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User's Manual for LPGS

A Computer Program for Calculating Radiation Exposure Resulting from Accidental Radioactive Releases to the Hydrosphere

Prepared by J. E. White, K. F. Eckerman

Oak Ridge National Laboratory

Prepared for
U.S. Nuclear Regulatory Commission

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Technical Data Mangement Center
Engineering Physics Division

USER'S MANUAL FOR LPGS: A COMPUTER PROGRAM FOR CALCULATING RADIATION EXPOSURE RESULTING FROM ACCIDENTAL RADIOACTIVE RELEASES TO THE HYDROSPHERE

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PREFACE

The LPGS computer program was developed to examine the radiological consequences of accidental releases of radionuclides to the hyrdosphere at light water reactor (LWR) sites. In the past, accidental releases to the hydrosphere had received limited attention because of the focus on airborne releases at LWR accident sites. In the early 1970s, the nuclear industry proposed the development of offshore floating nuclear power plants (FNPs). With that proposal, attention turned towards the consequences of releases to the hydrosphere and the observation that a core melt accident at an FNP might represent a risk substantially different from that deemed acceptable at land-based plants (LBPs). The need for an analysis tool to evaluate the potential difference in the consequences resulting from accidental releases from FNPs and LBPs led to the code development.

The original undocumented computer code evolved out of the Liquid Pathway Generic Study (LPGS) performed by staff members of the Nuclear Regulatory Commission (NRC) and reported in NUREG-0440, Impacts of Accidental Radioactive Releases to the Hydrosphere from Floating and Land-Based Nuclear Power Plants, by D. L. Schreiber, H. Berkson, G. L. Chipman, Jr., R. B. Codell, K. F. Eckerman, O. D. T. Lynch, Jr., A. R. Marchese, and P. F. Riehm (February 1978). Eckerman, who was responsible for the radiological assessment, designed, developed, and applied the LPGS calculational system. Development of the hydrologic transport models was largely the work of R. B. Codell (NRC/NRR/Hydrology-Meteorology Branch). J. E. White, of the NRC-sponsored Technical Data Management Center (TDMC), assumed the task of documenting the work, which resulted in giving the "collection of routines" a better sense of unification and completeness.

The main purpose of this manual is to provide a user's guide to the preparation of input for LPGS and to make available in one document essential information for its understanding and use. A description of the hydrologic models, published in NUREG-0440, excerpted from the NRC report and revised to include only the material implemented in the current computer program is included as appendices with the permission of the Technical Information and Document Control Division (TIDC), Office of Administration (ADM), U. S. Nuclear Regulatory Commission. Background information on the use of the original LPGS code development is referenced.

We acknowledge with deep appreciation the technical guidance of Sarbeswar Acharya (NRR/NRC) and the encouragement and advice of the contract monitor, Myrna Steel (TIDC/ADM/NRC), throughout the LPGS revision and documentation process. Since documentation, as well as code development, is subject to change following critical examination and usage, we solicit feedback from the user community.

We are also pleased to acknowledge the work of Ms. Alice F. Rice in the preparation of this document for publication, a task which was complicated by the necessity of working with a newly installed word processing system.

The Authors

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USER'S MANUAL FOR LPGS:
A COMPUTER PROGRAM FOR CALCULATING RADIATION EXPOSURE RESULTING FROM ACCIDENTAL RADIOACTIVE RELEASES TO THE HYDROSPHERE

John E. White* and Keith F. Eckerman**

ABSTRACT

The LPGS computer program was developed to calculate the radiological impacts resulting from radioactive releases to the hydrosphere. The hydrosphere is represented by the following types of water bodies: estuary, small river, well, lake, and one-dimensional (1-D) river. The program is principally designed to calculate radiation dose (individual and population) to body organs as a function of time for the various exposure pathways. The radiological consequences to the aquatic biota is estimated. Several simplified radionuclide transport models are employed with built-in formulations to describe the release rate of the radionuclides. Optionally, a tabulated user-supplied release model can be input. Printer plots of dose versus time for the various exposure pathways are provided.

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John E. White* and Keith F. Eckerman**

1. INTRODUCTION

The computer program LPGS was developed for use in assessing the risks resulting from accidental releases of radionuclides to the hydrosphere. The name LPGS was derived from the Liquid Pathway Generic Study for which the original computer program was used primarily as an analytic tool in the assessment process. Because of the diverse nature of the hydrosphere, no generic modeling approach can address all types of water bodies. Consequently, the approach taken is one of defining hydrologic models suitable for describing the various types of water bodies in a generic sense. The software developed then serves to drive the hydrologic transport model for the water body and the radiological assessment models by a radionuclide release module.

The effort reported here was guided by the following considerations: a) to improve transportability, b) to implement flexible dimensioning techniques, c) to make tabulated printouts more readable, d) to remove constants buried in the FORTRAN, e) to make the calculational units consistent, f) to revise and improve the edits of the input parameters, g) to ease input data preparation, and h) to perform general FORTRAN "clean-up."

1.1. GENERAL DESCRIPTION OF CODE

The mathematical modeling contained in LPGS evaluates the time dependent radiation impact arising from the population utilizing water associated pathways. The exposure modes considered are (if pathway exists): 1) consumption of drinking water, 2) consumption of aquatic foods, and 3) recreational exposure through swimming and shoreline activities. Dose information is derived as a function of time for both the population and an individual in the immediate vicinity of the release. The contribution of each radionuclide pathway is indicated at two time periods, one of which is specified by the user on input. The user also must specify the extent of utilization of environmental media for the selected locations in the environment. These locations are generally taken to be regions of the water bodies (reaches) over which waterborne concentrations can be averaged. In some instances, the identification of such regions should be based on the applicabilty of dispersion models. For example, a lake is described by a near field 2-dimensional model as well as a mixed tank model to represent the whole lake.

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Hydrologic dispersion models were formulated for a class of water body types including: a small river, large river, lake, estuary, and groundwater (aquifer). The concentration of a released radionuclide, as a function of time, in regions of these bodies is determined by convolution of the release rate function and the response function of the water body to a unit instantaneous release. Usage of pathway media contaminated by the radionuclides resident in the water body determine the extent to which man is exposed to released radionuclides.

Radionuclides can enter surface water bodies either through their direct release to the water body or as a result of influx from another water body. For example, material introduced into groundwater may appear in surface water if the groundwater flows into the surface water body. The initial release at the source may be instantaneous or time dependent. The LPGS code has provisions to define the release as:

1) instantaneous,
2) a constant release rate over some time period,
3) a fractional release rate per unit of time, i.e., release rate proportional to activity present,
4) a release rate expression defined as the sum of three decaying exponentials, i.e., release rate = $A_1e^{-B_1T} + A_2e^{-B_2T} + A_3e^{-B_3T}$, and
5) a user supplied table of release rate vs time.

These features were felt to cover the range of possible release rate information which might be available for assessment activities.

The modeling as implemented here does not include consideration of ingrowth of daughter nuclides from the released radionuclides. The omission should be noted and may limit the utilization of the code in some applications. Potential users should consider the significance of this omission to the problem at hand. A description of the models developed for LPGS is appended (A, B, and C).

2. ORGANIZATION AND CALCULATIONAL FLOW

The LPGS program design is highly structured to permit coupling of numerous options at various stages of the analysis. A schematic diagram illustrating the main calculational blocks is provided in Fig. 1. Subroutine DOSIT plays the role of the executive routine which controls the calculational sequence for a problem. Within a loop over the number of nuclides in the source term, DOSIT selects the appropriate calculational block. Before we begin the discussion on the calculational flow, the manner in which LPGS treats the time domain of the calculation merits some attention.

Depending on the problem, the time character can be quite variable, e.g., instantaneous release to a swift river or release to a slow moving aquifer entering a lake. The user is requested to supply an upper limit

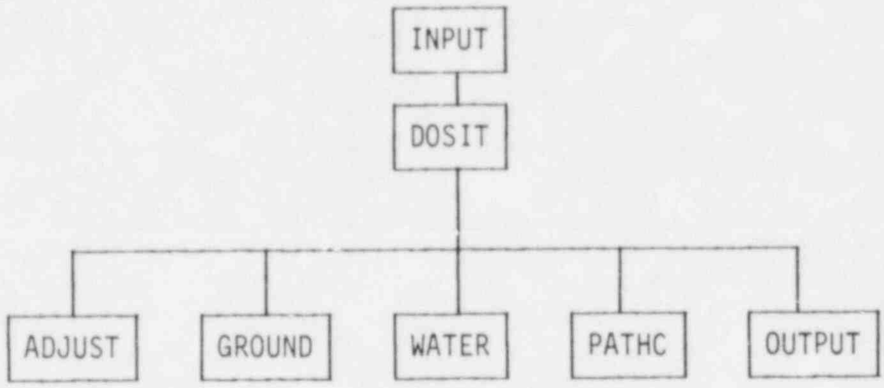


Fig. 1. Schematic diagram illustrating the main calculational blocks.

on the time frame for the analysis. To treat each radionuclide with sufficient detail, recourse was made to logarithmic time grid. Each radionuclide is handled on its own characteristic time grid, with the upper time limit not exceeding the user specified value. Thus as each radionuclide is addressed its time domain is estimated, i.e., when it would enter and exit from a water body. These estimates are rather crude at best and thus provisions are made for additional time steps to ensure an adequate characterization of the radionuclides resident in the water body. After calculations of the radionuclide concentration are complete for the nuclides, this information is used to compute dose data which are then superimposed on a master time grid of 100 logarithmic steps. This method of treating time ensures an adequate level of accuracy in the numerical integration and avoids unnecessary computations on a very dense time grid.

In the discussion which follows, the general calculational flow is given. First, a call is made to Subroutine ADJUST. Using the nuclide specific information provided via card input, ADJUST prepares a release rate history for the nuclide. The Job Control variable JC4 defines the release rate model. Five specific models are available ranging from an instantaneous release to several mathematical functions. As an alternative to the built-in release models, the capability exists to input a user-prepared table of release rate versus time. All dispersion models, including the groundwater model, are driven by a release rate versus time tabulation. Thus, in coupling the hydrological models; the release rate history for the next stage of transport must be defined.

The sequence of subroutines called following return from ADJUST depends on the user's problem. If a direct release to a surface water body is indicated (see JC5), a call is made to WATER. If the release is directed to groundwater, GROUND is called. Unless one is interested in concentrations in the aquifer at locations of wells (JC1 = 3), the purpose of the call to GROUND is simply to determine the release rate vs time into the surface water. In this case GROUND sets up the computational parameters and the time scale (when the nuclide will reach the surface water body and when it's input will end) for the computation of the ground water transport. Subroutine PGRND performs the actual transport calculations.

Once the release time history to the surface water body is established, WATER is called. This subroutine sets up the computations of concentrations in surface water bodies and calls the appropriate routines implementing the dispersion models for the surface water body.

Once the time-dependent waterborne concentration of the radionuclide has been calculated, a call to PATHC is made to determine concentrations vs time in the various pathways, e.g., fish, invertebrates, and sediments along the shoreline. These data are used to estimate doses through man's use of these pathway media. The calculations to this stage are based on the specific time grid for the radionuclide and are then, using interpolation, placed on the master time grid. The master time grid has as its upper bound the user defined maximum time and as a lower bound, the earliest time any radionuclide entered the surface water; if the release was directly to the surface water an initial time step of one day is assumed. NOTE: An initial time step of zero is not admissible in defining a logarithmic grid.

Nuclide specific dosimetric information is tabulated at two time periods, one year and at the user defined upper time step. Tabulations and page plots in time are presented for each pathway for both an individual in the near field (located in the first region) of the water body as well as the exposed population. Provisions are included for both forward and backward integration of the dosimetric information. Use of the backward integration is an aid in estimating the population dose if interdiction measures were implemented at various times e.g., restricting fishing or use of the shoreline for various time periods.

Several levels of detail printing are available to the user. The utility of these details depends on the extent of insight into the analysis the user is seeking. In order to become familiar with the capabilities and limitations of LPGS, full use of the various options is suggested.

3. PROGRAMMING INFORMATION

LPGS is programmed in IBM 360/370 system FORTRAN IV language. The program, originally developed on an IBM computer, has been implemented with modifications and enhancements on an IBM 3033 computer under the MVS operating system. Specifically designed to facilitate transportability, LPGS contains no machine-dependent features. In addition, an effort was made to implement flexible dimensioning techniques, make the printout more readable, remove constants buried in the FORTRAN, remove any inconsistency in calculational units, provide for shoreline erosion, and perform general FORTRAN "clean-up."

For the most part, the input data control is centralized with a clearly readable edit of the input parameters provided. In addition, the free-field FIDO input method (See Section 3.6) is employed for user convenience. An important advantage of the free-field method can be readily seen when preparing card image input data via a computer terminal.

3.1. INPUT DATA PREPARATION

This section describes the input data requirements for preparing LPGS problems. With the exception of two title cards (case title and source title), all data is input using the free-field FIDO format. FIDO arrays or sets of arrays which are not needed should not be entered. In general, the number of data entries is explicitly stated. When given, the quantity in brackets [ ] is the array dimension and the expression in braces { } is the condition requiring that array or set of arrays.

The following variables are required to execute LPGS problems. A brief description of each variable, including units and default values if appropriate, is given below.

Case Title (20A4)

Data Block 1

1$$ General Problem Description [15 entries]

JC1 - Liquid pathway transport model

- 0 - small river
- 1 - estuary
- 2 - dry site (deactivated)
- 3 - wells only
- 4 - lake site
- 5 - 1-D river

JC2 - Selects computation of individual dose

- 0 - compute individual dose
- 1 - do not compute

JC3 - Selects computation of population dose

- 0 - compute population dose
- 1 - do not compute

JC4 - Description of radionuclide release model where QDOT = $\dot{Q}$

- 0 - instantaneous release of Q curies C
- 1 - constant release rate QDOT Ci/yr over DLECH days
- 2 - QDOT = F(T) as a user-supplied source
- 3 - QDOT = FRELS*Q*EXP (-(FRELS*LAMBDA)*T)
- 4 - QDOT = Q + EXP(-LAMBDA*T)*$[A_1e^{-B_1T} + A_2e^{-B_2T} + A_3e^{-B_3T}]$

JC5 - Selects release path for radionuclide source

- 0 - direct release to JC1
- 1 - release to groundwater which enters JC1

JC6 - Special groundwater option flag

0 - no effect
1 - only compute groundwater transport (you must select JC5=1; used to obtain concentration and activity as a function of time and their integrals at wells or the interface to surface water bodies).

JC7 - Selects detailed printout of nuclide concentration data

0 - print nuclide concentration data
1 - no print

JC8 - Selects printout of nuclide dose as a function of time

0 - print nuclide breakdown of dose
1 - no print

JC9 - Index for current reach (internal use only; default = 0)

JC10 - Total number of reaches over which concentration data is averaged (internal use only; default = 0)

JC11 - Select printer plots of dose versus time

0 - plot data
1 - suppress plots

JC12 - Selects mode of integration; recommend forward integration

0 - forward integration of dose rate
1 - backward integration of dose rate

JC13 - Selects printout of dose factors

0 - no print
1 - print dose factors

JC14 - Numbe. of radionuclides in source term

JC15 - not used

2** General floating point parameters [4 entries]

DTIM - period of the evaluation (days)
DLECH - period of chronic release (leach source) (days)
CLSWB - surface water limit (pCi/l) (default = 1.0E-10)
CLGRD - groundwater limit (pCi/l) (default = 1)

3** Surface water parameters [8 entries]

- DEPTH - depth of water body (ft)
- WIDTH - width of water body (ft); for lake model WIDTH=0.
- RVEL - current velocity (ft/sec); needs a nonzero entry for lake model.
- EX - X dispersion coefficient (ft^2/sec)
- EY - Y dispersion coefficient (ft^2/sec)
- CRDIS - cross river distance (ft)
- VOL - volume of water body (ft^3)
- QQ - water outflow rate (ft^3/sec)

4** Hydrological parameters [6 entries]

- SEDF - sediment fraction (default = 1.0E-20)
- SEDR - sediment rate (ft/sec)
- BEDE - bed dispersion coefficient (ft^2/sec)
- BEDU - bed velocity (ft/sec)
- ZLAKET - transfer rate to sediment - KF value (ft/sec)
- SHRER - shoreline erosion rate (1/sec)

5$$ NDIS - number of reaches

T Terminate Data Block 1

Data Block 2

Surface water usage parameters

6** DIST [NDIS] - midpoint of reach (miles)

7** POP [NDIS] - drinking water usage (number of people)

8** AREA [NDIS] - area of water body (acres)

9** URCH [NDIS] - flow rate, if not RVEL (default = RVEL)

10** RWIDE [NDIS] - width of water body, if not WIDTH (default = WIDTH)

11** SR1 [NDIS] - shoreline usage rate (user-day/day)

12** SR2 [NDIS] - swimming usage rate (user-day/day)

13** Additional surface water usage parameters [4 entries]

- C1 - commercial fish harvest (kg/acre/day)
- C2 - recreational fish harvest (kg/acre/day)
- CX1 - commercial invertebrate harvest (kg/acre/day)
- CX2 - recreational invertebrate harvest (kg/acre/day)

14** Ground water parameters [12 entries]

X1 - X coordinate at point of interest (ft)
Y1 - Y coordinate at point of interest (ft)
Z2 - Z coordinate at point of interest (ft)
Z22 - source depth at X=0, Y=0, Z=Z22 (ft)
AX1 - dispersivity in the X-direction (ft)
AY1 - dispersivity in the Y-direction (ft)
AZ1 - dispersivity in the Z-direction (ft)
U1 - ground water velocity (ft/day)
DEP1 - depth of aquifer (ft)
BDEN1 - bulk density (g/cc)
TOTRO1 - total porosity
EFFPR1 - effective porosity

T Terminate Data Block 2

Data Block 3 {JC4=4}

16** Coefficients for the exponential leach release model [13 entries]

Release rate = $A_1e^{-B_1T} + A_2e^{-B_2T} + A_3e^{-B_3T}$
A [6] - A coefficients
B [6] - B coefficients
TSTEP, time at which the second term is used

T Terminate Data Block 3

Source Term Title Card (20A4)

Data Block 4

17** Misc. source term parameters [2 entries]

FRELS - fraction released if JC4=3 (default =1.)
UML - multiplier for source (Q); used to convert activity units to curies

T Terminate Data Block 4

The following data blocks (5 and 6 as required) are repeated for each radionuclide source. Note that the chemical symbol is input via Hollerith characters.

Data Block 5

18** - Radionuclide source description [11 entries]

IAA - chemical symbol
MASS - mass number (use negative value to denote isomeric state)

Q - activity (curies)
CFF - bioaccumulation factor for finfish
CFI - bioaccumulation factor for shellfish
RET1 - biological retention in aquatic biota (days)
ZKD1 - aquifer distribution coefficient (K_d)
ZKD2 - surface water distribution coefficient (K_d)
FWAT1 - fractional water treatment transfer (default=1.)
R30 - relative leach rate (default = 1.)
NSS - number of time intervals used to describe the source term if JC4=2

T Terminate Data Block 5

Data Block 6 {JC4=2}

Tabulated representation of source term. NOTE: Linear interpolation is assumed for the release rate data.

19** THIST [NSS] - time (sec)

20** CHIST [NSS] - release rate (curies/sec)

T Terminate Data Block 6

This concludes input specification for LPGS.

3.2. DESCRIPTION OF SUBROUTINES

Brief descriptions of the subroutines and function subprograms are provided below with cross reference information. No distinction is made between a subroutine and a function subprogram.

ADJUST computes release rate as a function of time for the nuclide currently being considered. The user can select the functional form of the release rate or make use of a table of release rate vs time.
Routines called: EXFCT, EXFCT1, SIMPUN
Called from: DOSIT

BLOCK DATA initializes array containing chemical symbols.
Routines called: none
Called from: none

BREAKM prints detailed tables by nuclide of maximum and collective dose including a breakdown by exposure pathway.
Routines called: none
Called from: OUTPUT

CLEAR zeroes L locations of an array.
Routines called: none
Called from: DOSIT, INPUT, OUTPUT

COLAKE two dimensional dispersion routine used to calculate concentrations in the near field of a discharge to a lake. Convolution integral employed (see CONCXY).
Routine called: FP5
Called from: WATER

CONC one dimensional dispersion routine for calculation of concentration in an estuary. Convolution employed as discussed with CONCXY.
Routine called: FP2
Called from: WATER

CONCXY calculates concentrations in a channel using a two dimensional dispersion model. The arbitrary release rate specified in ADJUST (if release is into surface water) or specified by the output of PGRND, if release was to groundwater, is convoluted with the analytical formulation for an instantaneous release.
Routine called: FP3
Called from: WATER

COPYIT a generalized copy routine for manipulating a single precision array.
Routines called: none
Called from: FFREED, FIDAS1

DFISH computes decay factor for transit time of aquatic foods from harvest to consumption.
Routines called: none
Called from: OUTPUT

DOPYIT a generalized copy routine for manipulating a double precision array.
Routines called: none
Called from: FIDAS1

DOSIT control routine for the various calculational sequences. Contains the loop over all nuclides in the source term. Also, controls the printing of results.
Routines called: ADJUST, CLEAR, EXFCT, GROUND, PATHC, OUTPUT, WATER
Called from: INPUT

EXFCT evaluates the function (1.-EXP(-X)).
Routines called: none
Called from: ADJUST, DOSIT, EXFCT1

EXFCT1 evaluates the function (EXP(-X) - EXP(-Y)).
Routine called: EXFCT
Called from: ADJUST

FFREED	free-field FIDO read routine. Routine called: COPYIT Called from: FIDAS1
FG2	concentration response function for groundwater system. Routines called: none Called from: PGRND
FG3	activity response function for groundwater system. Routines called: none Called from: PGRND
FIDAS1	control routine for FIDO input system. Routines called: COPYIT, DOPYIT, FFREED, FIDEL Called from: INPUT, SOURCE
FIDEL	performs arithmetic operations associated with entering FIDO data into array locations by invoking the '@' option. Routines called: none Called from: FIDAS1
FP2	concentration response function for estuary model. Routines called: none Called from: CONC
FP3	concentration response function for 2-d dispersion model. Routines called: none Called from: CONCXY
FP5	concentration response function for near field dispersion in a lake. Routines called: none Called from: COLAKE
FP8	concentration response function for the mixed lake model. Routines called: none Called from: MLAKE
GROUND	sets up computational parameters and time scale for radionuclide movement through the groundwater. Computations are computed by PGRND. Routines called: PGRND, SIMPUN Called from: DOSIT
INPUT	main control routine for card input parameters. Routines called: CLEAR, DOSIT, FIDAS1, SITE, TRANS. Called from: MAIN

LP — stores chemical symbol into INTEGER*4 variable.
Routines called: none
Called from: SOURCE

MLAKE — calculates concentration in a well mixed lake using the convolution integral over the input rate function.
Routine called: FP8
Called from: WATER

OUTPUT — prints detailed tables of the individual and collective dose as a function of distance, pathway, nuclide, and organ.
Routines called: BREAKM, CLEAR, DFISH, OUTPU2, SIMPUN
Called from: DOSIT

OUTPU2 — prints detailed tables of results and calls page plotting routine.
Routine called: PLOTER
Called from: OUTPUT

PATHC — computes the concentration in pathway other than water, i.e., aquatic biota and shoreline sediment.
Routine called: SIMPUN
Called from: DOSIT

PGRND — calculates the amount of material entering a surface water body following its injection into the ground water. The arbitrary release rate specified in ADJUST is convoluted with the analytical formulation for an instantaneous release.
Routines called: FG2, FG3
Called from: GROUND

PLOTER — printer page plotting routine.
Routines called: none
Called from: OUTPU2

REDDF — reads dose-rate conversion factor data library.
Routines called: none
Called from: INPUT

SIMPUN — numerical integration routine which employs a combination of quadratic and trapezoidal integration over unevenly spaced points.
Routines called: none
Called from: ADJUST, GROUND, OUTPUT, PATHC

SITE — edits problem and site specific input data.
Routines called: none
Called from: INPUT

SOURCE reads source term input data.
Routine called: FIDAS1
Called from: INPUT

TRANS stores input data into appropriate labeled common areas.
Routines called: none
Called from: INPUT

WATER sets up computational parameters and time scales for computing concentrations in surface water bodies. The calls are made to the appropriate surface water dispersion model indicated during input.
Routines called: CONCXY, CONC, COLAKE, MLAKE
Called from: none

YLAG performs a Lagrangian interpolation.
Routines called: none
Called from: OUTPUT

3.3. OUTPUT INFORMATION

The end result of an LPGS calculation is a set of tables — and, if requested, printer plots — containing dose (individual and population) as a function of time (days) for the various pathways. A breakdown of dose by nuclide, pathway, and organ is provided as an option. All input parameters are displayed by variable name with a brief description. Also, the dose factors can be edited upon input option.

3.4. MISCELLANEOUS USEFUL INFORMATION

3.4.1. Estimation of Array Size

Most of the improvements to the original undocumented code were implemented by employing flexible dimensioning techniques. The default value of 1000 for COMMON/DATA/ in the main program is ample for most problems. If the array size needs to be expanded, the following prescription can be used to estimate COMMON/DATA/ storage.

200 + 7*NDIS + JC14*(2*NSS)

Note that NDIS, JC14, and NSS are variable input parameters.

3.4.2. Restriction on the Range of Variables

LPGS can accommodate 60 radionuclides in a source term. This is considered a reasonable number for routine assessments. If the user is faced with a source term containing a larger number of radionuclides, it is suggested that multiple runs be made to determine the radionuclides contributing significantly to the assessment.

3.4.3. Overlay Structure

The overlay structure is provided in Fig. 2.

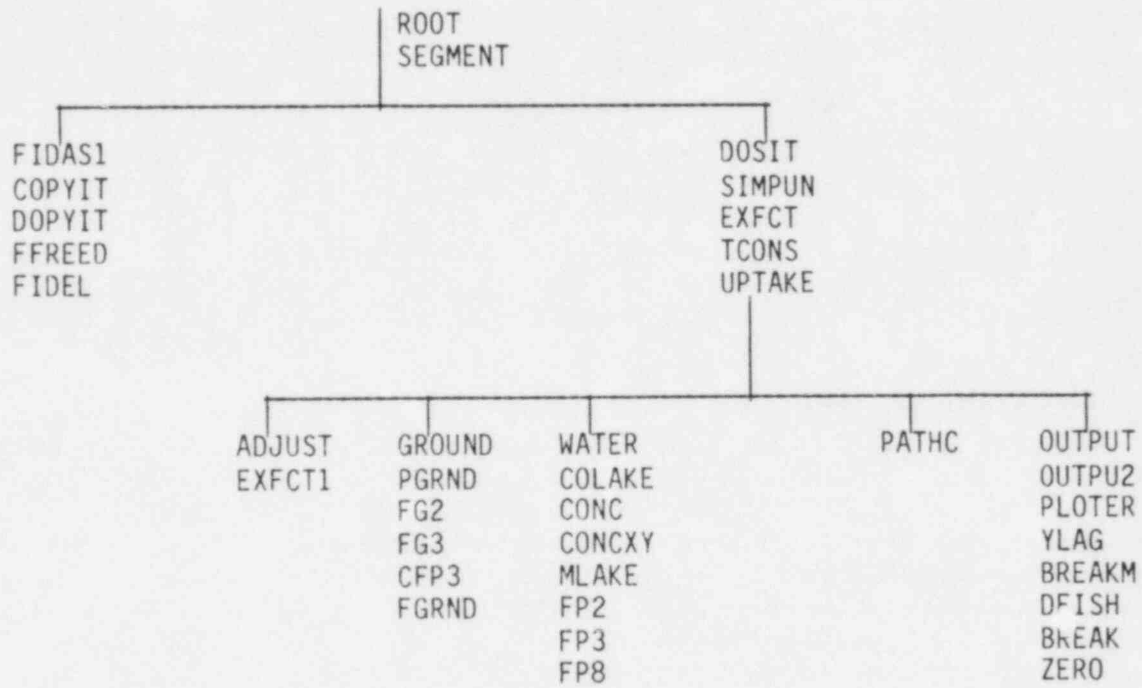


Fig. 2. LPGS overlay structure.

3.4.4. I/O Assignments

LPGS requires the following I/O units.

5	-	standard input
6	-	standard output
12	-	dose-factor library

3.5. EXTERNAL DATA FILES

LPGS requires a dose-rate conversion factor data library for adults. The data file consists of internal radiation dose factors for the following seven organs: bone, liver, total body, thyroid, kidney, lung, and gastrointestinal tract (large lower intestine). In addition, external exposure due to immersion in water (swimming) and exposure to contaminated sediments are needed. The absorbed energy per nuclear disintegration in fish and invertebrates is also required. The current version uses dose factors obtained from NRC Regulatory Guide 1.109 Rev., Calculations of Annual Doses to Man From Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance With 10 CFR Part 50, Appendix I, (October 1977).

3.6. FIDO (Floating Index Data Operation) Input System

The FIDO (Floating Index Data Operations) input method is especially devised to allow the entering or modifying of large data arrays with minimum effort. Special advantage is taken of patterns of repetition or symmetry wherever possible. Developed by W. A. Rhoades and W. W. Engle at Atomics International in the early 1960s for use in a one-dimensional discrete ordinates code (DTF-II), FIDO was patterned after an input method used with the early FLOCO coding system at Los Alamos Scientific Laboratory. Since that time, numerous features requested by users have been added, a free-field option has been developed, and FIDO applications are widespread. The I/O package implemented in LPGS contains all available developments, including extensive improvements made by James Marable of the Oak Ridge National Laboratory.

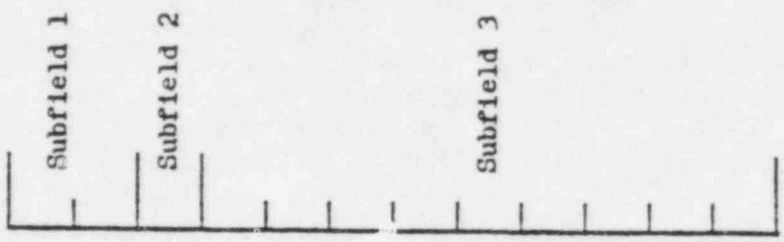
Use of FIDO provides powerful and attractive advantages to the programmer as well as the user. The programmer may insure efficient core utilization as well as relieve himself of the worry with read statements and associated formatting. The user is relieved of the burden of formatted input requirements (when using free-field input) and may enter or modify lengthy data arrays with a minimum of effort.

Efficient core utilization is achieved by the flexible dimensioning inherent with FIDO usage. With flexible dimensioning, the length of arrays read by FIDO in one block may be determined by parameters read in a previous block. This is similar to FORTRAN run-time dimensioning. The important feature is that one contiguous area of storage is available for all data arrays and is denoted as an array whose dimension is the length of that area. The displacements needed for referencing subarrays are also stored contiguously in the same or a different area. When the 'N' array is read by FIDO, the origin of that array is determined by the N-th displacement and the length is determined by the difference between the N+1 and N-th displacements. This feature is further enhanced when dynamic storage allocation is possible.

A group of one or more arrays read with a single call to the FIDO package forms a block, and a special delimiter is used to signify the end of each block. Arrays may be read in any order within a block, but an array belonging to one block should not be placed within another block. An array can be entered more than once within a block, in which case the last value read for each location within the array is stored. If no entries to the arrays within a block are required, the delimiter alone satisfies the input requirement. Arrays may be read as fixed-field, free-field, or user-field input.

3.6.1. Fixed-Field Input

Each card is divided into six 12-column data fields, each of which is divided into three subfields. The following sketch illustrates a typical data field. The three subfields always comprise 2, 1, and 9 columns, respectively.



To begin the first array of a block, an array originator field is placed in any field on a card:

Subfield 1: An integer array identifier < 100 specifying the data array to be read in.

Subfield 2: An array-type indicator:
"$" if the array is integer data
"*" if the array is real data

Subfield 3: Blank

Data are then placed in successive fields until the required number of entries has been accounted for. An example illustrating the format and flexibility of the input will follow the description of the data operators.

In entering data, it is convenient to think of an "index" or "pointer" which is under control of the user, and which specifies the position in the array into which the next data entry is to go. The pointer is always positioned at array location #1 by entering the array originator field. The pointer subsequently moves according to the data operator chosen. Blank fields are a special case, in that they do not cause any data modification and do not move the pointer.

A data field has the following form:

Subfield 1: The data numerator, an integer < 100. We refer to this entry as N_1 in the following discussion.

Subfield 2: One of the special data operators listed below.

Subfield 3: A nine-character data entry, to be read in F9.0 format. It will be converted to an integer if the array is a "$" array or if a special array operator such as Q is being used. Note that an exponent is permissible but not required. Likewise, a decimal is permissible but not required. If no decimal is supplied it is assumed to be immediately to the left of the exponent, if any; and otherwise to the right of the last column. This entry is referred to as N_3 in the following discussion.

A list of data operators and their effect on the array being input follows:

Operator	Description
blank	"Blank" indicates a single entry of data. The data entry in the third subfield is entered in the location indicated by the pointer, and the pointer is advanced by one. However, an entirely blank field is ignored.
+	"+" of "-" indicates exponentiation. The data entry in the third field is entered and multiplied by $10^{\sim N_1}$, where N_1 is the data numerator in the first subfield, given the sign indicated by the data operator itself. The pointer advances by one. In cases where an exponent is needed, this option allows the entering of more significant figures than the blank option.
&	"&" has the same effect as "+".
R	"R" indicates that the data entry is to be repeated N_1 times. The pointer advances by N_1.
I	"I" indicates linear interpolation. The data numerator, N_1, indicates the number of interpolated points to be supplied. The data entry in the third subfield is entered, followed by N_1 interpolated entries equally spaced between that value and the data entry found in the third subfield of the next non-blank field. The pointer is advanced by $N_1 + 1$. The field following an "I" field is then processed normally, according to its own data operator. In "$" arrays, interpolated values will be rounded to the nearest integer.
L	"L" indicates logarithmic interpolation. The effect is the same as that of "I" except that the resulting data are evenly separated in log-space.
Q	"Q" is used to repeat sequences of numbers. The length of the sequence is given by the third subfield, N_3. The sequence of N_3 entries is to be repeated N_1 times. The pointer advances by $N_1 * N_3$. If either N_1 or N_3 is 0, then a sequence of $N_1 + N_3$ is repeated one time only, and the pointer advances by $N_1 + N_3$.
N	The "N" option has the same effect as "Q", except that the order or the sequence is reversed each time it is entered.

M "M" has the same effect as "N" except that the sign of each entry in the sequence is reversed each time the sequence is entered. For example, the entries.

1 2 3 2M2

would be equivalent to

1 2 3 -3 -2 2 3.

Z "Z" causes $N_1 + N_3$ locations to be set to 0. The pointer is advanced by $N_1 + N_3$.

C "C" causes the position of the last array item entered to be printed. This is the position of the pointer, less 1. The pointer is not moved.

O "O" causes the print trigger to be changed. The trigger is originally off. Successive "O" fields turn it on and off alternately. When the trigger is on, each card image is listed as it is read.

S "S" indicates that the pointer is to skip N_1 positions leaving those array positions unchanged. If the third subfield is blank, the pointer is advanced by N_1. If the third subfield is non-blank that data entry is entered following the skip, and the pointer is advanced by $N_1 + 1$.

A "A" moves the pointer to the position, N_3, specified in the third subfield.

F "F" fills the remainder of the array with the datum entered in the third subfield.

E "E" skips over the remainder of the array. The array length criterion is always satisfied by an E, no matter how many entries have been specified. No more entries to an array may be given following an "E", except that data entry may be restarted with an "A".

The reading of data to an array is terminated when a new array origin field is supplied, or when the block is terminated. If an incorrect number of positions has been filled, an error edit is given, and a flag is set which may later abort execution of the problem. FIDO then continues with the next array if an array origin was read.

A block termination consists of a field having "T" in the second subfield. All entries following "T" on a card are ignored, and control is returned from FIDO to the calling program.

Comment cards can be entered within a block by placing an apostrophe (') in column 1. Then columns 2–80 will be listed, with column 2 being used for printer carriage control. Such cards have no effect on the data array or pointer.

Note that the sample data sheet below is for illustrative purposes only and is not meant to represent a collective set of meaningful data.

Name ______ Charge ______ Date ______ Page ______

		IDENTIFICATION	REMARKS (DO NOT PUNCH)
24$			Begin 24$ array
F	1		Fill 24$ array with ones
2*			Begin 2* array
10R	1.0		enter 1.0 ten times - (Repeat)
A	20		Enter next data in 20th position
10R	1.0		Enter 1.0 in positions 20 through 29
10S	1.0		Skip positions 30-39 and enter 1.0
4*			Begin 4* array
4I	0.0		Enter 0.0, 1.0, 2.0, 3.0, 4.0
	5.0		and 5.0 in successive locations
1*			Begin 1* array
16R			Enter zero 16 times (note exception to
			restriction 2)
25*	10		Begin 25* array with 11th
			member of that array
T			Terminate
	1.234		Enter 1.234 in all cases
	.1234 +1		
	123.4E-2		
	1234 -5		
	12.34E-01		
1+	.1234		

R - REPEAT I - INTERPOLATE S - SKIP T - TERMINATE

3.6.2. Free-Field Input

With free-field input, data are written without fixed restrictions as to field and subfield size and positioning on the card. The options used with fixed-field input are available, although some are slightly restricted in form. In general, fewer data cards are required for a problem, a card listing is more intelligible, the cards are easier to keypunch, and certain common keypunch errors are tolerated without affecting the problem. Data arrays using fixed- and free-field input can be intermingled at will within a given block.

The concept of three subfields per field is still applicable to free-field input, but if no entry for a field is required, no space for it need be left. Only columns 1–72 may be used, as with fixed-field input. The <u>array originator field</u> can begin in any position. The array identifiers and type indicators are used as in fixed-field input. The type indicator is entered twice, to designate free-field input (i.e., "$$" or "**"). The blank third subfield required in fixed-field input is not required. For example:

31**

indicates that array 31, a real-data entry, will follow in free-field format.

Data fields may follow the array origin field immediately. The data field entries are identical to the fixed-field entries with the following restrictions:

(1) Any number of blanks may separate fields, but at least one blank must follow a third subfield entry if one is used.

(2) If both first- and second-subfield entries are used, no blanks may separate them, i.e., 24S, but not 24 S.

(3) Numbers written with exponents must not have imbedded blanks, i.e., 1.0E+4, 1.0E4, 1.0+4, or even 1+4, but not 1.0 E4.

(4) In third-subfield data entries, only 9 digits, including the decimal but not including the exponent field, can be used, i.e., 123456.89E07, but not 123456.789E07.

(5) The Z entry must be of the form: 738Z, not Z738 or 738 Z.

(6) The + or - data operators are not needed and are not available.

(7) The Q, N, and M entries are restricted: 3Q4, 1N4, or M4, but not 4Q, 4N, or 4M.

3.6.3. User-Field Input

If the user follows the array identifier in the array originator field with the character "U" or "V", the input format is to be specified by the user. If "U" is specified, the FORTRAN format to be used must be suppled in columns 1–72 of the next card. The format must be enclosed by the usual parentheses. Then the data for the entire array must follow on successive cards. The rules of ordinary FORTRAN input as to exponents, blanks, etc., apply. If the array does not fill the last card, the remainder must be left blank.

"V" has the same effect as "U" except that the format read in the last preceding "U" array is used.

Example of FIDO Free-Field Input

1$$ F0	Zero out the 1$ array
2** 12.34-1 4Z	Enter 1.234 and 4 zeroes in the 2* array
3** A5 60 E	Enter 60.0 as the 5th entry of the 3* array
4$$ 2I1 2R4 2Q5	Enter 1,2,3,4,4,1,2,3,4,4,1,2,3,4,4
T	Terminate the block

3.6.4. Features of the Improved Version of FIDO

Recent improvements to FIDO include the reading of formatted or unformatted pieces of arrays from various I/O devices, reading Hollerith characters, reading numbers to an arbitrary base (e.g., octal, binary, and hexidecimal), modifying (by multiplication, etc.) numbers already in storage, entering double precision arrays, and other changes. It is important to note that these improvements have been incorporated without changing the previous definitions. Old FIDO input decks will still be read correctly.

The characters and the corresponding operation instructions are listed in Table 1. Operator characters with superscript 'a' denote operations which ignore the first subfield value N1. Operator characters with a superscript 'b' denote operations for which it is not possible to enter a third subfield (using free-field input). These characters terminate the field, and a new field starts immediately regardless of whether there is space or not. Except for these operations a field is terminated by a space following the third subfield.

In general, no space is allowed between the first and second subfields, and spaces are allowed but are not required between the second and third subfields. Between fields spaces are allowed and one is required (except for above exceptions associated with superscript 'b').

Table 1. Characters and Their Corresponding Operation Instructions

Character	Operation Instruction
$\$\$^b$	Designate array N1 to be an integer array and set the pointer to the first location in array N1. When interpolated, etc., numbers are always rounded off to the nearest integer value.
$**^b$	Designate array N1 to be a floating point array and set the pointer to the first location in array N1.
$\#\#^b$	Designate array N1 to be a double precision array and set the pointer to the first location in array N1.
$/^{a,b}$	Skip to the next card ignoring all comments following the slash.
(blank)[a]	Enter the third and only nonblank subfield into the location indicated by the pointer and then advance the pointer by 1.
Z^b	Enter 0 |N2| times and advance the pointer by |N1|. If zero or blank is entered for N1 it is replaced by 1.
R	Enter the third subfield |N1| times, and with alternating sign if N1 is negative. Increase the pointer by |N1|. If a zero or blank is entered from N1, it is replaced by 1.
I	Determine N1+2 numbers by linear interpolation starting with the third subfield of this field and ending with the third subfield of the next field. Enter the first N1+1 numbers.
$T^{a,b}$	Terminate this call for FIDO input and return to the calling program.

Table 1. (continued)

Character	Operation Instruction
L	Determine N1+2 numbers determined by logarithmic interpolation starting with the third subfield of this field and ending with the third subfield of the next field. Enter the first N1+1 numbers. (The logarithms of the numbers entered are uniformly spaced.)
W	Repeat the sequence of $\lvert N3 \rvert$ numbers immediately preceding the pointer $\lvert N1 \rvert$ more times, multiplying on each repetition each number of the sequence by 10 or 0.1 according as N3 is positive or negative. If N1 is negative the sign of the sequence changes on each repetition.
Q	Repeat the sequence of $\lvert N3 \rvert$ numbers immediately preceding the pointer $\lvert N1 \rvert$ more times. If N1 is negative change the sign of each number of the sequence on each repeat. If N3 is negative reverse the order of the sequence for each repetition. The pointer is finally advanced by $\lvert N1 \rvert * \lvert N3 \rvert$.
N	This is equivalent to the operation Q with a first subfield N1 and a third subfield $-\lvert N3 \rvert$.
M	This is equivalent to the operation Q with a first subfield $-\lvert N1 \rvert$ and a third subfield N3.
C[a,b]	Print the pointer value of the last array item entered. This is one less than the pointer position.
0[b]	If N1>0 the print trigger is turned on. If N1=0 the print trigger if flipped. If N1<0 the print trigger if turned off. When the print trigger is on, each card image is printed as it is read.

Table 1. (continued)

Character	Operation Instruction
S^b	Add N1 to the pointer value. N1 may be negative thereby descreasing the pointer value.
A^a	Place the pointer at N3.
F	Fill the remaining locations of the array with the third subfield entry. If N1 is negative the entries alternate in sign. Set the pointer after the last location.
$E^{a,b}$	Skip over the remainder of the array by placing the pointer after the last location.
H	Enter the N1 hollerith characters which are in the third subfield. Advance the pointer by the number of words required to store these N1 characters. (number of words = (N1+NCPW-1)/NCPW where NCPW is the number of characters per word).
G	Read N3 words from I/O device with data set reference number N1 according to the format to be specified in the next field which is hollerith. If the next field is 0 or hollerith blank the field is unformatted. Advance the pointer by N3.
Y	If N1 is positive, change the input unit so as to read the succeeding card images from unit N1 until a delimiting T operation appears (or until a similar Y instruction appears). If N3>0 change all FIDO edit to unit N3. If N3<0 change card image listing (see 'O' operation) only to unit |N1|. After a delimiter T appears the next call to FIDO resets the input and output unit numbers to the original value.
$@^b$	Ordinarily FIDO enters data - interpolated, sequence repeat, etc. - by entering each "raw entry" into the proper location obliterating the previous "old number"

Table 1. (continued)

Character	Operation Instruction
	in that location. By means of the operation denoted by character '@' FIDO changes its mode of entering data according to the value of the first subfield N1 preceding the operation character '@'. The various manipulations are performed on the "old number" in storage and the "raw entry" in order to obtain the final number which is stored. Let A be the "old number" previously stored, let B be the "raw value" determined by the usual FIDO entry. We have the following possibilities for the number finally stored according to the value of N1.

N1	Number Entered
0	B (the default mode)
1	A+B
2	A-B
3	A*B
4	A/B
5	B/A
6	B*EXP(A)
7	B*LN(A)
8	EXP(B)
9	LN(B)

Each time an array is designated by an array designator field the default mode (N1=0) is reactivated causing raw data to be entered directly into array storage.

4. SAMPLE PROBLEMS

This section contains a description of the input data and the computer printout for two sample problems. The first problem was chosen to illustrate a release to an estuary by way of groundwater. This is a typical application of LPGS in an assessment problem. Note that detailed output information has been suppressed. Both the format and amount of output for runs using different release models or other surface water bodies are similar to this case. On the other hand, the second problem demonstrates an application of LPGS to examine the transport of the released radionuclides through the groundwater system. This case provides information on the time dependent rate at which the radionuclides cross an interface down gradient of the release in the groundwater.

The sample problems, together with annotated printout listings, are given on the following pages. Note that the free-form FIDO input system discussed in Section 3.6 was used to prepare the input data.

Sample Problem 1

In this problem, four nuclides (^{90}Sr, ^{106}Ru, ^{131}I, and ^{137}Cs) are assumed to be instantaneously released into the groundwater which interfaces with an estuary (surface water body) at 1500 ft from the release. Individual and population dose are computed. The individual is assumed to be at the midpoint of the first region of the estuary (at 5 miles). The population dose represents contributions from the population usage of the estuary as represented by four regions. The contributions of each radionuclide and pathway to the individual population dose is indicated.

The input data for problem 1 is provided below with the computer printout immediately following the input.

```
SAMPLE PROBLEM NO. 1    :    RELEASE TO AN ESTUARY VIA GROUNDWATER
1$$ 1 A4 0 1 0 2R1 A13 1 3 E
2** 2R 36525. E
3** 32.8 4875. 0.1 1500. E
4** 0.01 8.33-10 E
5$$ 4 T
6** 5. 20. 40. 75.
8** 5.91+3 2R1.18+4 2.95+4 E
11** 3.+3 2R6.+3 1.5+4 E
12** 800. 1.6+3 1.6+3 4.+3 E
13** 0.0053 0.052 0.016 0.0039 E
14** 1500. 0. 30. 15. 2. 2. .3 6.7 32.8 1.7 .42 0.2 T
  SOURCE TERM FOR SAMPLE PROBLEM NO. 1
17** 1.-3 1. T
18** 2HSR 90 5.+6 2. 20. 1.16+2 2. 7.+2 0.2 1. E  T
18** 2HRU 106 5.+6 3. 1.+3 1.16+2 2. 1.+2 0.5 .1 E  T
18** 2HCS 137 8.6+6 40. 25. 34.6   20. 1.+3 0.9 1. E  T
/*
```

```
1$ ARRAY       15 ENTRIES READ
2* ARRAY        4 ENTRIES READ
3* ARRAY        6 ENTRIES READ
4* ARRAY        8 ENTRIES READ
5$ ARRAY        1 ENTRIES READ

0T
6* ARRAY        4 ENTRIES READ
8* ARRAY        4 ENTRIES READ
11* ARRAY       4 ENTRIES READ
12* ARRAY       4 ENTRIES READ
13* ARRAY       4 ENTRIES READ
14* ARRAY      12 ENTRIES READ

0T
```

FIDO input routines produce these messages

L I Q U I D   P A T H W A Y   S T U D Y

CASE TITLE : SAMPLE PROBLEM NO. 1   :   RELEASE TO AN ESTUARY VIA GROUNDWATER → *Case title*

JOB CONTROL :

JC(1) = 0/1/2/3/4/5	SMALL RIVER/ESTUARY/DRY SITE/WELLS ONLY/LAKE SITE/1-D RIVER	1
JC(2) = 0/1	COMPUTE INDIVIDUAL DOSE/NO EFFECT	0
JC(3) = 0/1	COMPUTE POPULATION DOSE/NO EFFECT	0
JC(4) = 0/1/2/3/4	INSTANTANEOUS RELEASE OF Q CURIES /CONSTANT RELEASE RATE QDOT CI/YR OVER DLECH DAYS /QDOT=F(T) AS USER INPUT SOURCE /QDOT=FRELS*Q*EXP(-(FRELS*LAMBDA)*T) /EXPONENTIAL LEACH RELEASE MODEL	0
JC(5) = 0/1	DIRECT RELEASE TO JC(1)/RELEASE TO GROUND WATER WHICH ENTERS JC(1)	1
JC(6) = 0/1	NO EFFECT/GROUND WATER RELEASE DATA ONLY	0
JC(7) = 0/1	NO EFFECT/NUCLIDE CONCENTRATION DATA SUPPRESSED	1
JC(8) = 0/1	NO EFFECT/NUCLIDE DOSE BREAKDOWN SUPPRESSED	1
JC(9) =	INTERNAL USE ONLY (DEFAULT = 0)	0
JC(10) =	INTERNAL USE ONLY (DEFAULT = 0)	0
JC(11) = 0/1	NO EFFECT/SUPPRESS PLOTS	0
JC(12) = 0/1	FORWARD INTEGRATION OF DOSE/BACKWARD INTEGRATION	0
JC(13) = 0/1	NO EFFECT/PRINT DOSE FACTORS	1
JC(14) =	NUMBER OF RADIONUCLIDE SOURCES	3
JC(15) =	NOT USED	0

1$$ array

DTIM	DOSE PERIOD (DAYS)	3.6525E 04
DLECH	LEACH PERIOD (DAYS)	3.6525E 04
CLSWB	SURFACE WATER LIMIT (PCI/L)	2.8330E-09
CLGRD	GROUND WATER LIMIT (PCI/L)	1.0000E 00

2$$ array

SITE DATA :

NUMBER OF HYDROLOGICAL REGIONS   4 → *Estuary represented by 4 regions*

Midpoint of region   *Surface water area*   *Recreational usage*

REGION	DIST(MILES)	POPULATION	AREA(ACRES)	FLOW(FT/SEC)	WIDTH(FT)	SHORELINE USAGE (USER-D/D)	SWIMMING USAGE (USER-D/D)
1	5.00000E 00	0.0	5.91000E 03	1.00000E-01	4.87500E 03	3.00000E 03	8.00000E 02
2	2.00000E 01	0.0	1.18000E 04	1.00000E-01	4.87500E 03	6.00000E 03	1.60000E 03
3	4.00000E 01	0.0	1.18000E 04	1.00000E-01	4.87500E 03	6.00000E 03	1.60000E 03
4	7.50000E 01	0.0	2.95000E 04	1.00000E-01	4.87500E 03	1.50000E 04	4.00000E 03

These data are used in conjunction with the area values for the regions

COMMERCIAL FISH HARVEST (KG/A/D)	5.3000E-03	RECREATIONAL FISH HARVEST (KG/A/D)	5.2000E-02
COMMERCIAL INVERTEBRATE HARVEST (KG/A/D)	1.6000E-02	RECREATIONAL INVERTEBRATE HARVEST (KG/A/D)	3.9000E-03

```
SURFACE WATER BODY HYDROLOGICAL PARAMETERS :
 DEPTH    DEPTH OF WATER (FT)                          3.2800E 01
 WIDTH    WIDTH OF WATER BODY (FT)                     4.8750E 03
 RVEL     CURRENT FLOW RATE (FT/SEC)                   1.0000E-01
 EX       X DISPERSION COEFFICIENT (FT**2/SEC)         1.5000E 03
 EY       Y DISPERSION COEFFICIENT (FT**2/SEC)         0.0
 CRDIS    CROSS RIVER DISTANCE (FT)                    0.0
 SEDF     SEDIMENT FRACTION                            1.0000E-02
 BEDE     BED DISPERSION COEFFICIENT (FT**2/SEC)       0.0
 BEDU     BED VELOCITY (FT/SEC)                        0.0
 SEDR     SEDIMENT RATE (FT/SEC)                       8.3300E-10
 ZLAKET   TRANSFER RATE TO SEDIMENT :KF  (FT/SEC)      0.0
 SHRER    SHORELINE EROSION RATE (1/SEC)               0.0
 VOL      WATER VOLUME  (FT**3)                        0.0
 QQ       WATER OUTFLOW RATE  (FT**3/SEC)              0.0
```

*3** and 4** arrays*

```
GROUND WATER HYDROLOGICAL PARAMETERS :
 DEPTHG    DEPTH OF AQUIFER (FT)                       3.2800E 01
 UGRND     GROUND WATER VELOCITY (FT/DAY)              6.7000E 00
 X         X COORDINATE AT POINT OF INTEREST           1.5000E 03
 Y         Y COORDINATE AT POINT OF INTEREST           0.0
 Z         Z COORDINATE AT POINT OF INTEREST           3.0000E 01
 Z1        SOURCE DEPTH (FT) AT X=0, Y=0, Z=Z1         1.5000E 01
 BDEN      BULK DENSITY (G/CC)                         1.7000E 00
 TOTPOR    TOTAL POROSITY                              4.2000E-01
 EFFPOR    EFFECTIVE POROSITY                          2.0000E-01
 ALPHAX    DISPERSIVITY IN THE X-DIRECTION (FT)        2.0000E 00
 ALPHAY    DISPERSIVITY IN THE Y-DIRECTION (FT)        2.0000E 00
 ALPHAZ    DISPERSIVITY IN THE Z-DIRECTION (FT)        3.0000E-01
```

*14** array*

```
17* ARRAY      2 ENTRIES READ

 0T
18* ARRAY     11 ENTRIES READ

 0T
18* ARRAY     11 ENTRIES READ

 0T
18* ARRAY     11 ENTRIES READ

 0T
```

More FIDO messages

DESCRIPTION OF SOURCE TERM

SUBTITLE :   SOURCE TERM FOR SAMPLE PROBLEM NO. 1 → *Source term title card*

```
UML    CONVERT ACTIVITIES TO CURIES        1.0000E 00
FRELS  FRACTION RELEASED IF JC(4)=3        1.0000E-03
```

*17** array*

		BIOTA PARAMETERS			DISTRIBUTION COEFFICIENT		WATER TREATMENT
	ACTIVITY	BIOACCUMULATION		RETENTION	AQUIFER	STREAM	TRANSFER
NUCLIDE	(CI)	FISH	INVERTEBRATE	(DAY)	(CC/G)		
38SR 90	5.0000E 06	2.0000E 00	2.0000E 01	1.1600E 02	2.0000E 00	7.0000E 02	2.0000E-01
44RU106	5.0000E 06	3.0000E 00	1.0000E 03	1.1600E 02	2.0000E 00	1.0000E 02	5.0000E-01
55CS137	8.6000E 06	4.0000E 01	2.5000E 01	3.4600E 01	2.0000E 01	1.0000E 03	9.0000E-01

*18** array*

SUM OF ACTIVITIES =   1.8600E 07

Display dose factor data

ADULT DOSE FACTORS

NUCLIDE	LAMBDA	GROUND	SWIM	FISH	INVERT	BONE	LIVER	T. BODY	THYROID	KIDNEY	LUNG	GI-LLI
38SR 90	7.58E-10	0.0	5.40E-10	1.14E 00	1.14E 00	7.58E-03	0.0	1.86E-03	0.0	0.0	0.0	2.19E-04
44RU106	2.18E-08	1.80E-09	0.0	1.53E 00	1.44E 00	2.75E-06	0.0	3.48E-07	0.0	5.31E-06	0.0	1.78E-04
55CS137	7.31E-10	4.20E-09	1.00E-06	5.00E-01	2.67E-01	7.97E-05	1.09E-04	7.14E-05	0.0	3.70E-05	1.23E-05	2.11E-06

169 NUCLIDES READ FROM DOSE FACTOR LIBRARY

Time of arrival and departure for ^{90}Sr
at the 1500 ft interface

SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER

The value for biota represents the time beyond
the estuary clearance when the nuclide has
cleared from the biota

ACTIVITY-TIME RELATIONSHIPS

NUCLIDE	REGION	ACTIVITY (CURIES) INITIAL	RELEASE	SURFACE	TIME (DAYS) IN	OUT	BIOTA
SR 90		5.0000E 06	5.0000E 06	4.3751E 06	1.4135E 03	1.5158E 03	
	1				1.4143E 03	3.4607E 03	4.3310E 03
	2				1.4245E 03	3.6593E 03	4.5302E 03
	3				1.4503E 03	3.8190E 03	4.6904E 03
	4				1.5157E 03	4.0819E 03	4.9540E 03
RU106		5.0000E 06	5.0000E 06	1.0954E 05	1.4613E 03	1.3171E 03	
	1				1.4615E 03	2.8717E 03	3.5413E 03
	2				1.4641E 03	2.9056E 03	3.5753E 03
	3				1.4705E 03	2.9447E 03	3.6145E 03
	4				1.4868E 03	3.0087E 03	3.6787E 03
CS137		8.6000E 06	8.6000E 06	2.6999E 06	1.3127E 04	1.2340E 04	
	1				1.3128E 04	2.5594E 04	2.5859E 04
	2				1.3142E 04	2.5905E 04	2.6170E 04
	3				1.3178E 04	2.6126E 04	2.6391E 04
	4				1.3268E 04	2.6489E 04	2.6755E 04

Arrival and departure times for ^{137}Cs at the
midpoint of the region

Activity crossing the ground/surface water
interface at 1500 ft

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SAMPLE PROBLEM NO. 1   :   RELEASE TO AN ESTUARY VIA GROUNDWATER
*  O U T P U T   O F   R U N *

DOSES COMPUTED OVER THE TIME SPAN OF  1.41E 03 TO  2.68E 04 DAYS
NUMBER ON NUCLIDES    3
NUMBER OF HYDRO REGIONS    4
```

Longest departure time from above; cannot exceed the input value for DTIM

The earliest time a nuclide entered the surface water

The following set of four tables are intended to provide a picture of the contributions of each nuclide to the assessment. Data are tabulated at one year and at the maximum time.

SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER

Note units of rads

NUCLIDE BREAKDOWN OF MAXIMIUM INDIVIDUAL DOSE (REM) AT $\boxed{\text{3.65E 02 DAYS}}$ → *1 year interval*

	DRINKING WATER				AQUATIC FOODS				SHORE	SWIMMING	FISH	INVERT
NUCLIDE	T.BODY	BONE	THYROID	GI-LLI	T.BODY	BONE	THYROID	GI-LLI	T.BODY	T.BODY	RADS	RADS
SR 90	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
RU106	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CS137	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
TOTAL	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Note that no activity has reached the surface water after 1 year

SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER

NUCLIDE BREAKDOWN OF POPULATION DOSE (MANREM) AT 3.65E 02 DAYS

	DRINKING WATER				AQUATIC FOODS				SHORE	SWIMMING
NUCLIDE	T.BODY	BONE	THYROID	GI-LLI	T.BODY	BONE	THYROID	GI-LLI	T.BODY	T.BODY
SR 90	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
RU106	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CS137	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
TOTAL	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER

NUCLIDE BREAKDOWN OF MAXIMIUM INDIVIDUAL DOSE (REM) AT $\boxed{2.68E\ 04\ DAYS}$ ⟶ *Upper time period ≤ DTIM*

	DRINKING WATER				AQUATIC FOODS				SHORE	SWIMMING	FISH	INVERT
NUCLIDE	T.BODY	BONE	THYROID	GI-LLI	T.BODY	BONE	THYROID	GI-LLI	T.BODY	T.BODY	RADS	RADS
SR 90	7.63E 01	3.11E 02	0.0	8.99E 00	7.35E 01	3.00E 02	0.0	8.66E 00	0.0	3.80E-06	1.18E 01	1.18E 02
RU106	9.46E-04	7.48E-03	0.0	4.84E-01	9.99E-03	7.89E-02	0.0	5.11E 00	3.42E-01	0.0	4.87E-01	1.53E 02
CS137	7.78E 00	8.68E 00	0.0	2.30E-01	1.30E 01	1.45E 01	0.0	3.84E-01	2.14E 01	4.15E-03	7.03E 01	[illegible].35E 01
TOTAL	8.41E 01	3.20E 02	0.0	9.70E 00	8.65E 01	3.14E 02	0.0	1.41E 01	2.17E 01	4.15E-03	8.27E 01	2.95E 02

Note that this data assumes a single individual has been exposed over the 2.68×10^4 days

Population dose inlcudes contributions from all four regions.

SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER

NUCLIDE BREAKDOWN OF POPULATION DOSE (MANREM) AT 2.68E 04 DAYS

	DRINKING WATER				AQUATIC FOODS				SHORE	SWIMMING
NUCLIDE	T.BODY	BONE	THYROID	GI-LLI	T.BODY	BONE	THYROID	GI-LLI	T.BODY	T.BODY
SR 90	0.0	0.0	0.0	0.0	6.12E 06	2.49E 07	0.0	7.20E 05	0.0	1.14E 01
RU106	0.0	0.0	0.0	0.0	7.26E 02	5.74E 03	0.0	3.71E 05	1.15E 06	0.0
CS137	0.0	0.0	0.0	0.0	9.13E 05	1.02E 06	0.0	2.70E 04	9.00E 07	1.29E 04
TOTAL	0.0	0.0	0.0	0.0	7.03E 06	2.59E 07	0.0	1.12E 06	9.11E 07	1.29E 04

Drinking water population was zero for the estuary

Remaining tables show the time duration of the assessment for each pathway for both the maximum individual (located in the first region) and the population (over all regions)

SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER
RECREATIONAL AQUATIC FOOD PATHWAY

	* * * I N D I V I D U A L * * *				* * * P O P U L A T I O N * * *			
TIME	R E M				M A N R E M			
(DAYS)	T.BODY	BONE	THYROID	GI-LLI	T.BODY	BONE	THYROID	GI-LLI
1.41E 03	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.46E 03	8.80E-10	3.59E-09	0.0	1.04E-10	2.72E-06	1.11E-05	0.0	3.20E-07
1.50E 03	4.94E-08	2.01E-07	0.0	1.05E-08	1.55E-04	6.32E-04	0.0	3.85E-05
1.55E 03	1.83E-06	7.45E-06	0.0	3.14E-07	5.13E-03	2.09E-02	0.0	1.04E-03
1.59E 03	1.37E-05	5.57E-05	0.0	4.18E-06	4.42E-02	1.80E-01	0.0	1.69E-02
1.64E 03	1.21E-04	4.92E-04	0.0	4.11E-05	4.60E-01	1.87E 00	0.0	1.84E-01
1.69E 03	2.52E-03	1.03E-02	0.0	6.34E-04	8.98E 00	3.66E 01	0.0	2.78E 00
1.74E 03	1.62E-02	6.60E-02	0.0	3.99E-03	5.52E 01	2.25E 02	0.0	1.81E 01
1.79E 03	7.35E-02	3.00E-01	0.0	2.20E-02	2.87E 02	1.17E 03	0.0	1.13E 02
1.85E 03	3.17E-01	1.29E 00	0.0	8.91E-02	1.47E 03	5.98E 03	0.0	5.18E 02
1.90E 03	1.14E 00	4.66E 00	0.0	3.16E-01	5.29E 03	2.16E 04	0.0	1.93E 03
1.96E 03	3.30E 00	1.35E 01	0.0	8.55E-01	1.63E 04	6.63E 04	0.0	5.72E 03
2.02E 03	7.30E 00	2.98E 01	0.0	1.85E 00	4.28E 04	1.75E 05	0.0	1.40E 04
2.08E 03	1.40E 01	5.69E 01	0.0	3.39E 00	9.52E 04	3.88E 05	0.0	2.92E 04
2.14E 03	2.34E 01	9.54E 01	0.0	5.35E 00	1.79E 05	7.29E 05	0.0	5.10E 04
2.21E 03	3.39E 01	1.38E 02	0.0	7.40E 00	3.08E 05	1.26E 06	0.0	7.94E 04
2.28E 03	4.45E 01	1.81E 02	0.0	9.26E 00	4.85E 05	1.98E 06	0.0	1.12E 05
2.34E 03	5.35E 01	2.18E 02	0.0	1.08E 01	6.91E 05	2.82E 06	0.0	1.46E 05
2.41E 03	5.98E 01	2.44E 02	0.0	1.18E 01	9.32E 05	3.80E 06	0.0	1.81E 05
2.49E 03	6.46E 01	2.63E 02	0.0	1.25E 01	1.18E 06	4.82E 06	0.0	2.14E 05
2.56E 03	6.79E 01	2.77E 02	0.0	1.30E 01	1.42E 06	5.79E 06	0.0	2.45E 05
2.64E 03	7.00E 01	2.85E 02	0.0	1.33E 01	1.64E 06	6.67E 06	0.0	2.72E 05
2.72E 03	7.13E 01	2.91E 02	0.0	1.35E 01	1.81E 06	7.36E 06	0.0	2.92E 05
2.80E 03	7.22E 01	2.94E 02	0.0	1.36E 01	1.94E 06	7.90E 06	0.0	3.08E 05
2.89E 03	7.27E 01	2.96E 02	0.0	1.37E 01	2.03E 06	8.28E 06	0.0	3.20E 05
2.97E 03	7.31E 01	2.98E 02	0.0	1.37E 01	2.09E 06	8.50E 06	0.0	3.26E 05
3.06E 03	7.33E 01	2.99E 02	0.0	1.37E 01	2.12E 06	8.65E 06	0.0	3.30E 05
3.15E 03	7.34E 01	2.99E 02	0.0	1.37E 01	2.15E 06	8.75E 06	0.0	3.33E 05
3.25E 03	7.34E 01	2.99E 02	0.0	1.38E 01	2.16E 06	8.80E 06	0.0	3.35E 05
3.35E 03	7.35E 01	2.99E 02	0.0	1.38E 01	2.17E 06	8.83E 06	0.0	3.36E 05
3.45E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.17E 06	8.85E 06	0.0	3.36E 05
3.55E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.17E 06	8.86E 06	0.0	3.36E 05
3.66E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.17E 06	8.86E 06	0.0	3.37E 05
3.77E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
3.88E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
4.00E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
4.12E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05

4.25E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
4.37E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
4.51E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
4.64E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
4.78E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
4.93E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
5.07E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
5.23E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
5.38E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
5.55E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
5.71E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
5.89E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
6.06E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
6.25E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
6.44E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
6.63E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
6.83E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
7.04E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
7.25E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
7.47E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
7.69E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
7.92E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
8.16E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
8.41E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
8.66E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
8.92E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
9.19E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
9.47E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
9.76E 03	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.00E 04	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.04E 04	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.07E 04	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.10E 04	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.13E 04	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.17E 04	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.20E 04	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.24E 04	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.27E 04	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.31E 04	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.35E 04	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.39E 04	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.44E 04	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.48E 04	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.52E 04	7.35E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.57E 04	7.36E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.62E 04	7.36E 01	3.00E 02	0.0	1.38E 01	2.18E 06	8.87E 06	0.0	3.37E 05
1.67E 04	7.39E 01	3.00E 02	0.0	1.38E 01	2.19E 06	8.88E 06	0.0	3.37E 05
1.72E 04	7.48E 01	3.01E 02	0.0	1.38E 01	2.22E 06	8.91E 06	0.0	3.38E 05
1.77E 04	7.64E 01	3.03E 02	0.0	1.39E 01	2.28E 06	8.99E 06	0.0	3.40E 05
1.82E 04	7.90E 01	3.06E 02	0.0	1.39E 01	2.39E 06	9.11E 06	0.0	3.43E 05

1.88E 04	8.18E 01	3.09E 02	0.0	1.40E 01	2.54E 06	9.28E 06	0.0	3.48E 05
1.93E 04	8.41E 01	3.11E 02	0.0	1.41E 01	2.69E 06	9.44E 06	0.0	3.52E 05
1.99E 04	8.56E 01	3.13E 02	0.0	1.41E 01	2.80E 06	9.56E 06	0.0	3.55E 05
2.05E 04	8.62E 01	3.14E 02	0.0	1.41E 01	2.86E 06	9.63E 06	0.0	3.57E 05
2.11E 04	8.64E 01	3.14E 02	0.0	1.41E 01	2.88E 06	9.65E 06	0.0	3.58E 05
2.18E 04	8.65E 01	3.14E 02	0.0	1.41E 01	2.89E 06	9.66E 06	0.0	3.58E 05
2.24E 04	8.65E 01	3.14E 02	0.0	1.41E 01	2.89E 06	9.66E 06	0.0	3.58E 05
2.31E 04	8.65E 01	3.14E 02	0.0	1.41E 01	2.89E 06	9.66E 06	0.0	3.58E 05
2.38E 04	8.65E 01	3.14E 02	0.0	1.41E 01	2.89E 06	9.66E 06	0.0	3.58E 05
2.45E 04	8.65E 01	3.14E 02	0.0	1.41E 01	2.89E 06	9.66E 06	0.0	3.58E 05
2.52E 04	8.65E 01	3.14E 02	0.0	1.41E 01	2.89E 06	9.66E 06	0.0	3.58E 05
2.60E 04	8.65E 01	3.14E 02	0.0	1.41E 01	2.89E 06	9.66E 06	0.0	3.58E 05
2.68E 04	8.65E 01	3.14E 02	0.0	1.41E 01	2.89E 06	9.66E 06	0.0	3.58E 05

These values correspond to the total shown earlier

SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER
COMMERICAL AQUATIC FOOD PATHWAY

	* * * INDIVIDUAL * * *				* * * POPULATION * * *			
TIME	REM				MANREM			
(DAYS)	T.BODY	BONE	THYROID	GI-LLI	T.BODY	BONE	THYROID	GI-LLI
1.41E 03	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.46E 03	8.78E-10	3.58E-09	0.0	1.03E-10	4.92E-06	2.01E-05	0.0	5.80E-07
1.50E 03	4.92E-08	2.01E-07	0.0	1.01E-08	2.81E-04	1.14E-03	0.0	1.06E-04
1.55E 03	1.82E-06	7.43E-06	0.0	3.05E-07	9.29E-03	3.79E-02	0.0	2.68E-03
1.59E 03	1.36E-05	5.55E-05	0.0	3.95E-06	8.01E-02	3.27E-01	0.0	5.15E-02
1.64E 03	1.20E-04	4.90E-04	0.0	3.88E-05	8.33E-01	3.40E 00	0.0	5.66E-01
1.69E 03	2.51E-03	1.02E-02	0.0	6.04E-04	1.63E 01	6.63E 01	0.0	8.13E 00
1.74E 03	1.62E-02	6.50E-02	0.0	3.81E-03	1.00E 02	4.08E 02	0.0	5.37E 01
1.79E 03	7.33E-02	2.99E-01	0.0	2.08E-02	5.19E 02	2.12E 03	0.0	3.48E 02
1.85E 03	3.16E-01	1.29E 00	0.0	8.45E-02	2.66E 03	1.08E 04	0.0	1.56E 03
1.90E 03	1.14E 00	4.65E 00	0.0	3.00E-01	9.58E 03	3.91E 04	0.0	5.83E 03
1.96E 03	3.29E 00	1.34E 01	0.0	8.14E-01	2.95E 04	1.20E 05	0.0	1.72E 04
2.02E 03	7.28E 00	2.97E 01	0.0	1.77E 00	7.76E 04	3.17E 05	0.0	4.16E 04
2.08E 03	1.39E 01	5.67E 01	0.0	3.24E 00	1.73E 05	7.04E 05	0.0	8.51E 04
2.14E 03	2.33E 01	9.51E 01	0.0	5.11E 00	3.24E 05	1.32E 06	0.0	1.46E 05
2.21E 03	3.38E 01	1.38E 02	0.0	7.09E 00	5.58E 05	2.28E 06	0.0	2.21E 05
2.28E 03	4.43E 01	1.81E 02	0.0	8.90E 00	8.79E 05	3.58E 06	0.0	3.03E 05
2.34E 03	5.33E 01	2.17E 02	0.0	1.03E 01	1.25E 06	5.10E 06	0.0	3.81E 05
2.41E 03	5.96E 01	2.43E 02	0.0	1.13E 01	1.69E 06	6.88E 06	0.0	4.55E 05
2.49E 03	6.44E 01	2.62E 02	0.0	1.20E 01	2.14E 06	8.72E 06	0.0	5.23E 05
2.56E 03	6.77E 01	2.76E 02	0.0	1.25E 01	2.57E 06	1.05E 07	0.0	5.83E 05
2.64E 03	6.98E 01	2.85E 02	0.0	1.28E 01	2.97E 06	1.21E 07	0.0	6.34E 05
2.72E 03	7.11E 01	2.90E 02	0.0	1.30E 01	3.27E 06	1.33E 07	0.0	6.73E 05
2.80E 03	7.19E 01	2.93E 02	0.0	1.31E 01	3.51E 06	1.43E 07	0.0	7.02E 05
2.89E 03	7.25E 01	2.95E 02	0.0	1.32E 01	3.68E 06	1.50E 07	0.0	7.23E 05
2.97E 03	7.28E 01	2.97E 02	0.0	1.32E 01	3.78E 06	1.54E 07	0.0	7.35E 05
3.06E 03	7.30E 01	2.98E 02	0.0	1.33E 01	3.84E 06	1.57E 07	0.0	7.43E 05
3.15E 03	7.31E 01	2.98E 02	0.0	1.33E 01	3.89E 06	1.59E 07	0.0	7.49E 05
3.25E 03	7.32E 01	2.98E 02	0.0	1.33E 01	3.91E 06	1.59E 07	0.0	7.51E 05
3.35E 03	7.32E 01	2.99E 02	0.0	1.33E 01	3.92E 06	1.60E 07	0.0	7.53E 05
3.45E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.93E 06	1.60E 07	0.0	7.54E 05
3.55E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.60E 07	0.0	7.54E 05
3.66E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.60E 07	0.0	7.54E 05
3.77E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
3.88E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
4.00E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
4.12E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
4.25E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
4.37E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
4.51E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
4.64E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
4.78E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
4.93E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
5.07E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
5.23E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
5.38E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
5.55E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05

5.71E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
5.89E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
6.06E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
6.25E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
6.44E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
6.63E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
6.83E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
7.04E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
7.25E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
7.47E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
7.69E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
7.92E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
8.16E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
8.41E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
8.66E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
8.92E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
9.19E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
9.47E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
9.76E 03	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.00E 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.0[illegible] 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
[illegible] 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
[illegible] 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.13E 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.17E 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.20E 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.24E 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.27E 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.31E 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.35E 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.39E 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.44E 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.48E 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.52E 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.57E 04	7.33E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.62E 04	7.34E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.67E 04	7.37E 01	2.99E 02	0.0	1.33E 01	3.94E 06	1.61E 07	0.0	7.55E 05
1.72E 04	7.46E 01	3.00E 02	0.0	1.33E 01	3.95E 06	1.61E 07	0.0	7.55E 05
1.77E 04	7.62E 01	3.02E 02	0.0	1.34E 01	3.97E 06	1.61E 07	0.0	7.56E 05
1.82E 04	7.87E 01	3.05E 02	0.0	1.35E 01	4.00E 06	1.61E 07	0.0	7.56E 05
1.88E 04	8.15E 01	3.08E 02	0.0	1.35E 01	4.04E 06	1.62E 07	0.0	7.58E 05
1.93E 04	8.38E 01	3.11E 02	0.0	1.36E 01	4.08E 06	1.62E 07	0.0	7.59E 05
1.99E 04	8.53E 01	3.12E 02	0.0	1.37E 01	4.11E 06	1.63E 07	0.0	7.60E 05
2.05E 04	8.60E 01	3.13E 02	0.0	1.37E 01	4.13E 06	1.63E 07	0.0	7.60E 05
2.11E 04	8.62E 01	3.13E 02	0.0	1.37E 01	4.14E 06	1.63E 07	0.0	7.60E 05
2.18E 04	8.62E 01	3.13E 02	0.0	1.37E 01	4.14E 06	1.63E 07	0.0	7.61E 05
2.24E 04	8.62E 01	3.13E 02	0.0	1.37E 01	4.14E 06	1.63E 07	0.0	7.61E 05
2.31E 04	8.62E 01	3.13E 02	0.0	1.37E 01	4.14E 06	1.63E 07	0.0	7.61E 05
2.38E 04	8.62E 01	3.13E 02	0.0	1.37E 01	4.14E 06	1.63E 07	0.0	7.61E 05
2.45E 04	8.62E 01	3.13E 02	0.0	1.37E 01	4.14E 06	1.63E 07	0.0	7.61E 05
2.52E 04	8.62E 01	3.13E 02	0.0	1.37E 01	4.14E 06	1.63E 07	0.0	7.61E 05
2.60E 04	8.62E 01	3.13E 02	0.0	1.37E 01	4.14E 06	1.63E 07	0.0	7.61E 05
2.68E 04	8.62E 01	3.13E 02	0.0	1.37E 01	4.14E 06	1.63E 07	0.0	7.61E 05

SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER
TOTAL AQUATIC FOOD PATHWAY

These data are the sum of the recreational and commercial aquatic food harvests

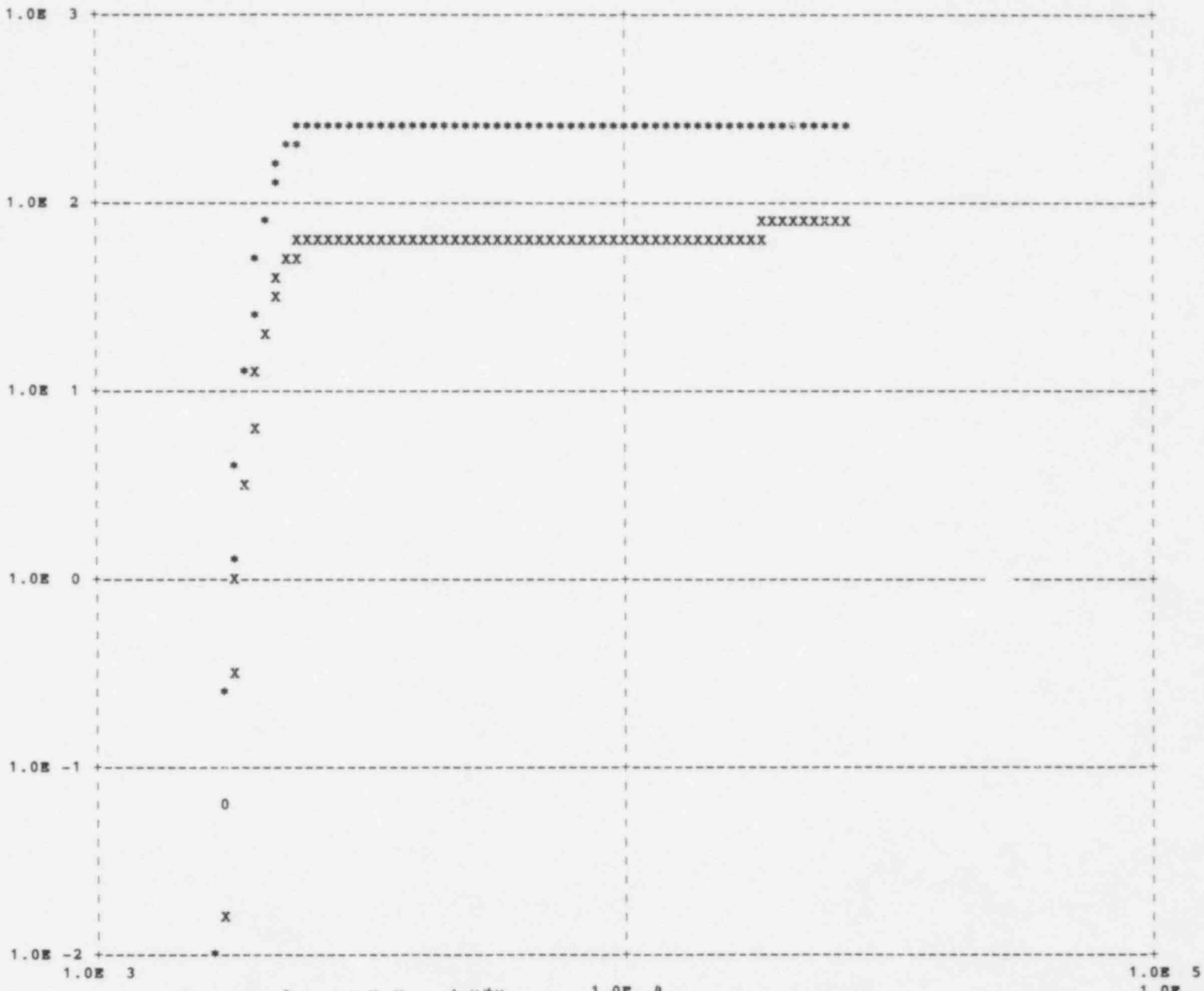
TIME	* * * I N D I V I D U A L * * *				* * * P O P U L A T I O N * * *			
	R E M				M A N R E M			
(DAYS)	T.BODY	BONE	THYROID	GI-LLI	T.BODY	BONE	THYROID	GI-LLI
1.41E 03	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1.46E 03	8.80E-10	3.59E-09	0.0	1.04E-10	7.64E-06	3.11E-05	0.0	9.00E-07
1.50E 03	4.94E-08	2.01E-07	0.0	1.05E-09	4.36E-04	1.78E-03	0.0	1.45E-04
1.55E 03	1.83E-06	7.45E-06	0.0	3.14E-07	1.44E-02	5.88E-02	0.0	3.72E-03
1.59E 03	1.37E-05	5.57E-05	0.0	4.18E-06	1.24E-01	5.07E-01	0.0	6.84E-02
1.64E 03	1.21E-04	4.92E-04	0.0	4.11E-05	1.29E 00	5.27E 00	0.0	7.49E-01
1.69E 03	2.52E-03	1.03E-02	0.0	6.34E-04	2.53E 01	1.03E 02	0.0	1.09E 01
1.74E 03	1.62E-02	6.60E-02	0.0	3.99E-03	1.55E 02	6.33E 02	0.0	7.18E 01
1.79E 03	7.35E-02	3.00E-01	0.0	2.20E-02	8.06E 02	3.29E 03	0.0	4.61E 02
1.85E 03	3.17E-01	1.29E 00	0.0	8.91E-02	4.12E 03	1.68E 04	0.0	2.08E 03
1.90E 03	1.14E 00	4.66E 00	0.0	3.16E-01	1.49E 04	6.06E 04	0.0	7.76E 03
1.96E 03	3.30E 00	1.35E 01	0.0	8.55E-01	4.57E 04	1.86E 05	0.0	2.29E 04
2.02E 03	7.30E 00	2.98E 01	0.0	1.85E 00	1.20E 05	4.91E 05	0.0	5.57E 04
2.08E 03	1.40E 01	5.69E 01	0.0	3.39E 00	2.68E 05	1.09E 06	0.0	1.14E 05
2.14E 03	2.34E 01	9.54E 01	0.0	5.35E 00	5.03E 05	2.05E 06	0.0	1.97E 05
2.21E 03	3.39E 01	1.38E 02	0.0	7.40E 00	8.66E 05	3.53E 06	0.0	3.01E 05
2.28E 03	4.45E 01	1.81E 02	0.0	9.26E 00	1.36E 06	5.56E 06	0.0	4.16E 05
2.34E 03	5.35E 01	2.18E 02	0.0	1.08E 01	1.94E 06	7.92E 06	0.0	5.27E 05
2.41E 03	5.98E 01	2.44E 02	0.0	1.18E 01	2.62E 06	1.07E 07	0.0	6.36E 05
2.49E 03	6.46E 01	2.63E 02	0.0	1.25E 01	3.32E 06	1.35E 07	0.0	7.37E 05
2.56E 03	6.79E 01	2.77E 02	0.0	1.30E 01	3.99E 06	1.63E 07	0.0	8.28E 05
2.64E 03	7.00E 01	2.85E 02	0.0	1.33E 01	4.60E 06	1.88E 07	0.0	9.06E 05
2.72E 03	7.13E 01	2.91E 02	0.0	1.35E 01	5.08E 06	2.07E 07	0.0	9.65E 05
2.80E 03	7.22E 01	2.94E 02	0.0	1.36E 01	5.45E 06	2.22E 07	0.0	1.01E 06
2.89E 03	7.27E 01	2.96E 02	0.0	1.37E 01	5.71E 06	2.33E 07	0.0	1.04E 06
2.97E 03	7.31E 01	2.98E 02	0.0	1.37E 01	5.86E 06	2.39E 07	0.0	1.06E 06
3.06E 03	7.33E 01	2.99E 02	0.0	1.37E 01	5.97E 06	2.43E 07	0.0	1.07E 06
3.15E 03	7.34E 01	2.99E 02	0.0	1.37E 01	6.04E 06	2.46E 07	0.0	1.08E 06
3.25E 03	7.34E 01	2.99E 02	0.0	1.38E 01	6.07E 06	2.47E 07	0.0	1.09E 06
3.35E 03	7.35E 01	2.99E 02	0.0	1.38E 01	6.09E 06	2.48E 07	0.0	1.09E 06
3.45E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.10E 06	2.49E 07	0.0	1.09E 06
3.55E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.11E 06	2.49E 07	0.0	1.09E 06
3.66E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.11E 06	2.49E 07	0.0	1.09E 06
3.77E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.11E 06	2.49E 07	0.0	1.09E 06
3.88E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
4.00E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
4.12E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
4.25E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
4.37E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
4.51E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
4.64E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
4.78E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
4.93E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
5.07E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
5.23E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
5.38E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
5.55E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
5.71E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06

5.89E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
6.06E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
6.25E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
6.44E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
6.63E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
6.83E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
7.04E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
7.25E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
7.47E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
7.69E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
7.92E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
8.16E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
8.41E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
8.66E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
8.92E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
9.19E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
9.47E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
9.76E 03	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.00E 04	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.04E 04	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.07E 04	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.10E 04	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.13E 04	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.17E 04	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.20E 04	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.24E 04	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.27E 04	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.31E 04	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.35E 04	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.39E 04	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.44E 04	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.48E 04	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.52E 04	7.35E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.57E 04	7.36E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.62E 04	7.36E 01	3.00E 02	0.0	1.38E 01	6.12E 06	2.49E 07	0.0	1.09E 06
1.67E 04	7.39E 01	3.00E 02	0.0	1.38E 01	6.13E 06	2.49E 07	0.0	1.09E 06
1.72E 04	7.48E 01	3.01E 02	0.0	1.38E 01	6.17E 06	2.50E 07	0.0	1.09E 06
1.77E 04	7.64E 01	3.03E 02	0.0	1.39E 01	6.25E 06	2.51E 07	0.0	1.10E 06
1.82E 04	7.90E 01	3.06E 02	0.0	1.39E 01	6.39E 06	2.52E 07	0.0	1.10E 06
1.88E 04	8.18E 01	3.09E 02	0.0	1.40E 01	6.59E 06	2.55E 07	0.0	1.11E 06
1.93E 04	8.41E 01	3.11E 02	0.0	1.41E 01	6.77E 06	2.57E 07	0.0	1.11E 06
1.99E 04	8.56E 01	3.13E 02	0.0	1.41E 01	6.91E 06	2.58E 07	0.0	1.11E 06
2.05E 04	8.62E 01	3.14E 02	0.0	1.41E 01	6.99E 06	2.59E 07	0.0	1.12E 06
2.11E 04	8.64E 01	3.14E 02	0.0	1.41E 01	7.02E 06	2.59E 07	0.0	1.12E 06
2.18E 04	8.65E 01	3.14E 02	0.0	1.41E 01	7.03E 06	2.59E 07	0.0	1.12E 06
2.24E 04	8.65E 01	3.14E 02	0.0	1.41E 01	7.03E 06	2.59E 07	0.0	1.12E 06
2.31E 04	8.65E 01	3.14E 02	0.0	1.41E 01	7.03E 06	2.59E 07	0.0	1.12E 06
2.38E 04	8.65E 01	3.14E 02	0.0	1.41E 01	7.03E 06	2.59E 07	0.0	1.12E 06
2.45E 04	8.65E 01	3.14E 02	0.0	1.41E 01	7.03E 06	2.59E 07	0.0	1.12E 06
2.52E 04	8.65E 01	3.14E 02	0.0	1.41E 01	7.03E 06	2.59E 07	0.0	1.12E 06
2.60E 04	8.65E 01	3.14E 02	0.0	1.41E 01	7.03E 06	2.59E 07	0.0	1.12E 06
2.68E 04	8.65E 01	3.14E 02	0.0	1.41E 01	7.03E 06	2.59E 07	0.0	1.12E 06

These values appear in an earlier summary tabulation

SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER

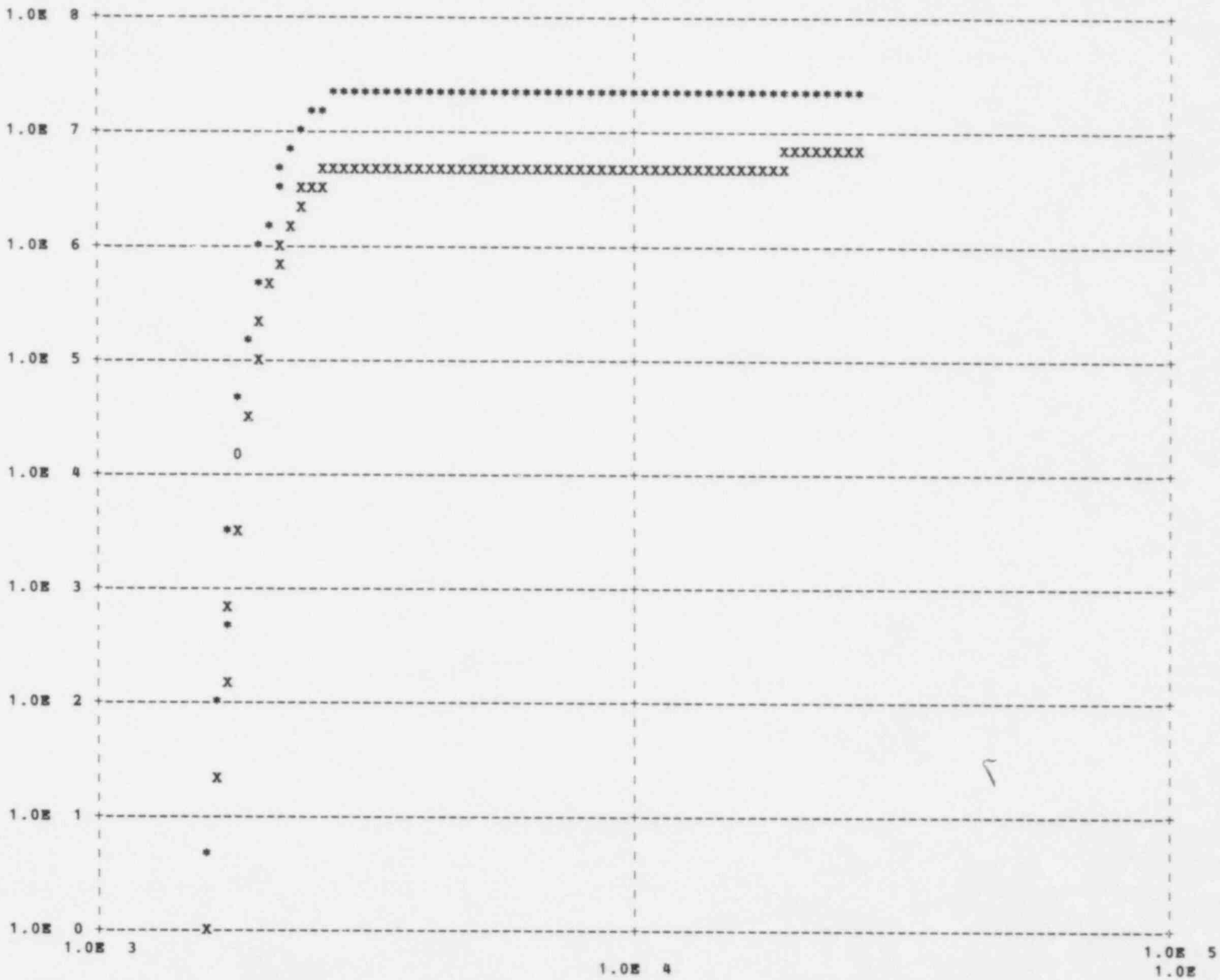
MAX INDIVIDUAL T.BODY (X) AND BONE (*) DOSE (REM) VIA AQUATIC FOOD VS TIME (DAYS)



0 - overlap of "x" and "*"

SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER

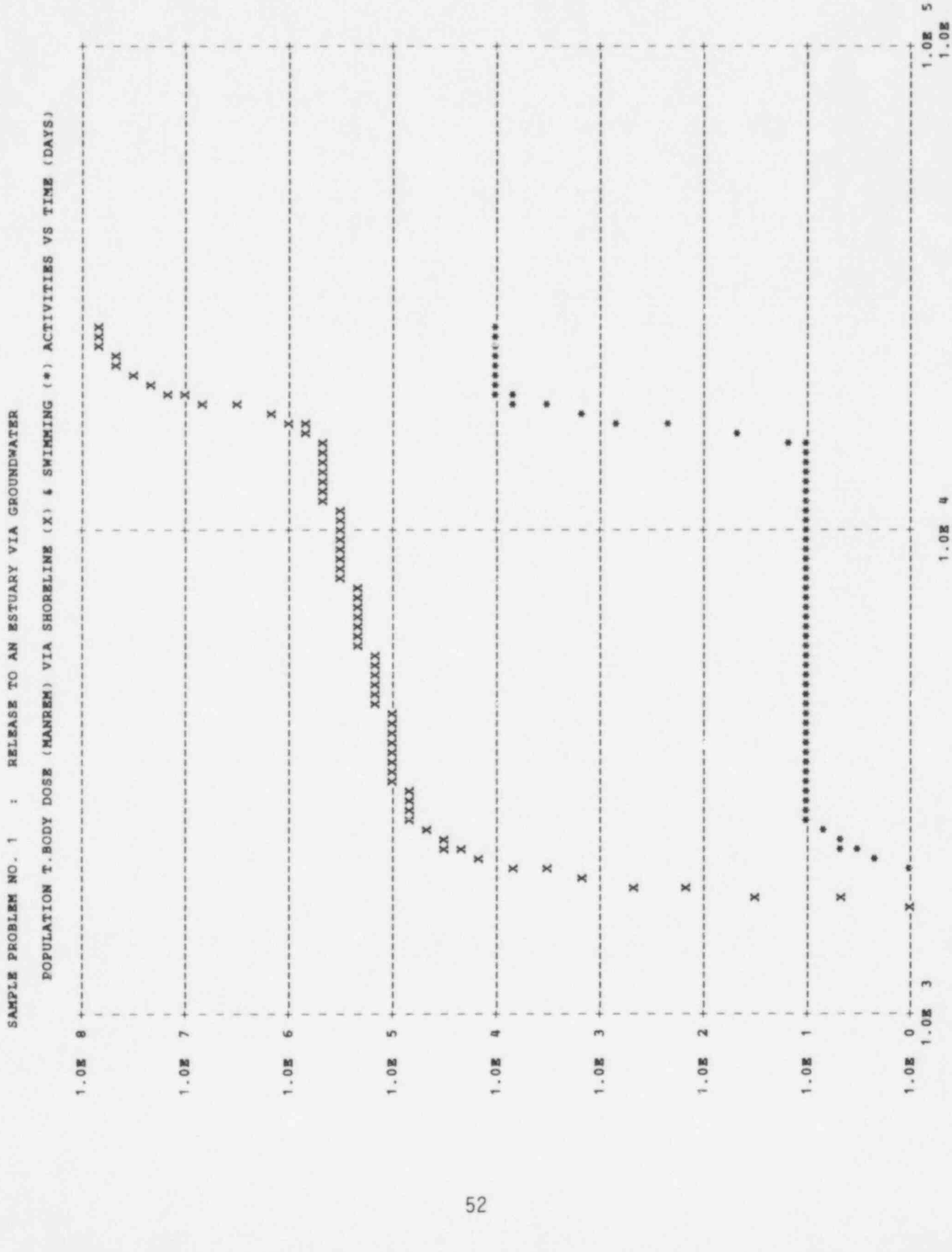
POPULATION T.BODY (X) AND BONE (*) DOSE (MANREM) VIA AQUATIC FOODS VS TIME (DAYS)



Total body dose

SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER
RECREATIONAL PATHWAY

TIME (DAYS)	* * SHORELINE * * INDIVIDUAL (REM)	POPULTN (MANREM)	* * SWIMMING * * INDIVIDUAL (REM)	POPULTN (MANREM)	TIME (DAYS)	* * SHORELINE * * INDIVIDUAL (REM)	POPULTN (MANREM)	* * SWIMMING * * INDIVIDUAL (REM)	POPULTN (MANREM)
1.41E 03	0.0	0.0	0.0	0.0	6.25E 03	6.93E-02	2.36E 05	3.80E-06	1.14E 01
1.46E 03	0.0	0.0	3.00E-16	9.32E-11	6.44E 03	7.18E-02	2.44E 05	3.80E-06	1.14E 01
1.50E 03	8.01E-12	7.86E-06	1.60E-14	5.06E-09	6.63E 03	7.43E-02	2.53E 05	3.80E-06	1.14E 01
1.55E 03	1.71E-10	1.72E-04	5.61E-13	1.60E-07	6.83E 03	7.70E-02	2.62E 05	3.80E-06	1.14E 01
1.59E 03	4.49E-09	4.62E-03	4.08E-12	1.33E-06	7.04E 03	7.97E-02	2.71E 05	3.80E-06	1.14E 01
1.64E 03	4.75E-08	5.18E-02	3.53E-11	1.35E-05	7.25E 03	8.26E-02	2.80E 05	3.80E-06	1.14E 01
1.69E 03	6.03E-07	6.97E-01	6.94E-10	2.48E-04	7.47E 03	8.55E-02	2.90E 05	3.80E-06	1.14E 01
1.74E 03	3.78E-06	4.76E 00	4.16E-09	1.44E-03	7.69E 03	8.85E-02	3.00E 05	3.80E-06	1.14E 01
1.79E 03	2.47E-05	3.33E 01	1.74E-08	6.96E-03	7.92E 03	9.15E-02	3.10E 05	3.80E-06	1.14E 01
1.85E 03	9.81E-05	1.47E 02	7.08E-08	3.32E-02	8.16E 03	9.47E-02	3.21E 05	3.80E-06	1.14E 01
1.90E 03	3.56E-04	5.75E 02	2.31E-07	1.11E-01	8.41E 03	9.80E-02	3.32E 05	3.80E-06	1.14E 01
1.96E 03	9.52E-04	1.73E 03	5.70E-07	3.03E-01	8.66E 03	1.01E-01	3.43E 05	3.80E-06	1.14E 01
2.02E 03	2.15E-03	4.28E 03	1.12E-06	7.10E-01	8.92E 03	1.05E-01	3.55E 05	3.80E-06	1.14E 01
2.08E 03	4.01E-03	9.01E 03	1.89E-06	1.42E 00	9.19E 03	1.08E-01	3.67E 05	3.80E-06	1.14E 01
2.14E 03	6.41E-03	1.60E 04	2.60E-06	2.37E 00	9.47E 03	1.12E-01	3.79E 05	3.80E-06	1.14E 01
2.21E 03	9.15E-03	2.49E 04	3.17E-06	3.60E 00	9.76E 03	1.16E-01	3.92E 05	3.80E-06	1.14E 01
2.28E 03	1.19E-02	3.47E 04	3.56E-06	5.01E 00	1.00E 04	1.20E-01	4.05E 05	3.80E-06	1.14E 01
2.34E 03	1.45E-02	4.45E 04	3.74E-06	6.44E 00	1.04E 04	1.24E-01	4.18E 05	3.80E-06	1.14E 01
2.41E 03	1.68E-02	5.38E 04	3.77E-06	7.91E 00	1.07E 04	1.28E-01	4.32E 05	3.80E-06	1.14E 01
2.49E 03	1.90E-02	6.22E 04	3.79E-06	9.15E 00	1.10E 04	1.32E-01	4.47E 05	3.80E-06	1.14E 01
2.56E 03	2.09E-02	6.98E 04	3.80E-06	1.02E 01	1.13E 04	1.37E-01	4.61E 05	3.80E-06	1.14E 01
2.64E 03	2.26E-02	7.66E 04	3.80E-06	1.08E 01	1.17E 04	1.41E-01	4.77E 05	3.80E-06	1.14E 01
2.72E 03	2.41E-02	8.26E 04	3.80E-06	1.11E 01	1.20E 04	1.46E-01	4.92E 05	3.80E-06	1.14E 01
2.80E 03	2.55E-02	8.79E 04	3.80E-06	1.13E 01	1.24E 04	1.51E-01	5.08E 05	3.80E-06	1.14E 01
2.89E 03	2.66E-02	9.26E 04	3.80E-06	1.14E 01	1.27E 04	1.56E-01	5.25E 05	3.80E-06	1.14E 01
2.97E 03	2.75E-02	9.65E 04	3.80E-06	1.14E 01	1.31E 04	1.61E-01	5.42E 05	3.80E-06	1.14E 01
3.06E 03	2.85E-02	9.99E 04	3.80E-06	1.14E 01	1.35E 04	1.66E-01	5.60E 05	3.80E-06	1.14E 01
3.15E 03	2.94E-02	1.03E 05	3.80E-06	1.14E 01	1.39E 04	1.71E-01	5.78E 05	3.80E-06	1.14E 01
3.25E 03	3.04E-02	1.06E 05	3.80E-06	1.14E 01	1.44E 04	1.77E-01	5.97E 05	3.80E-06	1.14E 01
3.35E 03	3.14E-02	1.10E 05	3.80E-06	1.14E 01	1.48E 04	1.83E-01	6.16E 05	3.91E-06	1.15E 01
3.45E 03	3.25E-02	1.13E 05	3.80E-06	1.14E 01	1.52E 04	1.89E-01	6.36E 05	4.89E-06	1.27E 01
3.55E 03	3.36E-02	1.17E 05	3.80E-06	1.14E 01	1.57E 04	1.97E-01	6.59E 05	1.11E-05	2.01E 01
3.66E 03	3.49E-02	1.21E 05	3.80E-06	1.14E 01	1.62E 04	2.11E-01	6.98E 05	4.22E-05	6.74E 01
3.77E 03	3.63E-02	1.25E 05	3.80E-06	1.14E 01	1.67E 04	2.52E-01	7.91E 05	1.[illegible]8E-04	2.39E 02
3.88E 03	3.78E-02	1.30E 05	3.80E-06	1.14E 01	1.72E 04	3.73E-01	1.10E 06	4.38E-04	8.03E 02
4.00E 03	3.94E-02	1.35E 05	3.80E-06	1.14E 01	1.77E 04	6.58E-01	1.89E 06	9.92E-04	2.04E 03
4.12E 03	4.10E-02	1.41E 05	3.80E-06	1.14E 01	1.82E 04	1.24E 00	3.68E 06	1.81E-03	4.15E 03
4.25E 03	4.27E-02	1.46E 05	3.80E-06	1.14E 01	1.88E 04	2.18E 00	6.90E 06	2.72E-03	6.91E 03
4.37E 03	4.44E-02	1.52E 05	3.80E-06	1.14E 01	1.93E 04	3.47E 00	1.17E 07	3.44E-03	9.50E 03
4.51E 03	4.61E-02	1.58E 05	3.80E-06	1.14E 01	1.99E 04	5.02E 00	1.79E 07	3.88E-03	1.14E 04
4.64E 03	4.79E-02	1.64E 05	3.80E-06	1.14E 01	2.05E 04	6.69E 00	2.50E 07	4.08E-03	1.24E 04
4.78E 03	4.98E-02	1.70E 05	3.80E-06	1.14E 01	2.11E 04	8.42E 00	3.25E 07	4.13E-03	1.28E 04
4.93E 03	5.17E-02	1.77E 05	3.80E-06	1.14E 01	2.18E 04	1.01E 01	4.02E 07	4.15E-03	1.29E 04
5.07E 03	5.37E-02	1.83E 05	3.80E-06	1.14E 01	2.24E 04	1.19E 01	4.77E 07	4.15E-03	1.29E 04
5.23E 03	5.57E-02	1.90E 05	3.80E-06	1.14E 01	2.31E 04	1.35E 01	5.52E 07	4.15E-03	1.29E 04
5.38E 03	5.78E-02	1.97E 05	3.80E-06	1.14E 01	2.38E 04	1.52E 01	6.26E 07	4.15E-03	1.29E 04
5.55E 03	6.00E-02	2.04E 05	3.80E-06	1.14E 01	2.45E 04	1.69E 01	6.99E 07	4.15E-03	1.29E 04
5.71E 03	6.22E-02	2.12E 05	3.80E-06	1.14E 01	2.52E 04	1.85E 01	7.71E 07	4.15E-03	1.29E 04
5.89E 03	6.45E-02	2.20E 05	3.80E-06	1.14E 01	2.60E 04	2.01E 01	8.41E 07	4.15E-03	1.29E 04
6.06E 03	6.68E-02	2.27E 05	3.80E-06	1.14E 01	2.68E 04	2.17E 01	9.11E 07	4.15E-03	1.29E 04



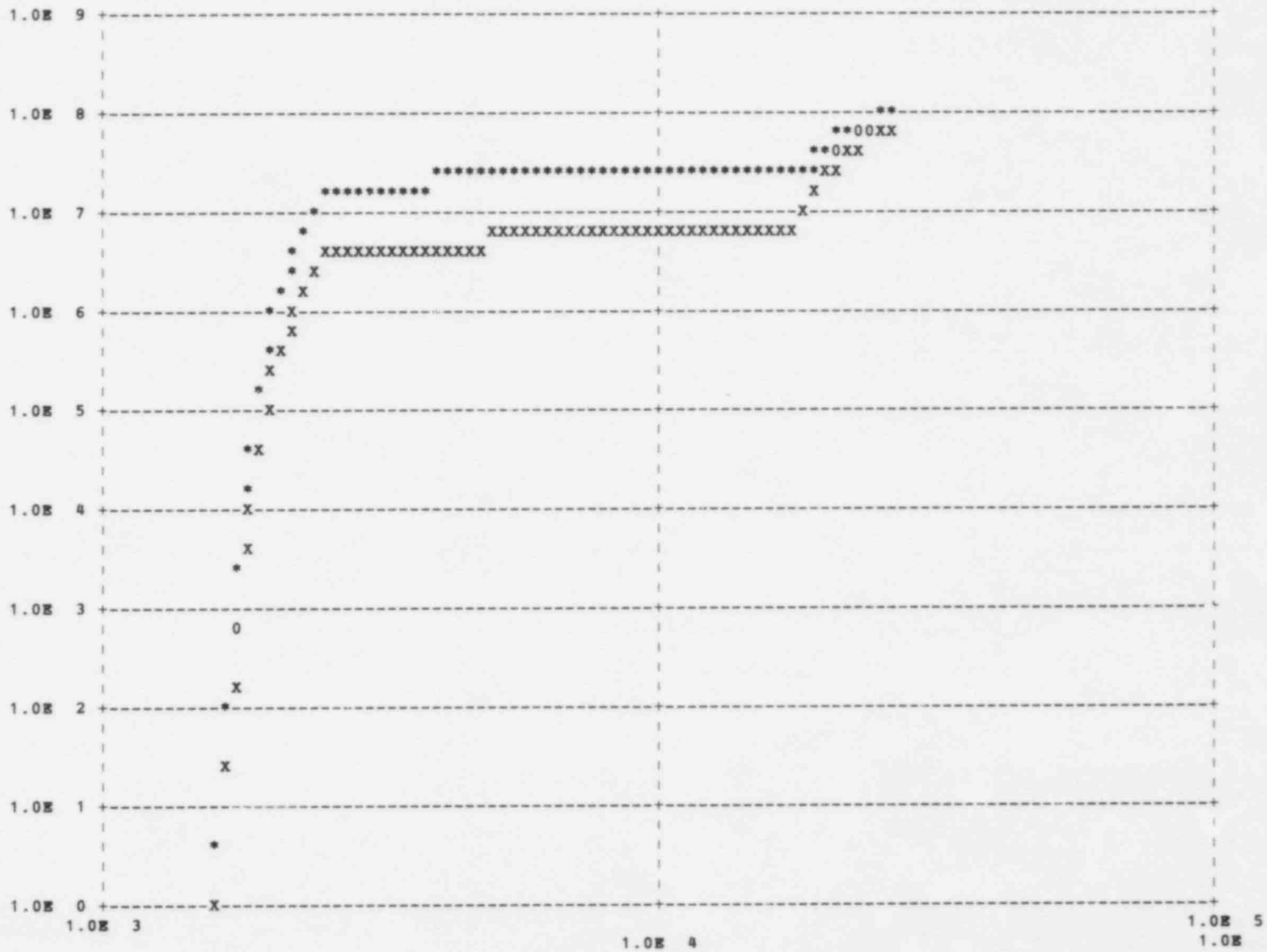
SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER
POPULATION T.BODY DOSE (MANREM) VIA SHORELINE (X) & SWIMMING (*) ACTIVITIES VS TIME (DAYS)
1.0E 8
1.0E 7
1.0E 6
1.0E 5
1.0E 4
1.0E 3
1.0E 2
1.0E 1
1.0E 0
1.0E 3
1.0E 4
1.0E 5
1.0E

SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER
TOTAL POPULATION DOSE (MANREM) ⟶ *Sum over all pathways*

(DAYS)	T.BODY	BONE	THYROID	GI-LLI	(DAYS)	T.BODY	BONE	THYROID	GI-LLI
1.41E 03	0.0	0.0	0.0	0.0	6.25E 03	6.35E 06	2.52E 07	2.36E 05	1.33E 06
1.46E 03	7.64E-06	3.11E-05	9.32E-11	9.00E-07	6.44E 03	6.36E 06	2.52E 07	2.44E 05	1.34E 06
1.50E 03	4.43E-04	1.78E-03	7.87E-06	1.53E-04	6.63E 03	6.37E 06	2.52E 07	2.53E 05	1.34E 06
1.55E 03	1.46E-02	5.90E-02	1.72E-04	3.89E-03	6.83E 03	6.38E 06	2.52E 07	2.62E 05	1.35E 06
1.59E 03	1.29E-01	5.11E-01	4.63E-03	7.30E-02	7.04E 03	6.39E 06	2.52E 07	2.71E 05	1.36E 06
1.64E 03	1.34E 00	5.33E 00	5.18E-02	8.01E-01	7.25E 03	6.40E 06	2.52E 07	2.80E 05	1.37E 06
1.69E 03	2.59E 01	1.04E 02	6.97E-01	1.16E 01	7.47E 03	6.41E 06	2.52E 07	2.90E 05	1.38E 06
1.74E 03	1.60E 02	6.28E 02	4.76E 00	7.66E 01	7.69E 03	6.42E 06	2.52E 07	3.00E 05	1.39E 06
1.79E 03	8.39E 02	3.32E 03	3.33E 01	4.94E 02	7.92E 03	6.43E 06	2.52E 07	3.10E 05	1.40E 06
1.85E 03	4.27E 03	1.70E 04	1.47E 02	2.22E 03	8.16E 03	6.44E 06	2.53E 07	3.21E 05	1.41E 06
1.90E 03	1.54E 04	6.13E 04	5.75E 02	8.33E 03	8.41E 03	6.45E 06	2.53E 07	3.32E 05	1.42E 06
1.96E 03	4.74E 04	1.88E 05	1.73E 03	2.46E 04	8.66E 03	6.46E 06	2.53E 07	3.43E 05	1.43E 06
2.02E 03	1.25E 05	4.95E 05	4.28E 03	5.99E 04	8.92E 03	6.47E 06	2.53E 07	3.55E 05	1.45E 06
2.08E 03	2.77E 05	1.10E 06	9.01E 03	1.23E 05	9.19E 03	6.48E 06	2.53E 07	3.67E 05	1.46E 06
2.14E 03	5.19E 05	2.07E 06	1.60E 04	2.13E 05	9.47E 03	6.50E 06	2.53E 07	3.79E 05	1.47E 06
2.21E 03	8.91E 05	3.56E 06	2.49E 04	3.26E 05	9.76E 03	6.51E 06	2.53E 07	3.92E 05	1.48E 06
2.28E 03	1.40E 06	5.60E 06	3.47E 04	4.50E 05	1.00E 04	6.52E 06	2.53E 07	4.05E 05	1.50E 06
2.34E 03	1.99E 06	7.97E 06	4.45E 04	5.71E 05	1.04E 04	6.54E 06	2.53E 07	4.18E 05	1.51E 06
2.41E 03	2.67E 06	1.07E 07	5.38E 04	6.90E 05	1.07E 04	6.55E 06	2.54E 07	4.32E 05	1.52E 06
2.49E 03	3.38E 06	1.36E 07	6.22E 04	8.00E 05	1.10E 04	6.56E 06	2.54E 07	4.47E 05	1.54E 06
2.56E 03	4.06E 06	1.64E 07	6.98E 04	8.97E 05	1.13E 04	6.58E 06	2.54E 07	4.61E 05	1.55E 06
2.64E 03	4.68E 06	1.88E 07	7.66E 04	9.82E 05	1.17E 04	6.59E 06	2.54E 07	4.77E 05	1.57E 06
2.72E 03	5.16E 06	2.08E 07	8.26E 04	1.05E 06	1.20E 04	6.61E 06	2.54E 07	4.92E 05	1.58E 06
2.80E 03	5.53E 06	2.23E 07	8.79E 04	1.10E 06	1.24E 04	6.62E 06	2.54E 07	5.08E 05	1.60E 06
2.89E 03	5.80E 06	2.34E 07	9.26E 04	1.14E 06	1.27E 04	6.64E 06	2.55E 07	5.25E 05	1.62E 06
2.97E 03	5.96E 06	2.40E 07	9.65E 04	1.16E 06	1.31E 04	6.66E 06	2.55E 07	5.42E 05	1.63E 06
3.06E 03	6.07E 06	2.44E 07	9.99E 04	1.17E 06	1.35E 04	6.68E 06	2.55E 07	5.60E 05	1.65E 06
3.15E 03	6.14E 06	2.47E 07	1.03E 05	1.19E 06	1.39E 04	6.69E 06	2.55E 07	5.78E 05	1.67E 06
3.25E 03	6.17E 06	2.48E 07	1.06E 05	1.19E 06	1.44E 04	6.71E 06	2.55E 07	5.97E 05	1.69E 06
3.35E 03	6.20E 06	2.49E 07	1.10E 05	1.20E 06	1.48E 04	6.73E 06	2.55E 07	6.16E 05	1.71E 06
3.45E 03	6.22E 06	2.50E 07	1.13E 05	1.20E 06	1.52E 04	6.75E 06	2.56E 07	6.36E 05	1.73E 06
3.55E 03	6.23E 06	2.50E 07	1.17E 05	1.21E 06	1.57E 04	6.78E 06	2.56E 07	6.59E 05	1.75E 06
3.66E 03	6.23E 06	2.50E 07	1.21E 05	1.21E 06	1.62E 04	6.82E 06	2.56E 07	6.98E 05	1.79E 06
3.77E 03	6.24E 06	2.50E 07	1.25E 05	1.22E 06	1.67E 04	6.92E 06	2.57E 07	7.91E 05	1.88E 06
3.88E 03	6.25E 06	2.51E 07	1.30E 05	1.22E 06	1.72E 04	7.27E 06	2.61E 07	1.10E 06	2.19E 06
4.00E 03	6.25E 06	2.51E 07	1.36E 05	1.23E 06	1.77E 04	8.14E 06	2.70E 07	1.89E 06	2.98E 06
4.12E 03	6.26E 06	2.51E 07	1.41E 05	1.23E 06	1.82E 04	1.01E 07	2.89E 07	3.69E 06	4.79E 06
4.25E 03	6.26E 06	2.51E 07	1.46E 05	1.24E 06	1.88E 04	1.35E 07	3.24E 07	6.90E 06	8.01E 06
4.37E 03	6.27E 06	2.51E 07	1.52E 05	1.24E 06	1.93E 04	1.85E 07	3.74E 07	1.17E 07	1.28E 07
4.51E 03	6.27E 06	2.51E 07	1.58E 05	1.25E 06	1.99E 04	2.49E 07	4.38E 07	1.80E 07	1.91E 07
4.64E 03	6.28E 06	2.51E 07	1.64E 05	1.26E 06	2.05E 04	3.20E 07	5.09E 07	2.50E 07	2.62E 07
4.78E 03	6.29E 06	2.51E 07	1.70E 05	1.26E 06	2.11E 04	3.96E 07	5.85E 07	3.26E 07	3.37E 07
4.93E 03	6.29E 06	2.51E 07	1.77E 05	1.27E 06	2.18E 04	4.72E 07	6.61E 07	4.02E 07	4.13E 07
5.07E 03	6.30E 06	2.51E 07	1.83E 05	1.27E 06	2.24E 04	5.48E 07	7.37E 07	4.77E 07	4.89E 07
5.23E 03	6.31E 06	2.51E 07	1.90E 05	1.28E 06	2.31E 04	6.23E 07	8.12E 07	5.52E 07	5.64E 07
5.38E 03	6.31E 06	2.51E 07	1.97E 05	1.29E 06	2.38E 04	6.97E 07	8.86E 07	6.26E 07	6.38E 07
5.55E 03	6.32E 06	2.51E 07	2.04E 05	1.30E 06	2.45E 04	7.70E 07	9.59E 07	6.99E 07	7.10E 07
5.71E 03	6.33E 06	2.51E 07	2.12E 05	1.30E 06	2.52E 04	8.41E 07	1.03E 08	7.71E 07	7.82E 07
5.89E 03	6.34E 06	2.51E 07	2.20E 05	1.31E 06	2.60E 04	9.12E 07	1.10E 08	8.41E 07	8.53E 07
6.06E 03	6.34E 06	2.52E 07	2.27E 05	1.32E 06	2.68E 04	9.82E 07	1.17E 08	9.11E 07	9.23E 07

SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER

POPULATION T.BODY (X) AND BONE (*) DOSE (MANREM) VIA ALL PATHWAYS VS TIME (DAYS)



Dose to aquatic biota in the first region; note that the data represents the integral of the dose rate to the various times; caution should be exercised in using this information over long time periods

SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER
FISH AND INVERTEBRATE DOSES

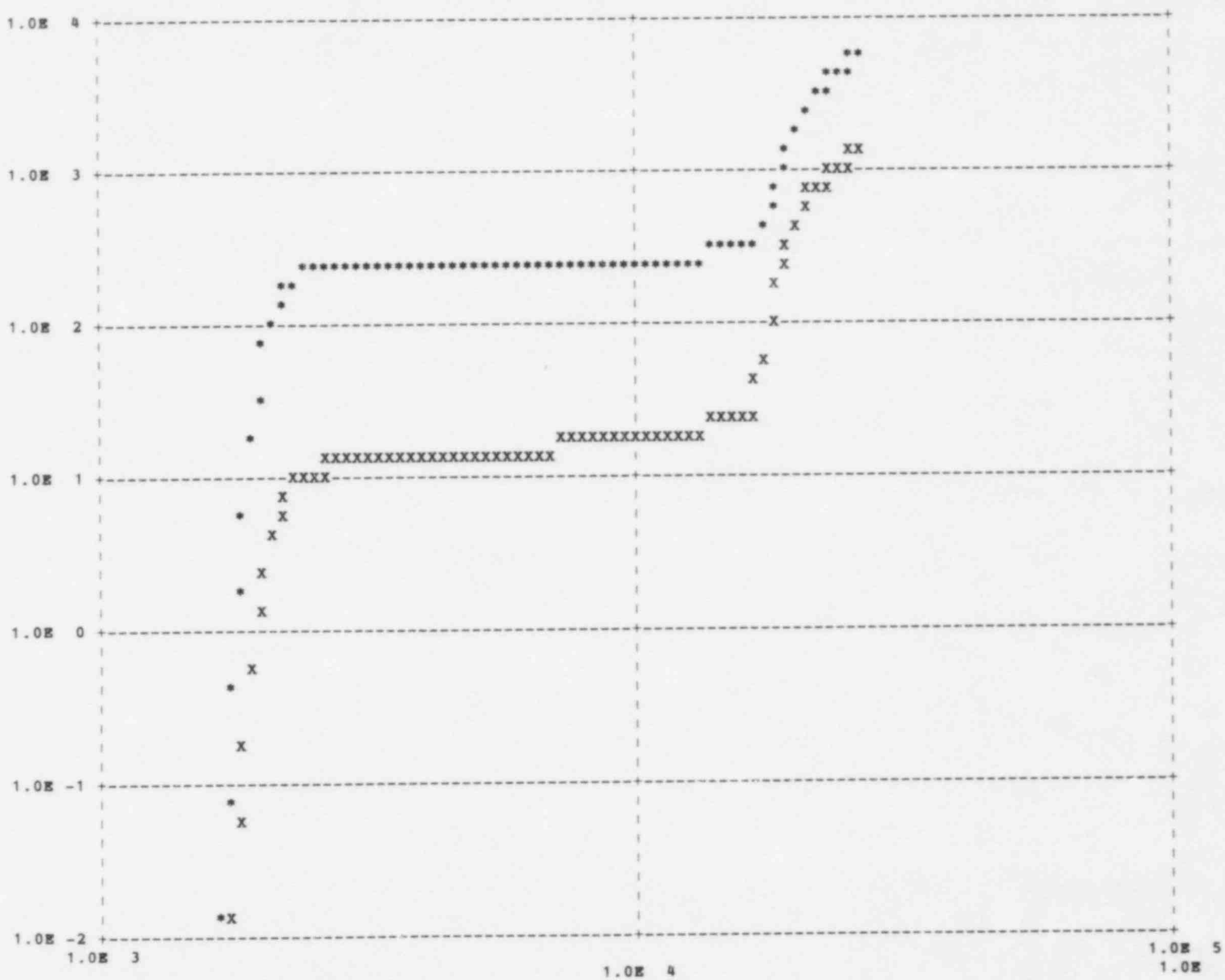
	* * * FISH * * *		* INVERTEBRATE *	
TIME (DAYS)	INTERNAL (RAD)	TOTAL (RAD)	INTERNAL (RAD)	TOTAL (RAD)
1.41E 03	0.0	0.0	0.0	0.0
1.46E 03	1.42E-10	1.42E-10	1.42E-09	1.42E-09
1.50E 03	8.40E-09	8.94E-09	2.19E-07	2.22E-07
1.55E 03	3.04E-07	3.16E-07	5.90E-06	5.94E-06
1.59E 03	2.45E-06	2.75E-06	9.88E-05	1.00E-04
1.64E 03	2.20E-05	2.52E-05	1.00E-03	1.01E-03
1.69E 03	4.38E-04	4.79E-04	1.41E-02	1.43E-02
1.74E 03	2.81E-03	3.06E-03	8.85E-02	8.95E-02
1.79E 03	1.31E-02	1.48E-02	5.17E-01	5.23E-01
1.85E 03	5.60E-02	6.26E-02	2.06E 00	2.08E 00
1.90E 03	2.01E-01	2.25E-01	7.26E 00	7.35E 00
1.96E 03	5.76E-01	6.40E-01	1.93E 01	1.95E 01
2.02E 03	1.27E 00	1.41E 00	4.15E 01	4.21E 01
2.08E 03	2.41E 00	2.68E 00	7.48E 01	7.59E 01
2.14E 03	4.02E 00	4.44E 00	1.15E 02	1.17E 02
2.21E 03	5.79E 00	6.40E 00	1.56E 02	1.59E 02
2.28E 03	7.55E 00	8.34E 00	1.92E 02	1.95E 02
2.34E 03	9.04E 00	1.00E 01	2.19E 02	2.23E 02
2.41E 03	1.01E 01	1.12E 01	2.38E 02	2.42E 02
2.49E 03	1.09E 01	1.21E 01	2.50E 02	2.55E 02
2.56E 03	1.14E 01	1.28E 01	2.59E 02	2.64E 02
2.64E 03	1.18E 01	1.33E 01	2.64E 02	2.70E 02
2.72E 03	1.20E 01	1.36E 01	2.67E 02	2.73E 02
2.80E 03	1.21E 01	1.38E 01	2.68E 02	2.75E 02
2.89E 03	1.22E 01	1.40E 01	2.70E 02	2.77E 02
2.97E 03	1.23E 01	1.41E 01	2.70E 02	2.78E 02
3.06E 03	1.23E 01	1.42E 01	2.71E 02	2.78E 02
3.15E 03	1.23E 01	1.43E 01	2.71E 02	2.79E 02
3.25E 03	1.23E 01	1.43E 01	2.71E 02	2.79E 02
3.35E 03	1.23E 01	1.44E 01	2.71E 02	2.79E 02
3.45E 03	1.23E 01	1.45E 01	2.71E 02	2.80E 02
3.55E 03	1.23E 01	1.46E 01	2.71E 02	2.80E 02
3.66E 03	1.23E 01	1.47E 01	2.71E 02	2.80E 02
3.77E 03	1.23E 01	1.48E 01	2.71E 02	2.81E 02
3.88E 03	1.23E 01	1.49E 01	2.71E 02	2.81E 02
4.00E 03	1.23E 01	1.50E 01	2.71E 02	2.82E 02
4.12E 03	1.23E 01	1.51E 01	2.71E 02	2.82E 02
4.25E 03	1.23E 01	1.52E 01	2.71E 02	2.83E 02
4.37E 03	1.23E 01	1.53E 01	2.71E 02	2.83E 02
4.51E 03	1.23E 01	1.54E 01	2.71E 02	2.83E 02

	* * * FISH * * *		* INVERTEBRATE *	
TIME (DAYS)	INTERNAL (RAD)	TOTAL (RAD)	INTERNAL (RAD)	TOTAL (RAD)
6.25E 03	1.23E 01	1.69E 01	2.71E 02	2.90E 02
6.44E 03	1.23E 01	1.71E 01	2.71E 02	2.90E 02
6.63E 03	1.23E 01	1.73E 01	2.71E 02	2.91E 02
6.83E 03	1.23E 01	1.75E 01	2.71E 02	2.92E 02
7.04E 03	1.23E 01	1.76E 01	2.71E 02	2.92E 02
7.25E 03	1.23E 01	1.78E 01	2.71E 02	2.93E 02
7.47E 03	1.23E 01	1.80E 01	2.71E 02	2.94E 02
7.69E 03	1.23E 01	1.82E 01	2.71E 02	2.95E 02
7.92E 03	1.23E 01	1.84E 01	2.71E 02	2.96E 02
8.16E 03	1.23E 01	1.86E 01	2.71E 02	2.96E 02
8.41E 03	1.23E 01	1.89E 01	2.71E 02	2.97E 02
8.66E 03	1.23E 01	1.91E 01	2.71E 02	2.98E 02
8.92E 03	1.23E 01	1.93E 01	2.71E 02	2.99E 02
9.19E 03	1.23E 01	1.95E 01	2.71E 02	3.00E 02
9.47E 03	1.23E 01	1.98E 01	2.71E 02	3.01E 02
9.76E 03	1.23E 01	2.00E 01	2.71E 02	3.02E 02
1.00E 04	1.23E 01	2.03E 01	2.71E 02	3.03E 02
1.04E 04	1.23E 01	2.06E 01	2.71E 02	3.04E 02
1.07E 04	1.23E 01	2.08E 01	2.71E 02	3.05E 02
1.10E 04	1.23E 01	2.11E 01	2.71E 02	3.06E 02
1.13E 04	1.23E 01	2.14E 01	2.71E 02	3.08E 02
1.17E 04	1.23E 01	2.17E 01	2.71E 02	3.09E 02
1.20E 04	1.23E 01	2.20E 01	2.71E 02	3.10E 02
1.24E 04	1.23E 01	2.24E 01	2.71E 02	3.11E 02
1.27E 04	1.23E 01	2.27E 01	2.71E 02	3.13E 02
1.31E 04	1.23E 01	2.30E 01	2.71E 02	3.14E 02
1.35E 04	1.23E 01	2.34E 01	2.71E 02	3.15E 02
1.39E 04	1.23E 01	2.37E 01	2.71E 02	3.17E 02
1.44E 04	1.23E 01	2.41E 01	2.71E 02	3.18E 02
1.48E 04	1.23E 01	2.45E 01	2.71E 02	3.20E 02
1.52E 04	1.24E 01	2.49E 01	2.71E 02	3.21E 02
1.57E 04	1.24E 01	2.55E 01	2.71E 02	3.23E 02
1.62E 04	1.29E 01	2.70E 01	2.71E 02	3.28E 02
1.67E 04	1.46E 01	3.14E 01	2.72E 02	3.39E 02
1.72E 04	1.92E 01	4.41E 01	2.73E 02	3.73E 02
1.77E 04	2.81E 01	7.22E 01	2.76E 02	4.52E 02
1.82E 04	4.17E 01	1.25E 02	2.81E 02	6.11E 02
1.88E 04	5.71E 01	2.03E 02	2.86E 02	8.66E 02
1.93E 04	6.96E 01	3.01E 02	2.90E 02	1.21E 03
1.99E 04	7.76E 01	4.13E 02	2.93E 02	1.63E 03

4.64E 03	1.23E 01	1.55E 01	2.71E 02	2.84E 02	2.05E 04	8.12E 01	5.28E 02	2.94E 02	2.08E 03
4.78E 03	1.23E 01	1.56E 01	2.71E 02	2.84E 02	2.11E 04	8.23E 01	6.44E 02	2.95E 02	2.54E 03
4.93E 03	1.23E 01	1.58E 01	2.71E 02	2.85E 02	2.18E 04	8.26E 01	7.59E 02	2.95E 02	2.99E 03
5.07E 03	1.23E 01	1.59E 01	2.71E 02	2.85E 02	2.24E 04	8.27E 01	8.73E 02	2.95E 02	3.45E 03
5.23E 03	1.23E 01	1.60E 01	2.71E 02	2.86E 02	2.31E 04	8.27E 01	9.85E 02	2.95E 02	3.90E 03
5.38E 03	1.23E 01	1.62E 01	2.71E 02	2.87E 02	2.38E 04	8.27E 01	1.10E 03	2.95E 02	4.34E 03
5.55E 03	1.23E 01	1.63E 01	2.71E 02	2.87E 02	2.45E 04	8.27E 01	1.21E 03	2.95E 02	4.78E 03
5.71E 03	1.23E 01	1.65E 01	2.71E 02	2.88E 02	2.52E 04	8.27E 01	1.31E 03	2.95E 02	5.21E 03
5.89E 03	1.23E 01	1.66E 01	2.71E 02	2.88E 02	2.60E 04	8.27E 01	1.42E 03	2.95E 02	5.64E 03
6.06E 03	1.23E 01	1.68E 01	2.71E 02	2.89E 02	2.68E 04	8.27E 01	1.53E 03	2.95E 02	6.07E 03

SAMPLE PROBLEM NO. 1 : RELEASE TO AN ESTUARY VIA GROUNDWATER

FISH (X) AND INVERTEBRATE (*) DOSE (RADS) VS TIME (DAYS)



IHC002I STOP 0

Sample Problem 2

A groundwater transport problem is demonstrated in this case. It is sometimes useful to screen a source term by performing radionuclide transport through the groundwater. Also, with regard to sensitivity analysis of the transport parameters, this type of calculation may be of interest to the hydrologist. An additional application lies with an assessment of transport and activity levels at well locations. The waterborne concentration and its integral may be used to compute the dose associated with drinking water usage at the location. In this example, nine radionuclides (^{3}H, ^{90}Sr, ^{106}Ru, ^{113m}Cd, ^{125}Sb, ^{129}I, ^{131}I, ^{134}Cs, and ^{137}Cs) are released into groundwater which reaches a well site.

The input data for problem 2 is shown below followed by the computer printout.

```
SAMPLE PROBLEM NO. 2  :   ACTIVITY VS TIME ENTERING SURFACE WATER BODY
1$$ 3 A4 0 1 1 2R1 A13 0 9 E
2** 2R 36525. E T
14** 1500. 0. 30. 15. 2. 2. .3 6.7 32.8 1.7 .42 0.2 T
  SOURCE TERM FOR SAMPLE PROBLEM NO. 2
17** 1.-3 1. T
18** 1HH      3 1.+6 2R0.9 34.6 2R0. 1. E T
18** 2HSR   90 1.+6 2. 20.    1.16+2 2. 700. 0.2  1. E T
18** 2HRU  106 1.+6 3. 1000. 1.16+3 2. 100. 0.5  1. E T
18** 2HCD -113 1.+6 3.+3 2.5+5 1.39+4 25. 1.25+3 0.6  1. E T
18** 2HSB  125 1.+6 540. 5.     1.16+3 15. 750. 0.8  1. E T
18** 1HI   129 1.+6 10.  50.    34.6   0.1 5.   0.8  1. E T
18** 1HI   131 1.+6 10.  50.    34.6   0.1 5.   0.8  1. E T
18** 2HCS  134 1.+6 40.  25.    34.6   20. 1.+3 0.9  1. E T
18** 2HCS  137 1.+6 40.  25.    34.6   20. 1.+3 0.9  1. E T
/*
```

```
 1$ ARRAY      15 ENTRIES READ
 2* ARRAY       4 ENTRIES READ

 0T
14* ARRAY      12 ENTRIES READ

 0T
```

FIDO input routines produce these messages

```
                    L I Q U I D   P A T H W A Y   S T U D Y

 CASE TITLE :  SAMPLE PROBLEM NO. 2   :   ACTIVITY VS TIME ENTERING SURFACE WATER BODY

JOB CONTROL :
  JC(1) = 0/1/2/3/4/5  SMALL RIVER/ESTUARY/DRY SITE/WELLS ONLY/LAKE SITE/1-D RIVER      3
  JC(2) = 0/1  COMPUTE INDIVIDUAL DOSE/NO EFFECT                                        0
  JC(3) = 0/1  COMPUTE POPULATION DOSE/NO EFFECT                                        0
  JC(4) = 0/1/2/3/4  INSTANTANEOUS RELEASE OF Q CURIES                                  0
                     /CONSTANT RELEASE RATE QDOT CI/YR OVER DLECH DAYS
                     /QDOT=F(T) AS USER INPUT SOURCE
                     /QDOT=FRELS*Q*EXP(-(FRELS*LAMBDA)*T)
                     /EXPONENTIAL LEACH RELEASE MODEL
  JC(5) = 0/1  DIRECT RELEASE TO JC(1)/RELEASE TO GROUND WATER WHICH ENTERS JC(1)       1
  JC(6) = 0/1  NO EFFECT/GROUND WATER RELEASE DATA ONLY                                 1
  JC(7) = 0/1  NO EFFECT/NUCLIDE CONCENTRATION DATA SUPPRESSED                          1
  JC(8) = 0/1  NO EFFECT/NUCLIDE DOSE BREAKDOWN SUPPRESSED                              1
  JC(9) =  INTERNAL USE ONLY  (DEFAULT = 0)                                             0
  JC(10) = INTERNAL USE ONLY  (DEFAULT = 0)                                             0
  JC(11) = 0/1  NO EFFECT/SUPPRESS PLOTS                                                0
  JC(12) = 0/1  FORWARD INTEGRATION OF DOSE/BACKWARD INTEGRATION                        0
  JC(13) = 0/1  NO EFFECT/PRINT DOSE FACTORS                                            0
  JC(14) = NUMBER OF RADIONUCLIDE SOURCES                                               9
  JC(15) = NOT USED                                                                     0

DTIM     DOSE PERIOD (DAYS)                    3.6525E 04
DLECH    LEACH PERIOD (DAYS)                   3.6525E 04
CLSWB    SURFACE WATER LIMIT (PCI/L)           2.8330E-09
CLGRD    GROUND WATER LIMIT (PCI/L)            1.0000E 00

SITE DATA :
 NUMBER OF HYDROLOGICAL REGIONS                    0

   REGION  DIST(MILES)  POPULATION  AREA(ACRES)  FLOW(FT/SEC)   WIDTH(FT)  SHORELINE USAGE (USER-D/D)  SWIMMING USAGE (USER-D/D)

      1     0.0          0.0          0.0          0.0          0.0                0.0                        0.0


 COMMERCIAL FISH HARVEST (KG/A/D)           0.0            RECREATIONAL FISH HARVEST (KG/A/D)           0.0
 COMMERCIAL INVERTEBRATE HARVEST (KG/A/D)   0.0            RECREATIONAL INVERTEBRATE HARVEST (KG/A/D)   0.0
```

SURFACE WATER BODY HYDROLOGICAL PARAMETERS :

DEPTH	DEPTH OF WATER (FT)	0.0
WIDTH	WIDTH OF WATER BODY (FT)	0.0
RVEL	CURRENT FLOW RATE (FT/SEC)	0.0
EX	X DISPERSION COEFFICIENT (FT**2/SEC)	0.0
EY	Y DISPERSION COEFFICIENT (FT**2/SEC)	0.0
CRDIS	CROSS RIVER DISTANCE (FT)	0.0
SEDF	SEDIMENT FRACTION	1.0000E-20
BEDE	BED DISPERSION COEFFICIENT (FT**2/SEC)	0.0
BEDU	BED VELOCITY (FT/SEC)	0.0
SEDR	SEDIMENT RATE (FT/SEC)	0.0
ZLAKET	TRANSFER RATE TO SEDIMENT :KF (FT/SEC)	0.0
SHRER	SHORELINE EROSION RATE (1/SEC)	0.0
VOL	WATER VOLUME (FT**3)	0.0
QQ	WATER OUTFLOW RATE (FT**3/SEC)	0.0

GROUND WATER HYDROLOGICAL PARAMETERS :

DEPTHG	DEPTH OF AQUIFER (FT)	3.2800E 01
UGRND	GROUND WATER VELOCITY (FT/DAY)	6.7000E 00
X	X COORDINATE AT POINT OF INTEREST	1.5000E 03
Y	Y COORDINATE AT POINT OF INTEREST	0.0
Z	Z COORDINATE AT POINT OF INTEREST	3.0000E 01
Z1	SOURCE DEPTH (FT) AT X=0, Y=0, Z=Z1	1.5000E 01
BDEN	BULK DENSITY (G/CC)	1.7000E 00
TOTPOR	TOTAL POROSITY	4.2000E-01
EFFPOR	EFFECTIVE POROSITY	2.0000E-01
ALPHAX	DISPERSIVITY IN THE X-DIRECTION (FT)	2.0000E 00
ALPHAY	DISPERSIVITY IN THE Y-DIRECTION (FT)	2.0000E 00
ALPHAZ	DISPERSIVITY IN THE Z-DIRECTION (FT)	3.0000E-01

```
17* ARRAY      2 ENTRIES READ

 0T
18* ARRAY     11 ENTRIES READ

 0T
18* ARRAY     11 ENTRIES READ

 0T
18* ARRAY     11 ENTRIES READ

 0T
18* ARRAY     11 ENTRIES READ

 0T
18* ARRAY     11 ENTRIES READ

 0T
18* ARRAY     11 ENTRIES READ

 0T
18* ARRAY     11 ENTRIES READ

 0T
18* ARRAY     11 ENTRIES READ

 0T
18* ARRAY     11 ENTRIES READ

 0T
```

More FIDO messages

DESCRIPTION OF SOURCE TERM

SUBTITLE : SOURCE TERM FOR SAMPLE PROBLEM NO. 2

UML CONVERT ACTIVITIES TO CURIES 1.0000E 00
FRELS FRACTION RELEASED IF JC(4)=3 1.0000E-03

		BIOTA PARAMETERS			DISTRIBUTION COEFFICIENT		WATER TREATMENT
	ACTIVITY	BIOACCUMULATION		RETENTION	AQUIFER	STREAM	TRANSFER
NUCLIDE	(CI)	FISH	INVERTEBRATE	(DAY)	(CC/G)		
1H 3	1.0000E 06	9.0000E-01	9.0000E-01	3.4600E 01	0.0	0.0	1.0000E 00
38SR 90	1.0000E 06	2.0000E 00	2.0000E 01	1.1600E 02	2.0000E 00	7.0000E 02	2.0000E-01
44RU106	1.0000E 06	3.0000E 00	1.0000E 03	1.1600E 03	2.0000E 00	1.0000E 02	5.0000E-01
48CD113M	1.0000E 06	3.0000E 03	2.5000E 05	1.3900E 04	2.5000E 01	1.2500E 03	6.0000E-01
51SB125	1.0000E 06	5.4000E 02	5.0000E 00	1.1600E 03	1.5000E 01	7.5000E 02	8.0000E-01
53I 129	1.0000E 06	1.0000E 01	5.0000E 01	3.4600E 01	1.0000E-01	5.0000E 00	8.0000E-01
53I 131	1.0000E 06	1.0000E 01	5.0000E 01	3.4600E 01	1.0000E-01	5.0000E 00	8.0000E-01
55CS134	1.0000E 06	4.0000E 01	2.5000E 01	3.4600E 01	2.0000E 01	1.0000E 03	9.0000E-01
55CS137	1.0000E 06	4.0000E 01	2.5000E 01	3.4600E 01	2.0000E 01	1.0000E 03	9.0000E-01

SUM OF ACTIVITIES = 9.0000E 06

SAMPLE PROBLEM NO. 2 : ACTIVITY VS TIME ENTERING SURFACE WATER BODY

NUCLIDE	H 3		
SOURCE (CI)		1.0000E 06	Activity present in source
RELATIVE LEACH RATE		0.0	
RETARDATION		1.0000E 00	See Appendix B Eq. (B-31)
CI RELEASED		1.0000E 06	
CI EXITED		9.6612E 05	Activity crossing groundwater interface

T(DAYS)	T(SEC)	PCI/SEC	Q(T) CI	PCI/L	PCI-D/L
1.5539E 02	1.3431E 07	6.0316E 00	0.0	2.0890E 00	0.0
1.6000E 02	1.3829E 07	2.9329E 02	5.9528E-05	1.0196E 02	2.3942E 02
1.6474E 02	1.4238E 07	1.0175E 04	1.5672E-03	3.5498E 03	6.3235E 03
1.6961E 02	1.4660E 07	2.5267E 05	[illegible].6990E-02	8.8429E 04	2.3071E 05
1.7464E 02	1.5094E 07	4.5010E 06	8.0763E-01	1.5798E 06	3.2780E 06
1.7981E 02	1.5541E 07	5.7684E 07	1.4707E 01	2.0300E 07	5.9862E 07
1.8514E 02	1.6001E 07	5.3298E 08	1.1928E 02	1.8802E 08	4.8647E 08
1.9062E 02	1.6475E 07	3.5577E 09	1.0886E 03	1.2576E 09	4.4500E 09
1.9626E 02	1.6963E 07	1.7178E 10	5.3166E 03	6.0838E 09	2.1764E 10
2.0208E 02	1.7466E 07	6.0106E 10	2.4730E 04	2.1320E 10	1.0141E 11
2.0606E 02	1.7983E 07	1.5255E 11	7.5688E 04	5.4179E 10	3.1069E 11
2.1422E 02	1.8515E 07	2.8105E 11	1.9116E 05	9.9923E 10	7.8550E 11
2.2057E 02	1.9064E 07	3.7610E 11	3.7464E 05	1.3382E 11	1.5405E 12
2.2710E 02	1.9628E 07	3.6566E 11	5.8404E 05	1.3017E 11	2.4027E 12
2.3383E 02	2.0210E 07	2.5829E 11	7.7450E 05	9.1969E 10	3.1873E 12
2.4075E 02	2.0808E 07	1.3252E 11	8.9146E 05	4.7185E 10	3.6692E 12
2.4788E 02	2.1424E 07	4.9362E 10	9.4288E 05	1.7571E 10	3.8810E 12
2.5522E 02	2.2059E 07	1.3337E 10	9.6277E 05	4.7451E 09	3.9629E 12
2.6278E 02	2.2712E 07	2.6111E 09	9.6519E 05	9.2829E 08	3.9728E 12
2.7056E 02	2.3385E 07	3.6986E 08	9.6619E 05	1.3136E 08	3.9769E 12
2.7858E 02	2.4078E 07	3.7849E 07	9.6611E 05	1.3426E 07	3.9766E 12
2.8683E 02	2.4791E 07	2.7930E 06	9.6612E 05	9.8927E 05	3.9767E 12
2.9532E 02	2.5525E 07	1.4829E 05	9.6612E 05	5.2432E 04	3.9767E 12
3.0407E 02	2.6281E 07	5.6487E 03	9.6612E 05	1.9934E 03	3.9767E 12
3.1307E 02	2.7059E 07	1.5402E 02	9.6612E 05	5.4236E 01	3.9767E 12
3.2235E 02	2.7861E 07	2.9967E 00	9.6612E 05	1.0527E 00	3.9767E 12

Waterborne concentration as a function of time at the location 1500 ft from the release

Total activity crossing a plane 1500 ft downgradient of the release

SAMPLE PROBLEM NO. 2 : ACTIVITY VS TIME ENTERING SURFACE WATER BODY

NUCLIDE	I 131	
SOURCE (CI)		1.0000E 06
RELATIVE LEACH RATE		1.0000E 00
RETARDATION		1.4048E 00
CI RELEASED		1.0000E 06
CI EXITED		1.8318E-06

About 10^{-12} of the release is indicated to pass the plane at 1500 ft

T(DAYS)	T(SEC)	PCI/SEC	Q(T) CI	PCI/L	PCI-D/L
2.8538E 02	2.4665E 07	1.1267E 00	0.0	3.9973E-01	0.0
2.9102E 02	2.5153E 07	.2895E 00	5.8946E-07	4.5790E-01	2.4209E 00
2.9678E 02	2.5651E 07	1.2653E 00	1.2406E-06	4.4965E-01	5.0971E 00
3.0265E 02	2.6158E 07	1.0644E 00	1.8318E-06	3.7850E-01	7.5283E 00

If the plane was to intercept a surface water body, this nuclide could be removed from the assessment due to the small level of activity entering the surface water.

APPENDIX A

Excerpted from Chapter 4: "Pathway Models" in NUREG-0440
Liquid Pathway Generic Study: Impacts of Accidental
Radioactive Releases to the Hydrosphere from Floating
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APPENDIX A

PATHWAY MODELS: BACKGROUND INFORMATION

Transport models have been developed for the Liquid Pathway Generic Study (LPGS)* to calculate the concentrations of radionuclides in surface water and groundwater resulting from accidental releases. These concentrations are subsequently used in dose assessment models. In some cases, the transport models have been formulated directly to assess doses, thus eliminating the intermediate concentration calculations.

A.1. GENERAL GUIDELINES

In developing the transport models, consideration was given to the relative complexity needed in terms of the various other models used in the study. Using a complicated time-dependent, multidimensional finite difference model of riverine transport, for example, would be unnecessary if concentrations averaged over large river reaches and long time periods were adequate for dose assessment. Often, a far simpler analytical model was found to be satisfactory.

The LPGS considered numerous scenarios of sites and accidents. Model parameters were continually being refined throughout the study. Since utilization of the transport and dose assessment models was always the last step in the analytical procedure, it was a matter of convenience for the computer codes to be unified into a master program so that reruns could be made if there was a major change to a parameter. Many of the transport models used are similar to those found in the literature for determining concentrations of various pollutants in field situations.[1-5] In some cases, no acceptable model was available, so models were developed specifically for this study. In general, the following guidelines were used in the selection and development of transport models:

1. The models actually employed for calculations were all closed-form analytical, for which some parameters were based upon empirical data. In order to generalize closed-form solutions for instantaneous sources to solutions for arbitrary sources, numerical integration of the closed-form solutions was sometimes used.

2. The effects of sediment in removing radionuclides from the water column have been established as being significant in some cases. The state-of-the-art of sediment-water interaction modeling is not as well developed as in other areas of transport modeling. Wherever possible, however, the effect of sediment was incorporated into the models. NRC has supported several advanced numerical modeling

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*References to the acronym LPGS in the appendices will refer to the actual generic study and not the computer program.

efforts in an attempt to better quantify the effects of sediment on radionuclide transport in surface water.[6-9] The results of these studies have not been used directly in the calculations, but were used to qualitatively support the validity of the far simpler computational models used in the LPGS.

3. Whenever possible, the validity of the models was tested by model-prototype comparison for real field situations. In some cases, the models were phenomenological, derived directly from field data.

4. Model parameters were usually chosen from actual field situations. This was desirable in order to avoid anomalous combinations of parameters which might have been caused by choosing averages or extremes.

5. Whenever a particular facet of a transport model was in doubt, the tendency was to opt for conservatism.

Two sets of transport models have actually been used for the LPGS. The original set of models was developed for the Draft LPGS.[10] Those models differ substantially from the present set in that the effects of sediments were neglected, and more emphasis was placed on fast-transient releases to cover a wide range of accident scenarios. The characteristic of fast-transient response is important for the accidents within design basis, since releases are always directly to the surface water, and are of short duration.

The present set of models includes the models used in the Draft LPGS but those models are used only for lower class accidents. For the core-melt accidents, greater emphasis was placed on models capable of simulating the slowly varying conditions which would typify that type of accident at an LBP. Such an accident would release radioactive material only through the groundwater pathway. The flux of material entering the surface water would, therefore, vary much more slowly than would be the case for the lower-class accident. The fact that this study is part of the National Environmental Policy Act (NEPA) environmental assessment process dictated the use of realistic models which consider the effects of sediment, flow variation, and other important variables for the highly important core-melt accident. Surface-water and groundwater models are briefly described below. They are developed in much more rigorous mathematical detail in Appendix B.

It must be stressed that the models and coefficients reported in this section are intended to be for representative site conditions only. The staff recognizes that many of the transport properties vary over wide ranges. There are undoubtably sites with potential for much more direct transport (e.g., plants with safety-related gravity underdrains, or plants built over cavernous limestone), as well as sites where there is virtually no transport capability other than through the normal circulating water system. Wherever possible, the sensitivity of the various model parameter has been determined to indicate the ranges of consequences to be expected.

A.2. RADIONUCLIDE TRANSPORT MODELS

A.2.1. Surface-Water Models

Surface waters covered in this study are classified as rivers, estuaries, Great Lakes, and oceans. Formulation of the analytical models is in the framework of generally simplified geometries and steady flow rates. The effects of sorption of radionuclides by bottom and suspended sediments are taken into account.

A.2.1.1. River Models

Model for Lower-Class Accidents

The river for this case was considered to have a uniform, rectangular cross section, with constant shore-parallel flow, and lateral and longitudinal dispersion representable with constant coefficients. Instantaneous or continuous discharge of radionuclides was assumed to take place from a vertical line source extending from the surface to the bottom of the river, so that there would be no variation of concentration in the vertical direction. This model was also used for the core-melt accident at a large river site.

Model for Core-Melt Accidents

This river model is based on the Clinch-Tennessee-Ohio-Mississippi River system. It is a steady-state model that accounts for the increase in flow in the downstream direction, the sorption of radioactivity by suspended and bottom sediments, and radioactive decay. The Clinch and Tennessee Rivers are extensively dammed. Each reservoir on the Tennessee River is represented by a perfectly mixed tank, which is similar to the mixed-tank model employed for the Great Lakes. A plug flow model was used for the Clinch River.

The effects of sorption are taken into account in the reservoir models by assuming scavenging by falling sediment and direct transfer from water to bottom sediment. There are no dams on the mainstem sections of the Ohio and Mississippi Rivers downstream of the last reservoir on the Tennessee River. Since the effect of sediment is relatively less important in a free running river than in a dammed river, the Ohio and Mississippi River sections account only for dilution. Neglecting sediment effects here is probably conservative.

The above-described river model is useful only for extended radioactive releases that would occur for large-scale accidents via the groundwater pathway, since this is the only way the radionuclides from the core-melt accident can enter the surface-water body. In these cases, only long-lived radionuclides would pass through the groundwater in appreciable amounts for the site parameters chosen, and the flux of radioactivity into the river would be changing very slowly. The radionuclide transport throughout the entire river would be on a much faster time scale than the flux of entering radioactivity, so a steady-state surface-water transport model is justified. The lack of sensitivity of the resulting dose to the steady-state assumption is demonstrated in Appendix B.

Model Parameters for Lower-Class Accidents

The parameters selected for use in this river model are similar to conditions in medium-sized rivers. A constant depth of 4.6 m (15 ft) and a constant width of 114 m (375 ft) were used. The flow rate selected was 81 m^3/sec (3400 cfs), which corresponds to an average downstream velocity of about 0.18 m/sec (0.6 ft/sec). Constant eddy diffusivities of 0.74 m^2/sec (8 ft^2/sec) and 0.056 m^2/sec (0.6 ft^2/sec) in the longitudinal and lateral directions, respectively, were considered representative. Smaller rivers are seldom selected for nuclear plant sites because of water supply problems during drought periods. There are nuclear plants located on larger rivers, but the medium size selected for this study was considered conservative from a dispersion standpoint. The river reach investigated was assumed to be 1287 km (800 mi) long. The staff recognizes that there are few, if any, undammed rivers that satisfy this condition, nor do any rivers maintain a constant flow rate over a long reach. These assumptions are clearly conservative.

Model Parameters for Core-Melt Accidents

Parameters for this river model were chosen to conform to conditions representative of the years 1960 and 1961 in the Clinch-Tennessee-Ohio-Mississippi River system. This time frame was chosen because it represents a period for which part of the system was under intensive study for transport of radioactive wastes discharged to the Clinch River from the Oak Ridge National Laboratory (ORNL).[11] Model parameters, such as reservoir dimensions and sediment loads, were gathered from published records or file documents of the U.S. Department of Agriculture (USDA),[12] the U.S. Geological Survey (USGS),[13-15] and the Clinch River Breeder Reactor (CRBR) Plant Docket.[16] Flow rates, gathered from published USGS water supply documents,[13] were averaged to partially correct for weighting of concentrations by flood and drought conditions. Further details are given in Appendix B.

Equilibrium distribution coefficients (Kd = curies/gram ÷ curies/ml), which are important to sorption considerations, were obtained directly from either the Clinch River Study[11] or a table prepared by Booth representing average fresh-water values.[17] These values of K_d were 0 for ^{3}H, 85,000 for ^{137}Cs, 2400 for ^{90}Sr, 32,000 for ^{60}Co, and 0 for ^{106}Ru. The 0-value for ruthenium considers only the complexed anionic fraction, and not the cationic form.

The model was calibrated with limited experimental data on radioactive releases to the Clinch River.[18] The method of calibration was to adjust a single coefficient called the sediment effectiveness factor, so that there was the most reasonable agreement between model and prototype for sediment/radionuclide concentrations measured in the Clinch and Tennessee Rivers. A value for ε of 10% gave the most satisfactory results. The sedimentation rate, which is an extremely important parameter in this model, was estimated by assuming that the gross annual sediment load to the reservoir was uniformly distributed over the entire surface area. In actuality, much of the sediment load from tributaries would fall out within a fairly short distance from the tributary mouth. Only fine sediments are transported large distances in large reservoirs.[19] Since the gross sediment load includes the entire range of sediments from clay to large rocks, the calculated sedimentation rate is too large. The 10% effectiveness factor reduces the calculated sedimentation rates to values far more typical of those actually observed in reservoirs and lakes, including the Great Lakes.[20-22] Further details of the calibration procedure are given in Appendix B. Values of physical properties such as volumes, sediment rates, and depths of reservoirs and river segments for the river system are given in Table A.2.1. An effective sediment depth of 10 cm (4 in.) and the coefficient of direct transfer of 0.4 m/yr (1.3 ft/yr) was chosen for all reservoir segments. Sensitivity to variation in these two coefficients is low.

The Clinch River portion of the model is represented as a uniform channel with a cross-sectional area of 520 m^2 (5600 ft^2) and a length of 33.4 km (20.8 mi). Sedimentation properties are assumed to be those of Watts Bar Reservoir.

Table A.2.1. Physical parameters of river system model

River segment name	Reservoir volume 10^6 ha-m	Length of segment km	Surface area ha	Average depth[1] m	Distribution of reservoir center from source[2] km	Sedimentation rate[3] m/yr	Flow out of segment m^3/sec
White Oak Creek to mouth of Clinch		33.3		9.27		0.0107	50
Watts Bar Lake	0.146[4]		15,800	9.27	33.3	0.0107	748
Chicamauga Lake	0.061	94.2	14,300	6.41	140	0.0128	925
Hales Bar Lake	0.0184	64		6.41	220	0.0128	970
Guntersville Lake	0.132	131	27,500	4.82	318	0.076	1,135
Wheeler Lake	0.131	119	27,200	4.82	443	0.061	1,277
Wilson Lake	0.080	24.8	6,280	12.84	514	0.0107	1,322
Pickwick Lake	0.137	84.3	174,500	7.87	569	0.0079	1,419
Kentucky Lake	0.35	295	64,900	5.37	758	0.0110	1,646
Kentucky Dam to Ohio River, Junction		35.8					6,100
Ohio River Junction to Memphis		13.6					11,170
Memphis to Vicksburg		363.					11,170
Below Vicksburg		301.					13,880

[1]Average depth = volume/area.
[2]Source is White Oak Creek (Clinch River km 33.3)
[3]Sedimentation rate = sedimentation load/area.
[4]Acutal volume used = 0.073 × 10^6 acre-ft (0.59 × 10^6 acre-ft).

Model Utilization for Lower-Class Accidents

For the lower-class accidents, the two-dimensional river model was used to calculate the concentration at points downstream from the release point. The release was specified as being located on the shoreline. Drinking water intakes were also specified as being on the shoreline, downstream from the discharge and on the same side of the river. Local concentrations were computed at points on the nearshore for the purpose of evaluating the fish consumption, sediment, and immersion pathways in the region within several miles of the source, before nearly uniform mixing across the river would occur.

Model Utilization for Core-Melt Accidents

Releases for core-melt accidents are routed through the groundwater pathway only. Groundwater seepage would probably be from a diffuse ill-defined source area from the sides and bottom of the river. Close to the source, mixing would be more complete than would be expected for surface releases. The Draft LPGS[10] indicated that the major contribution to the population dose occurs in reaches far downstream from the source, where there would be relatively uniform mixing across the river. Therefore, no separate near-field two-dimensional concentrations were computed for the core-melt accidents.

A.2.1.2. Estuary Models

Model for Lower-Class Accidents

A simple one-dimensional, tidally-averaged (currents) model was used to calculate the cross-sectionally averaged concentrations. The estuary was assumed to have a constant cross-section area and constant net downstream velocity. Tidal currents were not included explicitly as an advective mechanism, but were considered to be responsible for large-scale, longitudinal, Fickian dispersion with a constant dispersion coefficient. The effects of sediments were neglected for these accidents.

Model for Core-Melt Accidents

This model is similar to the lower-class accident model, except the effects of sediments are included. The assumption of intimate contact and, consequently, chemical equilibrium between water and sediment, was made, which allowed for a relatively simple closed-form solution.

Model Parameters for Lower-Class Accidents

The estuary model parameters selected for use are typical of conditions in large East Coast estuaries. The downstream fresh-water flow rate was taken to be 310 m^3/sec (13,000 cfs). For a constant cross-sectional area of 15,000 m^2 (160,000 ft^2), the corresponding fresh water velocity is 0.0205 m/sec (0.1 ft/sec). The corresponding eddy diffusivity was taken to be 139 m^2/sec (1500 ft^2/sec). The reach of estuary evaluated ranged from 24.1 km (15 mi) upstream of the plant to 161 km (100 mi) downstream.

Effective dispersion coefficients are much greater for estuaries than for rivers, because the simplified "tidally-averaged" estuary model treats tidal oscillations as being responsible for large-scale longitudinal dispersion (see Appendix B.).

Model Parameters for Core-Melt Accidents

The parameters for this model were the same as those for the lower-class accident model, except for those pertaining to sediment effects. Equilibrium distribution coefficients (K_d) were taken from observations in estuaries and marine environments and are presented in Table A.2.2. Reported values vary over a wide range for any isotope.[17,23-28] Values in the middle of the observed range were generally used.

Although the model was capable of accepting bed movement rates, the bed of the estuary was assumed to be stationary. Sedimentation in the estuary model serves only to bury radioactive bottom sediments and is thus a removal term. A separate treatment of sediment resuspension is provided in Appendix B. Rates measured in parts of Monsweage Bay,[27] an estuary in Maine, were 2.6 cm/yr (1 in./yr). Annual accumulations of 0.5 cm to 0.8 cm (0.2 to 0.3 in.) were observed in the upper portion of Chesapeake Bay.[29] A value of 0.8 cm/yr (0.3 in./yr) was used in the LPGS model. This value is near the lower limit of rates observed in this rather limited set of data and should be conservative.

Table A.2.2 Distribution coefficients (K_d) used for estuary model

Radionuclide	K_d
H	0
Sr	700
Y	400
Nb	20000
Ru	100
Rh	1250
Ag	1250
Cd	1250
Sn	750
Sb	750
Te	1000
I	5
Cs	1000
Ba	150
La	500
Ce	4000
Pr	3000

The effective depth of the contaminated layer was chosen to be 10 cm (4 in.) based on measurements of radioactive sediment depths in Monsweage Bay.[27] Sensitivity of population dose estimates to the coefficients of sedimentation rate and sediment depth used in the estuary model was shown to be low (see Appendix B.).

A.2.1.3. Great Lakes Models

Field studies in the Great Lakes have revealed that coastal currents are predominantly parallel to shore and have typical speeds of 10 to 20 cm/sec (0.33 to 0.66 ft/sec). Currents generally persist for several days, then, in direct response to wind shifts, they quickly reverse and persist in the opposite direction for several days.[3,30]

Each reversal of the coastal current is usually accompanied by large-scale mass exchange with offshore waters that effectively remove pollutants from the nearshore zone.[3]

Pollutants flushed from the nearshore zone are dispersed in the large-scale turbulence offshore, unimpeded until effects of the other shorelines are felt. Eventually the pollutants not picked up by sediments are nearly uniformly mixed in the lake. The characteristic mixing time for an instantaneous release is estimated to be from several weeks to several months, depending upon lake size and prevailing hydrologic conditions.[3-4]

Some pollutants are eventually flushed out of the lakes by the flow-through of fresh water. Others remain largely attached to sediments. The flushing time for the Great Lakes is on the order of years to tens of years, depending upon the particular lake volume and flow-through rate.

Nearshore Model

A simple two-dimensional model was used for the nearshore zone. The lake was assumed to have a straight shoreline and to be of constant depth, with steady, unidirectional flow parallel to shore. Release was postulated through a vertical line source extending from the water surface to the lake bottom. Dispersion was then postulated to occur in both the lateral and longitudinal directions with constant dispersion coefficients. This model was assumed to be useful for distances along the shore of up to 24.1 km (15 mi) from the source, which is approximately the distance a parcel of water can move in the nearshore region between current reversals.

Offshore Model

A phenomenological model based upon correlated instantaneous dye-release data was used for establishing concentrations needed for evaluating the fish consumption pathway in the offshore zone. This model is theoretically valid only for instantaneous releases offshore, where the shorelines of the lake would not significantly interfere with the radially symmetrical spreading of the patch of dye. The correlation developed in Appendix B is based upon data which were obtained for a period of only several days. It was assumed that the model would be valid for an instantaneous release from the time it entered the offshore zone (i.e., after current reversal), until the limits of the patch extended over the whole lake area, a period of about 13 days. Dispersion was assumed to be confined to the top 10 m (30 ft) of the lake, which is the estimated depth of the actual thermocline to be expected. The assumption of the presence of a thermocline was considered conservative from an effluent standpoint because a thermocline limits the volume of water available for dispersion, and is not always present in the lake.

The assumption that the patch of dye is radially symmetric and does not interfere with the shoreline is not entirely realistic. Interference with the near shore can be expected a short time after release. This model, however, is based upon the best available data for the Great Lakes, and represents a reasonable estimate of dispersion in a lake during the offshore phase.[4] Since the model was developed from data for instantaneous releases, it is not directly applicable to continuous releases.

Mixed-Tank Model for Lower-Class Accidents

The postulated mixed-tank model assumes that the released material has been uniformly distributed throughout the entire lake, and is being removed only by decay and the flow of fresh water through the lake. This model is justified in terms of the relatively slow flushing rate of the lakes compared to the mixing within the lake. The model is also capable of computing radionuclide concentrations in a series of lakes.

Dispersion is no longer assumed confined to the upper 10 m (30 ft) because, within the time scale for which this model is valid, there would be several seasonal overturns in the lake which would mix the released material with deeper waters. Thus, the entire volume of the lake is considered to be affected.

Mixed-Tank Model for Core Melt

This model includes the effect of radionuclide removal from the water column by interaction with sediments. It is based upon a four-compartment model,[16] simplified to conform to the general guidelines of Section A.1

The model has been tested, with reasonable results, against observed concentrations of ^{137}Cs and ^{90}Sr resulting from atmospheric fallout. Tests indicate that neglecting the effects of sediments in the Great Lakes can lead to a substantial overestimation of concentrations for ^{137}Cs and other highly sorbed materials.[31]

Great Lakes Model Utilization

Utilization of the lake models was not as straightforward as the river and estuary models. No single model was considered applicable to all dispersion regimes. In certain areas of the lake site simulation, it was necessary to estimate concentrations by interpolating the results of several models where none of the individual models were considered applicable. The largest contribution to population doses, however, resulted from the longterm concentrations uniformly distributed throughout the lake. This was adequately handled by the mixed-tank models. The mixed-tank model, which includes sediment interactions, was used only for the core-melt accident. Two types of releases were evaluated as discussed below.

Instantaneous Releases. Concentrations in the nearshore regime were calculated using the nearshore model from 0 to 3 days within 24.1 km (15 mi) of the source for the purpose of evaluating the drinking water, sediment, immersion, and fish consumption pathways. The effluent source, as well as the drinking water intakes, was presumed to be located on the shoreline. Concentrations at any point along the shoreline were halved since it has been shown that in the nearshore zone the currents have approximately equal distributions of amplitude and direction, upshore and downshore.[30]

The nearshore model does not predict the concentration buildup of long-term releases, so the results from that model were superimposed on the concentration buildup computed from the totally mixed model. Shoreline concentrations further than 24.1 km (15 mi) from the source were computed solely from the totally mixed model.

Concentrations for the fish consumption pathway were calculated using the offshore patch spreading model for the 10-day span between 3 and 13 days after release. Shoreline concentrations within 24.1 km (15 mi) of the source for the drinking water, immersion, and sediment pathways for the period from 3 to 13 days were estimated by interpolating between concentrations calculated from the nearshore model and the totally mixed model. Beyond 13 days after release, the totally mixed model was used for all concentration calculations.

Continuous Releases. For continuous releases, concentrations in the nearshore zone within 24.1 km (15 mi) of the source were calculated using the nearshore model, with the results multiplied by the factor of 0.5, as discussed above. After adding the background concentrations resulting from long-term buildup in the lake, the resultant nearshore concentrations were considered to be valid for as long as there was a continuous release.

The spreading patch model was considered to be valid only for instantaneous releases; for continuous releases, all concentrations outside of the nearshore zone were computed using the totally mixed model.

Great Lakes Parameter Selection

The physical properties such as currents, flow rates, volumes and depths were based largely upon those found in Lake Ontario.[32] Sediment-related properties such as equilibrium distribution coefficients and sedimentation rates were based upon data from the other Great Lakes as well.[20-22] Parameters for the various Great Lakes models used are discussed separately below.

Nearshore Model. Transport parameters necessary for this region of the lake include average effective depth, average current speed, and dispersion coefficients in the lateral and longitudinal directions. The nearshore model is applicable from about 0 to 3 days for distances up to about 24.1 km (15 mi) from the source.

The representative depth in the nearshore zone of Lake Ontario was estimated to be about 3 m (10 ft). This was considered to be a conservative value, but not unrealistic, because turbulence in the nearshore zone assures fairly complete mixing above the thermocline. The current speed parallel to the shore was estimated to be about 9.1 cm/sec (0.3 ft/sec) based upon field studies.[3-4,30] Corresponding eddy diffusivities in the lateral and longitudinal directions were estimated to be about 0.093 m^2/sec (1.0 ft^2/sec) and 0.372 m^2/sec (4.0 ft^2/sec), respectively.

Offshore Model. As stated above, the spreading patch model was considered applicable only to the instantaneous release sources for estimating concentrations necessary for evaluating consequences resulting from the fish consumption pathway, and for the 10-day time span between 3 and 13 days. As shown in Appendix B, the average concentration in the patch was a function of the amount (curies) of release, the area of the patch, the depth of the thermocline, the decay coefficient, and the amount of elapsed time after the release. An average depth of the thermocline of 10 m (30 ft) was considered applicable to Lake Ontario.[3,30] This value is conservative from the standpoint that the lake is vertically mixed during parts of the year. Variation in this value has little effect upon resultant population dose estimates.

Mixed-Tank Model. This model was used for all concentration calculations beyond 13 days after release. As shown in Appendix B, the input parameters necessary were the volume of the lake and the flow rate leaving the lake. The average values for Lake Ontario volume and flow rate,[32] 1.64×10^{12} m^3 (5.78×10^{13} ft^3) and 6630 m^3/sec (2.34×10^5 cfs), respectively, were used in this study.

Equilibrium distribution coefficients were taken from the literature or estimated from elemental compositions of sediment and water.[17] These values were 27,000 for ^{137}Cs, 2,400 for ^{90}Sr, zero for ^{3}H, and zero for ^{106}Ru (complexed anionic form only).

Sedimentation rates of 0.03 cm/yr to 0.08 cm/yr (0.012 to 0.031 in./yr) have been reported in the Great Lakes,[20-22] and an average value of 0.05 cm/yr (0.020 in./yr) was used. Lerman[20] reports that ^{90}Sr and ^{137}Cs occur in the upper 8 cm to 11 cm (3.1 to 4.3 in.) of sediment in Lakes Superior and Ontario, although much greater depths are reported in Lake Michigan. A value of sediment depth of 10 cm (4 in. was used for the LPGS model.[17] The coefficient of direct transfer of radionuclides from water to bottom sediment, K_f, has been measured by Lerman for several Canadian lakes.[22] An average value of 0.4 cm/yr (0.16 in./yr).[17] Sensitivity of population dose to the values of K_f and sediment depth is low.

Comparison of this model using the above-described coefficients against observed concentrations of fallout ^{137}Cs and ^{90}Sr in the Great Lakes has been good.[31,33]

A.2.1.4. Other Surface-Water Models

Two numerical modeling efforts were sponsored by NRC to investigate in greater detail the effects of sediment and other physical phenomena on transport of radionuclides in surface water. Such models would be extremely useful for site-specific analyses, but were used in the LPGS strictly for support of the analytical procedures actually used.

Onishi[6] applied a two-dimensional, vertical-longitudinal, finite-element model to radionuclide transport in the Clinch River below Melton Hill Dam. The model considers many actual physical properties of the river, including channel geometry and velocity profiles. Sediment transport in three size classes is modeled in detail, as well as nonequilibrium uptake of radionuclides by sediment.

The model was applied to the Clinch River in order to simulate, for a period of several days, the transport of radionuclides released from White Oak Creek and measured during the Clinch River Study.[11] Critical parameters necessary for the simulation were in some cases unavailable, but the model duplicated, at least qualitatively, many of the salient features of the transport of ^{137}Cs and ^{90}Sr actually observed.

Additional confirmation with the Onishi model is currently being pursued under NRC contract. Field observations and model runs are being made on a small river in western New York. The field experiments will involve the release of neutron-activation tracers in order to simulate as closely as possible the transport of readily sorbed radionuclides in a realistic accident situation.

Eraslan[7] applied a one-dimensional, fast-transient, discrete-element model to radionuclide transport in a 161-km (100-mi) reach of the Hudson River estuary. The model was developed with NRC support as part of the Unified Transport Model.[34] The model was run with tidal and flow-rate data for a 6-month period. It is capable of calculating flow rate, sediment, and dissolved constituent transport in estuaries, but unlike the Onishi model, does not incorporate the mechanisms for interchange of radionuclides between water and sediment. The model has nevertheless been useful for evaluating the extreme cases of transport of radionuclides with no sediment interaction and transport of tagged sediment with no dissolved radionuclides.

The model was applied to postulated releases in the Hudson River in the vicinity of Indian Point to simulate the transport of both tritium and tagged sediments. The basic transport model has previously been used for thermal effluent transport in the same estuary with good results.[35] Preliminary results for the transport of ^{137}Cs in the vicinity of Indian Point show good agreement with field data.[7]

Further models under development, but not extensively field tested, include a model similar to the estuary model described, but incorporating realistic mechanisms for radionuclide interchange between sediment and water,[8] and a two-dimensional (horizontal plane) model for radionuclide transport in estuaries, lakes, and oceans.[9]

A.2.2. Groundwater Models

Two models were used to evaluate the movement of radionuclides through the groundwater pathway. The first model (Point Concentration) simulates movement from a point source through an aquifer of constant thickness in order to calculate concentrations at some point downgradient from the release (e.g., at a well). The second model (Surface-Water Interface) computes the amount and time variation of radionuclides being released to the surface water. The groundwater models were only used for the core-melt accidents, because no lower-class accidents were postulated to have releases to the groundwater.

Both models were formulated for saturated flow in simple geometries, under assumed uniform properties of the medium (e.g., permeability, porosity, dispersion coefficients, and radionuclide sorption), and uniform groundwater velocities. The effects of sorption were taken into account in these models under the assumption that the concentration of the solid and liquid phases would be in chemical equilibrium.[36] Either continuous or instantaneous source-term functions can be used as input to the models.

A.2.2.1. Groundwater Model Utilization

The movement of released radionuclides through the groundwater pathway was computed using the two groundwater models. The release was assumed to be from a vertical line source in the aquifer, directly beneath the plant.

The Surface-Water Interface model was used for calculating the rate of radionuclide addition to a surface-water body downgradient from the source. Outputs from the Surface-Water Interface model were used directly as input to the river, Great Lakes, and estuary models for the core-melt accident.

The Point Concentration model was used for the evaluation of the core-melt accident at the dry site. Water usages in the dry site vicinity were assumed to be represented by a spatial continuum rather than discrete well locations. Computation in this case was carried through one additional integration to directly estimate the time-integrated consumption of accidentally released radionculides via the drinking water pathway. Both analytical and numerical integration of the Point Concentration model produced the following evaluations:

1. The population dose without interdiction or as a function of a downgradient exclusion distance within which use is denied.

2. The population dose when a fixed maximum permissible concentration level for consumption has been set.

3. The land areas within which concentration levels exceed the maximum permissible.

A.2.2.2. Groundwater Parameters

Point Concentration Model

The parameters chosen for this model were representative of the Idaho National Engineering Laboratory located on the Snake River Plain aquifer in southeastern Idaho. This site is one of the few where there has been extensive monitoring of actual radioactive releases to groundwater.[37-39]

Although the aquifer is quite complicated, experiments with this analytical model,[40] as well as more complicated numerical models,[37] demonstrate that acceptable verification can be obtained using the coefficients in Table A.2.3. Radioactive waste is assumed to enter the saturated groundwater instantaneously, even though there is a considerable unsaturated overburden at the prototype site. This is a conservative assumption.

Table A.2.3 Groundwater model parameters for dry site calculations and radionuclides of importance

Average groundwater velocity	=	1.32 m/day (4.32 ft/day)
Transversal dispersivity	=	127 m (450 ft)
Longitudinal dispersivity	=	91 m (300 ft)
Porosity	=	0.1
Aquifer thickness	=	96 m (250 ft)
Adsorption Retention Factors		
a	=	1.0 for ^{3}H
a	=	28 for ^{90}Sr
a	=	1.0 for complexed ^{106}Ru
a	=	253 for ^{137}Cs
Maximum permissible concentrations (10 CFR Part 20)		
MPC	=	3000 pci/ml for ^{3}H
MPC	=	0.3 pci/ml for ^{90}Sr
MPC	=	10 pci/ml for ^{106}Ru
MPC	=	20 pci/ml for ^{137}Cs

The characteristics of the selected site would tend to allow for an atypically large dispersion of released contaminants, since flow rate and longitudinal and transversal dispersion coefficients are large. Even with the great dispersion found at this site, the area impacted by the core-melt accident is estimated never to exceed more than several square miles. Smaller dispersion properties would result in even smaller affected areas, and in this case, smaller population doses. Therefore, the chosen site parameters are considered to be conservative. Sensitivity tests with smaller dispersivities and velocities are shown in Appendix B.

Because of the extremely slow movement of radionuclides through the hydrogeologic medium only the relatively long-lived radionuclides have to be evaluated. The analysis is thus restricted to ^{3}H, ^{90}Sr, ^{106}Ru, and ^{137}Cs.

Surface-Water Interface Model

Parameters for this model were chosen to be representative of unconfined local aquifers draining into surface-water bodies. Properties of such aquifers are quite different from the dry site. Longitudinal dispersivity was taken as 61 cm (2.0 ft). Dispersivity is a function of the aquifer material, including its nonhomogeneity. Reported dispersivity values may range from 10 cm (4 in.) or less in fine unconsolidated materials[41-42] to hundreds of meters in large regional aquifers, such as the Snake River Plain aquifer used in the dry site analysis.[37] The values selected for the surface-water interface model tend toward the low end of the range, but are not unrealistic.

The groundwater velocity of 204 cm/day (6.7 ft/day) and distance to the water body of 457 m (1500 ft) were taken from the WASH-1400 study.[43] These representative values are within the range of values found in current siting practice.

The adsorption retention factor is a function of the porosity and bulk density of the aquifer and the equilibrium distribution coefficient between the solid and the liquid phases for a particular ionic species. Adsorption retention factor values used correspond to about the same degree of realism as in WASH-1400.[43]

As demonstrated in Appendix B, sensitivity of population doses to changes in most of the above parameters in the model is low. Moderate variations in these parameters affect largely only the timing of the releases to the surface water, and not to a large degree the cumulative quantity. Short-lived radionuclides would decay beyond significance before reaching the surface water in the case studied. The longer-lived nuclides are the principle dose contributors for this case. All parameters used in this model are listed in Table A.2.4. Sensitivity studies on the parameters for this model are reported in Appendix B.

A.2.3. Models for Defining Areas of Environmental Impact

No acute fatalities to living organisms are expected from any liquid release from LBPs because the rate of release of radioactivity by way of groundwater is slow at most sites. In the case of FNPs however, concentrations of radionuclides from prompt releases are expected to be high enough to impact some aquatic life. The most susceptible forms of life are marine eggs. Eggs are planktonic and are associated with a particular water mass. Therefore, the eggs would travel with the radioactive plume

Table A.2.4 Parameters for surface water interface model

Average groundwater velocity	=	2.04 m/day (6.7 ft/day)
Longitudinal dispersivity	=	61 cm (2.0 ft)
Porosity	=	0.2
Adsorption retention factors		
a	=	1.0 for ^{3}H
a	=	9.2 for ^{90}Sr
a	=	1.0 for complexed ^{106}Ru
a	=	83 for ^{137}Cs

as it disperses. For the purposes of this analysis, only free-floating organisms are considered. It is recognized that other organisms would also be exposed, but those moving with the plume would accumulate the greatest exposure.

When a particular point in the patch accumulates 50 or more rads, it is assumed that any organism living there will be impacted. Once the organism is exposed to 50 or more rads, it is removed from the computation because it is assumed to have suffered maximum impact.

The above analysis is based on the applicant's instantaneous dispersion model for the offshore FNP and the staff's estuary model for the estuarine FNP. These models were reformulated in a frame of reference located at the center of the dispersing patch, as the patch travels with the ambient currents.

A.2.3.1. Estuary Model

The estuary FNP model assumes instantaneous release of radioactivity into the estuary, and subsequent transport by dispersion and net fresh-water flow in the downstream direction.

The estuary model does not include the effects of sediment, although it is recognized that bottom-dwelling organisms would be subjected to additional exposure by this pathway.

Dispersion of radioactive material in an estuary is a much more complicated phenomenon than represented by the simple tidally-averaged one-dimensional model used, but the model is a better representation far from the site, after initial dispersion within one tidal reach of the point of release has been accomplished. Computations for the example estuary case have shown that the zone of exposures greater than 50 rads is large enough, on the order of 100 km (62.5 mi), that it is not necessary to consider the more complicated aspects of dispersion close to the point of release.

A.2.4. Summary Comparison of Hydrologic Transport Properties

A summary comparison of the hydrologic transport properties of the various water bodies described in Section A.2 is presented in Table A.2.5.

Table A.2.5. Comparative radionuclide transport properties of water bodies

Type of water body	Dissolved phase transport	Sediment effects	Availability of radionuclides to ecosystem	Residence time (for instantaneous release)
Fast flowing rivers	Good. Moves at relatively high velocity.	Fine sediments, which contain most of sorbed radioactivity, will be suspended during high flow periods. Low flows will allow setting. Accounts for only small fraction of radioactivity.	Highly available during residence in river, but residence time relatively short. Contaminated sediments may be frequently resuspended by high flows.	Days to weeks to dissolved phase. Indefinite in sediment phase.
Reservoir on river	Relatively slower than fast flowing river.	Much sediment trapped, except during floods or dredging. Sediment is an important sink for some radioactivity.	Longer residence time than fast flowing river, but some radioactivity removed for dissolved phase by sediments. Bottom-dwelling organisms may be more greatly exposed, however. Sediments may also be occasionally resuspended.	Weeks to months in dissolved phase. Sediment phase indefinite, and probably longer than for fast flowing river.
Great Lakes	Rapid initial dispersion in large-scale turbulence and wind-driven currents until further dispersion limited to dimensions of lake. Flushing out of lake is then slow because of large volume and relatively small flow rate.	Nearly all sediment entering lake is trapped. Very effective sink for some radioactivity.	Residence time long, but substantial radioactivity removed by sediment. Bottom dwelling organisms may be exposed. Sediment resuspension minor - largely confined to coastal areas during storms.	Years to tens of years for dissolved phase once completely mixed in lake. Virtually forever (decay limited) for sediment phase.
Estuaries	Net removal out of system relatively slower than rivers, but dispersion from tidal flushing is significant.	Lower part of most estuaries are sediment traps, with virtually all sediment being deposited. Upper portions may have net downstream sediment transport. Sorption in brackish or salt water appears to be smaller, so sediments are not as important a sink as in fresh water.	High availability for biota but no drinking water pathway in saline portion. Residence time smaller than rivers. Major resuspension of sediment possible during storms.	Weeks to months in dissolved phase. Indefinite in sediment phase.
Oceans	Rapid dispersion in large-scale coastal turbulence: unconstrained by boundaries other than shoreline.	Sediment interaction minor because of low amount of sediments offshore and relatively low sorption potential in salt water. Transport of particle size core-debris possible during storms.	No drinking water pathway Dissolved phase relatively available, but residence time relatively short.	Hundreds of days on continental shelf.
Groundwater (dry site)	Poor dispersion, but radioactivity confined to relatively small region.	Much radioactivity sorbed on aquifer material - even those radionuclides not highly affected in surface water.	Drinking water pathway only; easily interdicted.	Years to hundreds of years. Limited by decay.
Groundwater (Surface water interface)	Slow transport because of small groundwater velocities and sorption.	Sorption slows transport of most radioactivity, allowing greater radioactive decay and protracting source to surface water body.	Assumed no direct usage until surface water encountered.	Months to hundreds of years.

A.3. EXPOSURE PATHWAYS AND DOSE MODELS

A.3.1. Consequence Analysis Considerations

In order to put the accident event spectra for LBPs and FNPs sited in various environments into perspective, numerical estimates were made of the radiation dose to man (consequences). Typically, predictive radiological impact assessments of postulated releases to the environment consider: (1) doses to individuals exposed in various pathways operating in the near field of the release — the so-called maximum exposed individual, and (2) the population or collective dose — a quantity which represents the sum over all contributing pathways, of the individual doses. Although these two quantities are not mathematically independent, i.e., the population is composed of individuals, they do represent distinctly different characterization of the impact of the release.

To obtain numerical estimates of consequences to man, the population dose, S_i, associated with the i^{th} exposure pathway can be expressed as:

$$S_i = \int_0^\infty H\, P_i(H)dH \ ,$$

where $P_i(H)dH$ is the number of individuals receiving a dose in the range H to H + dH through the pathway. It is often not necessary, for the purpose of evaluating the above integral, to accurately assess the contribution of large values of H provided they do not add significantly to the total integral. It is the large values of H which characterize the maximum exposed individuals.

Developing a complete description of the $P_i(H)dH$ distribution is a formidable task which requires detailed site-specific information. This task is particularly difficult if the pathway medium is a commodity which enters the commercial sector for distribution to the population. For these pathways the $P_i(H)dH$ distribution may not be governed by the predicted pathway media concentration distribution in the environment. For example, consider the development of the dose distribution associated with the milk pathway for a postulated airborne release of radioiodines. Given site-specific information as to the location of dairy cows the $P_i(H)hH$ distribution of the thyroid dose could be developed assuming one of the following:

1. the milk is consumed at the location of production;
2. the milk is completely pooled and mixed within the study region; or
3. the mathematical model is available of the dairy industry describing the market distribution.

The above three options would, in all probability, yield distinct $P_i(H)dH$ distributions. However, the population dose estimated by each option would not be expected to be significantly different as in each option the activity contained in the milk would have been consumed. Option 3 would require a considerable expenditure of resources in its development and would only be possible with detailed site-specific information. Thus for the purpose of estimating the population dose resulting from food pathway, the typical approach taken is to determine the total activity moving through the pathway and neglect the details as to the distribution of the commodity in the market. The maximum exposed individual dose serves as an indicator of the upper bound of the dose received by individuals in the population.

The maximum exposed individual is generally associated with pathways in the near field of the releases. In this region the predicted nuclide concentrations in the dispersing medium, and thus the pathway media, are strong functions of the spatial separation of the release and the pathway location. This dependence can be so pronounced that the assessment is largely independent of the general characterization of the environs. For example, the waterborne concentrations in the near field of a discharge into a lake are so dominated by the initial dispersion that for all practical purposes the concentration time history is similar to what might be expected on a river. Thus, the maximum exposed individual dose is insensitive to the general characterization of the environs.

The population dose, as noted above, is the result of an integration over the exposed population. This quantity reflects the physical and biotransport of the released radioactivity throughout the environs and man's usage of the environs. The population dose quantity possesses a high sensitivity to the various siting environments considered in this study and has been employed as the primary numerical index to judge the comparability or lack of it between FNPs and LBPs. It is important to appreciate that the population dose is only a representation of the total exposure of a group of people. Its significance in terms of possible consequences to the group is limited by the extent to which any effects are proportional to the dose received by individuals, i.e., the dose-effect relationship is linear and non-threshold. The population dose was used in this study to make a direct comparison of the exposure situation with another.

A.3.2. Exposure Pathways for the Liquid Environment

In making predictions of the consequences associated with releases of radioactivity, consideration is given to the potential pathways by which radioactivity might be expected to move to man. A number of potential exposure pathways for nuclides released to the hydrosphere are shown in Fig. A.3.1. In practice, it is generally found that a few pathways are so dominant that the multitude of alternative pathways that can be conceived are insignificant contributors to the total consequences for a given release. In this study, the dominant population exposure pathways were found to be the following:

1. ingestion of drinking water;

2. ingestion of aquatic foodstuffs;

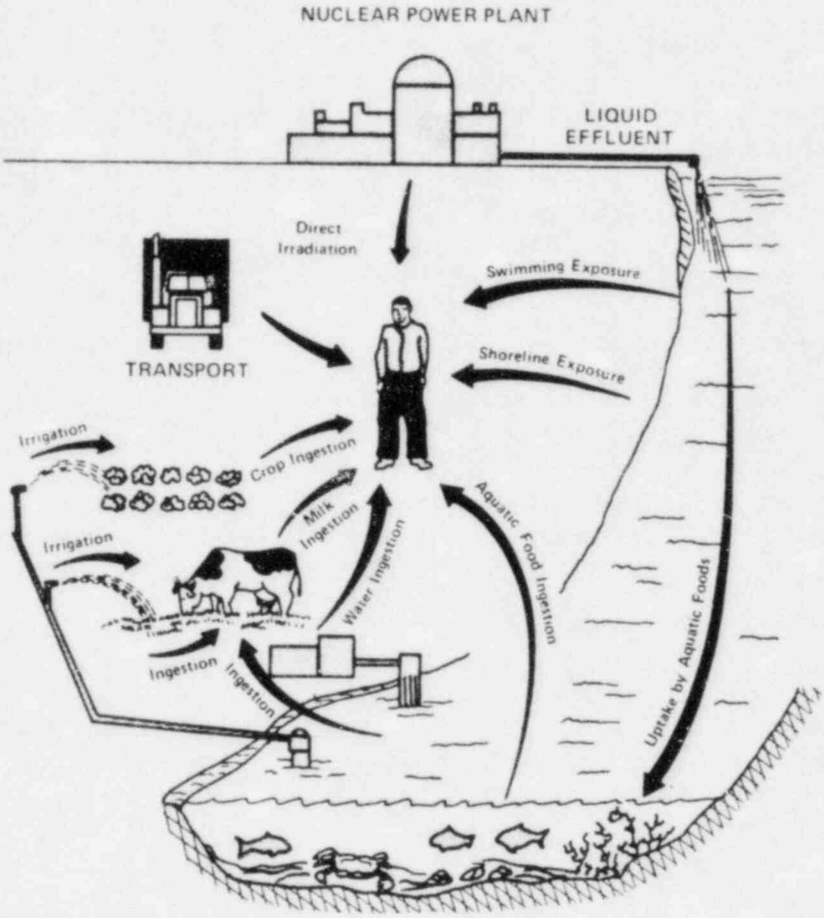

Fig. A.3.1. Exposure pathways to man from radioactivity released to the liquid environment.

3. external exposure to contaminated shoreline sediments (beaches); and

4. external exposure while immersed in the water during swimming.

A number of other pathways that might lead to exposure to a limited population include:

1. external exposure to items contaminated by adsorption of radionuclides from the water, e.g., fishing gear, marine buoys, dredged materials,

2. internal exposure from terrestrial foods contaminated through use of irrigation water or use of trash and spoil fish as fertilizers or animal feed; and

3. external exposure to irrigated land surfaces. These pathways were found not to represent a significant contribution to the population dose, largely as a consequence of the limited number of individuals participating in the pathway.

A.3.2.1. Drinking Water Pathway

The dominant exposure pathway for releases to the fresh-water receiving bodies is drinking water consumption. Factors affecting the dose through this pathway were recently discussed by Soldat.[44] The mathematical model employed in this evaluation is presented in Appendix C. The following generalizations are noted:

1. high usage rate -- liters per day;

2. little delay period within pathway; and

3. exposure duration governed by physico-chemical processes within the water body.

A.3.2.2. Aquatic Food Pathway

Aquatic organisms used by man as food provide a potentially significant exposure pathway for radioactivity released to the aquatic environment. To predict the movement of activity through this pathway relationship between the waterborne concentrations and the aquatic organisms of interest are required. A dynamic model of uptake and retention of the various nuclides was developed (see Appendix C). This model makes use of the bioaccumulation factor concept, which is widely used in the assessment of the aquatic food pathway under chronic release conditions, with the waterborne concentrations time function serving as the driving function for the model. The mathematical details of the models used to evaluate the aquatic food pathway are presented in Appendix C. The following features of the analysis are noted:

1. lower consumption rate than drinking water;

2. delay period between harvest and consumption significant, but highly variable;

3. radionuclides concentrated above water concentration; and

4. exposure duration governed in part by turnover of radionuclides in the aquatic organisms.

A.3.2.3. External Exposure Pathways

Exposure to the dispersing waterborne radionuclides (swimming pathway) and to shoreline or beach deposits of these radionuclides are pathways to external exposure to man. For the swimming pathway, the prediction of the dose is directly related to the waterborne concentration, as estimated by the hydrologic dispersion models. To predict the dose through the shoreline or beach pathway, a relationship between the waterborne concentrations and deposition onto the shoreline is required. The models used in the analysis are presented in Appendix C. The following general features of the direct exposure pathway analysis are noted:

1. usage rate is highly variable among the population;

2. exposure geometry is important — moving meters off the shoreline greatly reduces dose rate; and

3. a natural removal mechanism exists; however, the only mechanism modeled is radiological decay.

A.3.3. Pathway Usage at Fresh-Water Sites

A.3.3.1. River Sites

Two river sites were considered in the selection of representative sites. A large river site is considered where it is possible to construct both an FNP and an LBP. At the small river site, more representative of current river siting of LBPs, only an LBP is considered. The following discussion outlines the assignment of population usage for the river sites.

Large River

Because of the physical size of the FNP, only the lower reaches of major rivers could accommodate an FNP. The site was therefore assumed to be located with a downstream river reach of length 160 km (100 mi) and a width of 630 m (2100 ft). The population usage of a large river site is as follows.

Drinking Water. For purposes of evaluating the drinking water pathway, four public water supply intakes along the downstream river length were assumed. Each intake was considered to serve a population of 25,000 individuals for a total of 100,000 individuals participating in the pathway.

Aquatic Food Ingestion. Appendix D, Table D-8, establishes an annual recreational finfish harvest of 4.5 kg/ha (4 lb/acre) for streams and rivers. To estimate the commercial finfish harvest, the commercial harvest data of Table D-6, not including the aquaculture values, were simply added up and the total annual harvest distributed over the 45 million acres of rivers and streams within the contiguous 48 states. This approach yields an annual commercial finfish harvest density of 2.3 kg/ha (2 lb/acre).

For the large river, width 630 m (2100 ft) and length 160 km (100 mi), the annual recreational and commercial finfish harvests would be about 45,000 and 23,000 kg (100,000 and 50,000 lb) round weight, respectively. This harvest was considered to be uniformly distributed over the downstream reach of the river. The edible portion for both commercially and recreationally harvested fish was taken to be 50%. Shellfish in the river were not found to represent a significant population exposure pathway.

Direct Exposure Pathways. Population usage data were developed to dimension the estimated population exposure through shoreline and water contact recreational activities. The usage data were developed under the assumption that these activities are distributed uniformly throughout the year and among the various water bodies. In this manner, water contact and shoreline recreational activities densities are related to the water surface area in much the same manner as the aquatic food productivity.

Table D-12 of Appendix D provides estimated recreational participation for a number of shoreline/water activities. Swimming participation is estimated to be 1.7 billion user-days/yr in the U.S. Assuming 1 user-hr of actual water immersion per user-day, the above value corresponds to about 4.7 million user-hr/day. If this usage is assigned to the rivers and streams of the United States (area of 45 million acres), a fresh-water swimming usage of 0.2 user-hr/day (0.1 user-hr/acre-day) is indicated.

Boating, sailing, and canoeing activities are estimated, Table D-12, to provide an additional 420 million user-days/yr of recreational participation. Following the same procedure noted above and assuming 3 user-hr of actual activity on the water per user-day, a usage of 0.2 user-hr/ha-day (0.08 user-hr/acre-day) can be estimated. The exposure associated with a water surface activity such as boating can be approximated as one-half of the water immersion exposure (swimming), i.e., 2 π exposure geometry on the water surface vs 4 π exposure geometry while totally immersed. Thus, the boating and swimming population usage can be combined in the analysis with the usage being 0.1 (swimming) + 1/2 • 0.08 (boating) yield an equivalent swimming usage of 0.35 user-hr/ha-day (0.14 user-hr/acre-day).

Shoreline population [illegible] was estimated for the recreational activities of boating, swimming, [illegible]ing, and fishing (see Tables D-10 and D-12).

For the swimming activity, it was assumed that a user-day of participation in swimming would represent 3 user-hr of beach activity. Applying this usage totally to the rivers and streams of the U.S., the shoreline usage associated with swimming is considered to be approximately 0.7 user-hr/ha-day (0.3 user-hr/acre-day).

Shoreline usage associated with boating activities was assumed to be 1 user-hr per user-day of boating. The shoreline usage associated with boating is about one-third of the boating usage, about 0.05 user-hr/ha-day (0.02 user-hr/acre-day).

Waterfowl hunting is estimated to consume 25 million user-days/yr. If it assumed that this recreational usage is on streams and rivers with 6 user-hr of shoreline activity per user-day, a usage of 0.02 user-hr/ha-day (0.009 user-hr/acre-day) can be estimated.

For fresh-water recreational fishing, 590 million user-days/yr was assumed (Table D-10). Based on 6 user-hr of shore-line fishing per user-day of fishing and distributing the recreational fishing over the 18 million ha (45 million acres) of streams and rivers, a usage of 0.5 user-hr/ha-day (0.2 user-hr/acre-day) is indicated.

The total shoreline usage associated with swimming, boating, waterfowl hunting, and fishing is then 1 user-hr/ha-day (0.5 user-hr/acre-day). For the large river the annual population usage would be 1.3 million user-hr and 4.6 million user-hr for swimming and shoreline activities, respectively.

The above parameter assignment characterizes the population usage in the large river. These values were employed in the evaluation of both and FNP and an LBP in this environment.

Small River

A river site more representative of current river siting of LBPs was included in the evaluation. The following sets forth the population usage considered for a small river.

Drinking Water Pathway. The nuclear power reactor siting experience on rivers was reviewed with respect to drinking water usage. Table A.3.1 summarizes the usage as a function of downstream distance for a number of nuclear power reactors. The average values shown in the table were assigned to the small river site for evaluation of the drinking water pathway. For any river-plant location of Table A.3.1, the average values range from a factor of about 40 over estimate to a factor of 2 under estimate.

Table A.3.1. Drinking water population usage on rivers (in thousands)

River reaches (km)							
0-16	16-32	32-80	80-160	160-320	320-640	460-1300	TOTAL
2.3	-	2.3	1.0	17	-	-	22
-	15	-	-	-	-	-	15
-	28	0.20	1.5	-	-	-	30
-	0.55	27	8.1	9.2	71	-	120
29	-	8.6	260	36	81	-	410
33	20	51	400	44	-	-	540
-	-	-	-	-	670	26	700
-	300.	24	-	280	45	71	720
-	-	7.2	300	240	97	80	730
-	-	6.1	0.4	310	330	85	730
-	-	6.1	7.4	320	310	81	730
-	-	45	-	270	350	380	1,000
-	-	-	-	-	330	740	1,100
-	-	140	35	100	710	26	1,100
-	-	360	-	92	870	31	1,400
				Average			
4.3	28	45	67	110	260	100	620

Aquatic Food Ingestion Pathway. The finfish harvest densities discussed above were applied to the small river. For the river model used for the events within design basis, length 200 km (800 mi) and constant width of 110 m (375 ft), the annual recreational and commercial finfish harvests would be 68,000 kg and 34,000 kg (150,000 and 75,000 lb) round weight, respectively. This harvest rate was distributed uniformly among the various reaches of the river. The small river considered for the accident events beyond the design basis has a width that increases with downstream distance. The annual recreational and commercial finfish harvest is 7.7×10^5 kg and 3.9×10^5 (1.7×10^6 and 8.5×10^5 lb) round weight, respectively. This harvest rate is distributed according to the surface-water area of the various reaches of the river.

Direct Exposure Pathways. The shoreline and swimming usage density values were derived above. For the river model used in the events within design basis, length 1300 km (800 mi), constant width of 110 m (375 ft), the annual shoreline and swimming population usage would be 3.8×10^6 and 1.0×10^6 user-hr, respectively. For the small river considered in the analysis of events beyond the design basis, the annual shoreline and swimming population usage would be 8.8×10^7 and 2.2×10^7 user-hr, respectively.

A.3.3.2. Great Lakes Site

A Great Lakes site was considered in the representative siting spectra for LBPs. The following population usage values were used in the analysis.

Drinking Water Pathway

The nuclear power reactor siting experience on the Great Lakes was reviewed with respect to drinking water usage. Table A.3.2 summarizes the usage as a function of distance from various sites. The usage in any interval is the total for the interval on both sides of the facilities.

Table A.3.2 Near-field drinking water population usage on lakes* (in thousands)

Distance intervals (km)						
0-8	8-16	16-32	32-48	48-64	64-80	TOTAL
-	-	7.8	21	43	-	71
-	-	-	380	-	-	380
-	14	-	-	-	-	14
79	5	-	-	-	-	84
2	25	-	-	-	-	27
0.75	-	-	-	-	-	0.75
-	25	3.0	5.0	-	370	400
-	-	160	-	60	-	220
-	-	140	-	-	52	190
63**	190**	190**	190**	3,200**	2,700**	6,600**
Average*						
9.0	7.7	34	46	12	47	

*Note only the 0- to 32-km (0- to 20-mi) region is used in near-field model.

**Not used in computing the averages.

In determining the average usage for the various intervals or regions, the last entries (last row of data) were not considered. These data (last row) were discarded because they totally dominated the computed average; such a dominated average was concluded not to represent usage for a typical lake site.

The evaluation of the lake site utilized several hydrological dispersion models; for the drinking water pathway, near-field and mixed-lake models were used. As discussed in Section A.2.1.3, the near-field model covered the region out to 24 km (15 mi) from the release point. For this model then, the average values for the first three intervals were used. That is, the population usages of 9,000, 7,700, and 34,000 were evaluated at distances of 4, 12, and 24 km (2.5, 7.5, and 15 mi), respectively. For the mixed-lake model, where the release is mixed totally within the lake and removed only by the flow-through of fresh water into the lake, radiological decay and sedimentation, a total drinking water population usage for the lake of 2 million was assumed. This usage corresponds to the withdrawal of municipal water supplies on Lake Ontario,[45] Table A.3.3. As seen from Table A.3.3, the value of 2 million is about an order of magnitude higher usage than observed for Lake Superior, and about a factor of 5 lower than the Lake Michigan usage.

Table A.3.3. U.S. municipal water withdrawal (1970)

Lake	Withdrawal* $\times 10^6$ L/day	Population** (million)
Superior	180.	0.26
Michigan	7700.	11.
Huron	500.	0.71
Erie	6700.	9.5
Ontario	1400.	1.9

*Reference (Great Lakes Basin Commission 1975).
*Population computed use withdrawals and a per capita usage of 710 L/day (190 gal/day).

In addition to U.S. municipal water withdrawal on the Great Lakes, the potential exists for impact on Canadian water usage. For example, the Toronto metropolitan area of about 3 million derive their drinking water from Lake Ontario.

Aquatic Food Ingestion. Appendix D estimates a recreational finfish harvest of 17 kg/ha/yr (15 lb/acre/yr) from reservoirs and small lakes, 5.6 kg/ha/yr (5 lb/acre/yr) from Lake Erie, and 1.1 kg/ha/yr (1 lb/acre/yr) from the other Great Lakes (see Table D-6).

A recreational finfish harvest density of 5.6 kg/ha/yr (5 lb/acre/yr) for the typical Great Lake was used in this analysis. The total U.S. commercial harvest of edible finfish from the Great Lakes drainage basin is estimated to be 15×10^6 kg (32×10^6 lb) (see Table D-3). The Canadians harvest an additional 17×10^6 kg (37×10^6 lb) of fish for human food from the same Great Lakes waters, excluding Lake Michigan.

Assuming the U.S. harvest is uniformly distributed over the 25×10^6 ha (61×10^6 acres) of Great Lakes water surface area, a harvest of 0.6 kg/ha/yr (0.5 lb/acre/yr) is indicated. For the typical Great Lake used in the evaluation 2×10^6 ha (5×10^6 acres), the annual recreational and commercial finfish harvest would be 11×10^6 kg and 1.1×10^6 kg (25×10^6 and 2.5×10^6 lb), respectively. These values were employed in the evaluation over time periods when the mixed lake dispersion model was applicable.

The lake evaluation, in addition to the mixed lake dispersion model, made use of a near-field and patch-type model. The near-field region was divided into three sub-regions. In these regions, fishing was taken to extend out to 1 mile from shore. The harvest rate for the various regions was assigned based on the length of one region, the assumed width, and the conservatively estimated recreational and commercial harvest rates of 6 and 0.6 kg/ha/yr (5 and 0.5 lb/acre/yr), respectively. The shoreline water concentrations at the midpoints of the regions were used as the driving functions for the fish uptake model. Applying the nearshore waterborne concentration out to 1 mile results in an overestimate of the near-field contribution to the population exposure. However, the total population exposure from fish ingestion pathway is dominated by the estimates derived from the patch-type model and/or the mixed-lake model, and thus the overestimate in the near field is not of major concern.

For time periods when the patch-type model was applicable, the harvest rates were integrated over both time and patch area. For the generic lake, the time period for which the patch model was applicable was from about 3 days to 13 days following the accidental release. For evaluation periods in excess of 13 days, the mixed-lake model was employed with the entire lake harvest being considered.

Direct Exposure Pathways. Population shoreline and swimming usage densities were developed in Section A.3.3.1. These values were reduced by a factor of one-half, as a seasonal consideration on the Great Lakes. Thus, the annual participation in these pathways is 4.4×10^8 and 1.2×10^8 user-hr for the shoreline and swimming, respectively.

A.3.3.3. Dry Site

The dry site is an LBP site where the downgradient movement of the groundwater is away from any adjacent surface-water bodies or the site is far enough away that no radionuclides would ever reach the surface water. This site is a complement to the other land-based sites in that it represents an alternative to the assumption that the released activity intercepts the surface water. The dominant exposure pathway at this site is the consumption of water obtained from the aquifer by wells. Because of the long transport times in the aquifer, the affected wells will largely be restricted to wells within the immediate site region. For purposes of modeling, the wells are taken to be uniformly distributed serving 10 individuals per square mile. This density corresponds to a rural, farming-type environment.

A.3.4. Pathways Usage at Salt-Water Sites

A.3.4.1. Estuary

Analytical evaluations of the accident event release spectrum were carried out for both an FNP sited within and an LBP adjacent to this water body. Presented below are the population usage parameters employed in the evaluation. Note that in this environment, the drinking water pathway does not exist.

Aquatic Food Ingestion Pathway

The harvest, both commercial and recreational, of aquatic foods used in the analysis is based on the data of Appendix D and is tabulated in Table A.3.4. These values were employed for the analysis of both the FNP and the LBP in the estuarine environment.

Table A.3.4. Annual harvest from estuary

	Density (kg/ha)	Harvest* (million kg)
Commercial**		
Finfish	11	1.1
Crustacea	18	1.8
Mollusks	11	1.1
Recreational		
Finfish	93	9.5
Crustacea	11	1.1
Mollusks	11	1.1

*The harvest value is the annual production within the affected region of the estuary, 210-km long by 5-km wide (130-mi long by 3-mi wide).

**Note that in Appendix D no data basis was identified which permitted development of an estimate. In the consequent analysis, the commercial harvest was assumed.

Direct Exposure Pathway

Population shoreline and swimming usage densities were developed in Section A.3.3.1. These values were employed in the estuarine environment. For the estuary, length 210 km (130 mi), width 5 km (3 mi), the annual pathway usage is 2.6×10^8 and 7.3×10^7 user-hr for the shoreline and swimming, respectively.

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APPENDIX B

Excerpted from Appendix B:
"Radionuclide Transport Models" in NUREG-0440
Liquid Pathway Generic Study: Impacts of Accidental
Radioactive Releases to the Hydrosphere from Floating
and Land-Based Nuclear Power Plants

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APPENDIX B: RADIONUCLIDE TRANSPORT MODELS

B.1. INTRODUCTION

Radionuclide transport models were developed in accordance with the guidelines of Appendix A in Section A.1. All radionuclide transport models for surface water and groundwater are based on the principle of conservation of mass. In some cases, analytical solutions were formulated for instantaneous releases of radionuclides. Computations for more general releases are generated using the convolution integral:

$$C(t) = \int_0^t C_i(t - \tau)f(\tau)d\tau \ , \qquad \text{(B-1)}$$

where $C(t)$ is the concentration at time t, $C_i(t - \tau)$ is the analytical solution for concentration at time $t - \tau$ for an instantaneous release of 1 curie at time $t = 0$, and $f(\tau)$ is the function defining a noninstantaneous rate of release of radioactivity in curies/sec. Solution of Equation (B-1) is performed by numerical quadrature.

This appendix describes the transport models for groundwater, rivers, lakes, and estuaries.

B.2. SURFACE-WATER MODELS

All surface-water transport models are based on the solution of the convective diffusion equation in simple geometries, with steady unidirectional flow:

$$\frac{\partial C}{\partial t} + u\frac{\partial C}{\partial x} + v\frac{\partial C}{\partial y} + w\frac{\partial C}{\partial z} = D_x\frac{\partial^2 C}{\partial x^2} + D_y\frac{\partial^2 C}{\partial y^2}$$

$$+ D_z\frac{\partial^2 C}{\partial z^2} - \lambda C + W(t) - S(t) = 0 \ , \qquad \text{(B-2)}$$

where u,v,w are the velocities of water in the x, y, z directions, respectively; D_x, D_y, D_z are the dispersion coefficients in the x, y, and z directions, respectively; λ is the radioactive decay coefficient; $W(t)$ is a distributed source, and $S(t)$ is a distributed sink.

In general, each surface-water model is developed from a simplification of Eq. (B-2). $W(t)$ and $S(t)$ are source and sink terms for water-solid interactions when these effects are included in the models.

B.2.1. River Models

Large River

Equation (B-2) may be simplified for a vertically integrated two-dimensional model in a straight rectangular channel with flow parallel to the shore:

$$\frac{\partial C}{\partial t} + u \frac{\partial C}{\partial x} = E_x \frac{\partial^2 C}{\partial x^2} + E_y \frac{\partial^2 C}{\partial y^2} - \lambda C \ , \qquad \text{(B-3)}$$

where E_x and E_y are the vertically integrated dispersion coefficients in the x and y directions, respectively.

The resulting concentration in a straight rectangular channel of width B and cross-sectional area A, follows with steady flow (as depicted in Fig. B-1) corresponding to an instantaneous release at $t = 0$ of a unit quantity of material (1 curie) from a vertical line source at $x = 0$, $y = y_1$:

$$C_i = \frac{1}{\sqrt{4\ E_x t\ A}} \exp \left[- \frac{(x - ut)^2}{4E_x t} - \lambda t \right] \left[1 + 2 \sum_{n=1}^{\infty} \exp \left(- \frac{n^2 \pi^2 E_y t}{B^2} \right) \cos n\pi \frac{y_s}{B} \cos n\pi \frac{y}{B} \right] \qquad \text{(B-4)}$$

For a more general time-dependent release, the concentration may be computed using the convo ution integral, Eq. (B-1).

Small River

A radionuclide transport river model was developed to identify the most important transport features of a typical river system. This model is loosely based on the Clinch-Tennessee-Ohio-Mississippi River System. This system covers a wide range of conditions from the moderate size Clinch River and Tennessee River which are extensively dammed, to the very large Ohio and Mississippi River mainstreams which are undammed. This system is one of the few U.S. river systems which has been studied from the standpoint of the transport of radioactive waste.[1]

The model has been developed under the following assumptions:

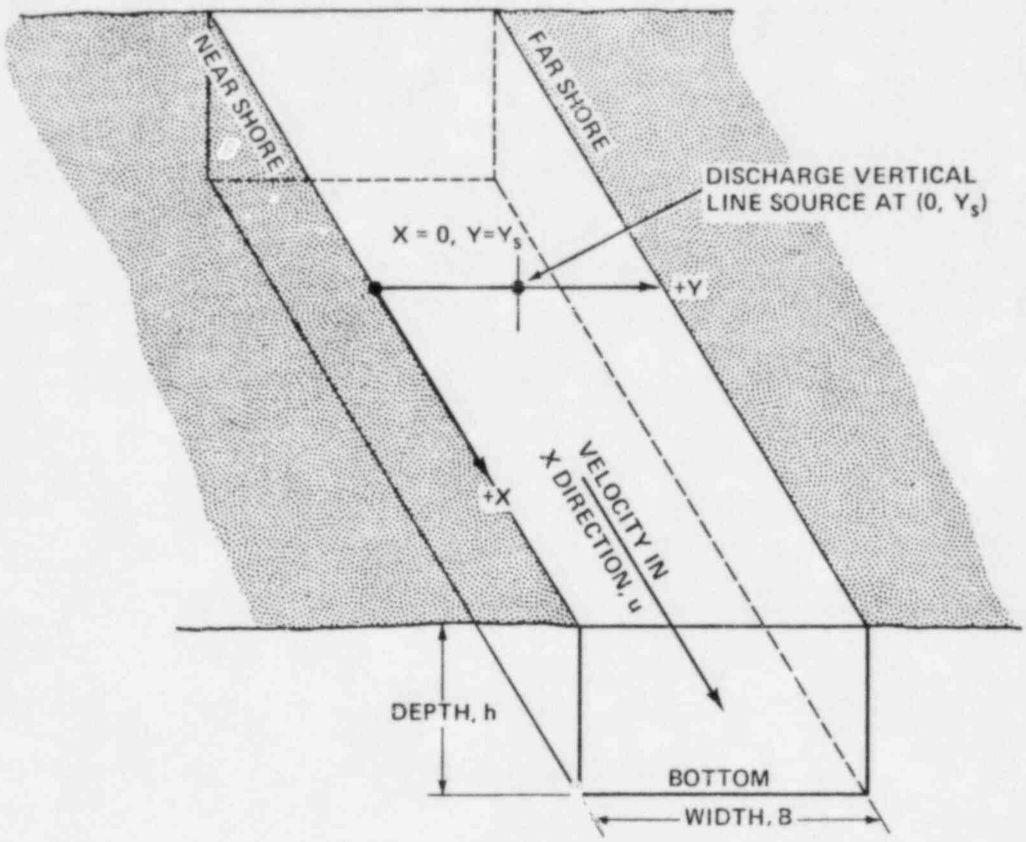

Fig. B-1. Geometry of simple 2-dimensional river model.

1. The source term is of long duration relative to the other dynamics of the river system. This is true only for the accident, where the contamination is released to the river through the groundwater pathway;

2. Sediments affect the transport of radionuclides by scavenging from the water column and by burial; and

3. Sediment effects are important in the reservoir segments of the system only, and not the relatively fast-flowing parts of the system.

Description of Model

The model is divided into two parts; a reservoir model and a river model. These parts are depicted schematically in Fig. B-2.

The reservoir model was applied to the Clinch and Tennessee Rivers, where the effects of sedimentation are considered to be highly significant. On the Ohio and Mississippi Rivers, however, concentrations were computed using only straight dilution (fully mixed conditions). This part of the river system is relatively fast flowing, so sedimentation and radioactive decay are presumed to be of less importance than in the

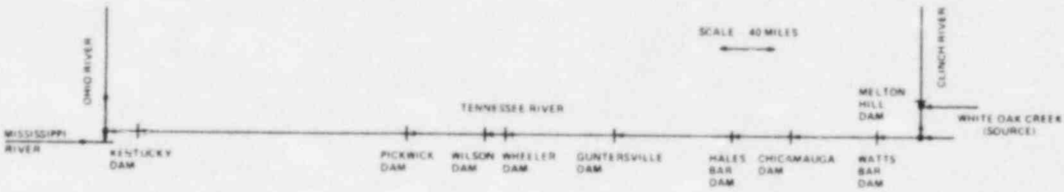

Fig. B-2. River system model representation.

upstream reservoirs. This assumption is probably conservative, since parts of the Mississippi River experience significant sediment buildup, as evidenced by shifting river channels and sandbars.

Reservoir Sections

Each reservoir in the Tennessee River is modeled as if it were perfectly mixed, which allows the use of a form of the Great Lakes mixed tank model. Details of the individual mechanisms of the model are described in B.2.3 and will not be repeated here. For the i^{th} reservoir, the concentrations of the water and sediment are described by the equations:

$$\frac{dC_i}{dt} = \frac{q_{i-1}\ C_{i-1}}{V_i} + C_{si}\lambda_{1i} - C_i\lambda_{2i}\ , \tag{B-5}$$

$$\frac{dC_{si}}{dt} = C_i\lambda_{3i} - C_{si}\lambda_{4i}\ , \tag{B-6}$$

where

C_i	=	the water phase concentration;
C_{si}	=	the sediment phase concentration;
q_{i-1}	=	the flow from the previous reservoir;
C_{i-1}	=	the concentration from the previous reservoir;
V_i	=	the volume of the reservoir;

$$\lambda_{1i} = \frac{K_f}{d_{1i}\ K_d}\ ,$$

$$\lambda_{2i} = \frac{q_i}{V_i} + \lambda + \frac{\varepsilon v_i\ K_d}{d_{1i}} + \frac{K_f}{d_{2i}}\ ,$$

$$\lambda_{3i} = \frac{\varepsilon v_i K_d + K_f}{d_{2i}} ,$$

$$\lambda_{4i} = \lambda + \frac{\varepsilon v_i}{d_{2i}} + \frac{K_f}{d_{2i} K_d} ,$$

K_f = the coefficient for direct transfer from the water to bottom sediment;

K_d = the equilibrium sorption coefficient;

ε = the sediment effectiveness factor;

d_{1i} = the average depth of the water layer in the reservoir;

d_{2i} = the average depth of the effective sediment layer in the reservoir;

λ = the radiological decay coefficient = ln 2/half life; and

v_i = the sedimentation rate.

If the rate a radionuclide enters the river system is constant, or at least changing very slowly, a considerable simplification can be performed by assuming that Eq. (B-5) and (B-6) are steady state. The equations can then be solved directly to give:

$$C_i = \frac{C_{i-1}\, q_{i-1}}{V_i} \Big/ \left(\lambda_{2i} - \frac{\lambda_{1i}\, \lambda_{3i}}{\lambda_{4i}} \right) . \tag{B-7}$$

The steady-state model is valid only for sources of long duration, e.g. the dynamics of the river system are fast in relation to the dynamics of the source term.

In the Clinch River, a slightly different form of the reservoir model was employed in order to get a more realistic representation of concentrations in the vicinity of the release, which was assumed to be located 33.3 km (20.8 mi) upstream of the mouth. For this case, a plug-flow rather than the mixed-tank assumption was employed, where the radionuclide from the source was considered to be moving as a plug in a uniform channel downstream to the mouth, as illustrated in Fig. B-3.

Where the Clinch River joins with the Tennessee River, the regular mixed-tank reservoir model is employed, except that only part of the Watts Bar reservoir volume is used (0.072×10^6 vs. 0.15×10^6 ha-m), since the volume above the Tennessee River junction with the Clinch River would be largely uncontaminated.

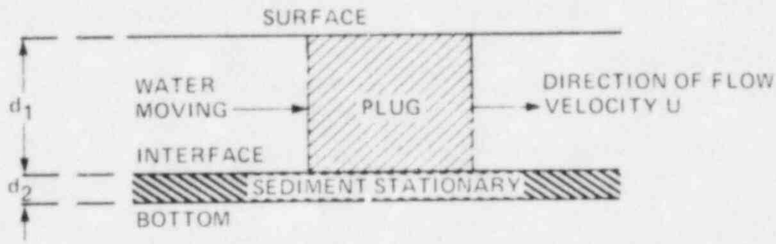

Fig. B-3. Plug flow river model.

In a steady, uniformly flowing, plug-flow channel with the same mechanisms as described for the mixed-tank model, the concentration of water and sediment can be described by the equations:

$$u \frac{dC}{dx} = \lambda_1 C_b - \lambda_2 C , \tag{B-8}$$

$$C_b = \frac{\lambda_3}{\lambda_4} C , \tag{B-9}$$

where

$$\lambda_1 = \frac{K_f}{d_1 K_d} ,$$

$$\lambda_2 = \lambda + \frac{\varepsilon v K_d}{d_1} + \frac{K_f}{d_1} ,$$

$$\lambda_4 = \lambda + \frac{\varepsilon v}{d_2} + \frac{K_f}{d_2 K_d} ,$$

u = average velocity in the channel,

and the other terms are as previously described.

Equations (B-8) and (B-9) may be solved to give an expression for the water-phase concentration

$$C = \frac{W(t)}{q} \exp \left[\left(\frac{\lambda_1 \lambda_3}{\lambda_4} - \lambda_2 \right) x/u \right] , \tag{B-10}$$

where

x = the distance from the source;
$W(t)$ = rate of release of radioactive material at the source; and
q = flow-rate past the source.

Selection of Parameters

Sedimentation Rate Fresh and contaminated sediments will accumulate primarily in the backwater reaches of reservoirs. Sediment accumulation in the TVA reservoirs has been measured, and an average sedimentation rate, v, calculated by assuming uniform deposition over the entire reservoir. The computed rate ranges from about 0.61 to 1.28 cm/yr (0.24 to 0.5 in./yr).

Preliminary computation of radionuclide concentrations in the Clinch and Tennessee Rivers resulting from radioactive releases from Oak Ridge National Laboratory indicated that removal rates of highly sorbed substances were greater than those actually observed. The error could not be explained solely on the basis of imprecise values of model coefficients such as K_d, K_f, reservoir volume, and depth. The probable cause of the error was the assumption of a steady fallout rate of sediment, uniformly distributed throughout the reservoir. Sedimentation in reservoirs is far from being uniform and steady. Sediment entering from the main upstream channel or from tributaries may fall out in an alluvial fan far from the impounding dam.[2] There may be a considerable segregation of particles, with the finest being transported most easily and the coarser ones only during floods. Periods of highest dissolved concentrations may not correspond to periods of highest suspended concentrations because of the high order dependence of sediment transport on flow rate. Finally, the ability of sediments to sorb radionuclides from solution is strongly dependent on particle size and mineral composition, with the clay-size fraction being the most effective.

The effectiveness factor, ε, is used to account for these complicated effects of sediment-radionuclide interaction in the river system. The river model was adjusted to match field measurement of radioactive sediment in the Tennessee River by adjustment of the single parameter ε. A value of $\varepsilon = 0.1$ was found to give the most satisfactory results with the other parameters chosen for the system. The model-prototype comparisons are given in Figs. B-4 through B-7. Although agreement is far from being perfect, the model was considered to be acceptable. The fact that the chosen effectiveness factor reduces the observed sediment rates to those more commonly found in lakes (i.e., 0.01-0.1 cm/yr) adds credibility to the model.[3-5]

Flow Rate

Since the river model is steady state, it can only accept constant flow rates. It is important to pick an average flow rate which is correctly weighted. Concentration of radionuclides in the dissolved phase in a

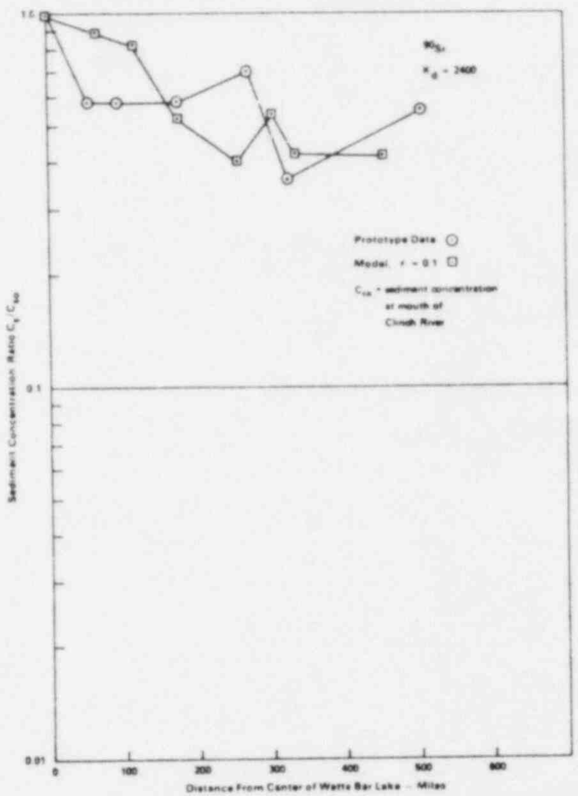

Fig. B-4 Model — prototype comparison for ^{90}Sr sediment concentrations in Tennessee River based on Clinch River study data. Source: P. H. Carrigan et al., Radioactive Material in Bottom Sediment of the Clinch River: Part A — Investigation of Radionuclides in Upper Portion of Sediment, ORNL-3721, Sup. 2a, Stat. Rep. 5, Union Carbide Corp., Nuclear Div., Oak Ridge Natl. Lab., Oak Ridge, Tenn., March 1967.

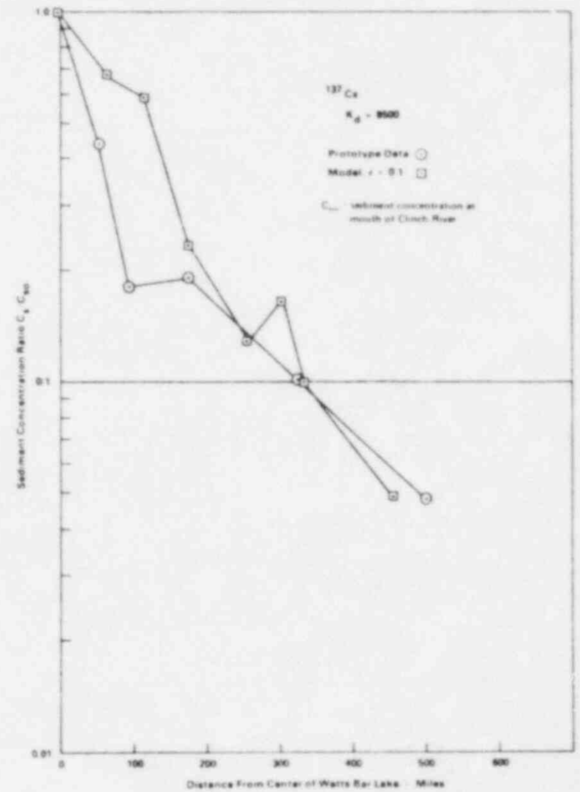

Fig. B-5 Model — prototype comparison for ^{137}Cs sediment concentrations in Tennessee River based on Clinch River study data. Source: *ibid.*

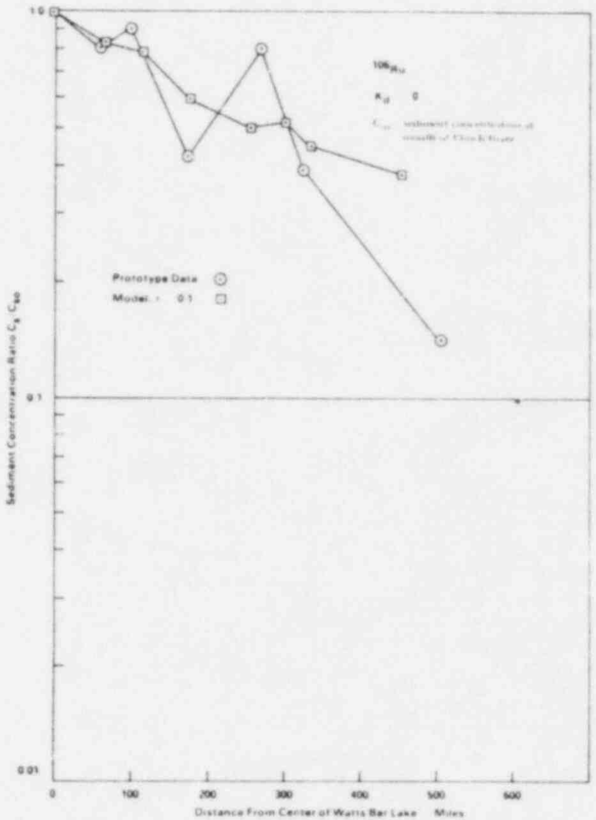

Fig. B-7 Model — prototype comparison for ^{106}Ru sediment concentrations in Tennessee River based on Clinch River study data. Source: *ibid.*

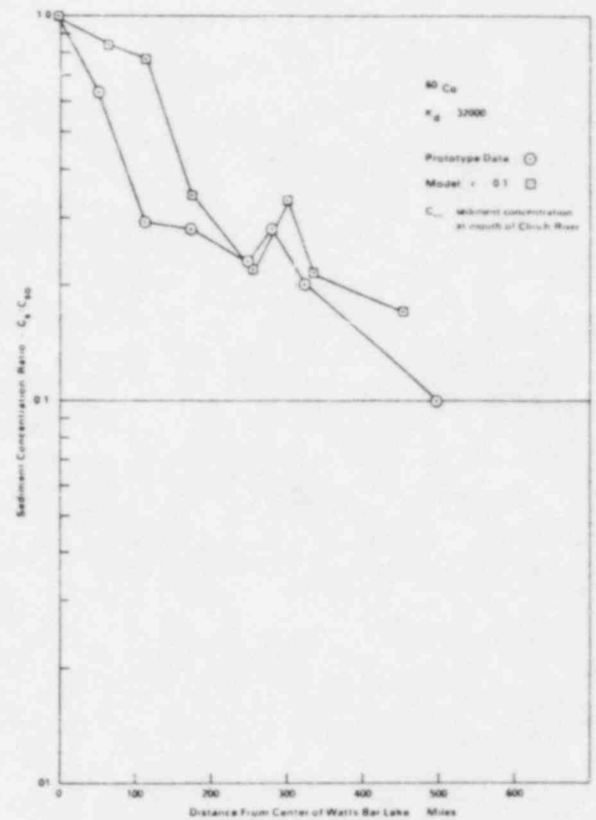

Fig. B-6 Model — prototype comparison for ^{60}Co sediment concentrations in Tennessee River based on Clinch River study data. Source: *ibid.*

river is strongly dependent on the reciprocal flow rate Q^{-1}. Sediment-phase transport, however, is a function of Q^n, where n is positive and greater than one.

For calculating average concentrations in the dissolved phase, the use of the arithmetic mean flow $\overline{Q}$ would yield optimistically high dilutions. $\overline{Q}$ is weighted more heavily by floods, when the dissolved concentrations are low, than by droughts, when the dissolved concentrations are high. A more conservative low flow would be based on the arithmetic mean of the reciprocal, $\overline{Q^{-1}}$. This average is always smaller, and would be more correctly weighted for floods and droughts. The technique must be used carefully, however, because it degenerates for flows close to zero.

Sediment is transported most readily during periods of high flow, so an average based more heavily on floods than droughts would be expected to be more representative of radioactivity carried by sediments. The concentration used in the dose computations was for both the dissolved and suspended phases. The choice of the most appropriate flow rate for the river model was handled in an arbitrary manner, by taking for all but the first segment, the average of both the arithmetic mean and the reciprocal of the inverse arithmetic mean flows, $\dfrac{\overline{Q}+\left(\overline{1/Q}\right)^{-1}}{2}$. Justification for this choice must rely partially on consideration of the fact that flow rates are among the best known of the coefficients used in the model. In addition, the difference between the two types of averages was usually less than a factor of 2.

For the Clinch River segment, radioactivity would be mostly in the dissolved phase, so only the reciprocal average flow rate was used.

Figure B-8 shows the average and reciprocal average flow rates for the river system as a function of distance from the presumed source for the period from October 1960 to October 1961. The reciprocal average flow rate appears to be anomalously low at one point. This was caused by several periods of very low flow from Kentucky Dam, which heavily weighted the average. This average is not consistent with the general trend of increasing flow with increasing downstream distance. For this and other physical reasons, this data point was discarded. The dashed curve shown was the reciprocal average flow actually used.

River-Like Sections

The Ohio River and Mississippi River sections of the model include only the effects of dilution with the increasing flow in the downstream direction. Sedimentation in these sections is considered to be far less important than in the reservoir sections. Radioactive decay would be relatively unimportant in these fast-flowing rivers for the radionuclides considered.

The complete model as it was used for subsequent computations is shown in Figure B-9, for ^{137}Cs and ^{3}H, in terms of dilution factor as a function distance from the source. Only these two curves are presented because they represent the extremes of sediment effects.

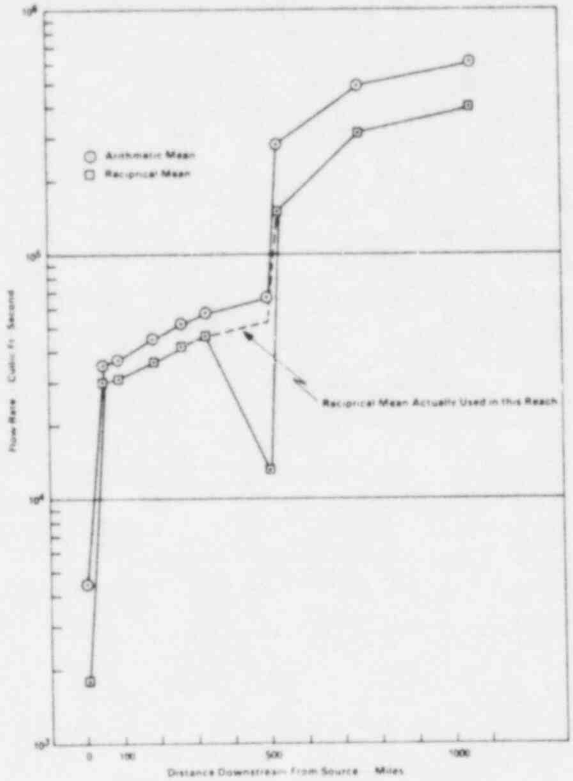

Fig. B-8. Arithmetic mean and reciprocal mean flow rates used in river model.

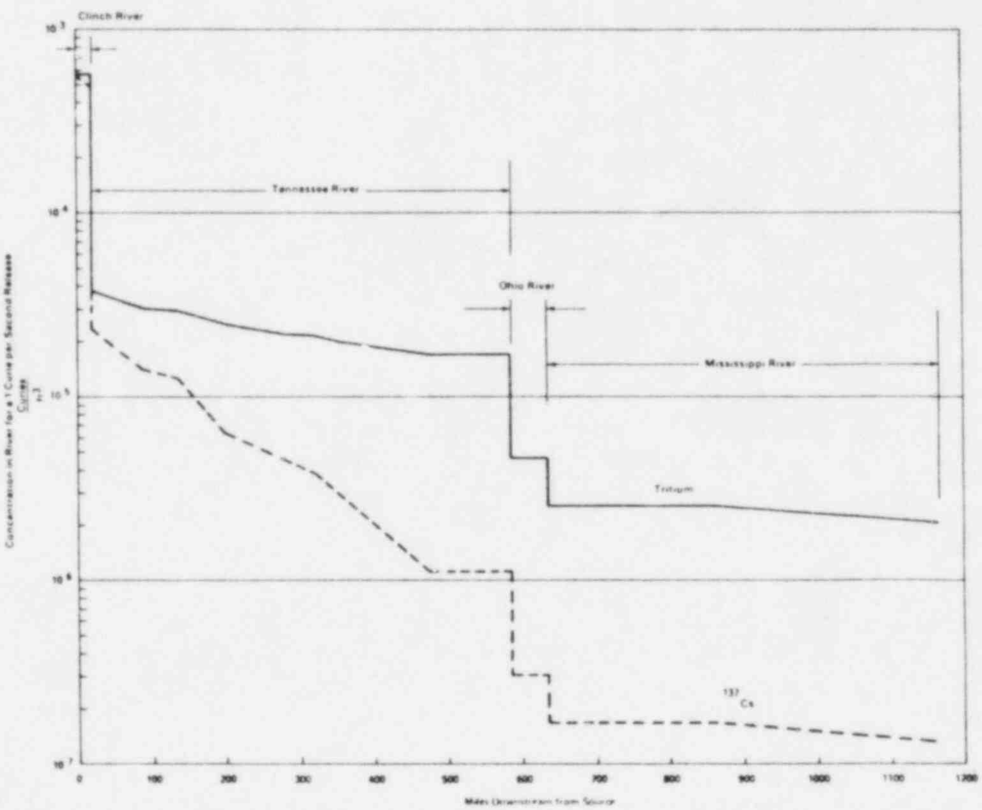

Fig. B-9. Concentrations in Clinch-Tennessee-Ohio-Mississippi River system for core melt model.

B.2.2. Estuary Models

Conserving Substance

A one-dimensional, sectionally averaged model was considered to be adequate for an estuary. There were assumed to be no potable water intakes along the shores of the estuary chosen for this study. Only concentrations averaged across the estuary sections were needed for the purposes of evaluating the exposure, so it was not necessary to evaluate lateral dispersion.

Equation (B-2), when simplified for the one-dimensional estuary with a constant cross section, as depicted in Fig. B-10, becomes:

$$\frac{\partial C}{\partial t} + U_f \frac{\partial C}{\partial x} = E_L \frac{\partial^2 C}{\partial x^2} - \lambda C \ , \qquad \text{(B-11)}$$

where U_f is the net downstream fresh-water velocity, and E_L is the longitudinal dispersion coefficient (assumed constant).

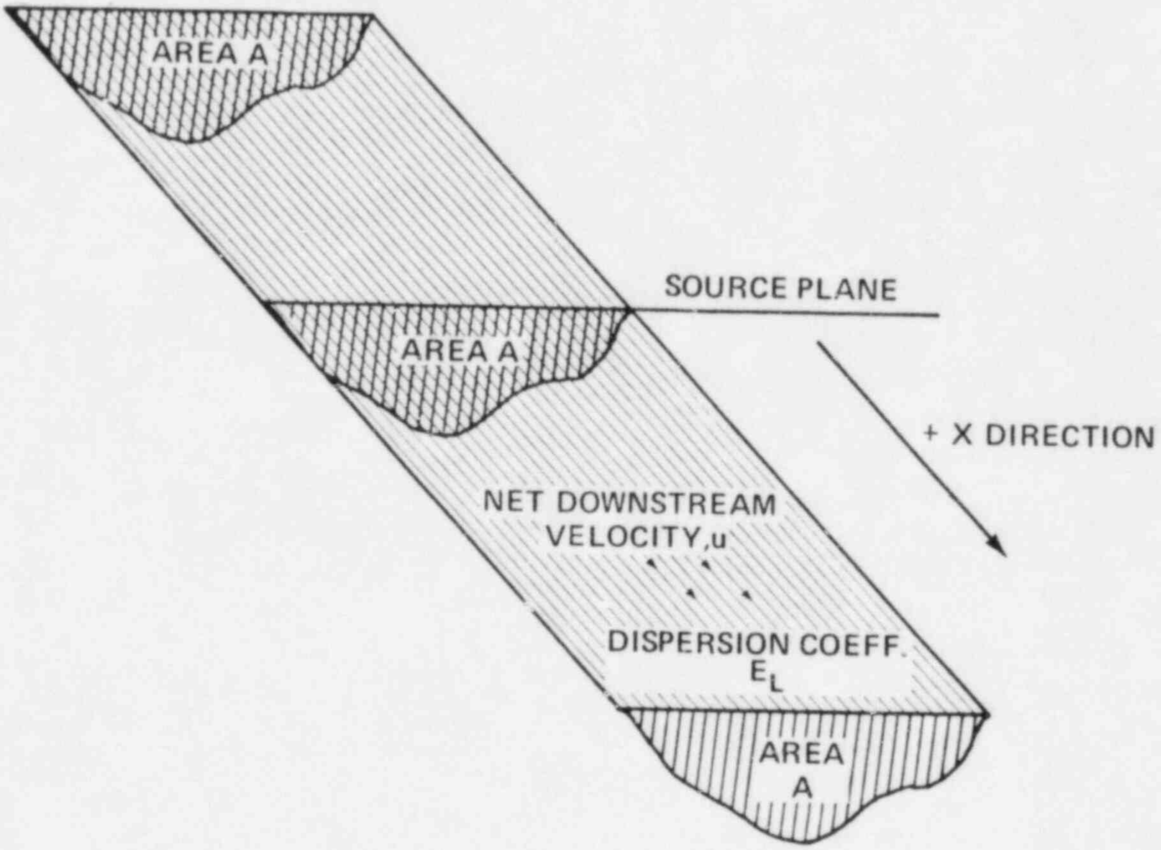

Fig. B-10. Uniform 1-dimensional estuary.

In this simple approach, the tidally averaged approximation is used, in which the tidal currents are not included explicitly as an advective mechanism, but are considered to be responsible for large-scale longitudinal dispersion. This technique, while simple, has been successfully used in studies of the dispersion of pollutants in several estuaries.[6-7] The low sensitivity of population dose to large changes in the coefficient E, as will be demonstrated, suggests that the appropriation is acceptable.

The analytical solution of Eq. (B-11) corresponding to an instantaneous release at t = 0 of a unit quantity of material uniformly over the cross section is:

$$C_i = \frac{1}{A\sqrt{4\pi E_L t}} \exp \left[- \frac{(x - U_f t)^2}{4E_L t} - \lambda t \right] , \tag{B-12}$$

where A is the estuary cross-sectional area.

For a more general time-dependent release, results may be obtained using the convolution integral, Eq. (B-1).

Model with Sedimentation

The estuary model above can be extended to account for the effects of sediments. As illustrated in Fig. B-11, a water layer of thickness d is in contact with a sediment layer of thickness d_2. The water layer is moving with a net downstream velocity (nontidal) of U_f, and the bed is moving with a net downstream velocity U_b. Diffusive transport from tidal oscillations in the water and sediment layers is assumed to be with constant longitudinal dispersion coefficients E and E_b, respectively. Sedimentation and burial occur uniformly at a rate v.

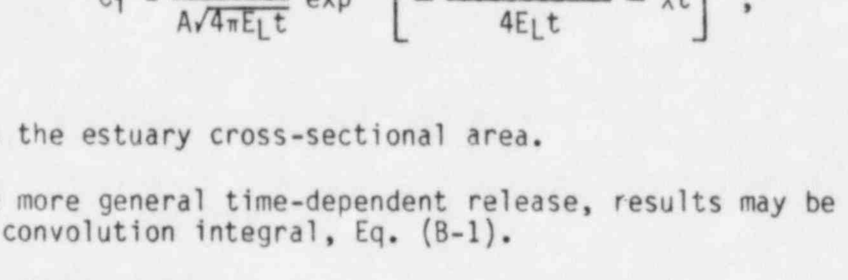

Fig. B-11. One dimensional estuary model with sedimentation.

The mechanisms of transport of radionuclides between the water and sediment phases in the estuary model are considered to be substantially different from those in the reservoir and Great Lakes models. Estuaries have substantially faster flowing water during part of the tidal cycle than the average flow would indicate, yet relatively small downstream net transport. This type of flow behavior would allow for resuspension and subsequent redepositions of fine sediment during each tidal cycle. Sediment and water would be in more intimate contact in an estuary than in a reservoir or lake. It was therefore assumed that the sediment and water are in chemical equilibrium, and their concentrations are related by the equilibrium distribution coefficient, K_d:

$$C_s = K_d \, C \, , \tag{B-13}$$

where C_s is the radionuclide concentration on the sediment, and C is the radionuclide concentration in the water.

The assumption of complete equilibrium represents the upper limit of sediment effects; whereby neglecting sediment effects, represents the opposite extreme. It is reasonable to expect that the correct model lies somewhere between the two extremes. Experience has indicated that population doses computed with both models indicate only minor differences, although there may be significant time dependent variations.

The differential equation describing the concentration in the water phase becomes:

$$\frac{\partial C}{\partial t} + U' \frac{\partial C}{\partial x} = E_L' \frac{\partial^2 C}{\partial x^2} - C\lambda \, , \tag{B-14}$$

where

$$U' = \frac{fU + (1 - f)U_b K_d}{f + (1 - f)K_d} \, ,$$

$$E_L' = \frac{fE + (1 - f)E K_d}{f + (1 - f)K_d} \, ,$$

$$f = \frac{d_1}{d_1 + d_2} \, ,$$

$$(1 - f) = \frac{d_2}{d_1 + d_2} \, ,$$

and other terms are as previously defined.

The solution to Eq. (B-14) for an instantaneous unit release at x = 0 is

$$C = \frac{1}{aA\sqrt{4\pi E_L' t}} \exp\left[-\frac{(x - U't)^2}{4\ E_L'\ t} - \lambda t\right], \tag{B-15}$$

where

$$a = f + (1-f)\ K_d.$$

The solution is generalized for arbitrary releases by using Eq. (B-1).

B.2.3. Great Lakes Models

Two models are used for dispersion estimates in the Great Lakes in order to cover the nearshore and totally mixed regimes. Sedimentation is not considered in the nearshore model.

Nearshore Model

A simple vertically integrated diffusion model for discharge into a lake having only an alongshore current may be formulated from Eq. (B-2):

$$\frac{\partial C}{\partial t} + u\frac{\partial C}{\partial x} = E_x \frac{\partial^2 C}{\partial x^2} + E_y \frac{\partial^2 C}{\partial y^2} - \lambda C\ . \tag{B-16}$$

In a lake having constant depth, h, a straight shoreline, constant dispersion coefficients, and a constant alongshore velocity (see Fig. B-12), the concentration resulting from an instantaneous release at t = 0 of 1 curie from a vertical line source at x = 0, y = y_s is:

$$C_i = \frac{1}{4\pi\sqrt{E_x E_y}\ th} \exp\left[-\frac{(x - ut)^2}{4\ E_x t} - \lambda t\right] \left\{\exp\left[-\frac{(y - y_s)^2}{4E_y t}\right] + \exp\left[-\frac{(y + y_s)^2}{4E_y t}\right]\right\}. \tag{B-17}$$

The case for a more general release may again be generated from the convolution integral, Eq. (B-1).

Mixed-tank Models

The mixed-tank transport model shown in Fig. B-13 is based on the simplification of Eq. (B-2) to yield an unsteady mass balance for interconnected, perfectly mixed lakes:

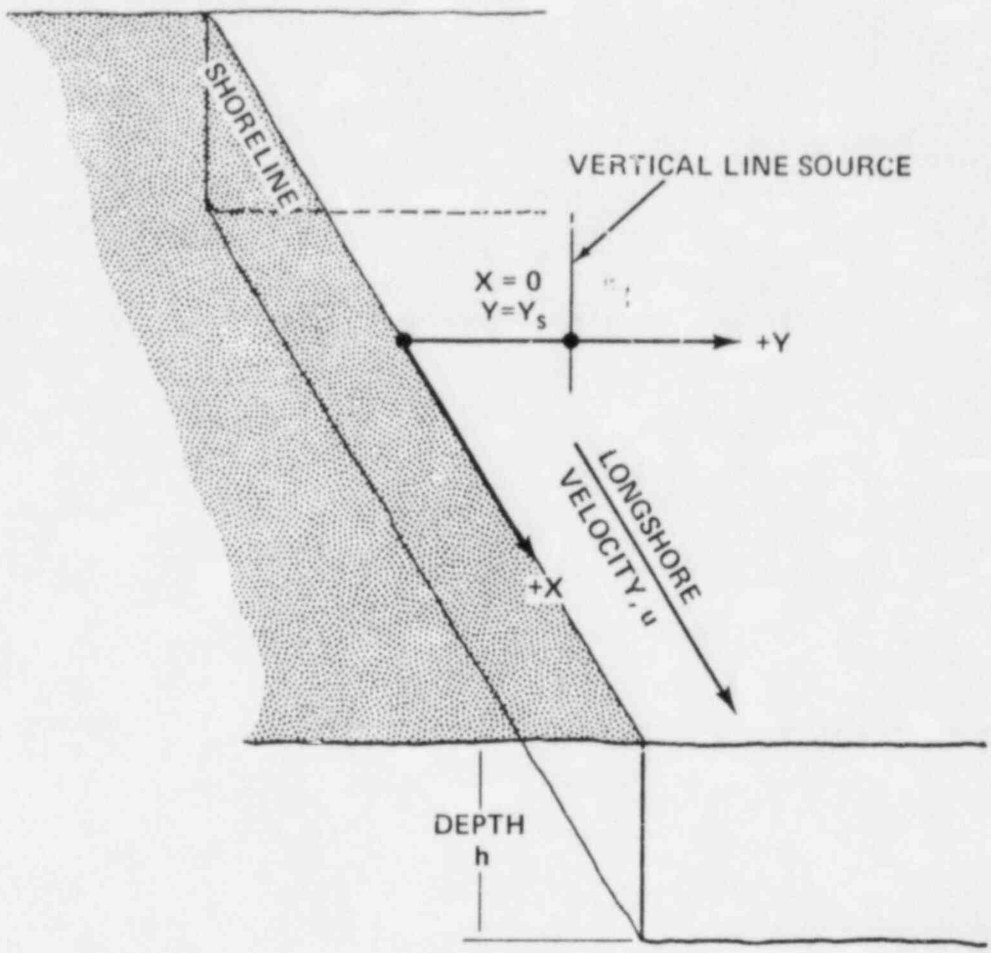

Fig. B-12. Near-field lake model.

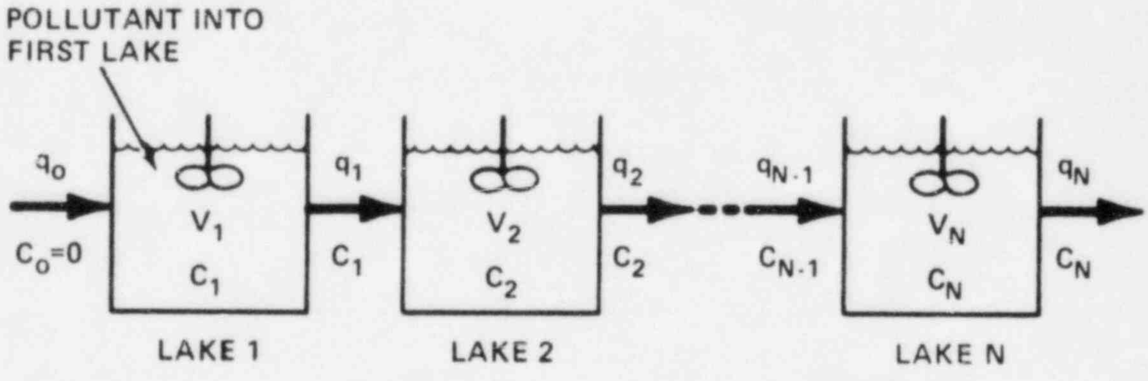

Fig. B-13. Perfectly mixed cascaded lake model.

$$\frac{dC_n}{dt} = \frac{q_{n-1}C_{n-1}}{V_n} - \left(\frac{q_n}{V_n} + \lambda\right) C_n \ , \qquad \text{(B-18)}$$

where

C_n = the concentration in the n^{th} lake;
q_n = the flow rate leaving the n^{th} lake and entering the $(n + 1)^{th}$ lake;
V_n = the volume of the n^{th} lake;

and the other terms are as previously defined.

The pollutant is assumed to enter the first lake in the series, for which the mass balance is:

$$\frac{dC_1}{dt} = \frac{f(t)}{V_1} - \left(\frac{q_1}{V_1} + \lambda\right) C_1 \ , \qquad \text{(B-19)}$$

where f(t) is the rate of release of the pollutant.

For an instantaneous release of M curies, the concentration in the L^{th} lake in the series is:

$$C_L = \frac{M \prod_{i=1}^{L-1} q_i}{\prod_{i=1}^{L} V_i} \left[\sum_{j=1}^{L} \frac{(-R_j)\exp(-R_j t)}{\prod_{\substack{k=1 \\ k \neq j}}^{L+1} (R_k - R_j)} \right] \ , \qquad \text{(B-20)}$$

where

$$R_j = \frac{q_j}{V_j} + \lambda \text{ and}$$

$$R_{L+1} = 0.$$

For a more general release, the convolution integral, Eq. (B-1), is used.

The long-term consequences of radionuclide releases to the large lakes may be overestimated if the effects of sedimentation are not taken into account. Observations of the concentration of ^{137}Cs in the Great Lakes from weapons testing fallout indicate that it vanishes from the water column at a rate much greater than predicted from radioactive decay and flushing alone.

Barry derived a model of Lake Michigan using an effective half life, empirically derived from observations of lake concentrations and atmospheric fallout, in order to calculate concentrations of normally released ^{137}Cs from power plants.[8] Booth derived a fourcompartment model for radionuclide exchange between free water, exchangeable sediment, irreversible sediment, and interstitial water.[9]

Booth's model is appealing because it defines the mechanisms of sediment interactions, appears to work well, and is conceptually simple. While the model is not very complicated, it involves the numerical solution of a stiff system of four simultaneous linear differential equations, thus representing a considerable complication over the other radionuclide transport models used here. In order to keep a closed-form solution, the model was simplified to two compartments, sediment and water. The lake is shown graphically in Fig. B-14, and the model shown schematically in Fig. B-15.

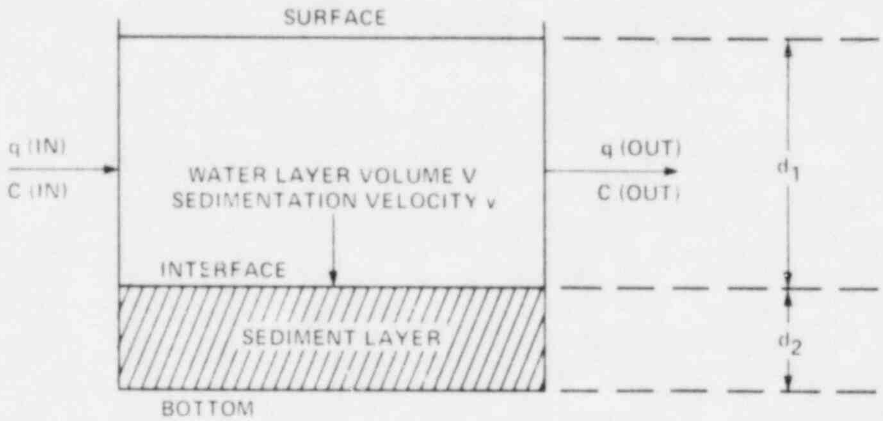

Fig. B-14. Lake or reservoir model with sediment.

The model is identical to that presented above for a single lake, except there is an additional source and sink term because of the effects of sediment. The model is described here in considerable detail, and is relevant to the discussion on the core-melt river transport model.

Dissolved material is flushed out of the lake at a rate:

$$R_F = \frac{q}{V} C , \tag{B-21}$$

where

q = the fresh-water flow rate (m^3/yr);
V = the lake volume (m^3); and
C = the concentration.

Falling sediment deposits on the lake bottom at a rate v, m/yr. It is assumed that this rate is uniform and that each sediment particle is in chemical equilibrium with the water through which it is falling. The rate at which sediment scavanges the water column is:

$$R_S = \frac{v\ K_d}{d_1} C \ , \qquad (B-22)$$

where K_d is equilibrium distribution coefficient, and d_1 is the depth of the water column.

Direct exchange from the water to the sediment layer occurs by a process similar to molecular diffusion. It is described for the purposes of this simplified model as being proportional to the concentration of the water:

$$R_D = \frac{K_f}{d_1} C \ , \qquad (B-23)$$

where K_f is the coefficient of direct transfer (m/yr).

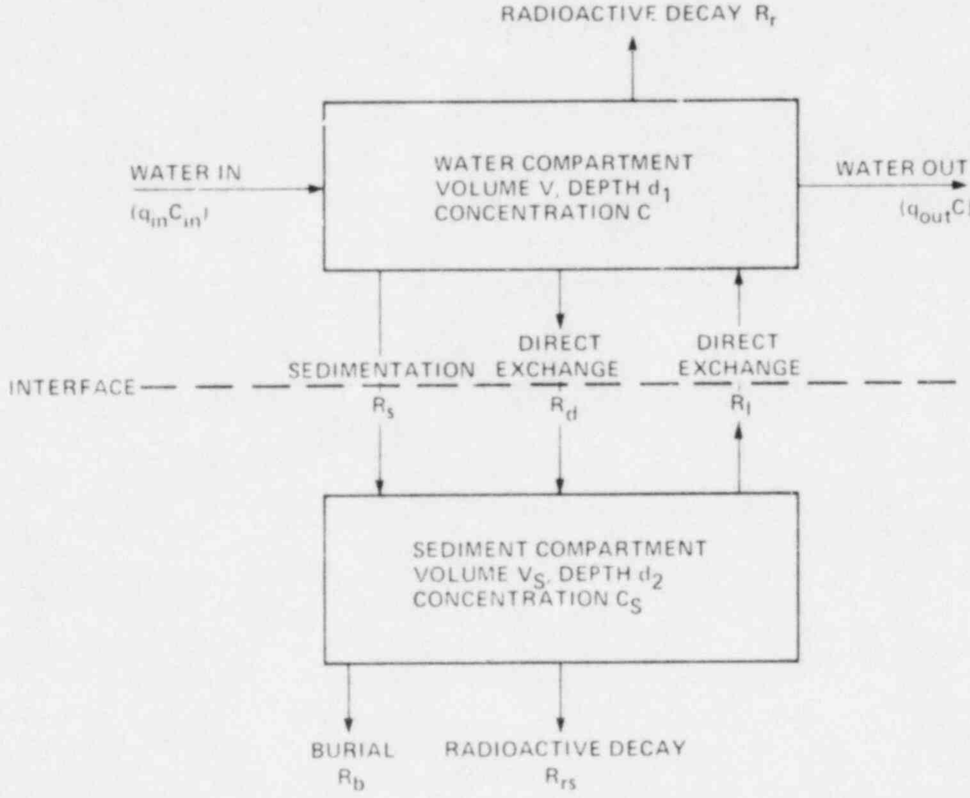

Fig. 15. Two compartment lake of reservoir model.

Direct exchange also occurs from the sediment layer to the water layer. It is assumed to be proportional to the concentration of water immediately surrounding the sediment particles, which is in turn assumed to be in equilibrium with the sediment itself:

$$R_L = \frac{K_f}{K_d \, d_1} C_s \ , \tag{B-24}$$

where C_s is the concentration in the bottom layer.

A fraction of adsorbed material deposited in the bottom layer will be available to release radionuclides back into the water column through the process of leaching. The sediment closest to the surface would be most effectively leached, and the effectiveness would decrease with increasing depth. For the purpose of this model, however, it was assumed that the sediment layer is of finite thickness, and that it is all effectively in contact with the water column. The sediment layer will continually be buried by fresh sediment. This burial and radioactive decay limit the effective layer thickness to a depth on the order of several centimeters as has been observed in all surface-water bodies.[1,3-5,10] The model assumes that the sediment layer thickness, d_2, remains constant. For this to be the case, a portion of the bottom layer was assumed to be removed by burial:

$$R_B = \frac{v}{d_2} C_s \ , \tag{B-25}$$

Radioactive decay occurs in both the surface and bottom layers:

$$R_R = \lambda C, \tag{B-26}$$

$$R_{RS} = \lambda C_s \ , \tag{B-27}$$

where λ = ln 2/half life.

Combining all terms leads to the following differential equations:

$$\frac{dC}{dt} = \frac{W(t)}{V} + C_s\lambda_1 - C\lambda_2 \ , \tag{B-28}$$

$$\frac{dC_s}{dt} = C\lambda_3 - C_s\lambda_4 \ , \tag{B-29}$$

where

$W(t)$ = the input rate of radioactive material, Ci/yr,

$$\lambda_1 = \frac{K_d}{d_1 K_d} ,$$

$$\lambda_2 = \frac{q}{V} + \lambda + \frac{v\ K_d}{d_1} + \frac{K_f}{d_1} ,$$

$$\lambda_3 = \frac{v\ K_d}{d_2} + \frac{K_f}{d_2} , \text{ and}$$

$$\lambda_4 = \lambda + \frac{v}{d_2} + \frac{K_d}{d_2\ K_d} .$$

For an instantaneous release of 1.0 Ci, the water-phase concentration can be solved from Eq. (B-28 and (B-29):

$$C_i = \frac{1}{V(S_1-S_2)} \left[(\lambda_4 + S_1)e^{S_1 t} - (\lambda_4 + S_2)e^{S_2 t}\right] , \qquad \text{(B-29a)}$$

where

$$S_{1,2} = \frac{-(\lambda_2 + \lambda_4) \pm \sqrt{(\lambda_2 + \lambda_4)^2 - 4(\lambda_2\lambda_4 - \lambda_3\lambda_1)}}{2} .$$

This instantaneous solution may be generalized using the convolution integral, Eq. (B-1).

B.3. GROUNDWATER MODELS

The following equation describes the three-dimensional dispersion of a radionuclide through a porous medium with constant dispersion coefficients, porosity, and ionic equilibrium distribution coefficient:

$$\frac{\partial C}{\partial t} + \frac{u}{a}\frac{\partial C}{\partial x} + \frac{v}{a}\frac{\partial C}{\partial y} + \frac{w}{a}\frac{\partial C}{\partial z} = \frac{D_x}{a}\frac{\partial^2 C}{\partial x^2} + \frac{D_y}{a}\frac{\partial^2 C}{\partial y^2} + \frac{D_z}{a}\frac{\partial^2 C}{\partial z^2} - \lambda C , \qquad \text{(B-30)}$$

where u, v, and w are the components of groundwater velocity in the x, y, and z directions, respectively; D_x, D_y, and D_z are the dispersion coefficients in the x, y, and z directions, respectively; and a is the "retention factor" resulting from ionic adsorption, defined as:

$$a = 1 + \frac{\rho b}{n} K_d \ , \tag{B-31}$$

where n is the total porosity; ρb is the bulk density (solids and voids) of the porous media; and K_d is the equilbrium distribution coefficient between the solid and liquid phases for the particular ionic species invloved.

B.3.1. Point Concentration Model

The first groundwater model is used for calculating the concentration at any point in the aquifer depicted in Fig. B-16, relative to the source location for any time during or after the release of radioactive material. This model was necessary to estimate concentrations in wells which are downgradient from the source.

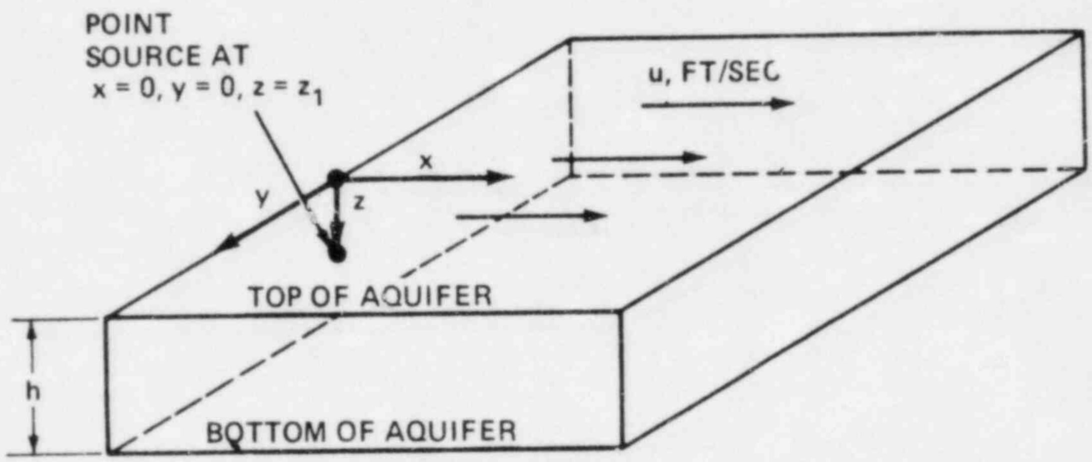

Fig. B-16 Idealized groundwater system for point concentration model, point source.

Equation (B-30) is solved for the aquifer represented in Fig. B-16 under the following limitations and assumptions:

1. constant thickness of aquifer, h;

2. constant, uniform velocity, u, in x direction only;

3. constant dispersion coefficients D_x, D_y, D_z;

4. constant porosity, n, and effective porosity, n_e;

5. point source discharge at $x = 0$, $y = 0$, $z = z_s$; and

6. equilibrium between solid and liquid phases.

The dispersion coefficients are estimated from the groundwater velocity by:

$$D_x = \alpha_x u \ , \tag{B-32}$$

$$D_y = \alpha_y u \ , \tag{B-33}$$

$$D_z = \alpha_z u \ . \tag{B-34}$$

where α_x, α_y, and α_z are the dispersivities (dispersion constants) in the indicated directions in the aquifer.[11]

Equation (B-5) is solved in terms of normalized influence or Green's functions:[12]

$$C_i = \frac{1}{n_e a} X\ (x,t)\ Y(y,t)\ Z(z,t) \ , \tag{B-35}$$

where C_i is the concentration at any point in space for an instantaneous one-curie release; X, Y, and Z are the Green's functions in the x, y, and z coordinate directions, respectively; and n_e is the effective porosity. Equation (B-35) has been developed for a variety of boundary and source configurations.[13]

It can be proven that for the aquifer used in this study, the pollutant would be vertically mixed over its entire thickness h within several thousand feet downgradient from the source. The vertically averaged form of Equation (B-35) would be:[13]

$$C_i = \frac{1}{n_e a} X_1\ Y_1\ Z_2 \ , \tag{B-36}$$

where

$$X_1 = \frac{1}{\sqrt{\frac{4\pi D_x t}{a}}} \exp\left[-\frac{\left(x - \frac{ut}{a}\right)^2}{\frac{4D_y t}{a}} - \lambda t\right] ,$$

$$Y_1 = \frac{1}{\sqrt{\frac{4\pi D_y t}{a}}} \exp\left[-\frac{y^2}{\frac{4D_y t}{a}}\right] , \text{ and}$$

$$Z_2 = \frac{1}{h} .$$

The case of the more general release is handled using the convolution integral, Eq. (B-1).

This model can be used to evaluate exposure at a well location. For this situation, the only operating pathway is the drinking water pathway.

B.3.2. Surface Water Interface Model

This model calculates the source term contribution to a surface water body from an arbitrary groundwater spill as depicted in Fig. B-17. It is assumed that all material entering the groundwater will eventually enter the surface water, except for the quantity lost through radioactive decay. All assumptions which pertain to the point concentration model described in the previous section also apply to this model.

In the unidirectional flow field assumed, the flux F (Ci/sec) of material crossing an area dA = dydz perpendicular to the x axis is described by the equation:

$$\frac{dF}{dA} = \left(uC + D_x \frac{\partial C}{\partial x}\right) n_e , \qquad \text{(B-37)}$$

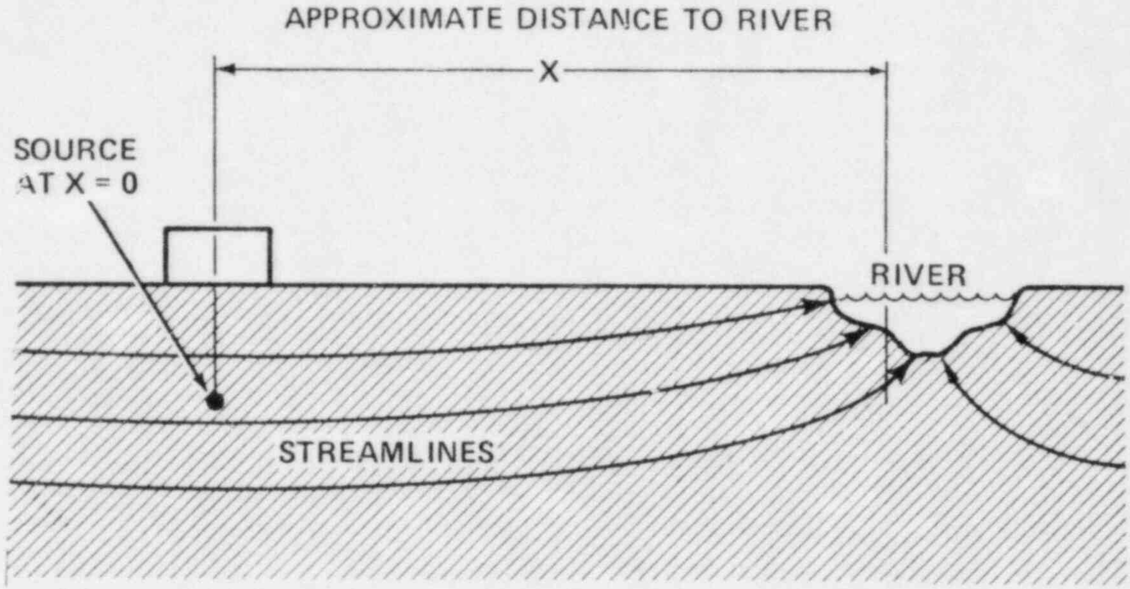

Fig. B.17. Groundwater — surface water interface, flux model.

where C is the concentration in the dissolved phase. The total flux across the plane would be:

$$F = n_e \int_{-\infty}^{\infty} \int_{-\infty}^{\infty} \left(uC + D_x \frac{\partial C}{\partial x}\right) dy\ dz \ , \tag{B-38}$$

If C_i is the concentration from an instantaneous release of one curie at $x = 0$ and time $t = 0$, as described by Eq. (B-36), then the resulting flux at distance x downgradient would be:

$$F_i = \frac{\left(x + \frac{u}{a}\ t\right)}{\sqrt{\frac{D_x}{a}\pi t^3}} \exp\left[-\frac{\left(x - \frac{u}{a}\ t\right)^2}{4\frac{D_x t}{a}} - \lambda t\right] \ , \tag{B-39}$$

Equation (B-39) may be generalized for an artibrary release using the convolution integral, Eq. (B-1).

REFERENCES FOR APPENDIX B

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APPENDIX C

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APPENDIX C: PATHWAY DOSE MODELS

C.1. INTRODUCTION

Pathway and dosimetry models for estimating radiation doses resulting from accidental releases to the aquatic environment largely follow from models developed for evaluation of chronic effluent releases.[1] The major additional consideration in the accident evaluation is the time variations of pathway concentrations arising from the time characteristics of the release. Thus in some sense, accident dose models are the general solutions of the pathway transport equations, and those for chronic release are the particular solutions.

The dose* to members of the population depends on the integrated pathway exposure. In the evaluation of chronic releases, equilibrium conditions in the environment are often assumed with the period of exposure taken to be a year. The dose, which is then, in fact, a dose rate (mrem/yr), can be compared to annualized radiation protection guidance, e.g., existing regulations. In this study, the dose is evaluated over exposure periods that range from a few days to in excess of several years. Over these exposure periods, the pathway radionuclide concentrations are time dependent, because equilibrium or steady-state conditions may not be obtained. The total exposure must be computed as the time integral of the pathway concentration functions.

C.1.1. List of Symbols

Symbols used frequently in this appendix are listed here. Equation or section numbers indicate where the symbol first appears or where additional clarification may be found. The units of hour, day, meter, liter, kilogram, and picocurie are abbreviated as hr, d, m, L, kg, and pCi, respectively.

Indexes

p	index denoting a particular exposure pathway, Eq. (C-1)
i	index denoting a particular radionuclide, Eq. (C-1),
j	index denoting particular hydrological subregion of the water body, Section C.3, Eq. (C-5)

* The term "dose" as used here implies committed dose equivalent, i.e., the total dose equivalent delivered over a 50-year period following the intake of a radionuclide. For radionuclides of short residence time in the body, or in the case of external exposure, no further irradiation will be experienced beyond that associated with the exposure period.

Variables

DI_p — individual radiation dose (rem) arising from exposure to the p^{th} pathway medium, Section C.2.1,

U_p — individual's usage of the pth pathway medium, units of (hr/d) for external exposure pathways and (kg/d or L/d) for ingestion pathways, Section C.2.2,

DF_{ip} — dose factor for the ith nuclide in the p^{th} exposure pathway, units of (rem-L/pCi-hr) for the water immersion pathway, (rem-m^2/pCi-hr), for the sediment exposure pathway, and (rem/pCi) for the ingestion pathways, Section C.2.3,

$C_{ip}(u)$ — the i^{th} radionuclide concentration function in the p^{th} exposure pathway medium (the independent variable u denotes time), units of (pCi/m^2) for the sediment exposure pathway, (pCi/kg or pCi/L) for the ingestion pathways, and (pCi/L) for the water immersion (swimming) pathway, Eq. (C-1),

$C_{if}(t)$ — the i^{th} radionculide concentration function in fish flesh (the independent variable t denotes time) units of (pCi/kg), Eq. (C-4),

$C_{iw}(t)$ — the i^{th} radionculide waterborne concentration function, units of (pCi/L), Section C.2.1.1,

$C_{id}(t)$ — the i^{th} radionuclide drinking water concentration function, units of (pCi/L) Section C.2.1.2,

$C_{is}(t)$ — the i^{th} radionuclide drinking water concentration function for shoreline sediment, units of (pCi/m^2), Eq. (C-4),

FT_i — fraction of the i^{th} radionculide activity passing through the drinking water treatment facility, Section C.2.1.2,

B_i — the bioaccumulation factor for the i^{th} radionuclide stable element analog, units of (ppm in fish flesh/ppm in water), Eq. (C-3),

λ_{bi} — the i^{th} radionuclide biological elimination rate constant in fish, units of (d^{-1}), Eq. (C-3),

λ_{ei} — the i^{th} radionuclide effective elimination rate constant in fish ($\lambda_{ei} = \lambda_{bi} + \lambda_i$), where λ_i is the radiological decay constant, units of (d^{-1}), Eq. (C-3),

F_{ip} — modifying factor accounting for decay of the i^{th} radionuclide as it is transported through the p^{th} exposure pathway and; for external exposure pathways geometry considerations, no units, Eq. (C-1), Section C.2.2,

P_{pj} — the population usage of the p^{th} exposure pathway medium in the j^{th} subregion of the water body, units of (man) for the drinking water pathway, (kg/d) fish flesh productivity for the fish ingestion pathway, (man-hr/d) for the shoreline and swimming exposure pathways, Eq. (C-5),

T — the fish life span, in units of (d), Eq. (C-3),

K — the transfer coefficient between water and sediment in units of (L/m^2-d), Eq. (C-4),

K_p — any conversion constant associated with the p^{th} exposure pathway, Eq. (C-5),

$C_{ipj}(u)$ — the i^{th} radionuclide concentration function in the p^{th} exposure pathway medium derived from the j^{th} hydrological subregion of the water body (the independent variable u denotes time), units of (pCi/m^2) for the sediment exposure pathway, (pCi/kg or pCi/L) for the ingestion pathways, and (pCi/L) for the swimming pathway, Eq. (C-5),

$C_{iwj}(u)$ — the i^{th} radionuclide waterborne concentration function for the j^{th} hydrological subregion, (pCi/L), Section C.3.2,

$C_{ifj}(u)$ — the concentration function for the i^{th} radionuclide in the flesh of fish harvested from the j^{th} hydrological subregion, (pCi/kg), Section C.3.3,

$C_{isj}(u)$ — the concentration function for the i^{th} radionuclide in shoreline sediments of the j^{th} hydrological subregion, (pCi/m^2), Section C.3.4,

P_{wj} — the population that obtains drinking water from the j^{th} subregion of the water body, Section C.3.2,

P_{fj} — the fish harvest assigned to the j^{th} subregion, edible weight (kg/d), Section C.3.3,

P_{sj} the population shoreline usage factor for the j^{th} subregion, (man-hr/d), Section C.3.4,

P_{rj} the population swimming usage factor for the j^{th} subregion, (man-hr/d), Section C.3.5, and

E_i the decay energy of the i^{th} radionuclide deposited within the aquatic biota, units of MeV/disintegration, Section C.4.3.

C.2. INDIVIDUAL DOSE MODELS

C.2.1. General Expression

The mathematical expression for the p^{th} exposure pathway contribution to an individual's dose (DI_p) can be generalized as:

$$DI_p = U_p \sum_{i=1}^{m} DF_{ip} \cdot F_{ip} \int_0^t C_{ip}(u)\, du \;. \qquad \text{(C-1)}$$

The concentration function, $C_{ip}(u)$, of radionuclide i in pathway p must be evaluated for each radionuclide released to the environment and for each pathway of concern. Note that in the above formulation, the usage factor, U_p, is taken to be constant throughout the evaluation. This may be a reasonable assumption for the minor releases; however, for major releases interdiction measures would alter this parameter. As the functional form of $C_{ip}(u)$ depends on a number of considerations, e.g., time characteristics of the release, hydrological dispersion parameters, etc., Eq. (C-1) must be evaluated using numerical integration methods. The following sections summarize the basic pathway concentration models.

C.2.1.1. Dispersing Waterborne Concentrations, $C_{iw}(t)$

The source term data (radionuclides and quantities of release, as well as release time character) and the radionuclide transport models of Appendix B were used to obtain the dispersing waterborne concentration functions, $C_{iw}(t)$, in the various water bodies. The dispersing waterborne concentration functions, in addition to determining the dose for pathways where water is the pathway medium (i.e., swimming), are the driving functions in the models used to predict the drinking water, aquatic foods, and sediment concentration functions.

C.2.1.2. Drinking Water Concentrations, $C_{id}(t)$

The concentration function of a radionuclide in the drinking water, $C_{id}(t)$, can be taken as the dispersing waterborne concentration, $C_{iw}(t)$.

The potential removal of radionuclides in the water treatment plant can be considered if one has information on the water treatment system. The fraction of the i^{th} radionuclide passing through these plants FT_i, has been presented by Soldat[2] (see Table C-1). As can be seen from the data, the reduction in the treated water concentration ranges from a factor of 5 to unity. The drinking water concentration function, $C_{id}(t)$ is given as:

$$C_{id}(t) = FT_i \; C_{iw}(t) \tag{C-2}$$

Table C-1. Removal of elements by drinking water treatment systems

Element	FT_i	Element	FT_i
H	1	Te	0.8
Sr	0.2	I	0.8
Y	0.2	Cs	0.9
Nb	0.7	Ba	0.4
Ru	0.5	La	0.2
Rh	0.5	Ce	0.2
Ag	0.7	Pr	0.2
Cd	0.6	Np	0.7
Sn	0.7	Pu	0.7
Sb	0.8		

C.2.1.3. Aquatic Food Concentrations, $C_{if}(t)$

The aquatic food concentration function of a radionculide was evaluated using the waterborne concentration function and the bioaccumulation factor as follows:

$$C_{if}(t) = \frac{B_i \cdot \lambda_{bi}}{1-\exp(-\lambda_{bi}T)} \exp(-\lambda_{ei}t) \int_0^t C_{iw}(u) \exp(\lambda_{ei} \cdot u)du. \tag{C-3}$$

A detailed discussion of the above model is presented in Section C.4.

C.2.1.4. Sediment Concentrations, $C_{is}(t)$

Shoreline sediment concentrations are estimated only for purposes of evaluating direct exposure. Models of radionuclide removal from the water column through sedimentation and leaching back from the sediments to the water column were considered in the aquatic transport. The model presented here applies only to shoreline considerations.

Shoreline sediment concentrations functions for each radionuclide are evaluated using the waterborne concentration functions as follows:

$$C_{is}(t) = K \exp(-\lambda_i t) \int_0^t C_{iw}(u) \exp(\lambda_i \cdot u) du \quad . \qquad \text{(C-4)}$$

The above equation is the integral equation for which the particular solution, assuming a time-independent water concentration, is given as Eq. (A-4) in Regulatory Guide 1.109.[1] Experimental evaluation on the Columbia River[3-4] indicate that K is approximately independent of radionuclide species with a value of about 70 L/m^2-day (200gal/ft^2-day). A K-value of 100 was used in the ocean-based offshore evaluation.[5] Either value and Eq. (C-3) should yield a conservative estimate of the pathway dose to man.

C.2.2. Pathway Usage Factors, U_p

The liquid pathway usage factors employed in the computer code for estimation of individual doses were taken from Regulatory Guide 1.109 and are listed in Table C-2 for completeness. The values in each case are for the maximum exposed individual (highest pathway usage) in each pathway, regardless of the age categories of Regulatory Guide 1.109.[1] In addition, Table C-2 presents the pathway distribution time, T_p, used to consider decay during distribution of radionuclides in the pathway medium. The modifying factor, F_{ip}, for the ingestion pathway is computed as $\exp(-\lambda_i \cdot T_p)$. For evaluation of commercial fish consumption, the correction factor is the summation of the factors for each type of processing, weighted by the fraction of the commercial harvest processed in that manner.

Table C-2. Pathway usage factors and distribution time

Pathway	Up	Tp
Drinking water	2 L/day (0.47 gal/day)	1 day
Fish ingestion	0.058 kg/day (0.13 lb/day)	
a recreational		1 day
b commercial*		8% fresh-3 days 34% frozen-25 days 58% canned-70 days
Shoreline use	0.18 hr/day	Not applicable
Swimming	0.068 hr/day	Not applicable

*Based on data of Appendix D.

The dose conversion factors developed for evaluation of the shoreline sediment exposure pathway are based on an infinite planer source geometry. The gamma exposure was evaluated at a height 1 m (3.3 ft) above the source plane. Factors, referred to as shore width factors,[1] have been developed to take into account a more realistic exposure geometry relative to the infinite plane. In evaluating the shoreline exposure pathway, the modifying factor, F_{ip}, of the generalized Eq. (C-10) was assigned the following values:

(a) River shoreline - 0.2;
(b) Lake shoreline - 0.3;
(c) Ocean shoreline - 0.5; and
(d) Estuary tidal zone - 1.0.

C.2.3. Dose Factors, DF_{ip}

The dose factors employed in the code are those of Regulatory Guide 1.109.[1] In the analysis, only the adult dose factors are used. The doses to individuals of other ages and adults whose diet and recreational preferences are different from those of Table C-2 would yield somewhat different dose estimates. Given the occurrence of the postulated accidents, other considerations (e.g., site-specific parameters, the course of the accident, the effectiveness of corrective and protective actions), would also strongly affect the actual dose experienced. Age-specific doses are not addressed in the computer code.

The dose factors are not presented here, but can be found in Regulatory Guide 1.109. Note that in the above reference, the factors have units of mrem, while in the formulation of Eq. (C-1), units of rem are indicated.

C.3. POPULATION DOSE MODELS

For the purpose of calculating the population dose (man-rem), the receiving water body was divided into a number of subregions over which pathway concentrations were averaged. Population usage data and other necessary parameters must be assigned to the subregions indicated by the available site-specific data.

C.3.1. General Expression

The mathematical expression for the p^{th} exposure pathway contribution to the population dose can be generalized as:

$$D_p = K_p \sum_{j=1}^{n} P_{pj} \sum_{i=1}^{m} DF_{ip} \cdot F_{ip} \int_0^t C_{ipj}(u)du \; . \qquad \text{(C-5)}$$

Equation (C-5) must be evaluated using numerical integration methods. The following sections outline the transformation of Eq. (C-5) for evaluation of the various exposure pathways.

C.3.2. Drinking Water Pathway

For the drinking water pathway, the receiving water body is divided into a number of subregions depending on the water body and the complexity of the hydrologic dispersion model. Drinking water populations are assigned to the subregions as indicated by the site data.

For the drinking water pathway, the factors of the generalized Eq. (D-4) are transformed as follows:

$$C_{ipj}(u) \longrightarrow C_{idj}(u)$$

$$P_{pj} \longrightarrow P_{wj}$$

$$K_p \longrightarrow K_w$$

$$F_{ip} \longrightarrow FT_i \cdot \exp(-\lambda_i \cdot T_p) .$$

C.3.3. Aquatic Food Ingestion Pathway

For the aquatic food ingestion pathway, commercial and recreational harvest values are assigned to the various subregions of the environment. The generalized Eq. (C-5) is transformed as follows for this pathway:

$$C_{ipj}(u) \longrightarrow C_{ifj}(u)$$

$$P_{pj} \longrightarrow P_{fj}$$

$$K_p \longrightarrow 1 \quad \exp(-\lambda_i \cdot T_p) \text{ - recreational}$$

$$F_{ip} \longrightarrow \Sigma w_j \exp(-\lambda_j T_{pj}) \text{ - commercial}$$

The basis for the modifying factor, F_{ip}, is given in Section C.2.2.

C.3.4. Shoreline Pathway

For the shoreline pathway, population usage data are assigned to the various subregions of the receiving water body. The generalized Eq. (C-5) is transformed in the following manner:

$$C_{ipj}(u) \longrightarrow C_{isj}(u)$$

$P_{pj} \longrightarrow P_{sj}$

$K_p \longrightarrow 1.$

$F_{ip} \longrightarrow$ see section C.2.2.

C.3.5. Swimming Pathway

For the swimming exposure pathway, population usage data are assigned the various subregions of the receiving water body. For this pathway the generalized Eq. (C-5) is transformed as follows:

$C_{ipj}(u) \longrightarrow C_{iwj}(u)$

$P_{pj} \longrightarrow P_{rj}$

$K_p \longrightarrow 1$

$F_{ip} \longrightarrow 1.$

C.4. BIOTA DOSE MODELS

Presented in this section are models employed for estimation of the dose to aquatic biota. The biota expsoure pathways considered are: (a) immersion in contaminated water, (b) exposure to contaminated sediments, and (c) radiation from internal deposited radionuclides. The calculational model employed follows the suggested procedure of Le Clare et al.[6]

C.4.1. Dose to Biota via Immersion

The immersion exposure component of the biota dose can be estimated in a similar manner to the swimming pathway for man. The factors of the generalized Eq. (C-1) are transformed as follows:

$C_{ip}(u) \longrightarrow C_{iw}(u)$

$F_{ip} \longrightarrow 1$

$U_p \longrightarrow$ 24 hours/day.

C.4.2. Dose to Biota via Contaminated Sediment

The component of the biota dose associated with exposure to contaminated sediments can be estimated in a similar manner to the shoreline pathway for man. The factors of the generalized Eq. (C-1) are transformed as follows:

$$C_{ip}(u) \longrightarrow C_{is}(u)$$

$$U_p \longrightarrow 24 \text{ hr/day}$$

$$F_{ip} \longrightarrow 2 \ .$$

Note that the modifying factor F_{ip}, which for this pathway represents exposure geometry considerations, has been assigned a value greater than unity. A value of two has been suggested[6] to take into account the increased gamma exposure at the surface relative to the 1-m height considered in the development of the dose conversion factors.

C.4.3. Dose to Biota via Internal Deposited Radionuclides

Estimates of the radionuclide concentrations in biota developed for assessment of dose to man can be utilized in evaluation of the internal component of the biota dose. The factors of the generalized Eq. (C-1) can be transformed as follows:

$$C_{ip}(u) \longrightarrow C_{it}(u)$$

$$F_{ip} \longrightarrow 1$$

$$DF_{ip} \longrightarrow \overline{E_i}$$

$$U_p \longrightarrow 5.1 \times 10^{-8}$$

The numerical constant 5.1×10^{-8} has units of rads-kg-disintegration per pCi-MeV-day. The numerical value 5.1×10^{-8} equals 2.2 disintegrations/minute/pCi times 1.6×10^{-6} ergs/MeV times 1×10^{-5} kg rad/erg times 1400 minutes/day. The $\overline{E_i}$ factor represents the i^{th} radionuclide decay energy, which is deposited considering the size of the organism. These data were obtained from Le Clare et al.[6]

C.5. AQUATIC FOOD CONCENTRATION MODE

C.5.1. The Model

In assessing the radionuclide concentration in aquatic foods arising from a chronic operational release to the aquatic environment, the concentration is taken to be directly proportional to the radionuclide concentration in water. The constant of proportionality is referred to as the bioaccumulation factor. This approach is appropriate if the organisms have been in a reasonably constant concentration field for a period of sufficient duration for trophic and biological exchange processes to approach equilibrium. Under such conditions, the bioaccumulation factor for a radionuclide species may be taken to be the ratio of the stable element analog concentration in fish flesh, C_f^*, to that of the water, C_w^*, i.e., $B = C_f^*/C_w^*$.

Determination of radioactivity concentrations in aquatic foods arising from acute accidental releases must consider, in some detail, the net results of biological uptake and elimination processes. The details of such processes are complex, involving feeding habits, osmoregulation, and metabolic considerations. These processes are, in turn, dependent on external environmental factors, such as water temperature, salinity, and the time of the year.

To avoid the nearly impossible task of assessing the details of all the biological, environmental, and ecological processes governing the uptake of radioactivity in aquatic organisms, a simple one-compartment metabolic model has been developed. The formulation of the model parallels the development by Peterson[7] and that recently cited by Vanderploeg.[8]

For a one-compartment organism, the differential equation describing the time rate of change of the radionuclide concentration in the organism, $C_f(t)$, subject to a time varying radionuclide concentration in its diet, $C_d(t)$, can be written as:

$$\frac{dC_f(t)}{dt} = I\ f\ C_d(t) - \lambda_e\ C_f\ , \qquad \text{(C-6)}$$

where f is the fractional absorption of the radionuclide across the organism's gut.

The solution to the above equation is given by:

$$C_f(t) = I\ f\ \exp(-\lambda_e t) \int_0^t C_d(u)\ \exp\ (\lambda_e u) du. \qquad \text{(C-7)}$$

The discussion leading to Eq. (C-7) can be applied to the stable element analog. In this case, the effective elimination constant, λ_e, would become the biological elimination constant of the element, λ_b. Assuming a time-independent stable element concentration C_d^* in the diet and integrating over the life span T, Eq. (C-7) becomes:

$$C_f^* = \frac{IfC_d^*}{\lambda_b} [1-\exp(-\lambda_b T)]\ , \qquad \text{(C-8)}$$

where C_f^* is the stable element concentration in the organism.

The bracketed term represents the approach of the organism's concentration to equilibrium with respect to the diet concentration. That is, for elements of long biological halftimes, the organism may not reach equilibrium within its lifetime. The flesh concentration of such elements continue to increase through the organism's lifetime.

Relating the diet concentration of the stable element analog to the waterborne concentration as, $C_d^* = KC_w^*$, (implied in the definition of the stable element bioaccumulation factor), and using the definition of the bioaccumulation factor, Eq. (C-8) can be solved for If as:

$$If = \frac{\lambda_b}{1-\exp(-\lambda_e t)} \cdot \frac{B}{K} \tag{C-9}$$

Using this information, Eq. (C-7) can be expressed in terms of the bioaccumulation factor as:

$$C_f(t) = \frac{B \cdot \lambda_b \cdot \exp(-\lambda_e t)}{1-\exp(-\lambda_b T)} \int_0^t C_w(u) \exp\,(\lambda_e u)\, du\ . \tag{C-10}$$

Note that the factor K of Eq. (C-9) does not appear in Eq. (C10), as the time-dependent radionuclide and stable element analog diet and water concentration are assumed to be related by the same constant. This assumption, in addition, implies that the concentrations in the lower trophic levels, which make up the diet of the organism, are closely coupled, in the temporal sense, to the waterborne concentrations.

Equation (C-10) was used to calculate the fish flesh concentration functions in this study. Values for λ_b were obtained from experimental data in the literature and supplemented by calculations based on stable element data. Before discussing these aspects (Section C.4.2), it may be instructive to examine Eq. (C-10) regarding its relationship to the more familiar formulation used to evaluate chronic effluent releases. For long-term chronic releases where the waterborne concentrations are time independent, i.e., $C_w(u) = C_w$, integration of Eq. (C-10) over the organism's lifetime, T, yields:

$$C_f = \frac{B \cdot \lambda_b}{1-\exp(-\lambda_b T)} \cdot \frac{1-\exp(-\lambda_e T)}{\lambda_e} \cdot C_w\ . \tag{C-11}$$

Case A

Consider a nuclide with a slow elimination rate (elimination halftime, $T_{1/2} = \ln(2)/\lambda_e$, greater than the organism's life span, e.g., $T_{1/2} > 10$ years). A Taylor series expansion of the exponentials yields:

$$C_f = B \cdot C_w \ . \qquad \text{(C-12)}$$

This is the definition of the bioaccumulation factor.

Case B

Consider a radionuclide with an effective elimination rate, λ_e, that is governed by the biological elimination rate, λ_b, i.e., $\lambda_e \sim \lambda_b$. Equation (C-11) then reduces to the bioaccumulation factor definition of Eq. (C-11).

Case C

Consider the case of a radionuclide for which the radiological half-life and biological elimination halftime correspond to halftime periods much less than T. Both terms containing exponentials approach unity, and Eq. (C-11) reduces to:

$$C_f = \frac{\lambda_b}{\lambda_e} \cdot B \cdot C_w \ . \qquad \text{(C-13)}$$

This is the expression suggested by Peterson[7] and Vanderploeg[8] to adjust the stable element bioaccumulation factor for the radionuclide half-life.

C.5.2. Model Data Base

To implement the model, values must be established for the biological elimination constants. Two approaches were used to establish these constants. The staff calculated these values using Eq. (C-7) and the stable element concentration data of Thompson et al.[9] The applicant's approach was to conduct a literature search and establish, where possible, the constants from uptake and retention studies.[5]

To solve Eq. (C-7) for λ_b (transcendental with respect to λ_b), two parameters must be estimated, namely, I and f. The following discussion outlines the approach taken for fish flesh.

The fish ingestion rate per unit mass, I, was taken to be 0.02 g/d/g-fish based on the food consumption rates of rainbow trout.[10]

The values for the fractional absorption of the elements across the gut, f, were taken to be the values established for man.[11] In addition, the fish were assumed to be piscivorous, with an age, T, of 3 years.

Table C-3 presents a listing of the bioaccumulation factors and elimination constants (expressed as halftimes) used in the fresh-water and salt-water environments considered in this study.

Table C-3. Selected stable element bioaccumulation factors* and biological halftimes for fresh-water and salt-water piscivorous fishes

Element	Salt water	Fresh water	Tb (days)
H	0.9	0.9	35
Sr	2	5	120
Y	25	25	7.8 (3)
Nb	3.0 (4)**	3.0 (4)	7.8 (3)
Ru	3	10	1.1 (3)
Rh	10	10	170
Ag	3.3 (3)	2.3	2.5 (3)
Cd	3.0 (3)	200	5.1 (3)
Sn	3.0 (3)	3.0 (3)	660
Sb	40	1.0	1.1 (3)
Te	10	1.0 (3)	140
I	10	15	35
Cs	40	400	35
Ba	10	4	660
La	25	25	7.8 (3)
Ce	10	25	7.8 (3)
Pr	25	25	7.8 (3)
Np	10	10	7.8 (3)
Pu	3	3.5	7.9 (3)

*Source of bioaccumulation factors: S. E. Thompson et al., Concentration Factors of Chemical Elements in Edible Aquatic Organism, UCRL-50564, Rev. 1, Lawrence Livermore Laboratory, 1972.

**Notation: 3.0 (4) should be read as 3.0×10^4.

For purposes of evaluating the invertebrate pathway, the above model was extended to aquatic invertebrate. The biological elimination constants for aquatic invertebrates, however, are largely unknown (only a few such values are available from the literature). As an estimator for this parameter, the biological elimination constants developed for fish have been employed. As, in general, the biological half-life of elements decreases with organism size,[12] the estimation procedure appears to be reasonable and conservative. Table C-4 provides a listing of the bioaccumulation factors used in the analysis of the invertebrate pathway.

Table C.4. Selected stable element bioaccumulation factors for fresh-water and salt-water invertebrates

Element	Bioaccumulation factor	
	Salt water	Fresh water
H	0.9	0.9
Sr	20	100
Y	1000	1000
Nb	100	100
Ru	1000	300
Rh	2000	300
Ag	3300	770
Cd	2.5 (5)	2000
Sn	1000	1000
Sb	5.0	10
Te	100	6100
I	50	5
Cs	25	1000
Ba	100	200
La	1000	1000
Ce	600	1000
Pr	1000	1000
Np	10	400
Pu	200	100

C.5.3. Discussion

The model presented above, essentially the model of Peterson,[7] provides a method for evaluation of the aquatic food pathway in a time varying concentration field. The model equation was shown to reduce to the general bioaccumulation factor approach used in evaluation of chronic releases to the liquid pathway. Intervening steps in the food chain between the organism and the water are considered in magnitude, as these processes are reflected in the bioaccumulation factor. In the temporal sense, these steps could lead to the organism's diet concentration lagging the time concentrations of the water. However, in general, the elimination coefficients of the prey (or forage) are generally much larger than the elimination coefficient of their respective predators. Thus, the concentrations in the lower trophic levels are expected to follow the time variation of the waterborne concentrations.

Radionuclides may enter the aquatic food web not only from the water concentration, but also from the bottom sediments. The extent of contamination of the bottom sediments and the availability of these nuclides for entrance into the food chain varies with sediment type and nuclide. Further understanding of the radionuclide-sediment transport processes and the extent of the sediment's influence on the organism's food web are needed to model this possible transfer.

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APPENDIX D

excerpted from Appendix D: "Fish Harvest Data," in NUREG-0440
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APPENDIX D: FISH HARVEST DATA

D.1. INTRODUCTION

Aquatic organisms used for food may provide a potentially important pathway to man for radioactivity resulting from an accidental release to the liquid pathway at a nuclear power plant. The importance of this pathway in evaluating radiation dose to the population has been assessed in this report on the basis of both commercially and recreationally harvested fish and shellfish. The possible intermediate pathway provided by industrial fish that are reduced to meal and subsequently fed to poultry and other domestic livestock used for food, has not been analyzed because of the low probability that it will result in significant doses to people. Similarly, the approximately 1.1 billion kg (2.4 billion pounds) of edible fishery products imported during 1973 (Table D.1) are not included in the radiological analyses presented in this report. Imported fishery products would be expected to be harvested from waters too distant to be affected by an accident at a U.S. power plant.

Table D.1. U.S. comsumption of aquatic foods (in millions of kg)

	U.S. commercial harvest-1971	U.S. recreational harvest-1970	Imports 1973
Finfish			
Marine	640	730	950
Fresh water	44	350	No estimate
Shellfish			
Marine	410	200 (assumed)	140
Fresh water	2.3	No estimate	No estimate
Fishmeal	270	Not applicable	60

Note: All weights are in round weights (live weights), except for imports and mollusks which are presented in terms of edible meat.

The purpose of this appendix is to provide a realistic basis for estimating the actual magnitude of the fish-to-man pathway. Because of the nature of this study, in which typical sites and habitats are compared on a generic basis, an attempt has been made to quantify the aquatic food harvest in terms of kg/ha/yr (lb/acre/yr). Information presented in these terms is readily combined with that from hydrologic determinations of effluent dispersion to quantify the significance of various aquatic food pathways.

This method of presentation also resolves the technical problem of translating fishery statistics that are reported in terms of harvest by geographic region to a specific area related to thr location of a certain plant. The data are developed here under the assumption that finfish and shellfish are distributed uniformly throughout their habitat during their time of occupancy. This assumption is necessary to normalize the patchy distribution of fish in space and time that results from schooling, feeding, migration, reproductive activity, microhabitat preferences, and physico-chemical changes in the environement. Most commercially exploited shellfish are more restricted in their mobility than finfish; some are even permanently attached throughout their life beyond larval stages. These organisms are also arranged in patches, but the distribution of these patches is random within their habitat; hence, we may also use the uniform distribution assumption in the shellfish case.

D.2. DATA SOURCES

The data shown in Tables D.2 through D.6 have been compiled chiefly from statistical reports of the National Marine Fisheries Service (NMFS), the Fish and Wildlife Service (FWS), and from personal communications with representatives of the Sports Fishing Institute (SFI). Other Federal and international publications were also consulted. Not all the data presented are equally reliable because only commercial marine harvests are carefully and fully recorded on a national basis by a Federal agency. Fresh-water commercial harvests are recorded only for the Great Lakes and the Mississippi River drainage areas. It is probable that limited commercial harvest of finfish also occurs in geographic areas not covered by available publications. The only commercial fresh-water invertebrates for which Federal statistics are readily available apply to crayfish from the Atchafalaya River. The recreational finfish harvest statistics are developed to a great extent on the basis of creel censuses, fisherman interviews, expert estimates, and extrapolation of observations on specific water bodies to other similar habitats (Tables D.7 through D.10). The data describing numbers of anglers and angler days were developed by the FWS based on the 1970 Bureau of Census survey. No data were found to quantify recreational fresh-water and salt-water harvests of invertebrates, even though large harvests of crayfish, shrimp, crabs, clams, and others are known to occur. However, limited information may exist on a state or local level. It is not unreasonable, based on topical reports, to assume that the recreational harvest of crustacea in estuarine and fresh waters equals that of the commercial fisherman.

Recreational finfishing is documented periodically by the FWS National Survey of Fishing and Hunting, a comprehensive study that is oriented towards examining the recreational and socioeconomic aspects of angling and hunting. The survey has been performed every five years since 1955. The most recent available report covers the 1970 survey. The SFI is also oriented towards recreational interests, but delves deeply into the biological aspects of sport fishing management and may be considered a source of authoritative data. The NMFS compiles statistics of recreational salt-water fishing according to species and the area caught. The

Table D.2. Commercial harvest from estuarine and marine fishing waters - 1974

	Atlantic	Gulf	Pacific (1973)
Area marine fishing waters 0-4.8 km (ha × 10^6)	5.9	6.9	1.9
Area estuarine water (ha × 10^6)	3.4	2.6	0.3
% estuarine	57%	37%	17%
Finfish			
Weight, 0-4.8 km (kg × 10^6)	71-85	24-33	76-79
Density, 0-4.8 km (kg/ha/yr)	12-14	3.4-4.7	41-42
Weight, exclusively estuarine (kg × 10^6)	10	2.2	7.3
Weight, estuarine-marine (kg × 10^6)	27-30	8.2-11	55
Density, estuarine (kg/ha/yr)	10-12	4.0-5.3	170[5]
Crustacea			
Weight, 0-4.8 km (kg × 10^6)	75	43	10
Density, 0-4.8 km (kg/ha/yr)	13	6.2	5.5
Weight, exclusively estuarine (kg × 10^6)	52	21	
Weight, estuarine-marine (kg × 10^6)	7.7	8.2	1.8
Density, estuarine (kg/ha/yr)	18	11	5.4
Mollusks			
Weight, 0-4.8 km (kg × 10^6)	39	6.8	8.2
Density, 0-4.8 km (kg/ha/yr)	6.6	1.0	4.4
Weight, exclusively estuarine (kg × 10^6)	26	6.8	3.6
Weight, estuar[illegible]e-marine (kg × 10^6)	11		0.9
Density, estuarine (kg/ha/yr)	11	2.6	14

Notes:

1. Values based on contiguous states only.

2. Values for finfish and crustacea are in kg of roung weight; values for mollusks are in kg of edible meat.

3. Values are calculated by summing total weight of species harvested exclusively from estuarine waters with the proportional weights of the other species which occur throughout the 0 to 4.8 km (3 mi) offshore zone. The porportionality factor is the ratio of estuarine waters (as identified by Cain in testimony before the U.S. House of Representatives) to the total area of marine fishing waters, 0-4.8 km (3 mi) (as identified in the National Survey of Needs for Hatchery Fish). Marine fishing waters, 0-4.8 km (3 mi) includes estuarine waters also.

4. Estuarine waters are differentiated from marine fishing waters because this study compares on offshore site with an estuarine site.

5. This value would be 17 kg/ha (14.9 lb/acre) if the harvests from Puget Sound and the Columbia River were excluded.

6. Sources: "Current Fisheries Statistics," 1974, NOAA; "Fisheries of the U.S.," 1974, NOAA; "National Survey of Needs of Hatchery Fish," FWS, USDI, 1968; and Cain, Stanley A., in "Estuarine Areas," U.S. House of Representatives, Serial No. 90-3, 1967.

Table D.3. Commercial harvest from estuarine and marine fishing waters — Atlantic Coast regions

	New England	Mid Atlantic	South Atlantic
Area marine fishing waters, 4.8 km (ha × 10^6)	0.89	2.1	3.0
Area estuarine waters (ha × 10^6)	0.16	1.9	1.3
% estuarine	18%	94%	43%
Finfish			
Weight, 0-4.8 km (kg × 10^6)	37-43	18-20	15-22
Density, 0-4.8 km (kg/ha/yr)	42-48	8.7-9.6	5.3-7.5
Weight, exclusively estuarine (kg × 10^6)	-	-	-
Weight, estuarine-marine (kg × 10^6)	6.8-7.7	14-15	3.2-6.4
Density, estuarine (kg/ha/yr)	42-48	8.8-9.5	7.7-11
Crustacea			
Weight, 0-4.8 km (kg × 10^6)	12	34	29
Density, 0-4.8 km (kg/ha/yr)	13	16	9.6
Weight, exclusively estuarine (kg × 10^6)	1.4	33	17
Weight, estuarine-marine (kg × 10^6)	1.8	0.9	5.0
Density, estuarine (kg/ha/yr)	20	17	17
Mollusks			
Weight, 0-4.8 km (kg × 10^6)	6.6	31	1.4
Density, 0-4.8 km (kg/ha/yr)	7.4	15	0.45
Weight, exclusively estuarine (kg × 10^6)	5.0	20	1.4
Weight, estuarine-marine (kg × 10^6)	0.45	10	-
Density, estuarine (kg/ha/yr)	34	16	1.0

Sources: See Note 6, Table D.2.

Table D.4. Commercial harvest from estuarine and marine fishing waters (0-4.8 km offshore) — Pacific Coast regions

	Puget Sound	Columbia River	Marine fishing river
Finfish (kg × 10^6)	43	6.8	27
Crustacea (kg × 10^6)	0.59	0.73	9.1
Mollusks (kg × 10^6)	1.5	-	6.7

Table D.5. Dominant aquatic food species from estuarine and marine waters (0-4.8 km)

Species	NE	MA	SA	Gulf	Pacific	Habitat	Feeding habit
Finfish							
Flounders	17%	10%			9%	Benthic	Benthic invertebrates
Sea herring	64%					Pelagic	Plankton
Striped bass		19%				Pelagic	Fish
Sea trout		17%	19%	13%		Pelagic-deep	Strongly bottom feeding
Bluefish		13%				Pelagic-deep	Fish
Scup		11%				Pelagic-bottom	Strongly bottom feeding
Catfish			40%			Benthic	Strict bottom feeding
Spot			23%			Pelagic-bottom	Strongly bottom feeding
Mullet				51%		Shallow water	Bottom feeding-plankton
Bonito					10%	Pelagic	Fish
Rockfishes					8%	Benthic	Bottom feeding
Salmon					45%	Pelagic	Fish
% Regional finfish catch	81%	70%	82%	64%	72%		
Crustacea							
Lobster	72%					Benthic	Benthos & Detritus
Shrimp	16%		37%	49%	70%	Benthic	Plankton-filter feeder
Crabs	11%	98%	61%	49%		Benthic	Benthos-Detritus
% Regional crustacea harvest	99%	98%	98%	98%	70%		
Mollusks							
Soft clams	51%					Benthic	Plankton-filter feeder
Hard clams	14%					Benthic	Plankton-filter feeder
Surf clams		32%				Benthic	Plankton-filter feeder
Oysters		41%	71%	99%	37%	Benthic	Plankton-filter feeder
Squid	17%				53%	Pelagic	Midwater fish
% Regional mollusk harvest	82%	73%	71%	99%	90%		

Source: Fishery Statistics of the United States, 1973; and Fisheries of the United States, 1973, 1974.

Table D.6. Commercial harvest from fresh water — 1971

	Finfish	Shellfish
Great Lakes drainage (kg × 10^6)	15	No estimate
Mississippi River & tributaries	29	2.3
Pond aquaculture (kg/ha)	1100–3300	No estimate

Table D.7. Recreational harvest from estuarine and marine fishing waters — 1970

	Atlantic	Pacific	Gulf	Chesapeake Bay	Total
Finfish (kg/ha/yr)	91	110	76	combined with Atlantic	
Crustacea (kg/ha/yr) estimate	11	16	8.8	29	
Mollusk (kg/ha/yr) edible meat	Not known — no basis for estimation				
Total weight finfish (kg × 10^6)	420	77	220	combined with Atlantic	730
Area (ha × 10^6)	4.5	0.73	2.9		8.1
Number of anglers (×10^5)	5	2.2	2.3		9.5
Harvest/angler (kg/yr)	82	36	96		
Number of angler days (×10^6)	61	17	36		130
Harvest/angler day (kg)	7	5	6		
Days fished/angler	12	7.8	16		

Notes: Values include only those for the contiguous United States.
Source: 1970 National Survey of Fishing and Hunting, Fisheries of the United States, 1974.

Table D.8. Recreational harvest from fresh water

Water body type	Finfish	Shellfish
Lakes, reservoirs and manmade ponds (kg/ha/yr)	17	*
Farm ponds (kg/ha/yr)	45	*
Streams and rivers (kg/ha/yr)	5	*
Lake Erie (kg/ha/yr)	6	*
Other Great Lakes (kg/ha/yr)	1	*
Total harvest in 1973 (kg × 10^6)	350	*
Number of anglers, 1970 (×10^6)	29	*
Harvest/angler (kg)	12	*
Number of angler days, 1970 (×10^6)	520	*
Harvest, angler day (kg)	0.6	*
Days fished/angler	20	*

*No basis for estimate.

Notes:

1. All weight values are estimated by the Sport Fishing Institute. These statistics are not authoritatively compiled by the United States government on a national basis.

2. Numbers of anglers include only those who spent $7.50 or more, or who reported three or more angler days during 1970 Bureau of Census survey.

3. Information is derived from various years.

Table D.9. Recreational fresh-water angling by water body type and geographical region (in thousands of anglers)

	Reservoirs	Man-made ponds	Natural lakes & ponds	Rivers & streams	Farm ponds
New England	130	40	570	410	410
Middle Atlantic	710	290	780	1200	630
East North Central	1200	760	3100	1600	1300
West North Central	810	550	1200	970	980
South Atlantic	1100	760	640	1500	1600
East South Central	890	630	190	670	1200
West South Central	1700	610	430	880	1300
Mountain	820	50	280	600	230
Pacific	950	200	820	1400	470
Total	8300	3900	8000	9200	7800

Note:
Anglers who fished in more than one water body type and/or region are represented in more than one category.

Source: 1970 Survey of Hunting and Fishing.

Table D.10. Recreational salt-water fishing by water body and geographic region (numbers and percent of anglers)

Water body type	Atlantic		Pacific		Gulf		Total U.S.	
	10^3	%	10^3	%	10^3	%	10^3	%
Surf	1800		580		580		2 960	
Bays and sounds	2900	70	840	54	1400	66	5 140	65
Tidal rivers and streams	1400		480		550		2 430	
Ocean	2700	30	1600	46	1300	34	5 600	35
Totals	8800		3500		3830		16 130	

Note:
1. Totals do not agree with those in Table D-7 because some anglers fished in several water body types and/or regions and are represented in more than one category.

Source: 1970 Survey of Hunting and Fishing.

NMFS does not include mollusks, crustacea, and other invertebrates, though they state, "In some coastal areas, recreational marine fisherman harvest significant quantities of these animals." (USDC 1975, p. 25)

Foreign and U.S. fish harvests from North Atlantic marine waters are recorded by the International Commission for the Northwest Atlantic Fishery (ICNAF) (Tables D.11 through D.13). The areas of interest are shown in Fig. D.1. The U.S. and Canadian Great Lakes harvest data are found in the "Fishery Statistics of the U.S." (see Table D.14).

D.3. DATA ANALYSIS

Data from various sources have been anlayzed to present information in terms of weight of edible aquatic food per surface acre per year according to water body type or by coast. The marine data have been evaluated to differentiate estuarine waters, as defined by Stanley Cain of the Department of Interior (Cain 1967), from marine fishing waters within the 0 to 4.8 km (0–3 mi) range for which statistics are collected. This differentiation is made because this study identifies estuarine and coastal sites as an alternative to the offshore site. The harvest from estuarine waters is determined simplistically by multiplying the ratio of the estuarine water area to the total marine fishing water area within 4.8 km (3 mi) by the total weight of edible fish harvested from the latter area. This procedure is necessary because the available fishery statistics present commercial marine source areas as 0–3 miles, 3–12 miles, 12–200 miles, and international waters. This procedure possibly may yield values that are low because the total catch of some species may be entirely from estuarine waters. In cases where species are known to be thus misrepresented (e.g., crabs from Chesapeake Bay) the correcting factor has not been applied.

Weights of all finfish and crustacea are given in round or live weights. According to the Department of Agriculture (Watt and Merrill 1963) the edible portions of the most commonly eaten finfish range from 31% (cod) to 65% (salmon). Crustacea range from 12% (crayfish) to 49% (shrimp). A weighted average based on the most abundantly harvested commercial fish indicates that 53% of the round (as caught) weight of marine finfish and 26% of the crustacea is edible, and that about 28% of the fresh-water finfish is edible by humans (see Table D.15). Weight of mollusks is given in terms of edible meats, and the values should not be corrected for assumed waste.

Recreational harvest of marine food organisms is taken to be entirely from estuarine and marine fishing waters within 4.8 km of shore under the conservative assumption that truly offshore fishing days are a relatively small percentage of total number of marine fishing days.

A survey performed in 1973 for the National Oceanic and Atmospheric Administration estimated the number of commercial and privately owned

Table D-11 Commercial United States and Foreign harvest of aquatic food from Georges Bank region - 14×10^6 ha (in metric tons, live weight)

	U.S. harvest	Foreign harvest	Total harvest	hg/ha	%U.S.
Finfish	54,720	347,993	402,713	29.4	13.6
Crustacea					
Shrimp	0	0	0	0	0
Crabs	180	0	180	0.01	100
Lobsters	1,166	178	1,344	0.1	86.8
Mollusks					
Oysters	0	0	0	0	0
Clams, mussels	0	0	0	0	0
Scallops	7,707	48,934	56,641	4.1	13.6
Squid	27	22,295	22,322	1.6	1.2

Note: Corresponds to ICNAF region 5Ze.

Table D-12 Commercial United States and Foreign harvest of aquatic food from mid-Atlantic region - 28×10^6 ha (in metric tons, live weight)

	U.S. harvest	Foreign harvest	Total harvest	hg/ha	%U.S.
Finfish	38,401	193,787	232,188	8.4	16.5
Crustacea					
Shrimp	0	0	0	0	0
Crabs	32,567	0	32,567	1.2	100
Lobsters	1,281	0	1,281	0.05	100
Mollusks					
Oysters	179,831	0	179,831	6.5	100
Clams, mussels	270,898	0	270,898	9.8	100
Scallops	15,662	0	15,662	0.6	100
Squid	1,288	22,035	23,323	0.8	5.5

Note: Corresponds to ICNAF areas 6a, 6b, and 6c.

Table D-13 Commercial United States and foreign harvest of aquatic food from New England region - 9.4×10^6 ha (in metric tons, live weight)

	U.S. harvest	Foreign harvest	Total harvest	hg/ha	%U.S.
Finfish	90,258	140,944	231,202	24.6	39
Crustacea					
Shrimp	7,964	0	7,964	0.8	100
Crabs	870	0	870	0.1	100
Lobsters	8,705	0	8,705	0.9	100
Mollusks					
Oysters	0	0	0	0	0
Clams, mussels	19,708	0	19,708	2.1	100
Scallops	1,894	0	1,894	0.2	100
Squid	1,090	4,462	5,552	0.6	20

Note: This area corresponds to ICNAF regions 5y and 5Zw.

Fig. D.1. Southwestern portion of the International Commission for the Northwest Atlantic Fishery area.

Table D.14. Commercial harvest from the Great Lakes — 1973

Lake	U.S. harvest			Canadian harvest		
	Area (ha × 10^3)	Weight (kg × 10^6)	Density (kg/ha)	Area (ha × 10^3)	Weight (kg × 10^6)	Density (kg/ha)
Erie	1,300	3.8	2.9	1,200	18	14
Huron	2,400	0.86	0.33	3,600	1.1	0.34
Michigan	5,800	6.5	1.1	--	--	--
Ontario	900	0.13	0.11	1,000	1.1	1.1
Superior	5,400	2.5	0.45	2,800	1.5	0.45

Sources: Fishery Statistics of the U.S., 1973; the 1969 World Almanac.

Table D.15. Harvest and edible portions of selected species

Species	Habitat	Harvest within 0–4.8 km (kg × 10^6)	% Edible	Weight of edible meat (kg × 10^6)
Commercially important:				
Salmon	marine	26	65%	17
Herring	marine	23	51%	11
Mullet	marine	12	53%	6.4
Flounders	marine	11	33%	3.8
Alewives	marine	8.2	49%	4.0
Buffalofish	freshwater	10	32%	3.1
Carp	freshwater	8.2	30%	2.5
Catfish & bullheads	freshwater	6.4	19%	1.2
Crabs	marine	86	14–25%	14
Shrimp	marine	45	28–49%	20
American lobster	marine	11	26%	2.9
Other commonly harvested species:				
Striped bass	marine	---	43%	---
Sea bass	marine	---	39%	---
Bluefish	marine	---	51%	---
Porgy (scup)	marine	---	41%	---
Snapper	marine	---	52%	---
Bonito	marine	---	58%	---
Sea trout (weakfish)	marine	---	48%	---
Ocean perch	marine	---	31%	---
Bass	freshwater	---	31%	---
Lake trout	freshwater	---	37%	---
Pickerel	freshwater	---	51%	---
Walleye pike	freshwater	---	57%	---
Smelt	freshwater	---	55%	---
Trout	freshwater	---	59%	---
Yellow perch	freshwater	---	39%	---

Source: U.S. Dept. of Agriculture Handbook No. 8, Composition of Foods, 1963.

recreational boats in the United States and determined their use for salt-water fishing by size class and type of water fished. Their findings are summarized in Tables D.16 and D.17. Of the privately owned recreational boats, about 44% of the total salt-water fishing days were in the ocean and 56% were in rivers, bays, and sounds. Approximately 4% of the total number of salt-water fishing days were in boats larger than 8 m (26 ft) long and 26% in boats between 5 and 8 m (16 and 26 ft) long. Many boats at the smaller end of this range would not be expected to fish frequently more than a few kilometers offshore because of safety and convenience reasons; hence, it is reasonable to assume that no more than 30% of the total salt-water fishing days are beyond 4.8 km from shore. It is probable, based on size characteristics and the factors of weather, safety, time, and expense involved with truly offshore fishing, that the actual number of salt-water fishing days from privately owned boats more than 4.8 km offshore is about 5 to 10% of the total salt-water fishing days.

Commercially owned recreational boats are more apt to fish offshore than privately owned boats. Table D.17 shows that 61% of the fishing days from these boats were the ocean. We cannot make any assumptions about where these vessels fished because the problems of size and safety limitations could not reasonably be applied on a generic basis. However, the relative number of participants apears to be low when compared with privately owned boats. Note that commercially owned boats spent about 544,000 fishing days, while the privately owned boats spent 21,400,000 fishing days. Even allowing for the larger numbers of anglers per commercially owned boat trip, the privately owned number of fishing days is overwhelmingly important.

D.4. FORECAST OF TRENDS

From 1945 through 1971, the total landings of commercially caught finfish and shellfish for human consumption ranged from a high of $1,500 \times 10^6$ kg (round weight) to a low of $1,000 \times 10^6$ kg. The range from 1965 through 1974 was $1,200 \times 10^6$ to $1,000 \times 10^6$ kg. There has been a variation from year to year in the relative amounts of each species harvested, but the overall total reflects a generally consistent level. The per capita fish consumption rate during these years has also remained relatively constant. Unless there is a drastic change in the eating habits of Americans, it should be realistic to use values recorded for the recent years to analyze the significance of this portion of the fish-to-man pathway. It is entirely possible, however, that the significance of this pathway may be increased as a result of utilizing currently unexploited stocks of fish for food and by increased use of fish as feed for meat-producing animals, especially chickens which may be fed a diet containing as much as 10% fish meal. Another possible source of increase in consumption is the development and possible adoption of fish meal concentrate as a larger part of the human diet. Technology currently exists for the large-scale manufacture and use of fish protein concentrate, but current Federal regulatory

Table D.16. Salt-water boat fishing days (in thousands) — privately owned vessels

Vessel length (m)	Atlantic		Gulf		Pacific		Total	
	Days	%	Days	%	Days	%	Number	%
Ocean								
<5	1,329	6	1,183	6	518	2	3,030	14
5–8	2,469	12	1,531	7	1,524	7	5,524	26
>8	522	2	125	1	136	1	783	4
Total	4,319	20	2,839	13	2,178	10	9,338	44
Rivers, bays, sounds								
<5	2,341	11	2,549	12	667	3	5,557	26
5–8	2,553	12	1,935	9	1,152	5	5,641	26
>8	579	3	170	1	111	1	860	4
Total	5,474	26	4,654	22	1,930	9	12,059	56

Source: Kenneth M. Bromberg, Determination of the Number of Commercial and Non-Commercial Recreational Boats in the U.S., Their Use and Selected Characteristics, 1973, prepared for NOAA; available from NTIS as report COM-74-11186.

Table D.17. Salt-water recreational fishing days (in thousands) — commercially owned vessels

Vessel length (m)	Atlantic	Gulf	Pacific	Total	
	Days	Days	Days	Days	%
Ocean					
<12	30	10	20		
12–20	112	41	77		
>20	22	9	10		
Total	163	59	108	330	61
Rivers, bays, sounds					
<12	25	7	17		
12–20	73	28	51		
>20	7	3	3		
Total	107	37	71	214	39

Notes:

1. Approximately 32% of all commercial recreational fishing vessels carried 6 or fewer anglers.

2. Source: See Table D.16.

policy denies its use in the United States. It is conceivable that this policy could change and that a major increase in direct fish ingestion could occur.

It is probable that the commercial and recreational harvest of marine invertebrates will increase in the future because of the abatement of estuarine water pollution, improved management procedures, and increasingly successful aquaculture ventures. Unfortunately, the amount of this expected increase cannot be quantified at this time.

The "National Survey of Needs for Hatchery Fish," (USDI 1968) forecasts recreational fishing activity for the period 1965 to 2000 for both fresh water and salt water (see Table D.18). According to this source, man-days of recreational fresh-water fishing will increase from about 390×10^6 in 1965 to about 840×10^6 man-days in 2000. Recreational salt-water fishing will increase from about 100×10^6 in 1965 to about 270×10^6 man-days in 2000. If we assume a consistent harvest per unit effort of fishing, the recreational fresh-water harvest in the year 2000 will about 2.0 times the 1970 harvest, and the recreational salt-water harvest will be about 2.3 times the 1971 harvest.

The "1970 National Survey of Fishing and Hunting," (USDI 1971) presents historical data for the years 1955, 1960, 1965, and 1970. These are summarized in Table D.19. These data tend to corroborate the predictions made above for salt-water fishing, but indicte that the fresh-water forecasts were too low by almost 40% in 1970, and probably will be as much or more in the future.

The recreational harvests of fresh- and salt-water invertebrates for human consumption are not recorded on a national basis and hence cannot be forecast. However, it will probably increase proportionately with the commercial harvest because of the increasing interest in outdoor recreation and because of increased areas of unpolluted estuarine waters.

REFERENCES FOR APPENDIX D

[1] Kenneth M. Bromberg, Determination of the Number of Commercial and Non-Commerical Recreational Boats in the U.S., Their Use, and Selected Characteristics, prepared for NOAA, NTIS Report No. COM-74-11186, 1973.

[2] Stanley A. Cain, Estuarine Areas, U.S. House of Representatives, Serial No. 90-3, p. 30, 1967.

[3] International Commission for the Northwest Atlantic Fishery, "Statistical Bulletin," Vol. 24 for the year 1974, Dartmouth, Canada.

[4] Sport Fishing Institute, personal communication with Robert Martin, 1976.

Table D.18. Forecasted number of anglers and recreation days of angling (in thousands)

	1965	1973	1980	2000
Number of anglers	42,000	53,000	63,000	93,000
Fresh water	29,000	36,000	43,000	64,000
Salt water	13,000	17,000	20,000	29,000
Recreation days of angling	490,000	620,000	740,000	1,100,000
Fresh water	390,000	490,000	570,000	840,000
Salt water	100,000	130,000	170,000	270,000

Source: USDI, "National Survey of Needs of Hatchery Fish," FWS, 1968.

Table D.19. Recorded number of anglers and recreation days of angling (in thousands)

	1955	1960	1965	1970
Number of anglers	21,000	25,000	28,000	33,000
Fresh water	18,000	22,000	24,000	29,000
Salt water	4,600	6,300	8,300	9,500
Recreation days of angling	400,000	470,000	520,000	710,000
Fresh water	340,000	390,000	430,000	590,000
Salt water	59,000	81,000	96,000	110,000

Source: "1970 National Survey of Fishing and Hunting," FWS, USDI.

Table D.20. Pathways through aquatic foods to humans

Recreational harvest	% of harvest
Fresh (consumed within 3 days of harvest)	100% (assumed)

Commercial harvest

1) Fresh — (consumed within 3 days of harvest) = 8%
2) Frozen — (consumed 3 days to 6 months of harvest = 34%
3) Canned & processed — (consumed 1 week to 2 years of harvest = 58%
4) Fish meal to chicken to human (total time in pathway is probably 3 to 12 months from harvest to human consumption). Approximate harvest from estuarine-coastal waters is 270×10^6 kg.

   a) The conversion rate of feed to body weight of broiler chickens is about 3:1.

   b) Fish meal may constitute up to 10% of total food ration.

Source: Fishery Statistics of the United States, 1971.

Table D.21. Other water contact recreation by people of age nine years and over — 1970

	Participants ($\times 10^6$)	Recreation days ($\times 10^6$)
Boating, sailing, canoeing	41	420
Swimming	77	1,700
Waterfowl hunting	3	25

Source: 1970 Survey of Hunting and Fishing.

[5] U.S. Department of Commerce, "Current Fisheries Statistics," Annual Summaries by States for 1974, published 1976.

[6] U.S. Department of Commerce, "Fisheries of the United States, 1974," 1975, p. 25.

[7] U.S. Department of Commerce, "Fishery Statistics of the United States, 1971," Statistical Digest No. 65, 1974.

[8] U.S. Department of Interior, "National Survey of Fishing and Hunting," 1970.

[9] U.S. Department of Interior, "National Survey of Needs for Hatchery Fish," 1968.

[10] R. K. Watt and A. L. Merrill, Composition of Foods, U.S. Department of Agriculture Handbook No. 8, 1963.

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The LPGS computer program was developed to calculate the radiological impacts resulting from radioactive releases to the hydrosphere. The hydrosphere is represented by the following types of water bodies: estuary, small river, well, lake, and one-dimensional (1-D) river. The program is principally designed to calculate radiation dose (individual and population) to body organs as a function of time for the various exposure pathways. The radiological consequences to the aquatic biota is estimated. Several simplified radionuclide transport models are employed with built-in formulations to describe the release rate of the radionuclides. Optionally, a tabulated user-supplied release model can be input. Printer plots of dose versus time for the various exposure pathways are provided.

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