFLUENCE - 7/17/75 - 8/11/76

FIGURE 90. Particulate Concentrations in Pc/m^3 Associated with Clay Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 91. Dissolved ^{239}Pu Concentrations Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 92. Total ^{239}Pu Concentrations Distributed in Time at the Confluence on South Mortandad Canyon

FIGURE 93. Simulated ^{239}Pu Bed Contamination on South Mortandad Canyon (7/17-75 - 8/11/76)

entire simulation period in Mortandad Canyon. The specific reason for the sediment resuspension and deposition in Mortandad Canyon for the entire simulation period is the same as that given for the single storm flow event (July 16, 1976) in Section 4.3.2. In general, when flow velocity increases the transport capacity in a section increases and sediment resuspension usually occurs. However, if flow velocity decreases because of a mild slope or increased cross-sectional area, the transport capacity also decreases. If the transport capacity drops below the amount of sediment being transported, then sediment deposition occurs.

As mentioned previously, Figures 57 through 93 present the simulation results for the entire simulation period of two years in South Mortandad Canyon. Since the August flow event (August 11, 1976) was very small in South Mortandad Canyon, the simulation results discussed below do not include those of the August event. Figures 57, 58, and 59 present the simulated flow rates at the head, midpoint (1215 meters downstream of the head), and the confluence of South Mortandad Canyon. The peak discharges at the midpoint and confluence are approximately 4.8 times larger than the peak discharges at the head of the channel (as predicted by ARM). This is because the lateral inflow at points downstream of the head of the channel is much larger than the seepage from the channel.

Figures 60 through 68 present the simulation results for the sediment flow rates in South Mortandad Canyon. Figure 60 presents the total sediment flux at the head of the canyon, while Figures 61, 62, and 63 present the corresponding sand, silt, and clay sediment flow rates. The composition of the total sediment flux at the head of the canyon was assigned to be 60% clay, 20% silt, and 20% sand. These flow rates represent the upstream boundary conditions as predicted by ARM. It is interesting to note that, given similar watershed characteristics and precipitation input, the flow rates and sediment loadings provided by ARM as an upstream boundary condition to the channel are almost linearly related to the size of the catchment. The flow rates and sediment loadings at the head of Mortandad Canyon are approximately 15 times larger than those at the head of South mortandad Canyon, while the size of the catchment feeding the head of Mortandad Canyon is also about 15 times larger than that feeding the head of South Mortandad Canyon.

Figures 64 and 65 present the simulated total sediment flow rates at the midpoint and confluence of South Mortandad Canyon, respectively, while Figures 66, 67, and 68 present the simulated sand, silt, and clay sediment flow rates at the confluence, respectively. There is a large increase in the total sediment flux from the head of the canyon to the midpoint, which is due to sediment erosion in the first two channel sections and to lateral inflow of sediment. The increase in the sediment flow rates from the midpoint to the confluence is also due to sediment erosion (in the last two channel sections) and lateral inflow of sediment. Since sand comprises approximately 85% of suspended sediment at the confluence and only 20% of the lateral inflow, the increase in suspended sediment appears to be due to erosion of the bed (98% of which is composed of sand). Suspended silt and clay amounts are relatively small as compared to the amount of sand being transported. Figure 69 graphically presents the simulated longitudinal variation of the net change in bed elevation. The reason for the sediment resuspension and deposition in South Mortandad Canyon for the entire simulation period is the same as that given for the single storm flow event (September 8, 1975) in Section 4.3.2. The total particulate ^{239}Pu flux, as predicted by ARM, and the corresponding particulate ^{239}Pu flux associated with sand, silt, and clay from Area C at the head of South Mortandad Canyon are presented in Figures 70, 71, 72, and 73, respectively. As discussed in Section 4.2.2, less than 1% of the available ^{239}Pu on the land surface reached the stream's edge during the simulation period. Furthermore, of the ^{239}Pu transported from Area C to the stream's edge, 99% was carried by the eroded sediment. Because of the high adsorptive capacity of clay, the simulation results reveal that clay carried approximately 83% of the particulate ^{239}Pu that reached the stream's edge, while silt and sand carried approximately 14% and 3%, respectively. The predicted total particulate ^{239}Pu flux and its corresponding particulate ^{239}Pu flux associated with sand, silt, and clay at the confluence of South Mortandad Canyon are presented in Figures 74, 75, 76, and 77. As discussed in Sections 3.6.2 and 4.3.2, the discretization of South Mortandad Canyon with seven cross sections was too coarse to adequately model the rapidly changing transport of the radionuclide. The results pertaining to the ^{239}Pu flux (as

related to mass balance) indicate a $\pm 100\%$ error. The particulate ^{239}Pu is divided among the sediment sizes at the confluence with approximately the same distribution as at the head of the channel: 83% associated with clay, 14% associated with silt, and 3% associated with sand. The distribution is nearly the same at the confluence as it is at the head of the canyon, because the affinity of ^{239}Pu to the cohesive sediments (silt and clay) is much larger than the affinity to sand. This is reflected in the K_d values associated with sand ($0.09 \text{ m}^3/\text{kg}$), silt ($2.55 \text{ m}^3/\text{kg}$), and clay ($30.00 \text{ m}^3/\text{kg}$). The distribution would probably shift if the rate of sand being resuspended increased significantly over what presently exists in the channel.

The simulated dissolved flux of ^{239}Pu at the head and confluence is presented in Figures 78 and 79, respectively, while the simulated total (dissolved and particulate) flux of ^{239}Pu at the head, midpoint, and confluence of the channel are presented in Figures 80, 81, and 82, respectively. These figures reveal that more than 99% of the ^{239}Pu is adsorbed by the sediment and that less than 1% is in the dissolved phase. This is true at all locations in the channel.

Figure 83 presents the simulated total particulate ^{239}Pu concentrations by weight at the confluence of South Mortandad Canyon, while Figures 84, 85, and 86 present the corresponding simulated particulate ^{239}Pu concentrations associated with sand, silt, and clay by weight. These figures present the degree of adsorption by sediment size. As the figures reveal, the majority of ^{239}Pu per unit weight of sediment is associated with the cohesive sediments (clay in particular). This is understandable, since the cohesive sediments have a higher adsorption capacity than sand. Figure 87 presents the predicted total particulate ^{239}Pu concentrations by volume at the confluence, while Figures 88, 89, and 90 present the corresponding predicted particulate ^{239}Pu concentrations associated with sand, silt, and clay per unit volume of water. Figures 88, 89, and 90 present the distribution of the total amount of ^{239}Pu by a sediment size fraction. These figures reveal that a majority of the total contamination is still being carried by the cohesive material. If the transport rate of sand increased significantly, then the sand could be the

major transporting medium of ^{239}Pu . This is not to say that the individual sand particles could adsorb more ^{239}Pu , but that more ^{239}Pu would be transported by sand because the amount of sand being transported has increased.

Figures 91 and 92 present the dissolved ^{239}Pu concentrations and the total (dissolved and particulate phases) ^{239}Pu concentrations per unit volume of water respectively, at the confluence of South Mortandad Canyon, according to the model prediction. These two figures indicate that the major transporting agent of ^{239}Pu is sediment, and that the dissolved concentrations are two to three orders of magnitude less than the particulate concentrations. This clearly indicates that deposition and resuspension of contaminated sediment may be an important mechanism in the migration and fate of ^{239}Pu in the aquatic environment. Since the half-life of ^{239}Pu is very long, bed contamination may become an important concern as a source of long-term contamination. Figure 93 presents the simulated longitudinal distribution of bed contamination by the sediment size fraction in South Mortandad Canyon. As expected, the degree and location of contamination followed the same pattern of the bed elevation change as presented in Figure 69. As indicated by Figures 69 and 93, contamination was greatest where the deposition of sediments was highest and the resuspension of sediments was lowest. Because silt and clay are generally being carried through the system and because sand is the major sediment type being deposited and resuspended, most of the contamination is associated with the sand in the bed. For Section 3, the model predicted that the percent contamination adsorbed by sand, silt, and clay is 67.4%, 4.8%, and 27.8%, respectively. For Section 4, the percent contamination adsorbed by sand, silt, and clay is 81.0%, 2.7%, and 16.3%, respectively. The contamination of ^{239}Pu in the other bed sections is two to three orders of magnitude less than in Sections 3 and 4. The large amount of ^{239}Pu adsorbed by clay in Sections 3 and 4 (27.8% and 16.3%, respectively) is due to the high adsorption capacity clay has for the ^{239}Pu . Overall, 0.8% of the upstream ^{239}Pu contamination ended up adsorbed to the bed sediments (sand, silt, and clay) above the confluence. All of the radionuclides transported past the

confluence were deposited in Mortandad Canyon above the Garden. The distribution between the three sediment types of ^{239}Pu in the bed for the entire channel reach (South Mortandad Canyon) had sand adsorbing 75.0%, silt adsorbing 3.6%, and clay adsorbing 21.4%.

REFERENCES

- Armstrong, N. E., and E. F. Gloyna. 1968. Numerical Solutions of Radionuclide Transport Equations, Chemical Engineering Progress Symposium Series. 64(90):47-58.
- Armstrong, N. E., and E. F. Gloyna. 1967. Radioactivity Transport in Water--Numerical Solutions of Radionuclide Transport Equations and Role of Plants in SR-85 Transport. EHE-12-6703, Technical Report No. 14 to the U.S. Atomic Energy Commission, University of Texas, Austin, TX.
- Bagnold, R. A. 1941. The Physics of Blown Sands and Desert Dunes, Methuen and Co., Ltd., London, England, 265 pp.
- Bhagat, S. K., and E. F. Gloyna. 1965. Radioactivity Transport in Water--Transport of Nitrosylruthenium in an Aquatic Environment. EHE-11-6502, Technical Report No. 9 to the U.S. Atomic Energy Commission, 144 pp.
- Booth, R. S. 1975. A System Analysis Model for Calculating Radionuclide Transport Between Receiving Waters and Bottom Sediment. ORNL-TM-4751, Stoudsburg, PA.
- Bramati, L., T. Marzullo, I. Rosa and G. Zara. 1973. "VADOSA: A Single Code for the Evaluation of Population Exposure Due to Radioactive Discharges," Proceedings of the Third International Congress of International Radiation Protection Association. CONF-7309007-P2.
- Brisbin, I. L., Jr., R. J. Beyers, R. W. Dapson, R. A. Geiger, J. B. Gentry, J. W. Gibbons, M. H. Smith, and S. K. Woods. 1974. "Patterns of Radiocesium in the Sediments of a Stream Channel Contaminated by Production Reactor Effluents," Health Physics. 27:19-27.
- Buckner, M. R., and D. W. Hayes. 1975. "1975 Pollutant Transport in Natural Streams." Presented at the Topical Meeting on Computational Methods in Nuclear Engineering, American Nuclear Society, Charleston, SC.
- Buckner, M. R., D. W. Hayes and J. R. Watts. 1975. Emergency Response Capability for Pollutant Release to Streams and Rivers. DP-MS-75-73, DuPont de Nemours (E.I.) and Co., Savannah River Laboratory, Aiken, SC.
- Chapman, R. S. 1977. A Model to Investigate the Influence of Suspended Sediment on the Mass Transport of Pollution Open Channel Flow. NASA-TM-X-94601.
- Chepil, W. S., and N. P. Woodruff. 1963. "The Physics of Wind Erosion and Its Control." Advances in Agronomy, 15:211-302.
- Chow, V. T. 1959. Open-Channel Hydraulics. McGraw-Hill Book Company, New York, Toronto, London.

- Chow, V. T. 1964. Handbook of Applied Hydrology, McGraw-Hill, New York.
- Churchill, M. A., J. A. Cragwall, R. W. Andrews and S. L. Jones. 1965. Concentrations, Total Sediment Loads and Mass Transport of Radionuclides in the Clinch and Tennessee Rivers. ORNL-3721, Suppl. 1, Oak Ridge National Laboratory, Oak Ridge, TN.
- Clanton, V. S., and E. F. Gloyna. 1964. Radioactivity Transport in Water—Sorption and Release of Radionuclides by Sediments of the Guadalupe River. EHE-10-6404, Technical Report No. 6 to the U.S. Atomic Energy Commission, University of Texas, Austin, TX.
- Crawford, N. J., and R. K. Linsley. 1966. Digital Simulation in Hydrology: Stanford Watershed Model IV. Technical Report 39, Department of Civil Engineering, Stanford University, Stanford, CA.
- Crawford, N. H., and A. S. Donigian, Jr. 1973. "Pesticide Transport and Runoff Model for Agricultural Lands." Office of Research and Development, U.S. Environmental Protection Agency, Washington DC, EPA 660/2-74-013.
- Croley, II, T. E., G. Whelan, P. Witinok, and S. C. Jain. 1979. Overland and Channel Kinematic Cascades of Water and Sediment, Symposium of Urban Runoff, University of Kentucky, pp. 297-305.
- Croley, T. E. 1977. Hydrologic and Hydraulic Computations on Small Programmable Calculators, Iowa Institute of Hydraulic Research, The University of Iowa.
- Dawson, G. W. (Project Manager). 1978. The Feasibility of Mitigating Kepone Contamination in the James River Basin. Submitted to the U.S. Environmental Protection Agency. Pacific Northwest Laboratory, Richland, WA.
- Donigian, A. S., and N. H. Crawford. 1976. Modeling Pesticides and Nutrients on Agricultural Lands. EPA-600/2-76-043.
- Donigian, A. S., and N. H. Crawford. 1979. "User's Manual for the Nonpoint Source (NPS) Model." Office of Research and Development, U.S. Environmental Protection Agency.
- Eraslan, A. H., E. J. Akin, J. M. Barton, J. L. Bledsoe, K. E. Cross, H. Diament, D. E. Fields, S. K. Fischer, J. L. Harris, D. M. Hetrick, J. T. Holdeman, K. K. Kim, M. H. Lietzke, J. E. Park, M. R. Patterson, R. D. Sharp, and B. Thomas, Jr. 1977. Development of a Unified Transport Approach for the Assessment of Power-Plant Impact. ORNL/NUREG/TM-89, Oak Ridge National Laboratory, Oak Ridge TN.
- Fields, D. E. 1976. CHNSD: Simulation of Sediment and Trace Contaminant Transport with Sediment/Contaminant Interaction. ORNL/NSF/EATC-19, Oak Ridge National Laboratory, Oak Ridge, TN.

- Fisher, H. B. 1967. "The Mechanics of Dispersion in Natural Streams," Journal of Hydraulic Division. 93(HY,6, ASCE):187-216.
- Fletcher, J. F., and W. L. Dotson (eds). 1971. Hermes-A Digital Computer Code for Estimating Regional Radiological Effects from the Nuclear Power Industry. USAEC Rep. HEDL-TME-71-1968.
- Fletcher, J. F., W. L. Dotson, D. E. Peterson and R. P. Betson. 1973. "Modeling the Regional Transport of Radionuclide in a Major United States River Basin," Environmental Behavior of Radionuclide Released in the Nuclear Industry. IAEA Vienna.
- Fread, D. L. 1973. "Technique for Implicit Dynamic Routing in Rivers with Major Tributaries." Water Resources Research. 9(4):918-926.
- Frere, M. H., C. A. Onstad and H. N. Holtan. 1975. ACTMO, An Agricultural Chemical Transport Model. Agricultural Research Service, USDA, ARS-H-3, 54 pp.
- Gallegos, A. F., B. J. Garcia and C. M. Sutton. 1980. Documentation of TRU Biological Transport Model (BIOTRAN). Los Alamos National Laboratory Report LA-8213-MS, Los Alamos National Laboratory, Los Alamos, NM.
- Gloyna, E. F., et al. 1963-1972. "Radioactivity Transport in Water." Technical Reports 1 through 23. U.S. AEC Contract No. AT(11-1)-490, The University of Texas, Austin, TX.
- Gloyna, E. F., and J. O. Ledbetter. 1969. Principles of Radiological Health. Marcel Dekker, Inc., New York, NY.
- Gromiec, M. J., and E. F. Gloyna. 1973. Radioactivity Transport in Water -- Final Report. EHE-73-01, Technical Report No. 23 to the U.S. Atomic Energy Commission, Center for Research in Water Resources, University of Texas, Austin, TX.
- Hakonson, T. E., et al. 1976. "Accumulation and Transport of Soil Plutonium in Liquid Waste Discharge Areas at Los Alamos." Transuranium Nuclides in the Environment, IAEA, Vienna.
- Haushild, W. L., H. H. Stevens, Jr., J. L. Nelson and G. R. Dempster, Jr. 1971. Radionuclides in Transport in the Columbia River from Pasco to Vancouver, Washington. Geological Survey. Open-file Report. U.S. Department of Interior, Western Resources Division, Portland, OR.
- Hjelmfelt, Jr., A. T. 1976. Modeling of Soil Movement Across a Watershed, Missouri Water Resources Center, University of Missouri, Completion Report for Project A-076-MO.

- Horst, T. W. 1979a. "A Review of Gaussian Diffusion-Deposition Models." Symposium on Potential Environmental and Health Effects of Atmospheric Sulfur Deposition, Gatlinburg, TN.
- Horst, T. W. 1979b. "The Annual-Average Effect of the Deposition-Resuspension Process on Airborne Contamination Near the Surface." Atmos. Environ., 13:791-796.
- Jenne, E. A. and J. S. Wahlberg. 1965. "Management and Iron Oxide Scavenging of Cobalt 60 in White Oak Creek Sediment, Oak Ridge, Tenn." Am. Geophys. Union Trans. 46(1):70 (Abstract).
- Kennedy, V. C. 1965. Mineralogy and Cation-Exchange Capacity of Sediments from Selected Streams. Geological Survey Professional Paper 433-D, U.S. Department of Interior.
- Kibler, D. F., and Woolhiser, D. A. 1970. The Kinematic Cascade as a Hydrologic Model, Hydrology Paper No. 39, Colorado State University, Fort Collins, Colorado.
- Kuh, H., D. L. Raddell, and E. A. Hiler. 1976. Two-Dimensional Model of Erosion from a Watershed, ASAE Paper 76-2539.
- Linsley, R. K., M. A. Kohler and J. L. H. Paulus. Hydrology for Engineers. Second Edition, McGraw-Hill, New York.
- Mahmoud, K. A., Y. A. Yousef, M. M. Ishak, W. E. Y. Adbel-Malik, S. R. Khalil, A. Shafik, R. M. K. El-Shinawy. 1973. "Transfer Mechanism and Concentrating Processes of Certain Radionuclides in the Aquatic Ecology of the Ismailia Canal," Environmental Behavior of Radionuclides Released in the Nuclear Industry. International Atomic Energy Agency, Vienna, Austria. pp. 205-215.
- Martin, J. A., Jr., C. Robbins, C. B. Nelson, R. D. Cousins, Jr., M. A. Culliton. 1976. A Computer Code (RADDOS) to Calculate Population Doses from Radioactive Liquid Effluents and on Application to Nuclear Power Reactors on the Mississippi River Basin. U.S. EPA, Office of Radiation Programs, Environmental Analysis Division.
- McElroy, A. D., et al. 1976. "Loading Functions for Assessment of Water Pollution From Nonpoint Sources." Office of Research and Development, U.S. Environmental Protection Agency, PB-253 325.
- Metcalf and Eddy, Inc. Storm Water Management Model. Volume 1-4, Palo Alto, CA.
- Meyer, L. D., and E. J. Monke. 1965. "Mechanics of Soil Erosion by Rainfall and Overland Flow," Transactions of the Americal Society of Agricultural Engineers, 8(4):572-577, 580.

- Nelson, J. L., R. W. Perkins, J. M. Nielsen, and W. L. Haushild. 1966. "Reactions of Radionuclides from the Hanford Reactors with Columbia River Sediments." Disposal of Radioactive Wastes into Seas, Oceans, and Surface Waters. International Atomic Energy Agency, Vienna, Austria. pp. 139-161.
- Onishi, Y. 1977. Mathematical Simulation of Sediment and Radionuclide Transport in the Columbia River. BNWL-2228, Pacific Northwest Laboratory, Richland, WA.
- Onishi, Y. 1980. "Section 4. Sediment Effects" of Aquatic Disposal of Radionuclides. Prepared by P. J. Ryan, M. Leonard, G. Segal and Y. Onishi for the Summer School of Health Physics Society.
- Onishi, Y., and S. E. Wise. 1978. "Mathematical Modeling of Sediment and Contaminant Transport in the James River Estuary." Proceedings of 27th Annual Hydraulic Division Specialty Conference of ASCE, pp. 303-310, College Park, MA.
- Onishi, Y., P. A. Johanson, R. G. Baca and E. L. Hilty. 1976. Studies of Columbia River Water Quality--Development of Mathematical Models of Sediment and Radionuclide Transport Analysis. BNWL-B-452, Pacific Northwest Laboratory, Richland, WA.
- Onishi, Y., D. L. Schreiber and R. B. Codell. 1979a. "Mathematical Simulation of Sediment and Radionuclide Transport in the Clinch River, Tennessee." Proceedings of ACS/CSJ Chemical Congress, Honolulu, HI, April 2-7, 1979, Contaminants and Sediment, edited by R. A. Baker, Ann Arbor Publisher, Inc.
- Onishi, Y., S. M. Brown, A. R. Olsen, M. A. Parkhurst, S. E. Wise, and W. H. Walters. 1979b. Methodology for Overland and Instream Migration and Risk Assessment of Pesticides. Battelle, Pacific Northwest Laboratories, Richland, WA.
- Onishi, Y., G. Whelan, M. A. Parkhurst, A. R. Olsen, and P. J. Gutknecht. 1980. Preliminary Assessment of Toxaphene Migration and Risk in the Yazoo River Basin, Mississippi. Submitted to the U.S. Environmental Protection Agency by Battelle, Pacific Northwest Laboratories, Richland, WA.
- Onishi, Y., R. J. Serne, E. M. Arnold, C. E. Cowan, and F. L. Thompson. 1981. Critical Review: Radionuclide Transport, Sediment Transport and Water Quality Mathematical Modeling, and Radionuclide Adsorption/Desorption Mechanisms. NUREG/CR-1322, PNL-2901, Pacific Northwest Laboratory, Richland, WA.
- Parker, F. L., M. A. Churchill, R. W. Andrew, B. J. Frederick, P. H. Carrigan, Jr., J. S. Cragwall, Jr., S. A. Jones, E. G. Struxness, and R. J. Morton. 1966. "Dilution, Dispersion and Mass Transport of Radionuclides in the Clinch and Tennessee Rivers." Proceedings of the Symposium on the Disposal of Radioactive Wastes into Seas, Oceans and Surface Waters. International Atomic Energy Agency, Vienna, Austria, pp. 33-54.

- Parkhurst, M. A., G. Whelan, Y. Onishi, A. R. Olsen. 1981. Simulation of the Migration, Fate and Effects of Diazinon in Two Monticello Stream Channels. Prepared for the U.S. Army Bioengineering Laboratory by Battelle, Pacific Northwest Laboratories, Richland, WA.
- Patterson, M. R., et al. 1974. "A User's Manual for the Fortran IV Version of the Wisconsin Hydrologic Transport Model", ORNL-NSF-EATC-7, Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Pickering, R. J., P. H. Carrigan, Jr., T. Tamura, H. H. Abee, J. W. Beverage and R. W. Andrew, Jr. 1966. "Radioactivity in Bottom Sediment of the Clinch and Tennessee Rivers." Disposal of Radioactive Wastes into Seas, Oceans, and Surface Waters. International Atomic Energy Agency, Vienna, Austria, pp. 57-86.
- Pruter, A. T., and D. L. Alverson. 1972. The Columbia River Estuary and Adjacent Ocean Waters--Bioenvironmental Studies. University of Washington Press, Seattle, WA.
- Reynolds, T. D., and E. F. Gloyna. 1963. Radioactivity Transport in Water--Transport of Strontium and Cesium by Stream and Estuarine Sediments. Technical Report No. 1 to the U.S. Atomic Energy Commission, University of Texas, Austin, TX.
- Roesner, L. A., et al. 1975. Agricultural Watershed Runoff Model for the Iowa-Cedar River Basins. Report for U.S. Environmental Protection Agency, Systems Development Branch.
- Rovey, E. W., D. A. Woolhiser and R. E. Smith. 1977. A Distributed Kinematic Model of Upland Watersheds, Hydrology Paper No. 93, Colorado State University, Fort Collins, Colorado.
- Rowe, D. R., and E. F. Gloyna. 1964. Radioactivity Transport in Water--The Transport of ^{65}Zn in an Aqueous Environment. EHE-09-6403, Technical Report No. 5 to the U.S. Atomic Energy Commission, Center for Research in Water Resources, University of Texas, Austin TX.
- Sayre, W. W., H. P. Guy and A. R. Chamberlain. 1963. Uptake and Transport of Radionuclides by Stream Sediments. Geological Survey Professional Paper 433-A.
- Sehmel, G. A., and F. D. Lloyd. 1976. "Particle Resuspension Rates." Proceedings of the Conference on Atmospheric-Surface Exchange of Particulate and Gaseous Pollutants, Richland, WA, September 4-6, 1974. ERDA Symposium Series 38, CONF-740921, NTIS, Springfield, VA, 988 pp.
- Shih, C. S., and E. F. Gloyna. 1967. Radioactivity Transport in Water--Mathematical Model for the Transport of Radionuclides. EHE-04-6702, Technical Report No. 12 to the U.S. Atomic Energy Commission, University of Texas, Austin, TX.

- Shih, C. S., and E. F. Gloyna. 1969. "Influence of Sediments on Transport of Solutes." J. Hydraulics Div. ASCE 95(HY4):1348-1367.
- Shih, C. S., and E. F. Gloyna. 1970. "Mathematical Model for the Transport of Radionuclides in Stream System." Environmental Surveillance in the Vicinity of Nuclear Facilities. Reinig, W. C. (ed.), Charles C. Thomas Publishing Co., Springfield, IL.
- Shull, R. D., and E. F. Gloyna. 1968. Radioactivity Transport in Water-- Simulation of Sustained Releases to Selected River Environments. EHE-04-6801, Technical Report No. 15 to the U.S. Atomic Energy Commission, University of Texas, Austin, TX.
- Smith, J. H., W. R. Mabey, N. Bohonos, B. R. Holt, S. S. Lee, T.-W. Chou, D. C. Bomberger and T. Mill. 1977. Environmental Pathways of Selected Chemicals in Freshwater Systems. Part I: Background and Experimental Procedures. EPA-600/7-77-113, Submitted to U.S. Environmental Protection Agency by SRI International, Menlo Park, CA.
- Soldat, J. K., N. M. Robinson and D. A. Baker. 1974. Models and Computer Codes for Evaluating Radiation Doses. BNWL-1754, Pacific Northwest Laboratory, Richland, WA.
- Stewart, K. 1967. "The Resuspension of Particulate Material from Surfaces." Surface Contamination, B. R. Fish, ed., Pergamon Press, New York.
- Tritremmel, Ch., G. Knollmayer, E. Wanderer, H. G. Heintschel and P. Stipanits. 1966. "Behavior of Radioisotopes Released to a Stream." Proceedings of the Symposium on the Disposal of Radioactive Wastes into Seas, Oceans, and Surface Waters. International Atomic Energy Agency, Vienna, pp. 89-105.
- U.S. Environmental Protection Agency. 1978. Kepone Mitigating Project Report, Standard and Criteria, Office of Water and Hazardous Materials, Washington, D.C.
- U.S. Nuclear Regulatory Commission. 1976. Regulatory Guide 1.113 - Estimating Aquatic Dispersion of Effluents from Accidental and Routine Reactor Releases for the Purpose of Implementing Appendix I.
- U.S. Nuclear Regulatory Commission. 1978a. Liquid Pathway Generic Study. Impacts of Accidental Radioactivity Releases to the Hydrosphere from Floating and Land-Based Nuclear Power Plants. NUREG-0440.
- U.S. Nuclear Regulatory Commission. 1978b. Revised Draft Environmental Statement Related to the Proposed Manufacture of Floating Nuclear Power Plants. NUREG-0127 (Revision 1)
- U.S. Nuclear Regulatory Commission. 1979. (Draft) 10 CRF 61: Disposal of Low-Level Radioactive Waste and Low-Activity Bulk Solid Waste.

- Vanderploeg, H. A., R. S. Booth and F. H. Clark. 1976. "A Specific Activity and Concentration Model Applied to Cesium-137 Movement in a Eutrophic Lake." Radioecology and Energy Resources. C. E. Cushing, Jr. (ed.) Dowden, Hutchinson & Sons, Inc.
- Vanoni, V. A., ed. 1975. Sedimentation Engineering. Prepared by the ASCE Task Committee for the Preparation of the Manual on Sedimentation of the Sedimentation Committee of the Hydraulics Division, ASCE.
- Watts, J. R. 1976. "Modeling of Radiation Doses from Chronic Aqueous Releases." Paper presented at the Health Physics Society Annual Meeting. San Francisco, CA.
- Wheeler, M. L. 1977. A Preliminary Evaluation of the Potential for Plutonium Release from Burial Grounds at Los Alamos Scientific Laboratory. LA-6694-MS, Los Alamos Scientific Laboratory, Los Alamos, NM.
- Whelan, G. 1980. Distributed Model for Sediment Yield, Master's Thesis Mechanics and Hydraulics, University of Iowa, Iowa City, Iowa.
- White, A., and E. F. Gloyna. 1969. Radioactivity Transport in Water--Mathematical Simulation. EHE-70-04, Technical Report No. 19 to the U.S. Atomic Energy Commission, University of Texas, Austin, TX.
- Witinok, P. M. 1979. Distributed Watershed and Sedimentation Model, Master's Thesis, Department of Geology, the University of Iowa, Iowa City, Iowa.
- Woodruff, N. P., and F. H. Siddoway. 1965. "A Wind Erosion Equation." Soil Sci. Soc. Amer. Proc., 29:602-608.
- Yevjevich, V., and K. Mohmood, (eds.). 1975. Unsteady Flow in Open Channels, Vol. I, Water Resources Publications, Ft. Collins, Colorado.
- Yousef, Y. A., and E. F. Gloyna. 1964. Radioactivity Transport in Water--The Transport of ^{58}Co in an Aqueous Environment. EHE-12-6405, Technical Report No. 7 to the U.S. Atomic Energy Commission, 121 pp.
- Yousef, Y. A., A. Kudo and E. F. Gloyna. 1970. Radioactivity Transport in Water--Summary Report. EHE-70-05, Technical Report No. 20 to the U.S. Atomic Energy Commission, 80 pp.

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