**Finite Element Models for
Sediment and Contaminant
Transport in Surface Waters**

**Transport of Sediments and
Radionuclides in the
Clinch River**

by
Yasuo Onishi

July 1977

**Prepared for the
Nuclear Regulatory Commission**

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FINITE ELEMENT MODELS FOR SEDIMENT AND
CONTAMINANT TRANSPORT IN SURFACE WATERS

TRANSPORT OF SEDIMENTS AND RADIONUCLIDES
IN THE CLINCH RIVER

by
Yasuo Onishi

July 1977

BATTELLE
PACIFIC NORTHWEST LABORATORIES
RICHLAND, WASHINGTON

SUMMARY

Radionuclide migration in rivers was simulated in this study to advance the state-of-the-art of computer modeling on radionuclide transport by including the effects of sediment-radionuclide interaction. Specifically, the finite element sediment and contaminant transport model, SERATRA, was modified and applied to the Clinch River in Tennessee to solve time-dependent, longitudinal and vertical distributions of sediments and radionuclides. Sediment transport was modeled for each sediment size fraction (i.e., sand, silt and clay), and radionuclide transport was modeled for dissolved and particulate nuclides. Furthermore, particulate radionuclides were solved for those adsorbed by each sediment size fraction.

Three radionuclides, cesium-137, strontium-90 and gold-198, were selected as sources of continuous and instantaneous releases because of their adsorption characteristics and field data availability. Agreement of predicted results and field data for continuous release cases was very good, while for instantaneous releases agreement was poor.

It was revealed that approximately 93 percent of the cesium-137 is in a particulate form, and only about 7 percent is dissolved. The model predicted that approximately 50 percent of the cesium 137 introduced in the Clinch River will be deposited on the river bed before it reaches the river mouth as a result of contaminated sediment deposition in slow moving areas of the river. Results on strontium-90 indicated the opposite trend, i.e., approximately 97 percent is in the dissolved form and only 3 percent was associated with the sediment; hence, the majority of strontium-90 moves with the water through the river system. Gold-198 was used for instantaneous release simulation, but since agreement between simulated results and data was not good no conclusions can be drawn for this case.

The results of this study indicate that suspended sediment has significant effects on radionuclide transport through radionuclide adsorption, deposition, and resuspension of contaminated sediments. Hence, in order

to obtain realistic predictions of radionuclide migration, interaction between radionuclides and river sediments must be taken into account. The present study demonstrates that the SERATRA code is capable of predicting the transport of both particulate and dissolved radionuclides by taking into account the radionuclide-sediment interaction. However, complete verification of the model is required before reliable predictions can be made concerning the migration of radionuclide-contaminated sediments within river systems.

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SYMBOLS

- A = the river cross-sectional area
- a = coefficient in Equation 24
- B = river width
- b = coefficient in Equation 24
- C_j = concentration of sediment of j th size fraction
- C_{ij} = sediment concentration of horizontal inflow for j th size fraction (no summation rule is applied)
- $D = \frac{\epsilon_z}{\Delta z^2}$
- e_i = subdomain of interest in Equation 22
- G_{Bj} = the radionuclide concentration per unit weight of sediment on the river bed for j th sediment size fraction
- G_j = particulate radionuclide concentration per unit weight of sediment in j th sediment size fraction
- G_{ij} = particulate concentration per unit weight of sediment in j th sediment size fraction in horizontal inflow (no summation will be applied)
- G_w = dissolved radionuclide concentration
- G_{wi} = dissolved radionuclide concentration in horizontal inflow
- h = water depth
- k_{dj} = rate of adsorption (or desorption) between dissolved nuclide and sediment (suspended and bed load sediments) of j th size fraction
- K_j = an empirical constant depending on the sediment type
- ℓ = longitudinal distance
- M_j = erodibility coefficient for sediment of j th size fraction
- n = the number of subdomains
- N = number of sediment size fractions considered. In this study, $N = 3$ (sand, silt and clay)

Q_i = horizontal inflow discharge equal to $U_i B \Delta z$
 Q_o = horizontal outflow discharge equal to $U_o B \Delta z$
 Q_v = vertical flow discharge equal to WA
 R = a domain of interest
 S_{Dj} = sediment deposition rate per unit area for j th sediment size fraction
 S_{Rj} = sediment erosion rate per unit area for j th sediment size fraction
 t = time
 U_o = horizontal outflow velocity
 U_i = horizontal inflow velocity
 v = coefficient in Equation 18
 v_i = Value v at i th node
 $\bar{v} = (v_i + 1/2 v_{i+1})$ shown in Equation 44
 $\tilde{v} = (1/2 v_i + v_{i+1})$ shown in Equation 44
 W = vertical flow velocity
 W_i = weighting function in Equation 20
 W_j = weighting function in Equation 19
 W_{sj} = fall velocity of sediment particle of j th size fraction
 z = vertical direction
 $[P]$ = symmetric tridiagonal matrix in Equation 40
 $[\tilde{P}]$ = symmetric tridiagonal matrix in Equation 45
 $[\bar{P}]$ = symmetric tridiagonal matrix in Equation 48
 $\{R\}$ = load vector in Equation 40
 $\{\tilde{R}\}$ = load vector in Equation 45
 $\{\bar{R}\}$ = load vector in Equation 47

$[S]$ = unsymmetric tridiagonal matrix in Equation 40
 $[\tilde{S}]$ = unsymmetric tridiagonal matrix in Equation 45
 $[\tilde{S}^*]$ = unsymmetric tridiagonal matrix in Equation 47
 α = coefficient in Equation 18
 β = coefficient in Equation 18
 γ = coefficient, i.e., probability that particle settling to the bed is deposited
 Δz = increment of vertical distance
 ϵ_z = coefficient in Equation 18
 ϵ_{zj} = diffusion coefficient of vertical (z) component for jth sediment size fraction
 ϵ_{zw} = vertical diffusion coefficient for dissolved radionuclides
 λ = decay rate of radioactive material
 τ_b = bed shear stress
 τ_{cdj} = critical shear stress for sediment deposition for jth sediment size fraction
 ϕ = variable in Equation 18
 $\bar{\phi}$ = approximation of ϕ shown in Equation 20
 $\dot{\phi}$ = time derivative of ϕ
 χ = residual error in Equation 21

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TRANSPORT OF SEDIMENT AND RADIONUCLIDES IN THE CLINCH RIVER

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1. INTRODUCTION

During normal operation of nuclear power reactors and other nuclear-related facilities small quantities of radioactive materials are released into receiving water bodies via plant cooling and service water discharges. These radionuclides are dispersed by mixing with river water, or can be adsorbed from solution onto sediments. Sorption by sediments is an important mechanism affecting the migration of radionuclides throughout a water body. This is seen by the deposition and resuspension of contaminated sediments on river beds.

In this study, a mathematical simulation of radionuclide migration in rivers was conducted to advance the state-of-the-art of radionuclide transport modeling by including the effects of sediment- radionuclide interaction. This study was conducted to assist the U.S. Nuclear Regulatory Commission (NRC) in hydrologic dispersion and radionuclide transport modeling as a part of their Liquid Pathway Generic Study.⁽¹⁾ To achieve this objective, the SERATRA code was applied to the Clinch River in Tennessee to compute sediment and radionuclide concentrations in the river and thereby verify the model. Although the field data are not sufficiently detailed to verify the computer code completely, the comparison of computer results with the field data shows a good agreement.

Radionuclides found in the Clinch River originated primarily from effluents discharged from early operations of the Oak Ridge National Laboratory (ORNL).^(2,3,4) The four major radioactive materials found are cesium-137 (^{137}Cs), strontium-90 (^{90}Sr), cobalt-60 (^{60}Co) and ruthenium-106

(^{106}Ru).^(5,6,7) Field measurements conducted in the early 1960's indicated that approximately 90% of ^{137}Cs released was adsorbed by the river's suspended sediments within a 10-mile reach downstream of the effluent discharge.⁽⁸⁾ At the same time, only about 5% of ^{90}Sr was associated with river sediments.⁽⁸⁾ The difference in behavior is an indication that if sediment effects on radionuclide transport are not considered, erroneous evaluations of radionuclide distributions in surface waters most likely will result.

Past studies concerned with sediment-related effects have not advanced because of the extremely complex physical and chemical characteristics of sediments and radionuclides. Because of the lack of existing verified generalized models,⁽⁹⁾ engineers have had to rely on field studies or experience to estimate the distribution of radionuclide concentrations.

The SERATRA model was developed as a means for more realistic predictions.^(10,11) By use of the finite element solution technique with the Galerkin weighted residual method, this model predicts temporal and spatial distributions (vertical and longitudinal) concentrations of both cohesive and noncohesive sediments, as well as dissolved radionuclides and radionuclides attached to each size fraction of river sediments. The model also predicts changes in river bed elevations and radionuclide inventory in the river bed. Sediment transport and adsorption were simulated independently for each sediment size fraction (e.g., sand, silt and clay) because important sediment characteristics, such as fall velocity, exchange capacity, and critical shear stresses for erosion and deposition, vary significantly with sediment particle sizes.

This report details the application of the SERATRA model to the Clinch River. Besides the simulation results, sections are included covering the model formulation and the code description.

2. MATHEMATICAL MODEL FORMULATION

Longitudinal and vertical distributions of sediment and radionuclide concentrations were computed by the SERATRA model. The modeling procedure involved simulating the transport of sediments within the water body. The results were then input to models of dissolved and particulate radionuclides in order to observe the interaction between sediment and radionuclide movements. Finally, changes in river bed conditions were recorded, including river bottom elevation change, ratio of sand, silt and clay sediments, and distribution of radionuclide concentration in the river bed.

This model is also applicable to other constituents, including nutrients, pesticides and heavy metals, which may undergo physical-chemical reactions with the sediments.

SEDIMENT TRANSPORT MODEL

Sand, silt and clay transport are modeled independently since sediment movement and adsorption capacity varies significantly with particle size. The model includes the effects of:

1. Convection and dispersion of sediments
2. Fall velocity and cohesiveness
3. Deposition on the river bed
4. Resuspension from the river bed (bed erosion and armoring)
5. Tributaries.

Sediment minerology and water quality effects are implicitly included through the above mentioned effects 2, 3, and 4.

Governing Equations

Mass conservation of sediment passing through the control volume leads to the following expression for the transport of sand, silt and clay sediments:

$$\underbrace{\frac{\partial}{\partial t} (C_j B \Delta z)}_{\text{rate of accumulation}} + \underbrace{(U_o C_j \Delta z B - U_i C_{ij} B \Delta z)}_{\text{horizontal convection}} + \underbrace{\frac{\partial}{\partial z} \{C_j (W - W_{sj}) B \Delta z\}}_{\text{vertical convection}} = \underbrace{\frac{\partial}{\partial z} (\epsilon_{zj} \frac{\partial C_j}{\partial z} B \Delta z)}_{\text{vertical diffusion}} \quad j = 1, 2, \dots, N \quad (1)$$

where (a)

- B = river width
- C_j = concentration of sediment of jth size fraction
- C_{ij} = sediment concentration of horizontal inflow for jth size fraction
(no summation rule is applied)
- ϵ_{zj} = diffusion coefficient of vertical (z) component for jth sediment size fraction
- ℓ = longitudinal distance
- N = number of sediment size fractions considered. In this study,
N = 3 (sand, silt and clay)
- t = time
- U_i = horizontal inflow velocity
- U_o = horizontal outflow velocity
- W = vertical flow velocity
- W_{sj} = fall velocity of sediment particle of jth size fraction
- z = vertical direction
- Δz = increment of vertical distance

This quasi-two-dimensional formulation neglects longitudinal diffusion and assumes lateral sediment concentrations to be uniform. However, the model does simulate longitudinal dispersion due to vertical variation of longitudinal velocity.

In terms of volumetric flow rates, Equation (1) may be written as:

$$\frac{\partial C_j}{\partial t} + \frac{Q_v}{A} - W_{sj} \frac{\partial C_j}{\partial z} = \epsilon_{zj} \frac{\partial^2 C_j}{\partial z^2} - \frac{Q_o C_j}{A \Delta z} + \frac{Q_i C_{ij}}{A \Delta z} \quad (2)$$

(a) The symbols defined above remain the same throughout the report

where

$$A = B\ell$$

$$Q_i = U_i B \Delta z$$

$$Q_o = U_o B \Delta z$$

$$Q_v = WA$$

Boundary conditions at the water surface ($z = h$) and river bed ($z = 0$) are:

$$\left(\frac{Q_v}{A} - W_{sj} \right) C_j - \epsilon_{zj} \frac{\partial C_j}{\partial z} = 0 \quad \text{at } z = h \quad (3)$$

$$-(1 - \gamma) W_{sj} C_j - \epsilon_{zj} \frac{\partial C_j}{\partial z} = S_{Rj} - S_{Dj} \quad \text{at } z = 0 \quad (4)$$

where

h = water depth

S_{Dj} = sediment deposition rate per unit area for j th sediment size fraction

S_{Rj} = sediment erosion rate per unit area for j th sediment size fraction

γ = coefficient, i.e., probability that particle settling to the bed is deposited.

In this study, γ is assumed to be unity, that is, for the same flow condition all suspended sediment settling to the river bed will stay on the river bed without going into the flow again.

Erosion and Deposition of Cohesive Sediments (Silt and Clay)

Sediment erosion and deposition rates, S_{Rj} and S_{Dj} , are also evaluated separately for each sediment size fraction because erosion and deposition characteristics are significantly different for cohesive and noncohesive sediments. Since only Partheniades⁽¹²⁾ and Krone⁽¹³⁾ formulas for erosion and deposition rates, respectively, are presently available, these formulas were adopted in this study:

$$S_{Rj} = M_j \left(\frac{\tau_b}{\tau_{cRj}} - 1 \right) \quad (5)$$

$$S_{Dj} = \frac{2W_{sj}C_j}{h} \left(1 - \frac{\tau_b}{\tau_{cDj}} \right) \quad (6)$$

where

M_j = erodibility coefficient for sediment of jth size fraction

τ_b = bed shear stress

τ_{cDj} = critical shear stress for sediment deposition for jth sediment size fraction

τ_{cRj} = critical shear stress for sediment erosion for jth sediment size fraction

Values of M_j , τ_{cDj} and τ_{cRj} must be determined by field and/or laboratory tests for a particular river regime. PNL has been conducting laboratory flume tests by using Columbia River cohesive sediments to examine these values under fresh and saline water environments for a modeling study on sediment and radionuclide transport in the Columbia River. (11)

When the fall velocity, W_{sj} , depends on sediment concentration and no aggregation occurs, the fall velocity may be assumed: (13)

$$W_{sj} = K_j C_j^{4/3} \quad (7)$$

where

K_j = an empirical constant depending on the sediment type.

Erosion and Deposition of Noncohesive Sediments (Sand)

Erosion and deposition of noncohesive sediments are affected by the amount of sediment the flow is capable of carrying. For example, if the amount of sand being transported is less than the flow can carry for given hydrodynamic conditions, the river will scour sediment from the stream bed to increase the sediment transport rate. This occurs until the actual sediment transport rate becomes equal to the carrying capacity of the flow. Conversely, the river deposits sand if its actual sediment

transport rate is above the flow's capacity to carry sediment. Toffaleti's formula⁽¹⁴⁾ was used to estimate the flow capacity, Q_s , which was then compared with the actual amount of sand, Q_{sa} , being transported in a river water. Hence:

$$S_{Rj} = \frac{Q_s - Q_{sa}}{A} \quad (8)$$

$$S_{Dj} = \frac{Q_{sa} - Q_s}{A} \quad (9)$$

where

A = the river bed surface area.

DISSOLVED RADIONUCLIDE TRANSPORT MODEL

A study conducted by Nielson and Perkins⁽¹⁵⁾ concluded that the association of dissolved radionuclides with suspended sediments is the primary mechanism of radionuclide uptake, while biota accounts for a small percentage of the depletion. Laboratory flume experiments conducted by Gloyna et al.⁽¹⁶⁾ also revealed that a small amount of a radionuclide become associated with the bottom sediments by direct sorption, and that the larger amount found in the bed sediments is attributed to the settling of suspended sediments. For this study the radionuclide transport model considers only the effects of adsorption (or desorption) of dissolved radionuclide materials (or other dissolved contaminants) by suspended and bed load sediments. In addition to adsorption mechanics, the model includes the effects of:

1. Convection and dispersion of radionuclides (or other contaminants) within the river
2. Adsorption (uptake) of dissolved radionuclides (or other contaminants) by sediments (suspended and bed load sediments) or desorption from sediments into water
3. Radionuclide (or other contaminants) decay
4. Tributaries. (Radionuclide contributions from nuclear facilities, fallout and groundwater to a river may be treated as a part of tributary contributions).

Effects of water quality (e.g., pH, water temperature, salinity, etc.) and clay minerals are taken into account through changes in adsorption or desorption rate, K_{dj} . Mass conservation of dissolved radionuclides (and other contaminants) may be expressed as:

$$\begin{aligned}
 & \underbrace{\frac{\partial}{\partial t} (G_w B \Delta z)}_{\text{rate of accumulation}} + \underbrace{(U_o G_w B \Delta z - U_i G_{wi} B \Delta z)}_{\text{horizontal convection}} + \underbrace{\frac{\partial}{\partial z} (W G_w B \Delta z)}_{\text{vertical convection}} \\
 &= \underbrace{\frac{\partial}{\partial z} \left(\epsilon_{zw} \frac{\partial G_w}{\partial z} B \right) \Delta z}_{\text{vertical diffusion}} - \underbrace{\lambda G_w B \Delta z}_{\text{decay term}} - \sum_j \underbrace{k_{dj} C_j G_w B \Delta z}_{\text{adsorption (or desorption)}}, \quad (10)
 \end{aligned}$$

j = 1, 2, ..., N

In addition to the previously defined symbols:

- ϵ_{zw} = vertical diffusion coefficient for dissolved radionuclides
- G_w = dissolved radionuclide concentration
- G_{wi} = dissolved radionuclide concentration in horizontal inflow
- k_{dj} = rate of adsorption (or desorption) between dissolved nuclide and sediment (suspended and bed load sediments) of jth size fraction
- λ = decay rate of radioactive material.

It was assumed that radionuclide uptake is the first order reaction and that the rate of radionuclide uptake is proportional to the product of concentrations of dissolved radionuclides and sediments, as shown in Equation (10). Longitudinal diffusion is considered to be negligible when compared to convection. Longitudinal dispersion due to vertical variation of longitudinal velocity was, however simulated in this study.

In terms of volumetric flow rates, Equation (10) may be rewritten as:

$$\begin{aligned}
 & \frac{\partial G_w}{\partial t} + \frac{Q_v}{A} \frac{\partial G_w}{\partial z} + \frac{Q_o}{A \Delta z} + \lambda + \sum_j k_{dj} C_j G_w = \epsilon_{zw} \frac{\partial^2 G_w}{\partial z^2} \\
 & + \left(\frac{Q_i G_{wi}}{A \Delta z} \right), \quad j = 1, 2, \dots, N \quad (11)
 \end{aligned}$$

Boundary conditions at the water surface and river bed are:

$$\frac{Q_v G_w}{A} - \epsilon_{zw} \frac{\partial G_w}{\partial z} = 0 \quad \text{at } z = h \quad (12)$$

$$\frac{\partial G_w}{\partial z} = 0 \quad \text{at } z = 0 \quad (13)$$

PARTICULATE RADIONUCLIDE TRANSPORT MODEL

The transport model of radionuclides (or other contaminants) attached to sediment includes the effects of

1. Convection and dispersion of particulate radionuclides
2. Adsorption (uptake) of dissolved radionuclides by sediments or desorption from sediments into water
3. Radionuclide decay
4. Deposition of particulate radionuclides to the river bed or resuspension from the river bed
5. Tributaries. (Radionuclide contributions from nuclear facilities, fallout and groundwater to a river may be treated as a part of the tributary contributions.)

As in the transport of sediments and dissolved radionuclides, the conservation of radionuclides adsorbed by each sand, silt and clay sediment may be expressed as:

$$\begin{aligned} & \frac{\partial}{\partial t}(G_j C_j B \Delta z) + (U_o G_j C_j B \Delta z - U_i G_{ij} C_{ij} B \Delta z) + \frac{\partial}{\partial z}\{(W - W_{sj})G_j C_j B \Delta z\} \\ & \quad \text{rate of} \quad \quad \quad \text{horizontal} \quad \quad \quad \text{vertical} \\ & \quad \text{accumulation} \quad \quad \quad \text{convection} \quad \quad \quad \text{convection} \\ & = \frac{\partial}{\partial z} \left(\epsilon_{zj} \frac{\partial G_j C_j B}{\partial z} \right) \Delta z - \lambda G_j C_j B \Delta z + k_{dj} C_j G_w B \Delta z \\ & \quad \quad \quad \text{vertical} \quad \quad \quad \text{decay} \quad \quad \quad \text{adsorption} \\ & \quad \quad \quad \text{diffusion} \quad \quad \quad \text{term} \quad \quad \quad \text{(or desorption)} \end{aligned} \quad (14)$$

where

G_j = particulate radionuclide concentration per unit weight of sediment in jth sediment size fraction

G_{ij} = particulate concentration per unit weight of sediment in jth sediment size fraction in horizontal inflow (no summation will be applied).

Longitudinal dispersion was assumed to be negligible, as compared to longitudinal convection. However, as noted before, the longitudinal dispersion due to nonuniform vertical distribution of longitudinal velocity was simulated in the model.

Subtraction of Equation (1) from Equation (14) yields the following expression in terms of volumetric flow rates:

$$\begin{aligned} \frac{\partial G_j}{\partial t} + \left(\frac{Q_v}{A} - W_{sj} - \frac{2\epsilon_{zj}}{C_j} \frac{\partial C_j}{\partial z} \right) \frac{\partial G_j}{\partial z} + \left(\frac{Q_i C_{ij}}{C_j A \Delta z} + \lambda + k_{dj} \right) G_j \\ = \epsilon_{zj} \frac{\partial^2 G_j}{\partial z^2} + \left(\frac{Q_i C_{ij} G_{ij}}{C_j A \Delta z} + k_{dj} G_w \right) \end{aligned} \quad (15)$$

The boundary condition at the water surface is:

$$G_j C_j \left(\frac{Q_v}{A} - W_{sj} \right) - \epsilon_{zj} \frac{\partial G_j C_j}{\partial z} = 0 \quad \text{at } z = h$$

Noting Equation (3), the above equation becomes:

$$\frac{\partial G_j}{\partial z} = 0 \quad \text{at } z = h \quad (16)$$

The boundary conditions at the river bed is:

$$-(1 - \gamma) W_{sj} G_j C_j - \epsilon_{zj} \frac{\partial G_j C_j}{\partial z} = G_{Bj} S_{Rj} - G_j S_{Dj} \quad \text{at } z = 0$$

where G_{Bj} equals the radionuclide concentration per unit weight of sediment on the river bed for j th sediment size fraction, and γ was assumed to be unity as previously defined.

Noting Equation (4), this equation may be expressed as

$$\frac{\partial G_j}{\partial z} = - \frac{G_{Bj} - G_j}{C_j \epsilon_{zj}} S_{Rj} \quad (17)$$

The finite element technique with the Galerkin weighted residual method was used to solve transport equations of sediments, dissolved radionuclides and particulate radionuclides previously described.

FINITE ELEMENT TECHNIQUE

High speed digital computers have enabled engineers to employ various numerical discretization techniques for approximating solutions to complex mathematical equations. The finite element method is one such technique.⁽¹⁷⁾ This method has recently gained popularity for solving both linear and nonlinear partial differential equations because of its increased solution accuracy, and ready accommodation to various boundary geometrics.^(10,17-20)

To apply the finite element method to a partial differential equation, an alternate but equivalent integral equation is imposed. The finite element method involves five basic steps:

1. Derive a partial differential equation
2. Divide the solution domain into a number of finite elements (polygons) defined by node points to form a computational network.
3. Approximate the dependent variables in terms of their unknown node point values within each element by selecting an approximate polynomial.
4. Develop finite element equations by minimizing error caused by the approximation.
5. Solve the resulting set of equations to get solutions at node points.

Since the governing equations of sediment and radionuclide transport (Equations 2, 11, and 15) have similar forms, the finite element technique is described here for a convection-diffusion equation of the general form:

$$L[\phi] = \frac{\partial \phi}{\partial t} + v \frac{\partial \phi}{\partial z} - \epsilon_z \frac{\partial^2 \phi}{\partial z^2} + \alpha \phi - \beta \quad (18)$$

The coefficients v , ϵ_z , α , β are defined to accommodate the specific form of the sediment and radionuclide transport equations.

Galerkin Weighted Residual Method

The governing differential equation can be recast as an integral problem employing the Galerkin method. This integral, referred to as the Galerkin functional, is formed by taking the product of $L[\phi]$ with some arbitrary set of weighting functions, W_j , which yields:

$$\chi = \int_R L[\phi] W_j dz \quad (19)$$

where

R = a domain of interest.

If ϕ is approximated by some polynomial:

$$\phi = \sum_{i=1} \phi_i W_i \quad (20)$$

then the quantity $L[\phi]$ represents the residual error. According to Galerkin's method, the approximating function W_i is chosen to be the same as the weighting function W_j . The Galerkin functional becomes an error distribution principle by which the nodal values ϕ_i can be determined so that the residual error over the domain R is small. This is achieved by imposing an orthogonality constraint so that $\chi = 0$.

By expanding the equation and integrating by parts, the reduced functional results:

$$\chi = \int_R \left[\frac{\partial \phi}{\partial t} W_j + v \frac{\partial \phi}{\partial z} W_j + \epsilon_z \frac{\partial \phi}{\partial z} \frac{\partial W_j}{\partial z} + \alpha \phi W_j - \beta W_j \right] dz \quad (21)$$

Note that Equation 21 was derived by assuming no net flux across the boundary. In cases where the boundary has positive or negative net flux across it, this flux was incorporated as a source or sink term.

The above integral can be partitioned so that:

$$\chi = \chi^e_1 + \chi^e_2 + \chi^e_3 + \dots + \chi^e_n = \sum_{i=1}^n \chi^e_i \quad (22)$$

where

n = the number of subdomains. This may be expanded to give:

$$\chi = \int_{z_1}^{z_2} [] dz + \int_{z_2}^{z_3} [] dz + \dots + \int_{z_{m-1}}^{z_m} [] dz \quad (23)$$

The contents in brackets in Equation 23 are the same as those in Equation 21. The integration of the Galerkin functional given above will yield a set of m ordinary differential equations in terms of ϕ_i .

Finite Element Equations

The system of equations for a typical finite element can be developed by using a single functional and particular polynomial approximation for ϕ . For simplicity, a linear approximation was chosen:

$$\phi = \bar{\phi} = a + bz \quad (24)$$

The following requirements are imposed:

$$\phi_i = a + bz_i \quad (25)$$

$$\phi_{i+1} = a + bz_{i+1} \quad (26)$$

The coefficients a and b are:

$$\begin{Bmatrix} a \\ b \end{Bmatrix} = \frac{1}{\Delta z} \begin{bmatrix} z_{i+1} & -z_i \\ -1 & 1 \end{bmatrix} \begin{Bmatrix} \phi_i \\ \phi_{i+1} \end{Bmatrix} \quad (27)$$

This allows an approximation to ϕ over the subdomain:

$$\phi(z,t) = [w_1, w_2] \begin{Bmatrix} \phi_i \\ \phi_{i+1} \end{Bmatrix} \quad (28)$$

The weighting functions are given by:

$$w_1 = \frac{1}{\Delta z} (z_{i+1} - z) \quad (29)$$

$$w_2 = \frac{1}{\Delta z} (z - z_i) \quad (30)$$

The individual terms in the functional are approximated by:

$$\frac{\partial \phi}{\partial t} = [w_1, w_2] \begin{Bmatrix} \dot{\phi}_i \\ \dot{\phi}_{i+1} \end{Bmatrix} \quad (31)$$

$$\frac{\partial \phi}{\partial z} = [-1/\Delta z, 1/\Delta z] \begin{Bmatrix} \phi_i \\ \phi_{i+1} \end{Bmatrix} \quad (32)$$

$$\frac{\partial W_i}{\partial z} = [-1/\Delta z, 1/\Delta z] \quad (33)$$

where

$\dot{\phi}$ represents the time derivative.

Substituting these quantities into the functional for a typical subdomain, a set of integral equations is obtained. The integrals are evaluated using the following formula:

$$\int_{z_i}^{z_{i+1}} W_1^\gamma W_2^\beta dz = \frac{\gamma! \beta! \Delta z}{(\gamma + \beta + 1)!} \quad (34)$$

The results for individual terms are given below.

1. Time Dependent Term:

$$\int_{z_i}^{z_{i+1}} \begin{bmatrix} W_1 W_1 & W_2 W_1 \\ W_1 W_2 & W_2 W_2 \end{bmatrix} dz = \frac{\Delta z}{6} \begin{bmatrix} 2 & 1 \\ 1 & 2 \end{bmatrix} \begin{Bmatrix} \dot{\phi}_i \\ \dot{\phi}_{i+1} \end{Bmatrix} \quad (35)$$

2. Advective Term:

$$\begin{aligned} \int_{z_i}^{z_{i+1}} \frac{1}{\Delta z} \begin{bmatrix} -(v_i W_1^2 + v_{i+1} W_1 W_2) & (v_i W_1^2 + v_{i+1} W_1 W_2) \\ -(v_i W_1 W_2 + v_{i+1} W_2^2) & (v_i W_1 W_2 + v_{i+1} W_2^2) \end{bmatrix} dz \begin{Bmatrix} \phi_i \\ \phi_{i+1} \end{Bmatrix} \\ = \frac{1}{6} \begin{bmatrix} -(2v_i + v_{i+1}) & (2v_i + v_{i+1}) \\ -(v_i + 2v_{i+1}) & (v_i + 2v_{i+1}) \end{bmatrix} \begin{Bmatrix} \phi_i \\ \phi_{i+1} \end{Bmatrix} \end{aligned} \quad (36)$$

3. Diffusion Term:

$$\frac{\epsilon_z}{\Delta z^2} \int_{z_i}^{z_{i+1}} \begin{bmatrix} 1 & -1 \\ -1 & 1 \end{bmatrix} \begin{Bmatrix} \phi_i \\ \phi_{i+1} \end{Bmatrix} dz = \frac{\epsilon_z}{\Delta z} \begin{bmatrix} 1 & -1 \\ -1 & 1 \end{bmatrix} \begin{Bmatrix} \phi_i \\ \phi_{i+1} \end{Bmatrix} \quad (37)$$

4. Decay Term:

$$\int_{z_i}^{z_{i+1}} \alpha \begin{bmatrix} w_1 w_1 & w_2 w_1 \\ w_1 w_2 & w_2 w_2 \end{bmatrix} \begin{Bmatrix} \phi_i \\ \phi_{i+1} \end{Bmatrix} dz = \frac{\alpha \Delta z}{6} \begin{bmatrix} 2 & 1 \\ 1 & 2 \end{bmatrix} \begin{Bmatrix} \phi_i \\ \phi_{i+1} \end{Bmatrix} \quad (38)$$

5. Source Term:

$$\int_{z_i}^{z_{i+1}} \beta \begin{Bmatrix} w_1 \\ w_2 \end{Bmatrix} dz = \beta \Delta z \begin{Bmatrix} 1/2 \\ 1/2 \end{Bmatrix} \quad (39)$$

Summing up and gathering terms give the matrix equation for a typical finite element:

$$[P] \mathbf{e}_j \begin{Bmatrix} \phi_i \\ \phi_{i+1} \end{Bmatrix} + [S] \mathbf{e}_j \begin{Bmatrix} \phi_i \\ \phi_{i+1} \end{Bmatrix} = \{R\} \mathbf{e}_j \quad (40)$$

where

$$[P] = \begin{bmatrix} 1/3 & 1/6 \\ 1/6 & 1/3 \end{bmatrix}, \{R\} = 1/2 \begin{Bmatrix} \beta \\ \beta \end{Bmatrix} \quad (41) \text{ and } (42)$$

$$[S] = \begin{bmatrix} (D - \frac{\bar{V}}{3\Delta z} + \frac{\alpha}{3}) & -(D - \frac{\bar{V}}{3\Delta z} - \frac{\alpha}{6}) \\ -(D + \frac{\tilde{V}}{3\Delta z} - \frac{\alpha}{6}) & (D + \frac{\tilde{V}}{3\Delta z} + \frac{\alpha}{3}) \end{bmatrix}, \quad (43)$$

$$D = \frac{\epsilon_z}{\Delta z^2}, \quad \bar{V} = (v_i + \frac{1}{2} v_{i+1}), \quad \tilde{V} = (\frac{1}{2} v_i + v_{i+1}) \quad (44)$$

The summation given in Equation (22) gives the rules for constructing the matrices for all finite elements; the matrices $[P]$ and $[S]$ become tridiagonal matrices. The final set of equations is a generalization of Equation (40).

Time-Dependent Solution

The final set of equations derived above can be expressed by:

$$[\tilde{P}] \frac{d\phi}{dt} + [\tilde{S}] \{\phi\} = \{\tilde{R}\} \quad (45)$$

where

$[\tilde{P}]$ matrix = a symmetric tridiagonal,

$[\tilde{S}]$ matrix = an unsymmetric tridiagonal

$\{\tilde{R}\}$ = a load vector.

This final system of equations is approximated by a Crank-Nicholson scheme: ⁽²¹⁾

$$[\tilde{P}] \frac{\{\phi\}^{n+1} - \{\phi\}^n}{\Delta t} = \frac{1}{2} [\tilde{S}] \{\phi\}^{n+1} + \{\phi\}^n + \{\tilde{R}\} \quad (46)$$

Solving for the $n+1$ value of ϕ , this expression can be rearranged to obtain:

$$[\tilde{P}] \{\phi\}^{n+1} = [\tilde{S}] \{\phi\}^n + \{\tilde{R}\} \quad (47)$$

where

the new coefficient matrices are:

$$[\bar{P}] = [\tilde{P}] + \frac{\Delta t}{2} [\tilde{S}] \quad (48)$$

$$[\bar{S}] = [\tilde{P}] - \frac{\Delta t}{2} [\tilde{S}] \quad (49)$$

$$\{\bar{R}\} = \Delta t \{\tilde{R}\} \quad (50)$$

This type of approximation is second order correct, unconditionally stable and easily solved by most available tridiagonal solution schemes.

A computer program has been written in FORTRAN IV to implement the model.

3. CODE DESCRIPTION

The computer program described in this section is designed to obtain solutions of the differential Equations (2), (11), and (15) with appropriate boundary and initial conditions. The program consists of subroutines which are managed by the executive routine, SERATRA.

COMPUTATION PROCEDURE

The procedure for solving the equations is as follows:

1. Read input data
2. Assign new river reach
3. Assign new time
4. Compute vertical velocity distribution and bed shear stress
5. Solve sediment transport equation to compute vertical distributions of sediment concentrations for
 - a. sand
 - b. silt
 - c. clay
6. Solve dissolved radionuclide transport equations to compute vertical distributions of dissolved contaminant
7. Solve particulate radionuclide transport equation to compute vertical distributions of concentrations of contaminant attached to
 - a. sand
 - b. silt
 - c. clay
8. Update river bed history
9. Repeat Steps 3 through 8 until all computation time is covered.
10. Repeat Steps 2 through 9 until all river reaches (longitudinal distance) are covered.
11. Print computer results.

EXECUTIVE PROGRAM AND SUBROUTINE DESCRIPTION

The computer code consists of an executive routine called SERATRA and 20 subroutines. A list of the primary duties served by each of these routines follows:

SERATRA Executive Routine

1. Reads data for program control
2. Calls subroutines for data input, problem set up and initialization, and program execution.
3. Performs minor computations such as presetting variables, converting input data to appropriate units, etc.

Subroutines

The subroutines called from the main program are:

INPUT
SHEAR
TRIB
AVERAGE
SEGMTX
NUCCON
TRANSP
PRINTS

Subroutine INPUT

General data input routine

- Reads input data from cards or teletype
- Reads restart file if required
- Converts portions of input data to appropriate quantities
- Calls subroutines

PROFIL
SHEAR

Subroutine PROFIL

Computes vertical distribution of flow discharge assuming the velocity distribution is logarithmic.

Subroutine SHEAR

Computes bed shear stress and shear velocity for a sediment laden flow.

Subroutine TRIB

Sets up input quantities of tributaries

- Computes vertical discharge distribution from vertically averaged value, assuming the velocity is logarithmically distributed.
- Computes vertical distributions of contaminants from vertically averaged values.

Subroutine AVERAGE

Calculates inflow concentrations of sediments and contaminants into a river reach by adding tributary input to the main river contributions to the reach.

Subroutine SEGMTX

Converts output of upstream reach to input of immediate downstream reach by taking into account possible different numbers of vertical computation elements for these two reaches.

Subroutine NUCCON

Obtains total concentration of a contaminant.

- Sums up concentrations of contaminant attached to sand, silt and clay in water.
- Sums up concentration of contaminant attached to bed sediment for all sediment size fractions.

Subroutine TRANSP

Solves concentrations of sediment, dissolved contaminant and particulate contaminant. Calls subroutines for all transport equations.

- Calls subroutine FALL
- Calls subroutine SAND
- Calls subroutine SILCLA

- Calls subroutine SEDIME
- Calls subroutine RDSOL
- Calls subroutine RDSSC
- Calls subroutine BEDHST
- Calls subroutine SETUP
- Calls subroutine COMB
- Calls subroutine TRISOL

The code spends the majority of the execution time for this routine.

Subroutine FALL

Calculates the contribution of sediment fall velocity to the sediment transport equations.

Subroutine SAND

Calculates the amount of sand deposition to the river bed or resuspension from the bed. Availability of sand in the river bed is also checked if resuspension occurs. The Toffaleti's sediment discharge formula is used for computation of flow capacity to carry the noncohesive sediment for an assigned river reach.

Subroutine SILCLA

- Calculates the amount of silt deposition or erosion.
- Calculates the amount of clay deposition or erosion.

Availability of silt and clay on the river bed is checked if erosion occurs. The Partheniades and Krone formulas are used to calculate the rates of cohesive sediment deposition or erosion.

Subroutine SEDIME

Calculates coefficients of convection, diffusion, decay and source terms in the sediment transport convection-diffusion equations. This will be performed for each sediment size fraction.

Subroutine RDSOL

Calculates coefficients of convection, diffusion, decay and source terms in the dissolved contaminant transport convection - diffusion equation.

Subroutine RDSSC

Calculates coefficients of convection, diffusion, decay and source terms in the particulate contaminant transport convection - diffusion equation. This will be performed for a particulate contaminant attached to sediment of each sediment size fraction.

Subroutine BEDHST

- Maintains and updates a record of bed history for river bed elevation, bottom sediment size ratio and concentration of particulate contaminants.
- Calls subroutine LEFT

Subroutine LEFT

- Computes the amount of sediment in the top river bed layer for each sediment size fraction.
- Computes the amount of particulate contaminant attached to sediment of each size fraction in the top river bed layer.

Subroutine SETUP

Sets up finite element matrices, $[P]$, $[S]$, and $\{R\}$ in Equation (40) for equations of

- Sediment transport
- Dissolved contaminant transport
- Particulate contaminant transport

Subroutine COMB

Adds two terms in Equation (47)

- Multiplies the unsymmetric band matrix $[S]$ to $\{\phi\}^n$
- Adds the result to the load vector $\{R\}$

Subroutine TRISOL

Solves a system of equations $[A] \{X\} = \{B\}$ for $\{X\}$. The algorithm employs the Gaussian elimination technique.

Subroutine PRINTS

This is the primary output call routine. The computer results are printed out by line printer and simultaneously stored to a output file for concentration contour plotting.

LISTS OF INPUT DATA AND COMPUTER OUTPUT

The following summarizes the input data used for the SERATRA code:

Sediment Simulation

1. Flow and Channel Characteristics for Each River Reach
 - Discharge
 - Flow depth and water surface elevation
 - Channel geometry (i.e., river width, cross-sectional shape)
 - Diffusion coefficients
2. Sediment Characteristics for Each Sediment Size Fraction for Each River Reach
 - Diameter, density and fall velocity of sediment
 - Critical shear stress and erodibility coefficient for erosion of cohesive sediment in the river bed
 - Critical shear stress for deposition of cohesive sediment
3. Initial and Boundary Conditions
 - Concentration of total (suspended and bed load) sediment for each size fraction at each vertical element for a river reach
 - Sediment weight fraction in the river bed at each vertical depth for a river reach

Radionuclide Simulation

1. Radionuclide Characteristics

- Radionuclide type
- Adsorption and/or desorption rate for each river reach

2. Initial and Boundary Conditions

- Dissolved radionuclide concentration at each vertical element for a river reach
- Concentration of radionuclide attached to sediment for each sediment size fraction at each vertical element for a river reach
- Concentration of radionuclide attached to sediment in the river bed at each vertical depth for a river reach

With the data input described above the model will simulate the sediment and radionuclide movement in a river. A summary of the computer output (simulation) is as follows:

Sediment Simulation

1. For Each River Reach at Assigned Time

- Vertical distribution of sediment (suspended and bed load) concentration for each sediment size fraction
- Vertical distribution of sediment size weight fraction ratio in the river bed
- River bed elevation (changes due to sediment deposition and/or scour)

2. Longitudinal Distribution of Sediment Load for Each Sediment Size Fraction at Assigned Time

Radionuclide Simulation

1. For Each River Reach at Assigned Time

- Vertical distribution of dissolved radionuclide concentration

- Vertical distribution of radionuclide concentration adsorbed by sediment for each sediment size fraction
 - Vertical distribution of total radionuclide concentration in motion
 - Vertical distribution of radionuclide concentration in the river bed sediment for each sediment size fraction
 - Vertical distribution of total radionuclide in the river bed
2. Longitudinal Distributions of Dissolved, Particulate (for each sediment size fraction) and Total Radionuclide Concentrations at Assigned Time

4. COMPUTER SIMULATION RESULTS

The Clinch River (Figure 1) was selected for testing and verification of the SERATRA code for several reasons:

1. Its size is fairly representative of rivers near nuclear power plants in the United States. In addition, its characteristics are being examined as part of the Nuclear Regulatory Commission's Liquid Pathway Generic Study.⁽¹⁾
2. Through operation of the Oak Ridge National Laboratory (ORNL), measurable radioactive materials are present via the White Oak Creek (a tributary). ORNL has also collected a considerable amount of field data which were used for the verification of the SERATRA code.
3. The river is regulated by two upstream dams (the Norris and Melton Hill Dams) in the Clinch River and a downstream dam (Watts Bar Dam) in the Tennessee River and has three major tributaries, thereby encompassing a fairly wide range of hydraulic conditions.

To calculate sediment and radionuclide transport, the SERATRA code was applied to the reach at River Mile 23.1 immediately downstream of the Melton Hill Dam to the mouth of the river joining the Tennessee River. Measurable radioactive materials enter the Clinch River through the White Oak Creek (20.8 Clinch River Miles). In this study, both continuous and instantaneous radionuclide releases were modeled.

CONTINUOUS RELEASE OF RADIONUCLIDES

Of the four major radioactive materials found in the Clinch River, ^{137}Cs is more easily adsorbed by suspended sediments than ^{60}Co and ^{106}Ru , which in turn are more susceptible to sorption than ^{90}Sr . The two extremes, ^{137}Cs and ^{90}Sr , were selected to examine radionuclide transport and interaction with river sediments.

Four continuous release cases were modeled. Parameters are given in Table 1. The entire river stretch was divided into 33 equally-spaced river reaches. Reach 1 corresponds to the upstream end and Reach 33 corresponds to the downstream end of the study area. Figure 2 shows variations of water surface and bed elevations of the Clinch River for a discharge of $117.8 \text{ m}^3/\text{sec}$. This figure also indicates locations of tributaries. Cases 1 and 2 consist of field data collected in June 1962 by ORNL.⁽⁸⁾ There was no flow reversal in the study reach of the Clinch River during the period of field data collection conducted by ORNL.

Case 1: Verification Test of Cesium-137

This case dealt with the transport of dissolved and particulate (those adsorbed by sediment) forms of ^{137}Cs which were continuously released into the Clinch River through the White Oak Creek. The ^{137}Cs introduced from the White Oak Creek to the Clinch River was diluted by clear water and suspended sediments in the Clinch River and two tributaries (Poplar Creek and Emory River) before it reached the Tennessee River. The analysis included interaction between ^{137}Cs and river sediments through adsorption of ^{137}Cs by suspended sediments of each size fraction and erosion and deposition of clean and contaminated sediments.

The river sediments consist of sand, silt and clay. Their particle sizes were assumed to be 0.125, 0.030, and 0.002 mm, respectively.^(5,6) Based on Clinch River studies^(2,8) and the sensitivity analysis on the adsorption rate performed in this study, ^{137}Cs adsorption rates for sand, silt and clay were selected to be 200, 500, and $1500 \text{ m}^3/\text{kg-day}$ respectively. Values of parameters related to deposition and resuspension of cohesive sediments are indicated in Table 1. The decay rate of ^{137}Cs was selected to be 7.32×10^{-10} per second (or a half life of 30 years). Because field data were available for calibration and verification of the simulation model, a 6-day period in June 1962⁽⁸⁾ was used for this case. Average river flow discharges of the White Oak Creek, Clinch River, Poplar Creek and Emory River during this period were set at 1.3, 117.8, 7.4 and $45.0 \text{ m}^3/\text{sec}$,

TABLE 1. Test Conditions for Continuous Release of Radionuclide

	Case 1	Case 2	Case 3	Case 4
Type of Radionuclide	Cesium-137	Strontium-90	Cesium-137	Cesium-137
Discharge of Clinch River (m ³ /sec)	117.8	117.8	117.8	226.7
of Whiteoak Creek	1.3	0.4	1.3	1.3
of Poplar Creek	7.4	7.4	7.4	7.4
of Emory River	45.0	45.0	45.0	45.0
River Sediment Size;				
sand (mm)	0.125	0.125	0.125	0.125
silt	0.030	0.030	0.030	0.030
clay	0.002	0.002	0.002	0.002
Critical Shear Stress for Erosion (kg/m ²)				
silt	0.070	0.070	0.070	0.070
clay	0.073	0.073	0.073	0.073
Erodibility Coefficient (kg/m ³)				
silt	0.15	0.15	0.15	0.15
clay	0.15	0.15	0.15	0.15
Critical Shear Stress for Deposition (kg/m ²)				
silt	0.005	0.005	0.020	0.005
clay	0.0025	0.0025	0.010	0.0025
Radionuclide Adsorption Rate: ($\frac{m^3}{kg-day}$)				
sand	200	2.0	200	200
silt	500	5.0	500	500
clay	1500	15.0	1500	1500
Vertical Diffusion Coefficient m ² /sec	0.0003	0.0003	0.0003	0.0003
Radionuclide Decay Rate (1/sec)	7.32×10^{-10}	7.87×10^{-10}	7.32×10^{-10}	7.32×10^{-10}
Concentration of Released Radionuclide at the White Oak Creek,				
dissolved form pCi/m ³	2.5×10^5	1.26×10^6	2.50×10^5	2.50×10^5
particulate form pCi/kg				
attached to sand	0	0	0	0
attached to silt	7.42×10^7	2.25×10^6	7.42×10^7	7.42×10^7
attached to clay	7.42×10^9	2.25×10^7	7.42×10^8	7.42×10^8

respectively. It was assumed that there was originally no suspended sediments and no radionuclides in the entire study reach, and that only suspended silt and clay (7.5 ppm and 0.5 ppm) were entering the river reach from the upstream boundary (Melton Hill Dam) and from the White Oak Creek. All the computer results shown for this case occurred on the sixth day.

Figure 3 shows simulation results of longitudinal distribution of sediment concentrations (sum of suspended and bed load sediments) for each sediment size fraction, and the total sediment concentration. Since there were no measured sediment data available for this case, field data obtained in August 1975 for a river discharge of $76.0 \text{ m}^3/\text{sec}$ were included in this figure for the gross comparison with the computer results. These field data were obtained from E. H. Lesesne of Tennessee Valley Authority, Knoxville, TN, November 1976. The figure indicates that, for the uppermost river reach, minor river bed scouring causes an increase in concentration of suspended sand from zero to approximately 0.06 ppm. However, sand resuspended from the river bed is redeposited approximately 5 km downstream of the Melton Hill Dam because of decreased river velocity at that point. Suspended silt concentration also increases slightly in the uppermost river reach, then gradually decreases, again because of the decreased velocity.

When the river depth abruptly increases and the flow velocity decreases, as occurs approximately 16 km downstream of the Melton Hill Dam, silt concentration is reduced through deposition. When the flow reaches approximately 28 km downstream almost all suspended silt is deposited to the river bed. The suspended clay does not change its concentration until the flow reaches almost the downstream end where the flow velocity is extremely low (approximately 0.052 m/sec). The variation of total sediment concentration (sum of these three components) reflects these changes. The comparison of calculated results with the field data shows that they are within the same range of sediment concentrations.

Figure 3 reveals that SERATRA is capable of predicting the sediment transport with deposition and resuspension of river sediment. However, since there are no sediment field data obtained under the same flow conditions as this verification case, additional detailed field data are needed to conduct more complete model verification.

Figures 4, 5, and 6 show computed longitudinal distributions of radionuclides in each river reach (sum of dissolved and particulate forms), and dissolved and particulate ^{137}Cs concentrations, respectively, together with field data obtained by Churchill et al.⁽⁸⁾ Simulation results of total ^{137}Cs concentrations agree very well with the field data as shown in Figure 4. Comparison of computed dissolved concentration with the measured data indicates some discrepancy (see Figure 5), but the field data for such a low concentration do not allow a definite comparison between the predicted and measured data. In the case of the particulate radionuclide, a much higher concentration of ^{137}Cs is associated with the suspended clay than with the silt, as shown in Figure 6. This is because clay has a much greater cation exchange capacity of attracting radioactive materials than silt.^(7,22) Silt, in turn, has a higher radionuclide concentration than sand, resulting from its greater exchange capacity as compared with sand's capacity.

Predicted particulate concentrations at River Miles 14.5 and 4.4 are 10.5 and 4.0 pCi/l, respectively. Field data⁽⁸⁾ indicate that particulate concentrations at these locations are 12 and 5 pCi/l, respectively, which corresponds to 92 and 83 percent, respectively, of the total ^{137}Cs concentrations (sum of particulate and dissolved ^{137}Cs) measured there. Hence, agreement between the simulation results and measured data is evident. These three figures also demonstrate the capability of the SERATRA code to calculate both particulate and dissolved radionuclides by taking into account interaction with river sediments.

Vertical distributions of sediment and radionuclide concentrations in the water and the river bed are presented in Figures 7 through 22. Vertical

distributions of suspended sediments (i.e., sand, silt and clay) in Reach 1 (River Mile 22.4 to 23.1) are shown in Figure 7, with the vertical change in the percentage of each sediment size fraction to the total river bed sediment (by weight) shown in Figure 8. As stated above, it was originally assumed that no suspended sediments (sand, silt and clay) were in Reach 1 and that only suspended silt and clay were entering Reach 1 from the upstream boundary. Consequently all the suspended sand in Reach 1 was eroded from the river bed. It is also noticed that suspended sand has a higher concentration near the river bed, while silt and clay show almost uniform vertical distributions. This is due to the much smaller fall velocities of silt and clay. In the river bed the original sediment consisted of 35, 53, and 12 percent of sand, silt and clay, respectively. Although sand was resuspended from the top layer of the river bed, the amount of sand eroded from the river bed for 6 days is so small that no measurable change in sediment size fraction in the river bed occurs.

Figures 9 through 13 present simulation results of sediments and radionuclides in the water and the river bed for Reach 4 (River Miles 21.0 to 20.3) where the White Oak Creek joins the Clinch River. Suspended sand concentration in this reach (Figure 9) is less than that in Reach 1 (Figure 7), thus denoting the deposition of both clean and contaminated suspended sand to the river bed. This is because of the small reduction of flow velocity in Reach 4. However, this small reduction of velocity does not change the concentration of finer cohesive sediments. Vertical distributions of particulate and dissolved ^{137}Cs shown in Figures 10 and 11 indicate a dilution of radionuclide concentrations by both clean sediments and water in the Clinch River. River bed conditions reflect the behavior of sediments and radionuclides moving above it. Because some of the suspended sand was deposited on the river bed, only the sand constitutes the uppermost river bed layer which is made by the deposition

of suspended sediments, as shown in Figure 12. In the lower bed layers, the ratio of sand, silt and clay does not change from the original ratio of 35, 53 and 12 percent, respectively. The concentration of ^{137}Cs in the river bed sediments is given in Figure 13. Since it was assumed that there were originally no radionuclides in the river, ^{137}Cs associated with sand appears in the top layer is a result of deposition of contaminated suspended sand. No radionuclides associated with silt and clay appear in the river bed of Reach 4.

Figures 14, 15 and 16 illustrate vertical distributions of sediments and radioactive materials in the water for Reach 12 (River Miles 15.4 to 14.7). Practically all suspended sand is already deposited on the river bed before reaching Reach 12. Suspended silt reduced its concentration slightly. However, the adsorption of the radionuclide by suspended sediments causes the dissolved concentration of ^{137}Cs to decrease.

Simulation results for Reach 27 (River Miles 4.9 to 4.2) appear in Figures 17 through 20. In this reach, significant reduction of suspended silt concentration occurred (Figure 17). Furthermore, this silt deposition causes the increase of radionuclides associated with silt in the river bed (Figure 20).

Figures 21 and 22 describe vertical variations of suspended sediments and ^{137}Cs in the river bed for Reach 33 (River Mile 0.7 to 0). In this reach, significant deposition of clay occurs. Concentration of suspended silt is also practically zero. These phenomena also reflect the variation of radionuclide in the river bed (Figure 22).

Case 2: Verification Test of Strontium-90

As discussed above ^{137}Cs is easily adsorbed by suspended sediments, whereas only a small portion of ^{90}Sr is transported by suspended sediments. In Case 2 continuous release of ^{90}Sr to the Clinch River was examined. The computer results were then compared with the field data collected by

Churchill et al.⁽⁸⁾ Like Case 1, it was also assumed that initially there was no ^{90}Sr in the entire river reach under investigation. Detailed test conditions for Case 2 are listed in Table 1.

The computer results shown for this case were also obtained on the sixth simulation day. Figures 23, 24 and 25 show predicted longitudinal distributions of radionuclide amount in each river reach and dissolved and particulate ^{90}Sr concentrations, respectively, together with measured data.⁽⁸⁾ Comparisons of simulation results of total and dissolved ^{90}Sr concentrations with the field data reveal excellent agreement. For the particulate radionuclide, the highest concentration of ^{90}Sr is associated with the suspended clay, similar to Case 1. However, unlike ^{137}Cs , particulate ^{90}Sr constitutes an only minor portion of the total ^{90}Sr being transported in the river. Predicted particulate concentrations at River Miles 14.5 and 4.4 are 0.19 and 0.10 pCi/l, respectively. The measured data indicate that the particulate concentrations at these locations are 0.22 and 0.21 pCi/l, which corresponds to 5.6 and 7.0 percent of the total ^{90}Sr concentrations measured there. Since the concentrations of particulate ^{90}Sr are very small, as compared to dissolved concentrations, and are close to the accuracy of the field measurements (± 0.07 pCi/l), it is regarded that agreement between the simulation results and the field data is reasonably good.

Some vertical distributions of ^{90}Sr are depicted in Figures 26 through 30. Although individual values differ greatly from those computed for Case 1, the same trend for radionuclide concentrations prevails.

The verification tests of Case 1 (^{137}Cs) and Case 2 (^{90}Sr) demonstrate that generally there is a good agreement between the model prediction and the field data and that the SERATRA code is capable of calculating both particulate and dissolved radionuclides by taking into account the interaction with the river sediments. However, more detailed measured data are necessary to rigorously verify the SERATRA code.

Case 3: Cesium-137 with Greater Critical Shear Stress for Sediment Deposition

In order to investigate sensitivity of the critical shear stress of cohesive sediment deposition to sediment and radionuclide transport, this value was increased by a factor of 4, as compared to Cases 1 and 2. Simulation was conducted for 6 days, and the following results were obtained on the sixth day. Figure 31 gives the computed sediment concentrations for a river discharge of $117.8 \text{ m}^3/\text{sec}$ with the field data obtained in August 1975 for a river discharge of $70.0 \text{ m}^3/\text{sec}$. As compared to Figure 3, suspended sediments have similar concentrations in the upper river reach to those of Case 1, but suspended silt and clay deposited much earlier than Case 1. This, in turn, affects the radionuclide transport, as shown in Figures 32 through 34. There were two effects of the early sediment deposition on radionuclide transport: 1) Because of the large amount of deposition of contaminated sediments, less ^{137}Cs is transported downstream; 2) because the suspended sediment concentrations are less in the downstream area, dissolved radionuclides were not adsorbed by the suspended sediment as much as for Case 1. The net result of these two opposite effects on radionuclide transport for this case is the reduction of radionuclide transport in the downstream area.

Case 4: Cesium-137 with a Higher River Discharge

In order to investigate effects of river flow discharge on sediment and radionuclide transport, the flow rate was increased from $117.8 \text{ m}^3/\text{sec}$ to $226.7 \text{ m}^3/\text{sec}$. The simulation was conducted for 3 days. The following results are those obtained on the third day. In Case 4, ^{137}Cs was initially distributed in the river bed according to field data.⁽⁸⁾ Figures 35 through 38 are the computer results of sediment and radionuclide concentrations.

These results indicate that suspended sediment concentrations for Case 4 are significantly higher than previous cases. Accordingly, particulate cesium concentration is lower because of the dilution of

contaminated sediments by clear sediments eroded from the bed of the river reach upstream of the White Oak Creek mouth. Dissolved radionuclide concentration decreases very rapidly because of more dilution by the clean Clinch River water, and more adsorption due to higher sediment concentrations in the river.

Figures 39 through 42 present the vertical distributions of sediments and radionuclides for Reach 4. These figures illustrate the severe erosion of the river bed. Furthermore, silt and clay in the top river bed layer have been exhausted by the resuspension, leaving only sand. Variations of vertical distribution of ^{137}Cs in the river bed are shown in Figure 42.

INSTANTANEOUS RELEASE OF RADIONUCLIDES

In addition to the continuous release tests of radionuclides, four instantaneous radionuclide release cases (Cases 5 through 8) were tested to further identify effects of river sediments on the migration of radioactive materials and to examine the applicability of the SERATRA code to instantaneous release cases. For these cases, it was assumed that radionuclide transport modeling was performed for instantaneous releases of dissolved radionuclides introduced to the Clinch River through the White Oak Creek. Table 2 lists test conditions of these cases.

A verification test of instantaneous radionuclide (^{198}Au) release was made for Case 5. Cases 6, 7 and 8 were studied using hypothetical radionuclide releases to identify how radionuclide adsorption by sediments affects radionuclide transport. Case 6 involved dissolved ^{137}Cs without adsorption mechanics; Case 7, ^{137}Cs with adsorption; and Case 8, strontium with minor adsorption. For all three cases 1000 curies of dissolved radionuclides were assumed to be instantaneously released to the Clinch River from the White Oak Creek mouth. Flow discharge was assumed to be $96.3 \text{ m}^3/\text{sec}$ for these three cases. Instead of simulating true instantaneous releases of radionuclides at the White Oak Creek mouth, it was assumed that radionuclides were released for 30 minutes in Case 5 and 60 minutes in Cases 6,

TABLE 2. Test Condition for Instantaneous Release of Radionuclides

	Case 5	Case 6	Case 7	Case 8
Type of Radionuclide	Gold-198	Cesium-137	Cesium-137	Strontium-90
Discharge (m ³ /sec)				
Clinch River	226.7	96.3	96.3	96.3
Whiteoak Creek	0.2	0.4	0.4	0.4
Poplar Creek	7.4	7.4	7.4	7.4
Emory River	45.0	45.0	45.0	45.0
River Sediment Size (mm)				
sand	0.125	0.125	0.125	0.125
silt	0.030	0.030	0.030	0.030
clay	0.002	0.002	0.002	0.002
Critical Shear Stress for Erosion (kg/m ²)				
silt	0.070	0.070	0.070	0.070
clay	0.073	0.073	0.073	0.073
Erodibility Coefficient (kg/m ³)				
silt	0.15	0.15	0.15	0.15
clay	0.15	0.15	0.15	0.15
Critical Shear Stress for Deposition (kg/m ²)				
silt	0.005	0.005	0.005	0.005
clay	0.0025	0.0025	0.0025	0.0025
Radioactive Adsorption Rate ($\frac{m^3}{kg-day}$)				
sand	0	0	200	2.0
silt	0	0	500	5.0
clay	0	0	1500	15.0
Vertical Diffusion Coefficient (m ² /sec)	0.0003	0.0003	0.0003	0.0003
Radionuclide Decay Rate (1/sec)	2.97×10^{-6}	7.32×10^{-10}	7.32×10^{-10}	7.87×10^{-10}
Total Radionuclide Amount Released at the White Oak Creek (curies),				
dissolved form	7.5	1000	1000	1000
particulate form				
attached to sand	0	0	0	0
attached to silt	0	0	0	0
attached to clay	0	0	0	0

7 and 8. Simulations were conducted for 2 days in Case 5 and 6 days in Cases 6, 7 and 8. The results presented here are those that occurred at 48 hours.

Case 5: Verification Test of Gold-198

ORNL conducted a radionuclide dispersion field test on August 30-31, 1961, by discharging 7.5 curies of ^{198}Au tracer solution (gold chloride dissolved in a solution of hydrochloric and nitric acids) to the Clinch River from the mouth of the White Oak Creek.⁽⁵⁾ These data were used here for verification of the SERATRA code for instantaneous release. Since only the total concentrations of ^{198}Au were measured without dividing them into dissolved and particulate ^{198}Au concentrations, all ^{198}Au was treated as a dissolved form by setting the adsorption rate at zero. In this case, two computations were made by dividing the entire study area into (a) 33 equal-distance river reaches (Case 5.1), and (b) 11 equal-distance river reaches (Case 5.2).

Figures 43 and 44 show longitudinal distributions of dissolved concentrations of ^{198}Au at different times for Cases 5.1 and 5.2, respectively. These figures signify that Case 5.2 (with 11 reaches) dilutes radioactive material more rapidly than Case 5.1. Variations of ^{198}Au with time at River Miles 17.1, 14.5 and 4.4 are provided in Figures 45, 46, and 47, respectively. These figures also contain measured field data. Comparison of computer results with the field data is poor. The computer results reveal much larger longitudinal dispersion than the field data. An attempt was made to estimate the longitudinal dispersion coefficient for the field study case. However, the measured data are not complete enough to provide an accurate estimation. Results for Case 5.1 are closer to the measured data than those for Case 5.2. This indicates that the model may use smaller longitudinal dispersion coefficients, which can be changed by adjusting vertical distribution of longitudinal velocity, number of river reaches, and computation time steps.

Case 6: Dissolved Cesium-137 Without Adsorption

In order to establish a baseline case to which other cases can be compared, the transport of dissolved ^{137}Cs without adsorption by suspended sediments was simulated. Simulation results at 42 hours are shown in Figures 48 and 49 which present longitudinal distributions of total radionuclide in curies and dissolved radionuclide concentration in picocuries per liter (pCi/l). Since adsorption mechanics were not included in this case, the concentration of total radionuclide is equal to that of dissolved radionuclide. The radionuclide has its peak at approximately 20 km downstream at the Melton Hill Dam. These results were compared with those of Cases 7 and 8 to identify the effects of radionuclide adsorption by suspended sediments on the migration of radioactive materials.

Case 7: Cesium-137 with Adsorption

Sediment concentrations for Case 7 appear in Figure 50. Since the river flow for Cases 6, 7 and 8 is smaller than that of Case 1, the sediment deposition starts to occur around 15 km downstream from the Melton Hill Dam. This is much earlier than that of Case 1. Since this case includes the adsorption mechanics, suspended sediments adsorb some of the dissolved radionuclide. Comparison of Figure 48 and Figure 51 shows that the longitudinal distribution of total radionuclide which appeared in the upper half of the study reach (less than 20 km downstream from the Melton Hill Dam) is almost identical to that of Case 6. However, in the downstream half of the river, the total amount of ^{137}Cs for Case 7 is approximately 50 percent of Case 6. For example, at 15 km downstream of the Melton Hill Dam, the total concentration of ^{137}Cs is 2.55×10^7 pCi/m³, which is identical to Case 6. At 18.5 km downstream of the dam, the total concentration is 4.07×10^7 pCi/m³, which is slightly smaller than the concentrations of Case 6 (4.29×10^7 pCi/m³). However, at 25 km downstream from the dam, it becomes 1.97×10^7 pCi/m³, which is approximately 50 percent of Case 6 (3.82×10^7 pCi/m³). This reduction is a result of deposition of the adsorbed radionuclides to an area with slow velocity.

The predicted results shown in Figures 51 through 53 indicate that a large portion of the dissolved radionuclides are adsorbed by suspended sediments. Clay particles contain the highest ^{137}Cs concentration, approximately three times higher than those of silt particles. Sand particles contain only a trace of radionuclides. From Figures 50 through 53, the following estimation was made: at 15 km downstream from the Melton Hill Dam, dissolved ^{137}Cs concentration is $3.87 \times 10^5 \text{ pCi/m}^3$ while total particulate concentration (sum of concentrations of particulate ^{137}Cs attached to sand, silt and clay) is $2.51 \times 10^7 \text{ pCi/m}^3$. This implies that 98.4 percent of ^{137}Cs is transported as a particulate form and only 1.6 percent is in a dissolved form. Among the particulate ^{137}Cs , those attached to silt constitute 83.3 percent ($2.09 \times 10^7 \text{ pCi/m}^3$) of the total particulate concentration ($2.51 \times 10^7 \text{ pCi/m}^3$). Those associated with clay constitute 16.7 percent ($4.20 \times 10^6 \text{ pCi/m}^3$). Sand contains only a trace of the radionuclide.

At 18.5 km downstream of the dam, dissolved and total particulate concentrations are 2.81×10^5 and $3.79 \times 10^7 \text{ pCi/m}^3$, respectively. In this case the dissolved radionuclides constitute 6.9 percent and the particulate form 93.1 percent of the total radionuclide concentration. The small reduction of particulate radionuclides is due to the deposition of some contaminated sediments. Ratios of the amount of each particulate radionuclide associated with sand, silt and clay to the total amount of particulate radionuclides are almost the same as those of the 15 km case. When the radionuclides are transported to 20 km downstream of the dam, the dissolved and total particulate concentrations become 1.52×10^7 and $4.50 \times 10^6 \text{ pCi/m}^3$, respectively, i.e., dissolved radionuclide constitute 77.2 percent, while particulate radionuclides are only 22.8 percent of the total radionuclide concentration. Moreover, particulate ^{137}Cs attached to silt as well as to sand is negligibly small and all particulate radionuclides found there are those attached to clay. This is because almost all the contaminated silt as well as clean silt is deposited on the river bed and there is almost no silt being transported.

Figures 54 through 56 show vertical variations of suspended sediments and the radionuclides at River Mile 7.4 at 42 hours. These figures indicate

that only clay particles are suspended in water. This is reflected by the 1.75:1 ratio of dissolved to particulate radionuclides.

Case 8: Strontium-90 with Minor Adsorption

Figures 57 through 59 show predicted results of ^{90}Sr distributions. Comparison of this case with Cases 6 and 7 reveals that longitudinal distributions of total radionuclide amount and dissolved concentrations of ^{90}Sr are almost identical to those of Case 6 (^{137}Cs without adsorption) except for the very small reduction in the total and dissolved radionuclide amounts for ^{90}Sr . This is due to the fact that ^{90}Sr has a very small adsorption rate as compared to Case 7. The particulate form of ^{90}Sr (Figure 59) has a much smaller concentration than Case 7, reflecting the significantly different adsorption rates for these two cases. For example, at 15 km downstream of the Melton Hill Dam, the total concentration of radionuclides is 2.55×10^7 pCi/m³. Concentrations of dissolved and total particulate radionuclides are 2.44×10^7 and 1.14×10^6 pCi/m³, respectively, i.e., 95.5 percent of the radionuclides are in the form of the dissolved ^{90}Sr while only 4.5 percent of ^{90}Sr is attached to sediment.

Among the particulate ^{90}Sr those attached to silt constitute 83 percent, while those associated with clay are 17.0 percent. Sand has only a trace of attached radionuclides. Comparison of Case 8 with Case 7 indicates that sorbed partitioning between the dissolved and particulate radionuclides is significantly different for ^{90}Sr (Case 8) and ^{137}Cs (Case 7). The ratio of each particulate radionuclide attached to sand, silt and clay to the total particulate radionuclides is almost identical between Case 8 (^{90}Sr) and Case 7 (^{137}Cs). At 25 km downstream of the dam, 99.7 percent of radionuclides was in the dissolved form and only 0.3 percent was associated with sediments, which were almost all clay particles, reflecting the deposition of suspended silt on the river bed.

5. CONCLUSIONS AND RECOMMENDATIONS

The finite element sediment and contaminant transport model, SERATRA was modified and applied to the Clinch River in Tennessee to solve time-dependent, longitudinal and vertical distributions of sediments and radionuclides and their chemical and physical interactions. The modeling procedure involved simulating the movements of sediment for each sediment size fractions. These results were input to models of dissolved radionuclide and particulate radioactive materials attached to river sediments for each sediment fraction in order to observe the interaction between sediment and radionuclide movements. Finally, changes in river bed conditions were recorded, including river bottom elevation change, ratio of sand, silt and clay sediment and distribution of radionuclide concentration in the river bed.

Mathematical simulation tests were conducted for four continuous releases and four instantaneous releases of ^{137}Cs , ^{90}Sr and ^{198}Au . The main conclusions drawn from this investigation may be summarized as follows:

1. Results of verification tests on the continuous release of ^{137}Cs and ^{90}Sr (Cases 1 and 2) indicate that agreement between the predicted values and measured data for longitudinal distributions of total radionuclides are excellent. Also, dissolved and particulate radionuclide concentrations show reasonable to excellent agreement between computer and measured values. For ^{137}Cs (Case 1), the mathematical model predicted that 96.0 percent is attached to sediments at River Mile 14.4, and is thus subject to deposition in the downstream area of reduced river velocity. On the other hand, for ^{90}Sr (Case 2), 95.7 percent is transported as dissolved phase, and only 4.3 percent is associated with river sediments. Among the particulate radionuclides, the study shows that those attached to clay have the highest radionuclide concentration and those associated with sand the minimum. However, the amount of radionuclides (^{137}Cs and ^{90}Sr) attached to silt is the greatest because

of highest concentration of suspended silt in the river. The model predicted sediment concentrations for each sediment size fraction, i.e., sand, silt and clay. The range of sediment concentrations predicted agrees fairly well with the available field data. Vertical distributions of sediment and radionuclides in water and in the river bed appear reasonable. However, there are not enough field data to examine the predicted values.

2. Sensitivity analysis was conducted by changing the critical shear stress of cohesive sediment deposition (Case 3) and river discharge (Case 4). It was found that sediment transport is very sensitive to the critical shear stress, and selection of this value significantly influences the behavior of radionuclide movement through changing the deposition rate of the contaminated sediment. The change in river discharge (Case 4) indicates that suspended sediment concentrations are significantly higher than those of lower discharges (Cases 1, 2 and 3). Accordingly, particulate ^{137}Cs concentration of Case 4 is lower because of the dilution of contaminated sediments by clean sediments eroded from the bed of the river reach upstream of the White Oak Creek. However, it is anticipated that if the river discharge is higher than that of Case 4, then the river downstream of the White Oak Creek may be eroded, resulting in the resuspension of contaminated river bed sediments. Dissolved radionuclide concentration decreases very rapidly because of more dilution by the clean Clinch River water, and more adsorption due to higher sediment concentrations in the river.
3. For the instantaneous release of ^{198}Au (Case 5), predicted results and field data are poor. The computer model predicted more radionuclide dilution than measured in field because of the large difference between the longitudinal dispersion coefficients of the model and prototype. The reduction of longitudinal dispersion in the model can be made by adjusting vertical distribution of longitudinal velocity, number of river reaches, and computer time steps.

4. Comparison of Cases 6, 7 and 8 indicate the very important aspect of suspended sediments on radionuclide transport. For all three hypothetical cases, 1000 curies of dissolved nuclides, i.e., ^{137}Cs without adsorption (Case 6), ^{137}Cs with adsorption (Case 7), and ^{90}Sr with minor adsorption (Case 8), were released to the Clinch River. The model predicted that for Case 7, dissolved and particulate radionuclides constitute 6.9 and 93.1 percent, respectively, at approximately 15 km downstream from the released point. Among the particulate ^{137}Cs , those attached to silt and clay constitute 83.3 and 16.7 percent, respectively, although the radionuclide concentration attached to clay is the highest. Sand contains only a trace of the radionuclide. The total amount of radionuclides that appear in the lower half of the study area for Case 7 was approximately 50 percent of Case 6. This is because of the deposition of contaminated sediments in this slow moving area. On the other hand, Case 8 shows 97.0 and 3.0 percent attributed to dissolved and particulate radionuclides, respectively. Distribution of total radionuclides for this case is almost identical to Case 6 because of the very small adsorption rate for ^{90}Sr .

Examination of above eight cases clearly indicates that suspended sediments have significant effects on radionuclide transport through radionuclide adsorption, deposition, and resuspension of contaminated sediments. Hence, in order to obtain realistic predictions of radionuclide migration, interaction between radionuclides and river sediments must be taken into account. The present study demonstrates that the SERATRA code is capable of predicting the transport of both particulate and dissolved radionuclides by taking into account the radionuclide-sediment interaction. From the results of this study, it is judged that the SERATRA code is partially verified against field data. However, it became evident that detailed field data are not available for complete verification of the model. In particular, radionuclide distributions on the various sediment size fractions do not exist. Accordingly, complete field verification of the mathematical model is recommended before using it to accurately predict the migration of radionuclides in riverine environments.

FIGURE 1. Map of the Clinch River.

FIGURE 2. Variations of Water Surface and Bed Elevations of the Clinch River:
Discharge at 117.8 m³/sec

FIGURE 3. Longitudinal Distributions of Suspended Sediment, Case 1

FIGURE 4. Longitudinal Distribution of Total Amount of Cesium-137, Case 1

FIGURE 5. Longitudinal Distribution of Dissolved Cesium-137, Case 1

FIGURE 6. Longitudinal Distributions of Cesium-137 Adsorbed by Suspended Sediment, Case 1

FIGURE 7. Vertical Distributions of Suspended Sediment Concentrations for Reach 1 (River Miles 23.1 to 22.4), Case 1

FIGURE 8. Vertical Distribution of Sediment Fraction Ratio in the River Bed for Reach 1, Case 1

FIGURE 9. Vertical Distributions of Suspended Sediment Concentrations for Reach 4 (River Miles 21.0 to 20.7), Case 1

FIGURE 10. Vertical Distributions of Cesium-137 Adsorbed by Suspended Sediment for Reach 4, Case 1

FIGURE 11. Vertical Distributions of Dissolved and Total Cesium-137 for Reach 4, Case 1

FIGURE 12. Vertical Distributions of Sediment Size Fraction Ratio in the River Bed for Reach 4, Case 1

FIGURE 13. Vertical Distributions of Cesium-137 in the River Bed for Reach 4, Case 1

FIGURE 14. Vertical Distributions of Suspended Sediment Concentrations for Reach 12 (River Miles 15.4 to 14.7), Case 1

FIGURE 15. Vertical Distributions of Cesium-137 Adsorbed by Suspended Sediment for Reach 12, Case 1

FIGURE 16. Vertical Distributions of Dissolved and Total Cesium-137 for Reach 12, Case 1

FIGURE 17. Vertical Distributions of Suspended Sediment Concentrations for Reach 27 (River Miles 4.9 to 4.2), Case 1

FIGURE 18 Vertical Distributions of Cesium-137 Adsorbed by Suspended Sediment for Reach 27, Case 1

FIGURE 19. Vertical Distributions of Dissolved and Total Cesium-137 for Reach 27, Case 1

FIGURE 20. Vertical Distributions of Sediment Size Fraction Ratio in the River Bed for Reach 27, Case 1

FIGURE 21. Vertical Distributions of Suspended Sediment Concentrations for Reach 33 (River Miles 0.7 to 0), Case 1

FIGURE 22. Vertical Distributions of Cesium-137 in the River Bed for Reach 33, Case 1

FIGURE 23. Longitudinal Distribution of Total Amount of Strontium-90, Case 2

FIGURE 24. Longitudinal Distribution of Dissolved Strontium-90, Case 2

FIGURE 25. Longitudinal Distributions of Strontium-90 Adsorbed by Suspended Sediment, Case 2

FIGURE 26. Vertical Distributions of Strontium-90 Adsorbed by Suspended Sediment for Reach 12, Case 2

FIGURE 27. Vertical Distributions of Dissolved and Total Strontium-90 for Reach 12, Case 2

FIGURE 28. Vertical Distributions of Strontium-90 Adsorbed by Suspended Sediment for Reach 27, Case 2

FIGURE 29. Vertical Distributions of Dissolved and Total Strontium-90 for Reach 27, Case 2

FIGURE 30. Vertical Distributions of Strontium-90 in the River Bed for Reach 27, Case 2

FIGURE 31. Longitudinal Distributions of Suspended Sediment, Case 3

FIGURE 32. Longitudinal Distribution of Total Amount of Cesium-137, Case 3

FIGURE 33. Longitudinal Distribution of Dissolved Cesium-137, Case 3

FIGURE 34. Longitudinal Distributions of Cesium-137 Adsorbed by Suspended Sediment, Case 3

FIGURE 35. Longitudinal Distributions of Suspended Sediment, Case 4

FIGURE 36. Longitudinal Distribution of Total Amount of Cesium-137, Case 4

FIGURE 37. Longitudinal Distribution of Dissolved Cesium-137, Case 4

FIGURE 38. Longitudinal Distributions of Cesium-137 Adsorbed by Suspended Sediment, Case 4

FIGURE 39. Vertical Distributions of Suspended Sediment Concentrations for Reach 4, Case 4

FIGURE 40. Vertical Distributions of Cesium-137 Adsorbed by Suspended Sediment for Reach 4, Case 4

FIGURE 41. Vertical Distributions of Sediment Fraction Ratio in the River Bed for Reach 4, Case 4

FIGURE 42. Vertical Distributions of Cesium-137 in the River Bed for Reach 4, Case 4

FIGURE 43. Longitudinal Distributions of Dissolved Gold-198, Case 5.1

FIGURE 44. Longitudinal Distributions of Dissolved Gold-198, Case 5.2

FIGURE 45. Comparison of Computer Results of Cases 5.1 and 5.2 with the Field Data at River Mile 17.1

FIGURE 46. Comparison of Computer Results of Cases 5.1 and 5.2 with the Field Data at River Mile 14.5

FIGURE 47. Comparison of Computer Results of Cases 5.1 and 5.2 with the Field Data at River Mile 4.4

FIGURE 48. Longitudinal Distribution of Total Amount of Cesium-137, Case 6

FIGURE 49. Longitudinal Distribution of Dissolved Cesium-137, Case 6

FIGURE 50. Longitudinal Distributions of Suspended Sediment Concentrations, Case 7

FIGURE 51. Longitudinal Distribution of Total Amount of Cesium-137

FIGURE 52. Longitudinal Distributions of Dissolved Cesium-137, Case 7

FIGURE 53. Longitudinal Distributions of Cesium-137 Adsorbed by Suspended Sediment, Case 7

FIGURE 54. Vertical Distributions of Suspended Sediment Concentrations at River Mile 7.4, Case 7

FIGURE 55. Vertical Distributions of Cesium-137 Adsorbed by Suspended Sediment at River Mile 7.4, Case 7

FIGURE 56. Vertical Distributions of Dissolved and Total Cesium-137 at River Mile 7.4, Case 7

FIGURE 57. Longitudinal Distributions of Total Amount of Strontium-90, Case 8

FIGURE 58. Longitudinal Distributions of Dissolved Strontium-90, Case 8

FIGURE 59. Longitudinal Distributions of Strontium-90 Adsorbed by Suspended Sediment, Case 8

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