PNNL-16209

Pacific Northwest
National Laboratory
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U.S. Department of Energy

A Demonstration of the System Assessment Capability (SAC) Rev. 1 Software for the Hanford Remediation Assessment Project

P. W. Eslinger
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November 6, 2006

Prepared for the U.S. Department of Energy
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PACIFIC NORTHWEST NATIONAL LABORATORY
operated by
BATTELLE
for the
UNITED STATES DEPARTMENT OF ENERGY
under Contract DE-AC05-76RL01830

Printed in the United States of America

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Summary

The System Assessment Capability (SAC) is a suite of interrelated computer codes that provides the capability to conduct large-scale environmental assessments on the Hanford Site. Developed by Pacific Northwest National Laboratory for the Department of Energy, SAC models the fate and transport of radioactive and chemical contaminants, starting with the inventory of those contaminants in waste sites, simulating transport through the environment, and continuing on through impacts to the environment and humans. Separate modules in the SAC address inventory, release from waste forms, water flow and mass transport in the vadose zone, water flow and mass transport in the groundwater, water flow and mass transport in the Columbia River, air transport, and human and ecological impacts. The SAC supports deterministic analyses as well as stochastic analyses using a Monte Carlo approach, enabling SAC users to examine the effect of uncertainties in a number of key parameters.

The initial assessment performed with the SAC software (Bryce et al. 2002) identified a number of areas where both the software and the analysis approach could be improved. Since that time the following six major software upgrades have been made:

- An air pathway model was added to support all-pathway analyses.
- Models for releases from glass waste forms, buried graphite reactor cores, and buried naval reactor compartments were added.
- An air-water dual-phase model was added to more accurately track the movement of volatile contaminants in the vadose zone.
- The ability to run analyses was extended from 1,000 years to 10,000 years or longer after site closure.
- The vadose zone flow and transport model was upgraded to support two-dimensional or three-dimensional analyses.
- The ecological model and human risk models were upgraded so the concentrations of contaminants in food products consumed by humans are produced by the ecological model.

This report documents the functions in the SAC software and provides a number of example applications for Hanford problems. References to theory documents and user guides are provided as well as links to a number of published data sets that support running analyses of interest to Hanford cleanup efforts.

Acknowledgments

A team of people led by project manager Bob Bryce helped define the requirements for the SAC, developed the computer codes, and collected data for assessments. The authors are indebted to the following people who also assisted in this task: Rosanne Aaberg, Carmen Arimescu, Charlie Brandt, Amoret Bunn, Dave Engel, Vicky Freedman, Larry Gerhardstein, Beverly Kanyid, George Last, Terri Miley, Bruce Napier, Bill Perkins, Cynthia Rakowski, Van Ramsdell, Marshall Richmond, and Dennis Strenge.

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1.0 Introduction

The System Assessment Capability (SAC) is a suite of interrelated computer codes that provides the capability to conduct large-scale environmental assessments on the Hanford Site. The SAC was developed by Pacific Northwest National Laboratory for the Department of Energy. This report describes the SAC software and demonstrates the application of the SAC to several Hanford Site problems in the following discussions:

- Section 1.0 provides background on the Hanford Site and the history of Hanford Site assessments.
- Section 2.0 summarizes SAC capabilities, hardware requirements, and the modifications made to the SAC following publication of the initial assessment using the SAC (Bryce et al. 2002).
- Section 3.0 describes the functions of the major software modules and provides example calculations.
- Section 4.0 describes several example applications of the SAC.

1.1 Hanford Site Background

From its creation in the 1943 until the late 1980s, the Hanford Site was dedicated to the production of plutonium for national defense and management of the resulting waste. These activities produced about 2,600 waste sites on the Hanford Site. The severity of contamination at individual waste sites ranges from contaminated tumbleweeds to radioactive and chemical waste in underground liquid radioactive waste storage tanks. The majority of the waste was disposed of in the 100, 200, and 300 Areas. However, some waste was disposed of outside the operational areas at Gable Mountain Pond, Waste disposal trenches and caissons adjacent to Energy Northwest Property, the 300 North burial ground, and the Environmental Restoration Disposal Facility.

The location of the Hanford Site within the state of Washington and relative to geographic features including the Rattlesnake Hills, Columbia River, and Yakima River is shown in Figure 1.1. The core zone of the Central Plateau shown in Figure 1.1 is taken from the *Hanford Site End-State Vision* (DOE 2005, Figure 1.2). Groundwater contamination from past operations exists in the unconfined aquifer underlying the 100, 200, and 300 Areas. Substantial plumes of mobile contaminants have migrated to the north and to the east-southeast from inside or near the Core Zone. Commingling of contaminant plumes is possible between contaminant plumes with origins in the 200 Areas and those originating from the 100 and 300 Areas. Similarly, commingling is possible between plumes originating in the 200 West Area and those originating from 200 East Area waste sites. Accordingly, Hanford Site assessments can involve sources and their inventories in all operational areas of the site.

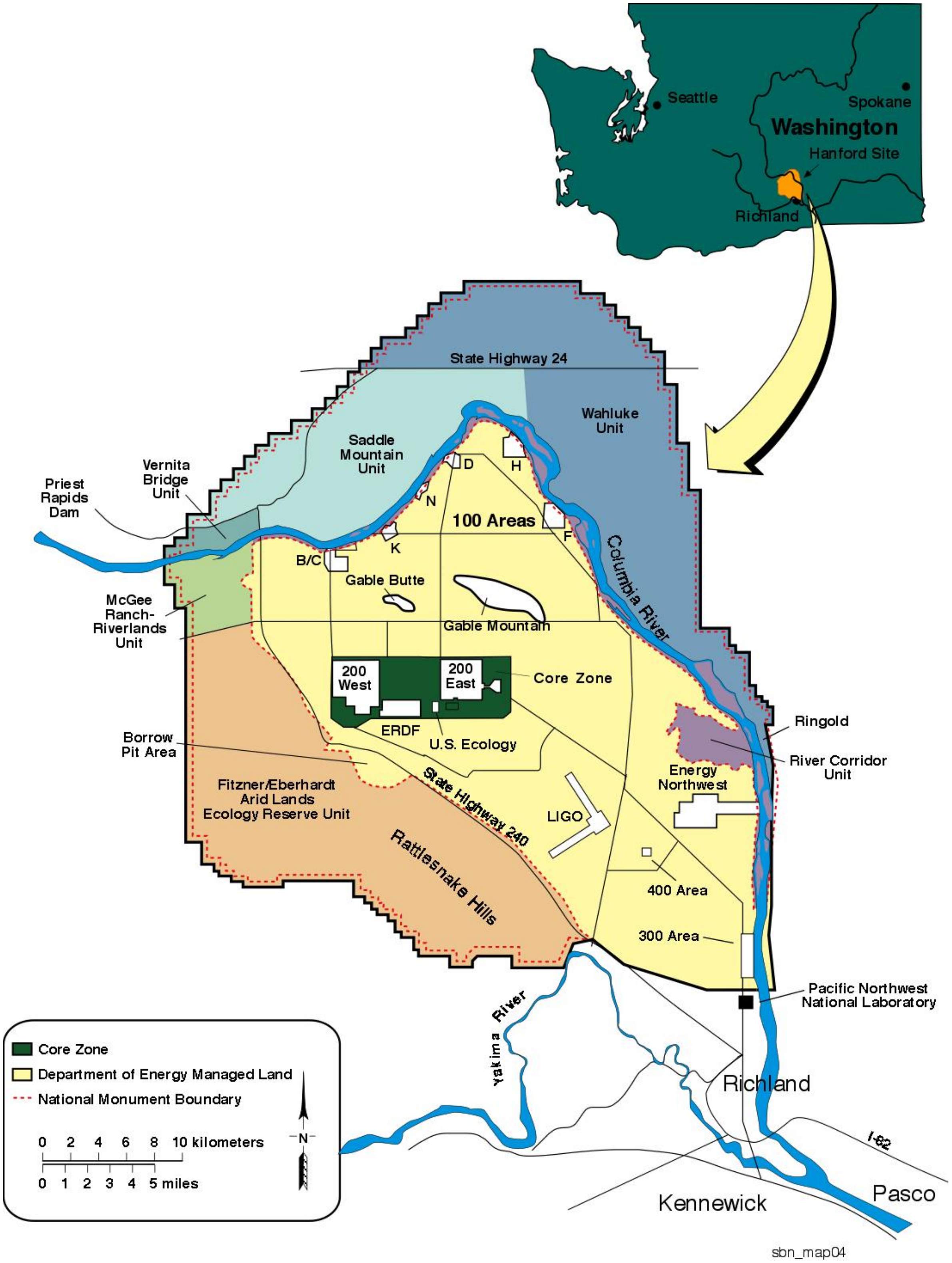


Figure 1.1. Hanford Site Location

1.2 Hanford Site Assessment History

Historically, the impact of contaminants from an individual waste site has been evaluated as part of the process to determine a remedy. During the late 1990s additional emphasis was placed on evaluating the impact of an individual waste site in the context of the waste sites located nearby and ultimately in the context of all waste that will remain at Hanford. This emphasis was first articulated by the DNFSB finding 94-2, which led to the requirement for a "Composite Analysis" in DOE Order 435.1. This DOE Order requires a Composite Analysis to ensure public safety through the management of active and planned low-level radioactive waste disposal facilities associated with the Hanford Site. A Composite Analysis is defined as "a reasonably conservative assessment of the cumulative impacts from active and planned low-level waste disposal facilities, and all other sources of radioactive contamination that could interact with the low-level waste disposal facility to affect the dose to future members of the public." At the Hanford Site, a Composite Analysis is required for continued disposal authorization for the immobilized low-activity waste (ILAW), spent vitrification plant melter components, low-level waste in the 200 East and 200 West Solid Waste Burial Grounds or the Integrated Disposal Facility to be located in the 200 East Area, and Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) waste in the Environmental Restoration Disposal Facility (ERDF).

A second effort that called for a holistic evaluation of Hanford's impact on the environment was the Columbia River Comprehensive Impact Assessment (CRCIA) document (DOE 1998a), which described the CRCIA group's vision for a complete assessment of Hanford impact on the Columbia River.

Based on those two efforts, the Groundwater Vadose Zone Integration Project undertook the development of the System Assessment Capability starting in 1998. The first year was dedicated to developing a set of requirements for an integrated assessment and subsequently the software and data sets needed to support the assessment. This effort included workshops with representatives of the regulatory agencies, Tribal Nations, State of Oregon, and stakeholders. It also involved peer review of the requirements, software design, and history matching results, as well as review by the Integration Project Expert Panel and the United States Geological Survey.

An initial assessment performed with the System Assessment Capability was documented in 2002 (Bryce et al. 2002). Following publication of that document, the software was modified to prepare it for use in an update of the Composite Analysis. The initial Composite Analysis was prepared in 1998, and the maintenance plan (DOE 2000) called for an update to the analysis every five years. In the fall of 2005 the decision was made by the Department of Energy that the next Composite Analysis would be drawn from the cumulative assessment performed for the Tank Closure and Waste Management Environmental Impact Statement. That analysis is being performed using other tools. A timeline of key events in the development of the SAC is presented in Table 1.1.

Table 1.1. Timeline of Key Events in the Development of the System Assessment Capability

Dates	Key Events
1999 (Fiscal Year)	• Meetings with stakeholders, regulators and Tribal Nations to develop the SAC approach
1999 October	• Software Design Started
2000 March	• Software Design Peer Review
2000 June	• Review by Integration Project Expert Panel
2000 September	• Software (Rev. 0) Assembled and Tested
2001 May	• Review by USGS
2001 June	• History Matching/Calibration
2001 September	• Review by Integration Project Expert Panel
2001 October	• History Match Peer Review
2002 September	• Initial Assessment Documented
2002 November	• Enhanced requirements identified
	• Reviewed Initial Assessment and Plans for the 2004 Composite Analysis with:
2003 February	❖ Nez Perce Tribe
2003 April	❖ Confederated Tribes of the Umatilla Indian Reservation
2003 May	❖ Yakama Indian Nation
2003 September	❖ Washington State Dept. of Ecology
2004 February	❖ U.S. Geological Survey
2004 March	❖ U.S. Environmental Protection Agency
2004 April	❖ Washington State Dept. of Ecology
2004 October	• Revised Software (Rev. 1) assembled
2005	• Performed Preliminary Composite Analysis Runs

2.0 Capability Description

The conceptual illustration of the System Assessment Capability (SAC) in Figure 2.1 portrays a linear flow of information. In general, an analysis begins with identifying the inventory of a selected set of waste sites and proceeds with the analysis of movement of contaminants from waste sites into and through the vadose zone, groundwater, atmosphere, and Columbia River. A soil environment is also modeled as impacted by groundwater, surface water, and atmospheric deposition. Contamination levels in the environment form a basis for estimating cumulative impacts on hypothetical future individuals and the ecology. In a few cases, the release of inventory occurs directly into the groundwater through a reverse well or into the Columbia River from the single-pass reactors. During operation of the chemical separations plant, there were direct releases into the atmosphere. The atmosphere, groundwater, Columbia River, and soil technical elements provide media-specific concentration estimates used in risk and impact assessments.

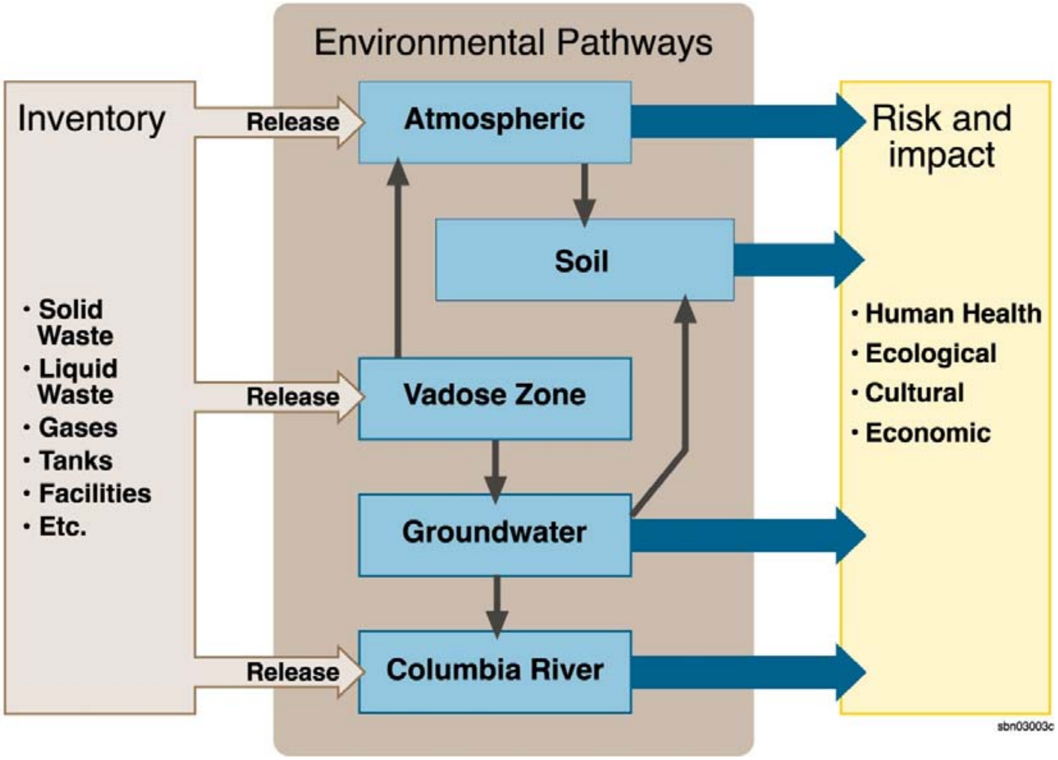


Figure 2.1. Conceptual Model of the System Assessment Capability

One of the challenges associated with performing an assessment is appropriately presenting how well the results predict what might actually occur. The challenge arises because the attributes of the site that affect transport of contaminants, the impact of contaminants on living systems, and the future conditions used in the assessment, as well as many other factors upon which the predictions depend, are not completely known or understood. SAC was developed to support stochastic risk assessments so the effect of uncertainties in inventory, release, and transport of contaminants in the environment, and the impacts to the ecology and man can be examined. This approach is consistent with recommendations made by the Nuclear Regulatory Commission (NRC) for hydrogeologic modeling and uncertainty analyses for nuclear sites (Neuman and Wierenga 2003). In general, other sources of uncertainty, such as conceptual model uncertainty (alternate concepts of chemical separations plant waste streams, for example), are not addressed within the SAC framework, but they can be discussed in the interpretation of the inventory and impacts from completed analyses.

2.1 Overview of SAC Capabilities

The SAC consists of an integrated suite of computer codes designed to model a Hanford-wide assessment. An assessment can:

- be deterministic (one realization) or stochastic (many realizations based on a Monte Carlo approach to quantifying uncertainty)
- address a single waste site or hundreds of waste sites
- simulate a single analyte or dozens of analytes
- simulate a portion of the environment or the entire gamut of solutions from inventory through transport to dose or risk.

The following major simulation steps are necessary for performing a complete Hanford-wide assessment. Such an assessment may involve several hundred waste sites, a couple dozen contaminants and a number of waste forms.

- Simulate the inventory.
- Simulate the time transient groundwater flow (to identify the thickness of the vadose zone as a function of time for subsequent analyses).
- Simulate contaminant release from waste forms in the vadose zone.
- Simulate vadose zone water flow and contaminant transport (including releases of volatile contaminants to the atmosphere).
- Simulate air transport from direct atmospheric releases and releases from the vadose zone.
- Simulate groundwater transport.
- Simulate transport in the river using background analyte concentrations from upstream.
- Simulate transport in the river using direct discharges from waste sites and output from the groundwater transport model (superimpose on background concentrations if desired).
- Simulate concentrations in seep water and soil in the riparian zone along the river.
- Simulate upland soil concentrations from air deposition of contaminants and use of contaminated irrigation water.
- Simulate accumulation of contaminants in ecological species and potential impacts to ecological species.
- Simulate risks or impacts to human health.
- A top-level diagram flow diagram identifying the major code modules of the SAC is provided in Figure 2.2. Four of the major codes (RATCHET2, STOMP, CFEST and MASS2) were written by

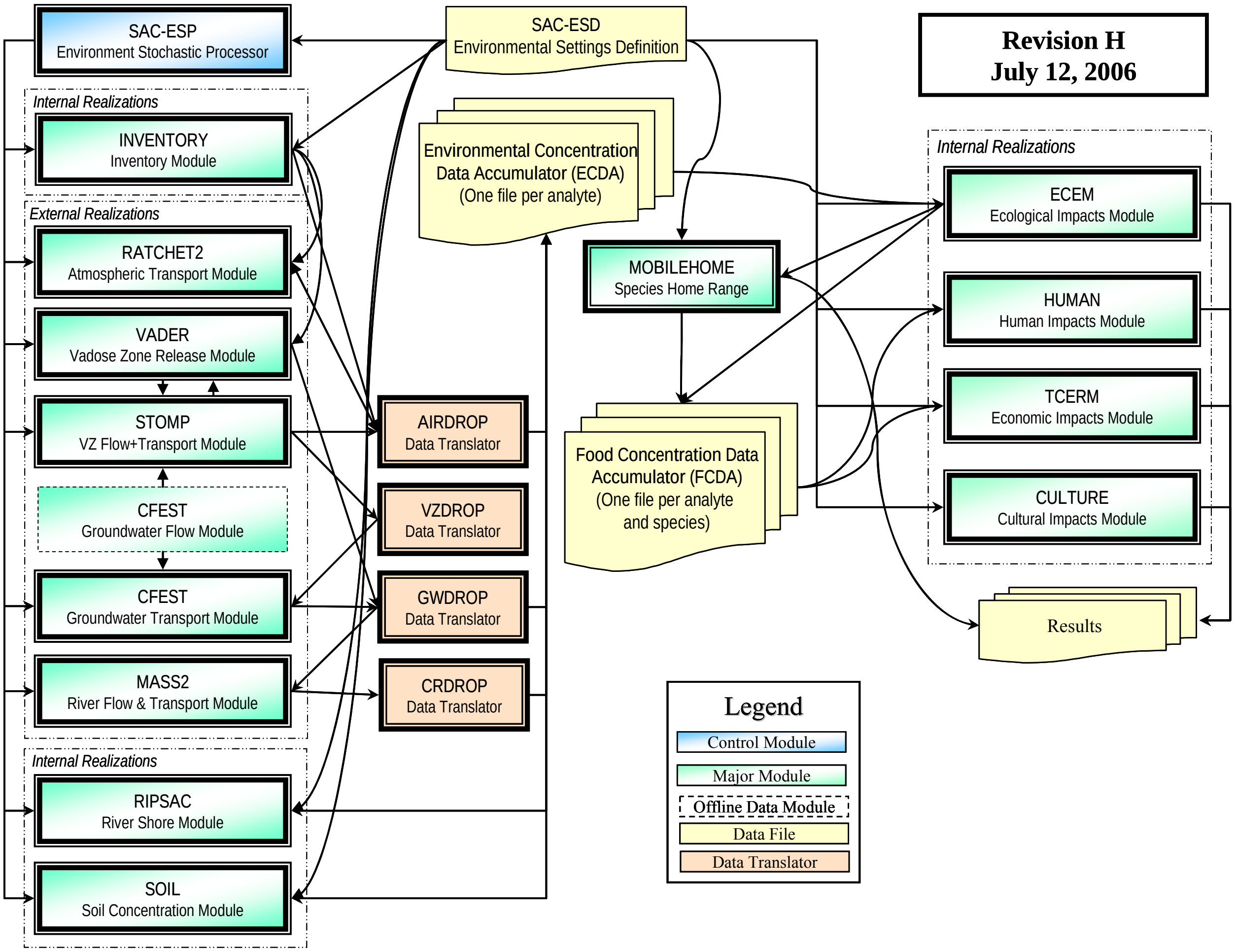


Figure 2.2. Flow Diagram for the SAC Rev. 1 Computer Codes

researchers for other projects or are publicly available, and they are embedded largely unmodified in the SAC. All of the data translation programs and a number of major modules were written specifically for the SAC. User instructions and supporting theoretical information are provided in the following documents:

- SAC-specific user guides (Eslinger et al. 2006a, 2006b, 2006c)
- Air transport model – RATCHET2 (Ramsdell and Rishel 2006)
- Vadose zone flow and transport model – STOMP (White and Oostrom 2000)
- Groundwater flow and transport model – CFEST (Freedman et al. 2006)
- River flow and transport model – MASS2 (Perkins and Richmond 2004a, 2004b)

A number of base assumptions guided the development of the SAC, and they have a bearing on how simulations are performed. Some of the major assumptions are the following:

- A simulation starts with a clean site and models the introduction and movement of contaminants. Thus, a typical Hanford assessment starts in 1944 and models forward in time.
- Subsurface movement of contaminants in the vadose zone is modeled separately for every waste site.

- Subsurface movement of contaminants in groundwater can be modeled on a spatial scale as large as the entire Hanford Site.
- Subsurface movement of contaminants in groundwater supports introduction of contaminants from multiple waste sites.
- Movement of contaminants in the Columbia River is modeled from the Vernita Bridge downstream to the confluence of the Columbia and Yakima Rivers.
- Air movement of contaminants is modeled on a spatial scale as large as the Hanford Site or larger.
- The set of SAC components that model movement of contaminants in the environment and the set of components that evaluate risk or impacts are separable. Either set of components can be run without running the other set.
- Impacts from upstream background concentrations and Hanford contributions in the Columbia River can be separated.
- The system supports performing both deterministic (single value) simulations and stochastic simulations, providing the capability to address uncertainty issues.
- The system supports analysis of the long-term effects from releases of contaminants. Risk or impacts results are based on models of chronic exposures rather than acute exposures.
- Contaminants can be radioactive or stable elements or compounds. In addition, they can be classified as organic and inorganic elements or compounds.
- A design philosophy was to make as many aspects of the SAC data driven as possible. For example, the number of waste sites, number of analytes, size of the groundwater domain, and inclusion or exclusion of a river component in the modeling system are all data driven. No software changes are required to switch between these different problem descriptions.

Simulation from 1944 Forward. Many prior studies of risk and impact (such as Wood et al. 1995; Wood et al. 1996, Mann et al. 2001) begin analysis with the waste and barrier system in place at site closure. Those that have addressed future remedial action alternatives (for example, CH2M HILL 2002) often start with the present-day contaminant profile underlying the past waste discharge or disposal facility in question. Because information on the position and levels of contamination in the Hanford vadose zone beneath most waste sites is not available, SAC assessments are designed to begin simulation in 1944 with a clean site and introduce waste discharges and disposals as they occur in time. The SAC system does allow insertion of an initial concentration condition in the groundwater module at any specified time, but such an action may cause mass conservation problems if not implemented with care.

Simulation of the Vadose Environment. The SAC can model releases from one or more waste sites using the governing equations for flow of water and transport of contaminants. The software is designed to facilitate tracking a dozen or more analytes through hundreds of different geologic profiles in the vadose zone. Although typical breakthrough curves may be generated, the approach is based on physical processes rather than convolution assumptions. Example simulation results provided in Section 4.0 use several hundred waste sites. The geology at different wastes sites at Hanford can vary dramatically, especially if sites on the Central Plateau and in the 100 Areas are simulated in the same analysis. Each vadose zone site is modeled separately from other sites and uses a unique description of the local geology. Vadose zone models can be run in 1D, 2D, or 3D and can include both liquid- and gas-phase physics.

Simulation of the Groundwater Environment. The SAC can use a groundwater model on a variety of spatial scales, but the historical applications have used a site-wide scale model. The modeling approach is based on physical processes rather than convolution assumptions based on a small set of representative runs. The groundwater flow system is transient in time; thus one cannot make the time-invariant assumptions required for a convolution approach. The groundwater system under the Hanford Site has natural hydraulic boundaries along Rattlesnake Mountain and at the Columbia River. The upper reaches of Dry Creek and Cold Creek drainages on Rattlesnake Mountain and Cold Creek valley define the upper elevations of the unconfined aquifer that underlies the site and discharges into the Columbia River as it passes Hanford to the north and east. In addition, the Yakima River is a natural boundary near the southern portion of the Hanford Site. Releases from possibly several hundred waste sites are integrated into a single input file for simulating contaminant migration in the groundwater. If desired, side analyses can be performed to examine the effect that releases from the vadose zone for a single waste site, or from a small group of waste sites, have on migration of the contaminant in the groundwater system.

Simulation of the Columbia River. The river model in the SAC can use a variety of spatial scales, but the applications reported in this document use a model from the Vernita Bridge downstream to the confluence of the Columbia and Yakima Rivers. The modeling approach is based on physical processes of water flow and contaminant transport. Vernita Bridge is located upriver of all Hanford operational areas and represents a logical upstream boundary for analyses. Discharges and background water quality indicative of releases from Priest Rapids Dam can be applied at Vernita Bridge and maintain model integrity. Communities near and downstream of the Hanford Site include Richland, Pasco, and Kennewick. The applications in this document simulate water concentrations at the Richland Municipal Water intake but do not include the Kennewick or Pasco water intake points.

Simulation of Air Transport. To address air transport occurring on a time scale of hours to days, the SAC can model chronic releases to the atmosphere from one or more waste sites. These waste sites may be facilities that release directly to the atmosphere, or they may be sites containing volatile contaminants that migrate from the subsurface to the surface. Example simulation results provided in Section 4.0 include several hundred waste sites, but relatively few waste sites release volatile contaminants to the air. The spatial domain of the current air model is discussed in detail by Ramsdell and Rishel (2006). The air transport domain is larger than the Hanford Site. On the west and north the air model domain extends a few miles outside the Hanford site boundaries. On the east the domain extends far enough to incorporate the region where the Snake and Walla Walla Rivers empty into the Columbia River, and on the south the domain extends roughly to Hermiston, Oregon.

Separable Environment and Risk/Impact Simulations. A key aspect of the SAC is the separability of movement of contaminants in the environment and calculation of risk or impact metrics. Results of contaminant migration within the environment were stored for a large number of discrete points in the simulated space and time. Risk or impact analyses can be performed using the stored concentration results. Thus, multiple points of exposure and multiple exposure scenarios can be analyzed without recalculating the release and environmental transport.

Separable Background and Hanford Contributions. The analysis of background and non-Hanford contributions to contamination are modeled in the river and upland soil environmental modules. The upland soil module is included because river water can be used for irrigation. This is accomplished by running one set of river simulations with only background levels of contamination (for example, contaminants entering the river system from radioactive fallout or other upstream sources) and another set of river simulations with Hanford Site contributions superimposed on the background levels. Background contributions from the Hanford Site in groundwater are not modeled. By simulating both background and Hanford Site contributions superimposed on background levels, analysts are able to evaluate and present the difference between the two results as the incremental contribution from Hanford sources.

Dual Stochastic and Deterministic Capability. The underlying framework of the SAC supports quantification of uncertainty using a Monte Carlo approach. However, the codes just as easily support deterministic simulations (one realization). The computational resources required to perform a stochastic analysis are significant for a Hanford-scale problem evaluating the impacts of multiple contaminants. The deterministic capability allows exploration of the basic response of the system without paying the premium in elapsed time and computer resources required for a full stochastic analysis.

Chronic Exposure Models. The environmental concentration data saved to support the risk and impacts modules are aggregated up to the annual level irrespective of the representative time scales needed to solve the governing equations of the underlying physical models. Therefore, the risk and impacts modules were developed using chronic exposure assumptions with an implicit time scale of one year.

Impacts Models. Exposures to humans from consumption of ecosystem products, including crops, meat, milk, and fish, are simulated by using results of the ecosystem model. An option exists in the software to evaluate ecological and human impacts based on specified media concentrations (such as a 1 pCi/L concentration of a radioactive analyte in groundwater) without running calculations for inventory or the environmental transport codes. This, the impacts codes can be used for a variety of analyses on CERCLA-related problems without running the entire system of codes. Analyses with just the impacts codes can be performed on a standalone desk-top computer and do not require utilization of the entire hardware cluster (see Section 2.4). In addition, impacts to ecological species can be calculated over an entire home range in addition to impacts at a discrete set of locations.

Contaminant Types. A large variety of contaminants were introduced into the environment across the Hanford site. Many of these were radioactive (tritium and technetium-99, for example) while others were stable compounds (chromium and nitrate, for example). In addition, a number of the contaminants are organic compounds (carbon tetrachloride and trichloroethylene, for example). The SAC is designed to handle all of these classes of contaminants.

2.2 Overview of Solutions Supported by the SAC

The large number of governing equations in the SAC provides the opportunity to extract a large number of solutions. Of course, the scope and purpose of the particular analysis dictates which solutions will be evaluated. Solutions available on individual components can often be aggregated into summary results. For example, the inventory of tritium could be aggregated over all waste sites with liquid releases, or the release of technetium-99 to the groundwater could be aggregated over all burial grounds. In addition, summary statistics or uncertainty plots can be developed for many solutions when a stochastic analysis is conducted. The following major solution types are available:

- Inventory Solutions
  - Volume and contaminant activity or mass of individual waste streams by disposal year
  - Volume delivered to individual waste sites by disposal year

  - Contaminant activity or mass delivered to individual waste sites by disposal year

- Waste Form Release Solutions
  - Release rate of each contaminant from individual waste forms as a function of time using eight different release models
  - Remaining inventory of each contaminant in individual waste forms as a function of time
- Vadose zone water flow and contaminant transport
  - Contaminant mass (or activity) flux release to the aquifer as a function of time for individual waste sites
  - Water mass flux to the aquifer as a function of time for individual waste sites
  - Concentration of contaminants in the vadose zone profile as a function of time for individual waste sites
  - Release of gas-phase contaminants to the ground surface as a function of time for individual waste sites
- Atmospheric Transport Model
  - Representative χ/Q values for unit releases at each waste site
  - Air concentrations as a function of location integrated over all releases
  - Air deposition as a function of location integrated over all releases
- Groundwater Flow and Transport Solutions
  - Hydraulic head at any point in the model domain by time step
  - Water velocity at any point in the model domain by time step
  - Contaminant concentration at any point in the model domain by time step
  - Contaminant mass flux transfer to the river at interface locations by time step
  - Water mass flux transfer to the river at interface locations by time step
  - Size of a groundwater plume above a user-specified threshold value
  - Probability (stochastic analyses) that a contaminant concentration in groundwater exceeds a user-specified threshold
- River Model Solutions
  - Contaminant concentration in the water column at any point in the model domain
  - Contaminant concentration in river bottom pore water at any point in the model domain
  - Contaminant concentration in river bottom sediment at any point in the model domain
  - Contaminant concentration in suspended sediment at any point in the model domain
- Riparian Soil Model Solutions
  - Concentration of each contaminant in seep water
  - Concentration of each contaminant in soil wetted by seep water

- Upland Soil Model Solutions
  - Concentration of each contaminant in soil for dry land assumptions
  - Concentration of each contaminant in soil using groundwater for irrigation
  - Concentration of each contaminant in soil using river water for irrigation
- Ecological Risk and Impacts Solutions
  - Body burden of contaminants (radioactive or stable elements or compounds; organic and inorganic elements or compounds) in ecological species
  - Dose from radioactive analytes to ecological species
  - Food crop concentrations for human risk modules
- Human Risk and Impacts Solutions
  - Radiation dose or risk for user-specified exposure scenarios
  - Cancer incidence or mortality risks for user-specified exposure scenarios
  - Hazard quotient from chemicals for user-specified exposure scenarios

Many solutions from the codes are stored for multiple locations at specified time steps. These solutions can be processed and displayed as contour plots or line plots for given times or as an animation of contour plots or line plots over time. The contaminant concentrations in the following environmental media can be displayed via animated plots: groundwater, river bank seep water, surface water, river bottom pore water, river bottom sediment, riparian zone soil, upland soil with no irrigation, upland soil with groundwater irrigation, upland soil with surface water irrigation, and air. The deposition rates of contaminants on the ground surface from air transport can also be animated. The outputs of the ecological and human impacts models are tied to the same time steps and locations as the environmental concentration data. Almost every detailed solution available from these models can also be displayed as contour plots or line plots for given times or as an animation of contour plots or line plots over time.

Because movement of contaminants through the vadose zone and the groundwater may take years, or even centuries, environmental concentration data saved to support the risk and impacts modules are aggregated up to the annual level. The environmental release and transport modules are allowed to operate on times scales appropriate to accurate solution of the equations representing the governing physical processes. For example, the wetting front in the vadose zone at the initiation of liquid releases to a waste crib may have a representative time scale on the order of seconds, while transport of a sorbed contaminant such as strontium-90 in the groundwater or cesium-137 in the vadose zone may have a representative time scale of years to decades. Data from the air transport and river transport models are averaged over time to annual values. Therefore, the risk and impacts modules use a representative time scale of one year.

A suite of postprocessors has been built to facilitate postprocessing the simulation results. These postprocessors facilitate addressing analysis questions at the scale of one waste site up to the scale of the entire Hanford Site. Some of the functions that have been automated are the following:

- Accumulate and report inventory by groups of waste sites and selected waste form types for specified periods of time. For example, the inventory of iodine-129 that went to ground in past leaks from waste tanks can easily be extracted.
- Accumulate and report remaining inventory in waste forms and release rate histories by groups of waste sites and selected waste forms for specified periods of time.
- Accumulate and report release from the vadose zone to the groundwater by groups of waste sites for specified periods of time. For example, summary information on past releases of contaminants to groundwater in the 200-ZP-1 groundwater operable unit can easily be obtained.
- Accumulate and report release from the groundwater to the river by river location for specified periods of time.

2.3 Overview of Problems Amenable to Solution Using SAC

The SAC is well suited to modeling a problem where there are a large number of waste sites with multiple waste forms releasing multiple contaminants into the atmosphere, the vadose zone and a river. The Monte Carlo approach in SAC makes it an excellent tool for evaluating the uncertainty in performance predictions.

The largest-scale problem where SAC is particularly well suited for providing solutions is a problem similar to the one present at the Hanford site. Stated another way, SAC is well suited to modeling a problem where there are a large number of waste sites with multiple waste forms releasing multiple contaminants directly into the atmosphere, the vadose zone, and a river. Waste entering the vadose zone can migrate down to an aquifer that eventually discharges into a river. Ecological and human impacts are then evaluated based on the movement of contaminants in the environment. Unlike many other modeling systems, the SAC incorporates a fully integrated Monte Carlo toolkit. Therefore, it is particularly well suited to evaluate the uncertainty resulting from model parameter uncertainty and arising from multiple processes that contribute to contaminant migration and human health predictions.

Because SAC is modular in approach, it is also well suited to a range of problems that are smaller in scope than an assessment of the entire Hanford Site. Some illustrative examples are the following:

- **Inventory**: SAC is well suited to evaluating the completeness of a site-wide or smaller scale (such as a process or operational area) inventory compilation or evaluating the uncertainty in an inventory compilation.
- **Release**: SAC is well suited for evaluating the releases of contaminants from multiple waste forms to the vadose zone environment at multiple sites.
- **Vadose Zone Transport**: SAC is well suited for evaluating the transport of contaminants in the vadose zone or the release of contaminants from the vadose zone to the aquifer at multiple sites. For example, the past release of technetium-99 to the groundwater for all waste sites in an operable unit can easily be calculated. In addition, estimates of the amount of technetium-99 still in the deep vadose zone for the same suite of sites can easily be obtained from the same set of runs.

- **Vadose Zone Flow**: SAC is well suited for evaluating the movement of artificial discharges of water through the vadose zone at multiple sites. This solution has proven valuable to calibration of groundwater flow models due to the large influence that artificial discharges have on the water table elevation.
- **Ecological and Human Impacts**: SAC is well suited for evaluating ecological and human impacts for a wide range of exposure scenarios and contaminants types (radioactive or stable elements or compounds, inorganic elements or compounds, or organic compounds). These impact modules operate as standalone entities or can be integrated in the SAC framework. The ecological model can provide reasonable estimates, as opposed to bounding estimates, of contaminant concentrations in food products to the human impacts model. These models are best suited to problems where a screening analysis has already indicated that a more detailed model should be applied.
- **Impacts from River Activities**: The SAC framework is well suited to integrating the effects of direct releases to a river and migration of contaminants from groundwater into the river. Concentrations of the contaminants in water and sediment downstream from the entry point (or multiple points) of the contaminants can be compared to benchmark values or used for ecological and human impact analyses. Identification of impacts from background concentrations versus impacts from contaminants derived from activities on the Hanford Site.

The long-term chronic exposure assumptions in the current SAC software imply that it is not the appropriate tool for a number of Hanford analyses. Software modifications would be required to address the following classes of problems:

- dose for a worker other than a chronic exposure scenario such as for a park ranger or industrial worker exposed to contaminated air or soil
- acute exposures of any type
- exposure pathways that depend on direct local intrusion into waste forms.

2.4 Overview of Hardware and System Software Configuration for SAC

Production runs using the SAC often examine the release and transport of a dozen or more contaminants at hundreds of vadose zone waste sites for times frames of up to 10,000 years in duration. The computational requirements are substantial enough that the SAC runs on a dedicated computer cluster. A recent photo of the cluster is provided in Figure 2.3. The cluster operating system is Red Hat Enterprise Linux WS release 3 (Taroon Update 7). The codes running on the cluster are all written in the Fortran language, and two compilers are used: Intel Fortran 8.1 and Lahey Fortran 95 version 6.2.

The SAC cluster is classified as a high performance computing system under the export control rules of the Department of Commerce. The cluster has 6.3% of the capability of PNNL’s current supercomputer as measured by MTOPS (Millions of Theoretical Operations per Second). The SAC cluster rates 1,207,134 MTOPS as compared to the 19,090,500 MTOPS of the supercomputer. A major difference between the machines is that the SAC cluster hardware is optimized to run a large number of small- to moderate-sized code runs on single CPUs, while the PNNL supercomputer is optimized to run large-memory jobs using a parallel memory architecture.

Figure 2.3. SAC Hardware Cluster

Functionally, the SAC cluster consists of a main processor (controller), three analysis work stations, a number of computational nodes and 6.2 terabytes of shared disk space. The main processor has dual 3.00-GHz (Intel Xeon) processors, 2 gigabytes of RAM, and the capability for disk access at 60 megabytes per second. The analysis work stations provide the capability for analysts to examine results and process moderate-sized jobs while the compute nodes are fully engaged in running a large simulation case. The analysis work stations and compute nodes have the following characteristics:

- Analysis work stations:
  - 2 dual 1.3-GHz Intel Pentium III, 4 gigabytes RAM, 1-gigabit Ethernet
  - 1 dual 2.8-GHz Intel Xeon, 4 gigabytes RAM, 1-gigabit Ethernet
- Computational nodes (214 total nodes):
  - 60 dual 1.0-GHz Intel Pentium III, 512 megabytes RAM, 100-Mbit Ethernet
  - 13 dual 2.66-GHz Intel Xeon, 1 gigabytes RAM, 1-gigabit Ethernet
  - 16 dual 2.80-GHz Intel Xeon, 1 gigabytes RAM, 1-gigabit Ethernet
  - 18 dual 3.20-GHz Intel Xeon, 2 gigabytes RAM, 1-gigabit Ethernet

Run times depend on the specific contaminants being examined and a large number of other input factors. Vadose zone transport, groundwater transport, and river transport combined typically takes well over 99% of the computing resources required to run an analysis. As an example, the following run times have been observed during a number of analyses:

- Vadose Zone Transport (STOMP Code)
  - ~ 2 minutes per run (1D, single-phase transport)
  - ~ 20 minutes per run (1D dual-phase [air, water] transport)
- Groundwater transport (CFEST Code)
  - ~ 7 days per 10,000-year run (unsorbed contaminant)
  - ~ 2 days per 1,000-year run (unsorbed contaminant)
- River Transport (MASS2 Code)
  - ~ 5 days per 10,000-year run
  - ~ 1 day per 1,000-year run

The number of individual code runs is substantial for a production run at the Hanford Site. A stochastic analysis using 10 analytes, 700 waste, sites and 100 realizations results in 700,000 runs of the STOMP code, 1,000 runs of the CFEST code, and 1,000 runs of the MASS2 code.

2.5 Overview of Configuration Management Controls

The SAC codes model complex phenomena, and development of data sets for analyses can require a substantial investment in manpower. A system of controls on software changes, data changes, and assessment simulations has been developed by the Hanford Remediation Assessment Project. The documents describing the controls are maintained as project records rather than publicly available documents.

The SAC software was developed under controls documented in a software configuration management plan. These controls were put in place as the code requirements were initially being developed and have guided the software activities through the design, implementation, and testing phases. The software is maintained in an automated configuration control system, and changes are initiated and tracked through a software change request process.

Compilation and documentation of complex data sets for analyses using the SAC are managed under the control of a data configuration and management plan. The controls in that plan help ensure that correct data are obtained, reviewed, and maintained for use in assessments. A number of data packages were developed under the control of this plan. For example, a vadose zone data package (Last et al. 2006a), an inventory data package (Kincaid et al. 2006), and a risk data package (Miley et al. 2006) have been published.

Conduct of individual analyses are performed under the control of an assessment documentation and verification plan. This plan helps ensure that the data selected are appropriate for an analysis and that checks have been made to ensure that the code runs executed were appropriate for the purposes of the analysis.

2.6 Overview of SAC Rev. 1 Software Enhancements

The initial assessment performed with the SAC software (Bryce et al. 2002) identified a number of areas where both the software and analysis approach could be improved. The six major software-related upgrades identified at that time were the following:

- An air pathway model should be added to allow an all-pathway analysis to be performed.
- Models for releases from a glass waste form, release of carbon-14 and chlorine-36 from buried graphite reactor cores, and releases from buried naval reactor compartments are needed.
- A multiple-phase model is needed to more accurately track the movement of carbon tetrachloride in the vadose zone.
- The capability to run 10,000-year or longer analyses is needed.
- The capability to more closely match modeling results from other projects requires incorporation of a two-dimensional or three-dimensional vadose zone transport model and the ability to accept results produced by others for specific waste sites.
- The ecological model needs to calculate upland animal and food crop concentrations, in part to support human risk estimates that include food crops and animal products.

The major enhancements in the Rev. 1 version of SAC discussed in this section address the six major items above as well as a number of other items. A large number of smaller enhancements were also included to improve such things as flexibility in the definition of specific analyses, system stability, disk space utilization, run time durations, ease of data entry, and ease of extraction and display of simulation results.

At the time the Rev. 1 enhancements to the SAC were incorporated, the plan was to use the SAC software to perform the Composite Analysis, an analysis required by DOE Order 435.1. Exposure to airborne contaminants must be included in this all-pathways analysis of the dose from radioactive analytes. Although the dose from future release of radioactive materials to the air is expected to be much lower than doses from historical releases, the SAC software suite was upgraded to handle modeling of an air release, transport, and exposure pathway. Upgrades to a number of models were made in addition to building in an air transport model based on the RATCHET code. SAC models upgraded to include an air pathway included the SAC system controller, the vadose zone transport model, the soil deposition model, and ecological and human impacts models.

Changes to the SAC system controller. In addition to handling the new air model, the simulation controller and associated modules were upgraded to handle simulation times as long as 1 million years after site closure. This relatively minor upgrade allows the SAC suite of codes to be used in an analysis of peak dose, even if the peak were to occur hundreds of thousands of years in the future. Another top-level function of the SAC system controller was upgraded to support selecting from multiple groundwater models at the time of problem execution. Given the Monte Carlo nature of the entire modeling system, this upgrade supports stochastic treatment of alternate groundwater models. A new controller function allows specification of a time history of contaminant release from a waste form or release of contaminant to the aquifer by models not included in the SAC framework. The purpose of this function is to allow detailed release or vadose zone transport results developed by projects dedicated to modeling a particular site or waste form to be incorporated into a system-level assessment without having to incorporate the computational capability of the project into the SAC software.

Changes to the inventory module. The major upgrade to the inventory module was to fully automate the application of fill-in rules, surrogate site rules, and uncertainty rules in a new inventory preprocessor named INPROC. In the Rev. 0 software the effects of these rules were processed by project analysts, and the augmented inventory data were folded back into the inventory database. The upgraded approach allows the inventory database to be dedicated to primary inventory data (data obtained from traceable sources such as reports, sampling activities, and other projects). Outputs from the inventory database are then processed through the INPROC inventory preprocessor as the first step of a specific analysis, allowing different inventory rules to be applied for different analyses without modification of the primary inventory data.

Changes to the release models. Three major changes were made in the SAC release models. First, an option was added to delay the onset of release after the time the waste form was placed in the environment. In essence, this capability allows for a waste package to provide complete containment for a period of time before it fails and allows environment releases to occur. Although the time delay is incorporated into the model, no embedded model of containment degradation and failure is provided. A new model was provided that allows for a container to be contaminated as well as holding contaminated materials. In addition, a new release model specific to reactor compartments was incorporated.

New air transport capability. The new air transport capability for the SAC was provided by an enhanced version of the RATCHET code. The RATCHET code was used for the air transport portion of the Hanford Environmental Dose Reconstruction Project (Ramsdell et al. 1994). New input and output data translators were written to support embedding this legacy model in the SAC code framework.

Changes to the vadose zone flow and transport module. A number of enhancements were made to the vadose zone flow and transport module. Some of these enhancements were achieved by activating more options in the parent code STOMP in the SAC framework. The multiple phase flow and transport capability of STOMP was one of two new capabilities activated in the SAC framework. This capability allows modeling of contaminants in both a liquid phase and a gaseous phase. For example, carbon-14 released from a graphite reactor core can partition into both liquid and gas phases, and some contaminant can migrate to the land surface while the rest migrates down the vadose zone to the groundwater. The other STOMP option activated in the SAC framework was the ability to model the vadose zone around a waste site in two or three dimensions. In SAC Rev. 0 only one-dimensional flow and transport was supported. The vadose zone module, in concert with the system controller and the release module, was upgraded to allow a stochastic representation of the natural infiltration into the deep vadose zone. Two aspects of the interface of the vadose zone module to the groundwater module were also upgraded. First, a previous run of the groundwater model is used to align the length of the vadose zone column to the time-varying depth to groundwater. This capability more accurately reflects the rise and fall of the groundwater elevation due to Hanford water disposal actions. In addition, a vadose zone data translator was upgraded to partition releases from the vadose zone into a number of nodes for the groundwater model rather than placing all of the releases into one groundwater node. This enhancement is based on the areas associated with the two models, and it improved the data handoff between the vadose zone model and the groundwater model. This is especially significant for waste sites with a large footprint (B Pond, for example) that overlay a groundwater model that uses a finely resolved computational mesh in the same locality.

Changes to the river flow and transport model. Two major enhancements were made to the river flow and transport model, and one previous capability was dropped. The capability to model transport of contaminants based on biotic uptake of contaminants was eliminated because analyses showed it was a very minor contributor to mass transport or dose. A model enhancement was incorporation of the ability to increase or decrease the number of computational cells representing the width of the river as the river bed varies. This change allows the model grid to accurately represent the effect that islands have on river flow without requiring the grid to be finely resolved in straight stretches of the river where no islands are present. Model run times were substantially reduced after this enhancement was incorporated. The other major enhancement to the river model improved the transfer of data from the groundwater model to the river model. The river model divides the water column into two vertical segments. One segment, called pore water, represents water flow in the cobble or gravel on the river bottom, and the other segment represents the free-moving water column. The method by which contaminant mass coming from the groundwater model was injected into the pore water segment was modified to account for the water volume as well as the contaminant mass coming from the groundwater model. Although this water volume is insignificant relative to the overall flow rate of the river, it is important in accurately defining contaminant concentrations in the river bed region that exhibits reduced water velocities.

Changes to calculation concentrations in surface soil. The model for calculating concentrations in surface soil was moved from the human impacts code into a standalone code. In addition, the pseudo steady-state model was upgraded to a time transient model that accounts for the onset time of irrigation and deposition of contaminants on the upper soil layer calculated by the air transport model. Three estimates of soil concentrations based on different water use assumptions (dry land, land irrigated with local groundwater, and land irrigated with remote surface water) are now calculated, and these values are passed to all of the impacts codes.

Changes to the ecological impacts model. A number of enhancements were made to the ecological impacts model. One of the major enhancements was to use the detailed ecological impacts model to calculate concentrations in food products and pass the concentrations to the human and economic impacts modules. Inputs from the air transport model were used in the ecological module to incorporate air deposition and air concentration pathways into the exposures for terrestrial plants and animals. In addition, a dust resuspension term based on soil concentrations was added for terrestrial animals. The models for terrestrial animals and plants were upgraded to support the concept of growing plants or grazing on irrigated fields. The model for organic contaminants was upgraded to match with other published model components. In addition, a postprocessor to the ecological modules was developed to account for the effect that home range has on mobile animal species. For example, a coyote may have a hunting range that covers 100 km^2 or more.

Changes to the human impacts module. Three significant functional enhancements were made to the human impacts module. First, an air exposure pathway was added. Second, concentrations in soil and food products were obtained from other SAC codes that utilize more detailed models rather than the simple models resident in the human impacts module in the Rev. 0 version of the code. Third, the model for calculating risk from radionuclides was changed from a simple multiplier applied to dose estimates to the direct calculation of risk as defined in Federal Guidance Report 13 (EPA 2002). Dose estimates from radionuclides continue to use the models and data from Federal Guidance Reports 11 (EPA 1988) and 12 (EPA 1993).

Changes to the cultural and economic models. The cultural model was upgraded to interface with the revised data structures for the Rev. 1 software, but no additional functional changes were made. The economic model was also upgraded to interface with the revised data structures for the Rev. 1 software, but no additional functional changes were made. No applications of the economic module have been made using the Rev. 1 software.

3.0 Capabilities of Individual SAC Modules

This chapter provides an overview of the capabilities of the individual modules of the SAC. Example applications of each module are included. Although the examples provided in this section were chosen to illustrate the component capabilities, a significant amount of work has been done to collect Hanford-specific data to support project or DOE programmatic analyses. Some of the areas where data have been collected and published are the following:

- Inventory data package (Kincaid et al. 2006)
- Geographic and operational site parameters (Last et al. 2006b)
- Atmospheric data package (Napier and Ramsdell 2005)
- Vadose zone hydrology data package (Last et al. 2006a)
- Groundwater data package (Thorne et al. 2006)
- Ecological and human risk data package (Miley et al. 2006)
- Ecological characterization data (Downs et al. 2004)
- Release model data package (Riley and Lopresti 2006).

3.1 Inventory Module

A schematic representation of historical operations at the Hanford Site related to generation of waste is shown in Figure 3.1. There were four distinct steps in the production process: fuel fabrication, fuel irradiation, chemical separations, and product finishing. During the first decades at the Hanford Site, it was common to locate waste disposal sites relatively close to waste-generating facilities. This practice resulted in numerous disposal sites of varying configurations. The highly radioactive liquid waste from chemical separations was stored in large underground tanks in the 200 Areas (Agnew et al. 1997; Kupfer et al. 1997). Large volumes of solid waste (for example, contaminated tools and protective clothing) were disposed in burial grounds, and large volumes of relatively low-level radioactive liquid waste were discharged to shallow subsurface cribs (drain fields), French drains, injection (or reverse) wells, and specific retention trenches. By 1990, all fuel fabrication, reactor operations, and chemical separations work ended and cleanup of past-practice units had begun. Low-level waste from ongoing laboratory and other waste management operations is sent to burial grounds in the 200 West and 200 East Areas. Most liquid discharges of radioactive waste have been discontinued, an exception being liquid discharges to the State-Approved Land Disposal Site, which receives treated water from the 200 Area Effluent Treatment Facility.

To determine an inventory estimate at a moment in time (now or at site closure, for example), the four steps identified in Figure 3.1 are amended to include two aspects:

1. The quantities of radionuclides and chemicals imported and exported from the Hanford Site are introduced or extracted at several points in the operation (for example, materials fed into the fuel fabrication process, chemicals fed into the reactor operation and chemical separations processes, and uranium and other special nuclear materials exported from the Hanford Site).

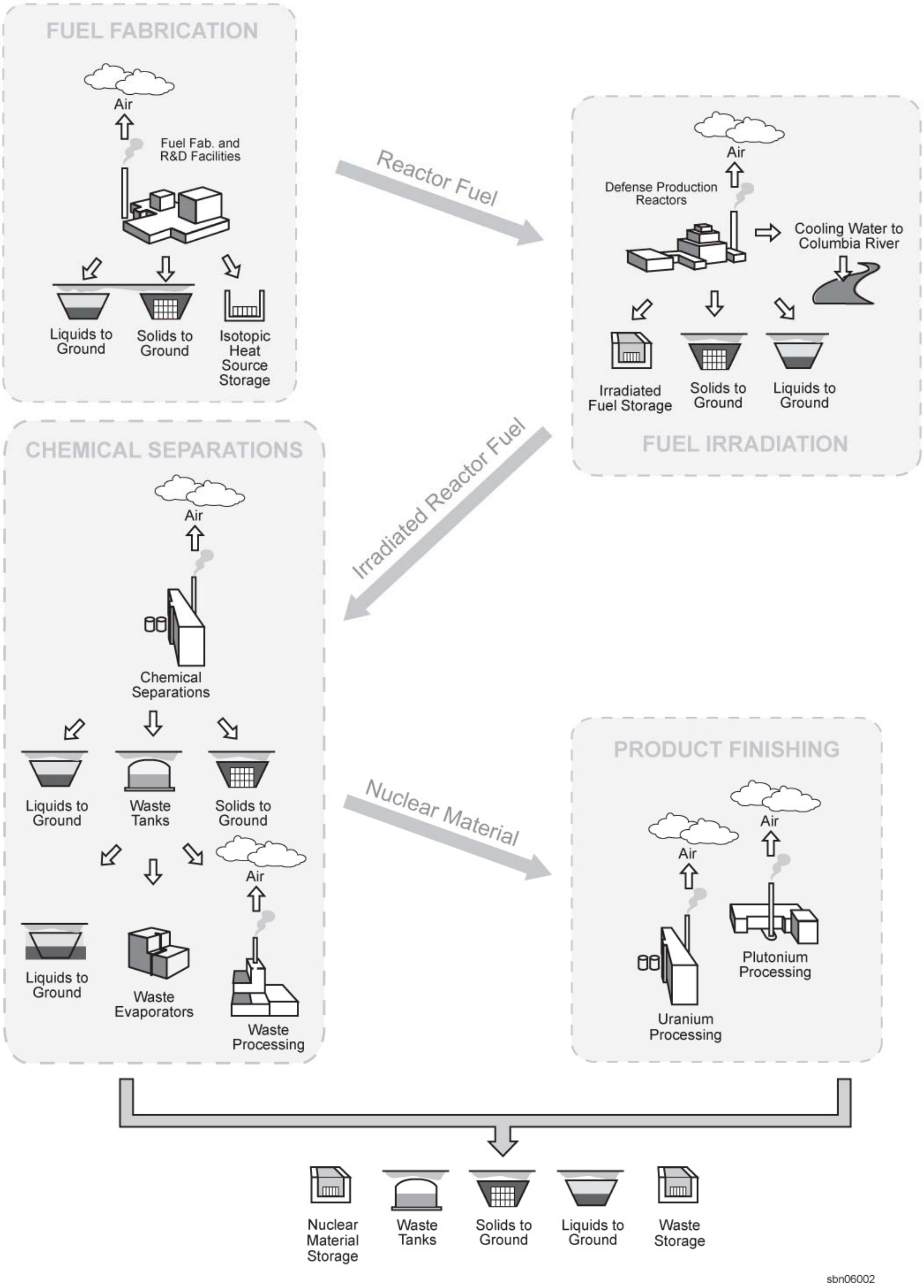


Figure 3.1. Conceptual Model of the Hanford Waste Generation and Disposition Process

2. Decisions regarding the remediation, decontamination and decommissioning, and disposal actions will impact many of the facilities and waste depicted in Figure 3.1. These cleanup actions will define the end-state configuration (that is, both location and stability or form) of the waste remaining at the Hanford Site.

Avenues by which contaminants enter and migrate through the environment to reach the accessible environment are shown in Figure 3.2. In this report, materials are not considered "waste" until they are placed in the environment and allowed to release (solid waste, for example) or are declared waste (spent fuel, for example). Thus, tank waste inventories currently *stored* in single-shell and double-shell tanks in the Central Plateau are not included in this inventory until they are recovered, treated, and disposed of.

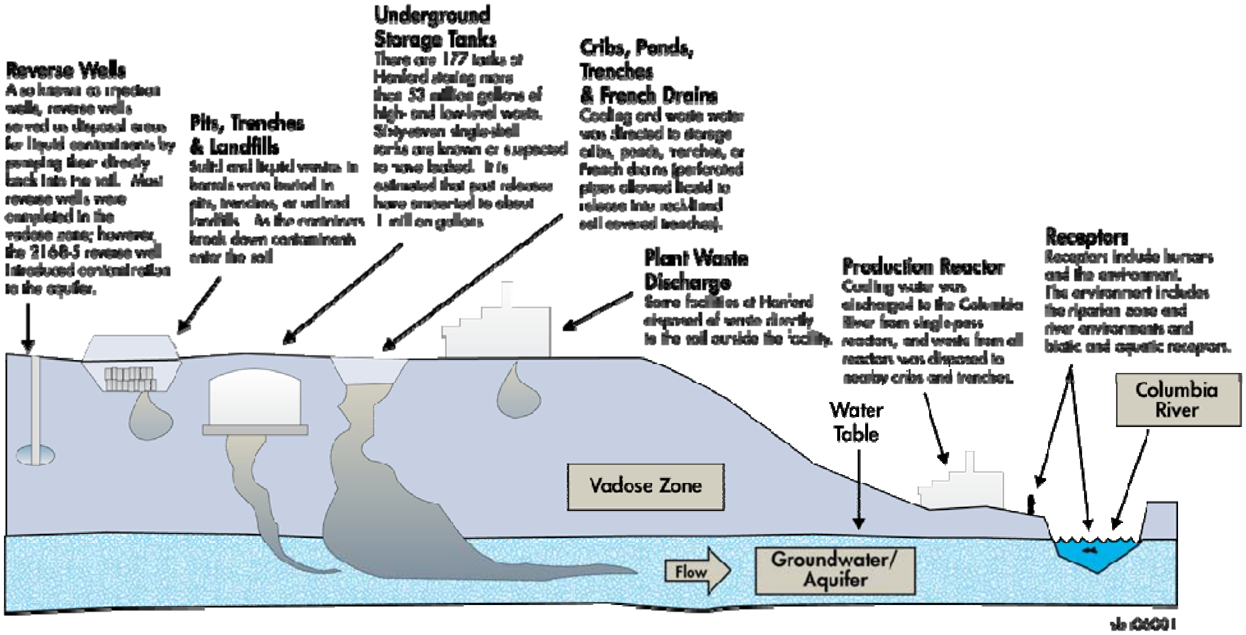


Figure 3.2. Avenues by which Hanford Waste Enters the Environment

The inventory module uses inventory information collected from a wide range of sources to develop inventory estimates for all releases. Release events are defined on an annual basis by a combination of information about location, waste form type, waste form volume, and analyte (possibly multiple analytes) concentrations or amounts. Detailed information on an inventory compilation and development of estimation and surrogate rules is provided by Kincaid at al. (2006). That data set forms the basis for most of the example applications provided in this document. Data on approximately 20 analytes and 700 waste sites are present in the inventory compendium and were collected from a large number of primary sources. The data are maintained in a database, and more than 1,470,000 records are generated when the data are exported in the format needed for further processing in the SAC framework.

Within the SAC framework there are three distinct stages in the inventory processing.

- **Inventory Database**: The inventory database is the first stage in the inventory processing. The database is the location where primary inventory-related data are stored and tracked. These primary data are derived from a wide range of sources including historical records and reports or data sets generated by other projects. More detailed information on the inventory data is provided by Kincaid at al. (2006). A data file of inventory information for selected sites and contaminants is exported by the inventory database for use in an analysis.

- **Inventory Rule Processing**: In the second stage of the inventory processing, a variety of rules are applied to the data set to create a file, called a disposal action file, specific to the particular analysis. The rules fall into three classes: estimation rules, surrogate rules, and uncertainty rules. *Estimation rules* can be used to fill in inventory estimates that are missing because of data gaps. Data gaps are common because few laboratory analyses provide data on all of the possible contaminants of interest for Hanford risk analyses. These rules are used to fill in values for missing contaminants based on the presence of other contaminants in the inventory data for that specific waste stream. *Surrogate rules* can be applied when no analyte-specific information exists in the records for a specific waste stream. In general, these rules use analyte-specific information from a similar waste stream and applies them to the waste stream under consideration. *Uncertainty rules* are used to assign uncertainty values to all data not already bearing uncertainty descriptions.
- **Inventory Data Generation**: The final stage of the inventory processing is to use the disposal action file to generate specific inventory values for the assessment. Data can be generated for a deterministic analysis (one realization) or for a stochastic analysis (multiple realizations). Waste streams are represented by annual values.

The Hanford Remediation Assessment Project has developed an inventory data set to support remedial investigations. This inventory contains data on a suite of 23 radionuclides (^{14}C, ^{36}Cl, ^{137}Cs, ^{152}Eu, ^{3}H, ^{129}I, ^{237}Np, ^{231}Pa, ^{226}Ra, ^{79}Se, ^{90}Sr, ^{99}Tc, ^{233}U, ^{234}U, ^{235}U, ^{238}U, ^{238}Pu, ^{239}Pu, ^{240}Pu, ^{241}Pu, ^{242}Pu, ^{241}Am and ^{243}Am) and has been documented by Kincaid et al. (2006). That report also documents the primary data sources used to estimate annual and cumulative inventories. The inventory includes data produced by the Hanford Tank Waste Operation Simulator (HTWOS) (Kirkbride et al. 2005). This simulator is used to estimate the inventories from the Hanford waste tanks as those tanks are cleaned out and the wastes processed for permanent disposal. The HTWOS simulation incorporated into this inventory is known as the Development Run executed on January 5, 2005, and documented by Kirkbride et al. (2005). The inventory also includes data from the Hanford Soil Inventory Model (SIM) that is documented by Corbin et al. (2005). The SIM model is used to estimate the more substantial past liquid discharges and unplanned liquid releases at the Hanford Site. It is based on process knowledge of chemical separations plants and waste transfer records from those plants to single-shell tanks, double-shell tanks, and liquid discharges (for example, to cribs, specific-retention trenches, French drains, and ponds), and unplanned releases including past tank leaks. The HTWOS data and the SIM data sets contain information on more contaminants than are incorporated into the inventory for the Hanford Remediation Assessment Project.

An example inventory calculation is provided in Table 3.1 for a set of 410 sites located on the Central Plateau of the Hanford Site. The data in the table are cumulative through the year 2007 and are decay corrected to 2007. This example is intended to represent the type of information that can be obtained rather than to definitively provide an inventory estimate. However, it was developed based on a specific set of limitations and assumptions. The sites were chosen to include or represent the following actions: past tank leaks, past (planned) liquid discharges, past solid waste disposals, past unplanned releases, and the State-Approved Land Disposal System (SALDS 600-211) facility. The following waste types and waste sites were not included in the summation: tank residuals, Environmental Restoration Disposal Facility, US Ecology site, Integrated Disposal Facility, graphite cores (production reactor blocks), Naval reactor compartments, buildings, PUREX tunnels, all waste sites in the 100, 300, and 400 Areas, waste

Table 3.1. Example Inventory Summation (Ci) for 410 Waste Sites by 2007 with Subtotals by Data Sources

Analyte	SIM Data	HTWOS Data	Record Data	Estimated Data	Surrogate Data	Subtotal
^{3}H	3.451E+03	2.198E+02	5.128E+03	0	3.760E-01	8.799E+03
^{14}C	1.951E+02	5.294E-05	5.056E+02	0	2.704E-02	7.008E+02
^{36}Cl	0	0	2.234E-01	0	0	2.234E-01
^{79}Se	2.146E+00	3.917E-07	1.073E-02	5.027E-01	1.285E-03	2.661E+00
^{90}Sr	9.511E+03	6.165E-06	1.436E+05	1.262E+04	1.782E+01	1.657E+05
^{99}Tc	6.862E+02	5.935E-04	2.338E+02	2.267E-01	2.039E-01	9.204E+02
^{129}I	4.686E+00	1.581E-05	5.556E-01	5.113E-01	8.248E-05	5.753E+00
^{137}Cs	4.405E+04	4.590E-06	2.151E+05	1.502E+04	2.182E+01	2.742E+05
^{152}Eu	1.297E-01	4.086E-08	1.786E-01	1.926E-02	3.807E-04	3.280E-01
^{233}U	2.251E+01	2.991E-05	5.257E-01	0	1.851E-02	2.306E+01
^{235}U	2.372E+00	9.002E-07	3.212E+01	2.772E-02	2.238E-01	3.475E+01
^{237}Np	5.376E+01	9.317E-05	1.144E-01	0	7.642E-03	5.389E+01
^{238}U	1.075E+02	4.313E-05	1.404E+03	6.273E-09	1.054E+01	1.522E+03

sites in the 600 Areas located off the central plateau and atmospheric releases. Detailed information on the inventory summarized in this table is provided by Kincaid at al. (2006). The HTWOS data come from the Hanford Tank Waste Operation Simulator (HTWOS) model documented by Kirkbride et al. (2005). The SIM data come from the Hanford Soil Inventory Model (SIM) documented by Corbin et al. (2005) and are provided as statistical distributions rather than point values. This table was generated using the mean values of the data distributions from the SIM.

3.2 Release Module

Release is the rate that radioactive and chemical contaminants find their way into the environment. The Release Module handles liquid releases and releases from solid waste forms. Liquid releases are handled as a simple pass-through to the vadose zone or to the Columbia River. The solid forms are primarily from solid waste burial grounds including past-practice sites (pre-1988), active sites (post-1988), and the Environmental Restoration Disposal Facility. Other solid waste includes residual waste in the single and double-shell tanks, naval reactor compartments, immobilized low-activity waste, the graphite cores of the retired production reactors, and concrete and cement waste associated with caissons and canyon buildings.

Waste containment facilities have a number of features that influence the rate at which contaminants can be released from waste. Those features are illustrated in Figure 3.3. The waste may be placed in a trench or reside in a tank. The trench, tank, or other engineered structure may have features that serve as barriers preventing infiltrating water from making contact with contaminants and transporting them from the waste to the vadose zone. Waste inside an engineered structure (such as a trench) may also be contained in a waste package (such as a metal drum or high-integrity concrete container). The drum or concrete container acts as an additional barrier preventing transport of the contaminants from the waste. Major containment materials for Hanford waste are concrete, steel, and bituminous layers and coatings. The stability and permeability of containment materials change over time. Time affects which features dominate the water or contaminant migration in containment materials. Surface covers on an engineered system and liners (geomembrane and geosynthetic) and leachate collection systems at the bottom further restrict infiltrating water from transporting contaminants to the vadose zone. Surface covers are particularly important because migration of infiltrating pore water may be limited as long as the cover maintains its integrity. Individual waste sites have one or more of the features shown in Figure 3.3. However, a typical waste site does not embody all of the features in the conceptual model. A number of key processes govern how much contaminant at any given time is released from the waste to the infiltrating water. One process is the affinity of contaminants to be retained by the waste (for example, sorption to soil or waste material). Another process is the ability of waste to dissolve, and in some cases, to form new precipitates, allowing some contaminants to be released to the infiltrating water while others remain trapped in the precipitated solids. Release from the waste may also be limited by the solubility of the contaminant in the infiltrating water.

Water infiltrating an engineered system may contact and react with fill materials (such as soil, basalt, or grout), containment materials in various states of degradation, and different types of waste. Reaction with these materials will change the water chemistry and physical and hydraulic properties over time. The water composition, pH, and redox state at any given time will influence the extent to which these processes influence contaminant release from the waste.

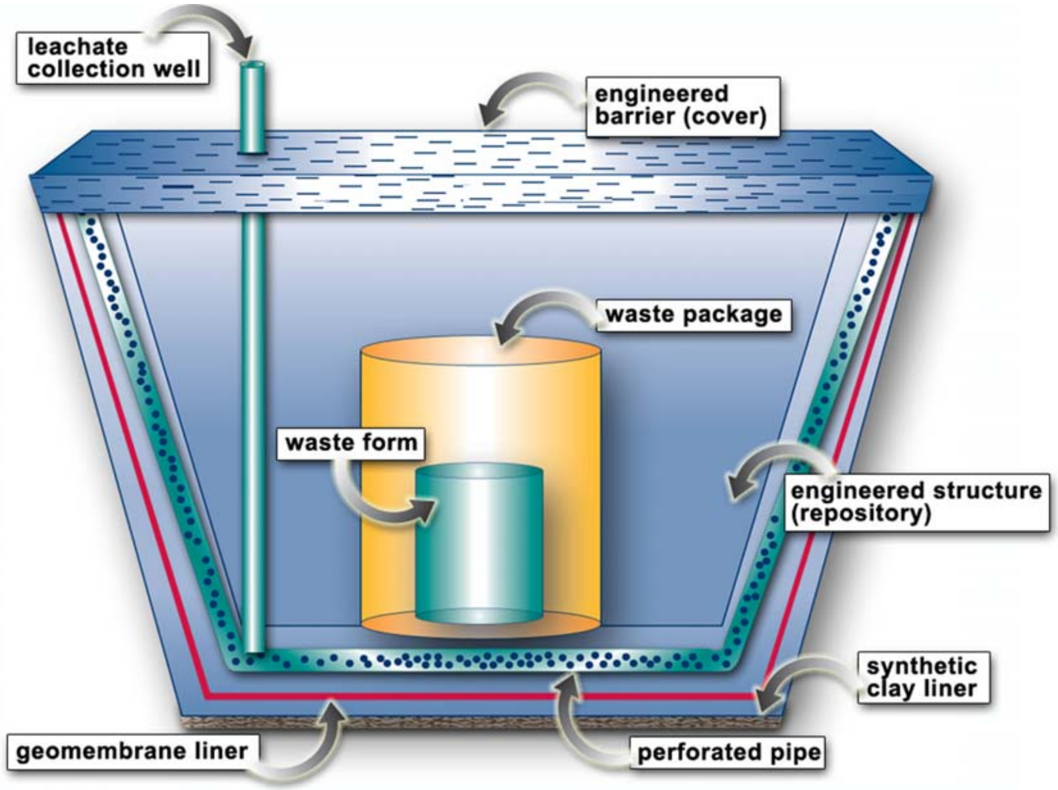


Figure 3.3. Conceptual Model for the SAC Release Module

The Release Module applies release models to waste inventory data from the Inventory Module and accounts for site remediation activities as a function of time. The resulting releases to the vadose zone, expressed as time-profiles of annual rates, become source terms for the Vadose Zone Module (see Figure 2.2). Radioactive decay, as appropriate, is accounted for in all inputs and outputs of the Release Module. The Release Module is implemented in the VADER (VADose zone Environmental Release) computer code. The location of the release module in the SAC processing sequence for contaminant transport is provided in Figure 3.4. This figure also illustrates the capability of VADER and STOMP to perform waste site remedial actions and move soil contamination and remaining waste forms to different waste sites (such as movement and burial of reactor cores or movement of contaminated soil to the Environmental Restoration Disposal Facility.

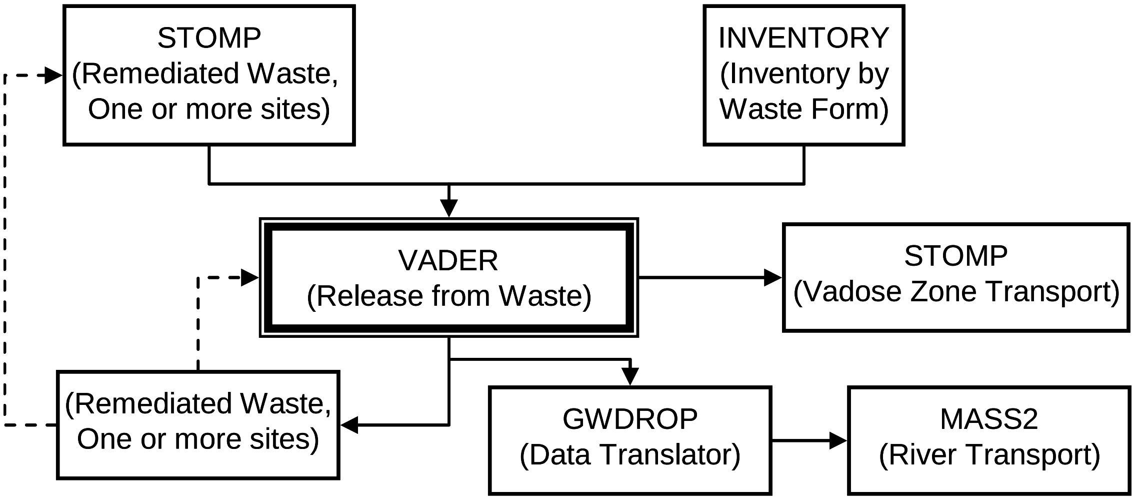


Figure 3.4. Location of the Release Module in the SAC Processing Sequence

The release model algorithms in VADER are summarized in Table 3.2. The mathematical derivations for the release models are documented in Appendix D of Kincaid et al. (1998), CCN 0512242 (BHI 1999), Riley and Lopresti (2004) and Eslinger et al. (2006a). The information provided here is intended only to highlight model assumptions and implementation features. The release models can be summarized into the following categories:

- Release is congruent with waste form matrix dissolution.
- Release is calculated from desorption of an analyte from soil.
- Release is calculated from diffusion of the analyte to the surface of the waste form.
- Release is a time-dependent empirical function.
- Release is a constant fraction of an initial quantity.

Currently only the water recharge rate past the waste form is implemented as a time-dependent parameter. Other parameters in the waste form models are time invariant.

Table 3.2. Summary of Waste Forms and Release Mechanisms Implemented in VADER

Waste Form	Characteristics	Typical Source	Release Mechanism	Routing
Soil-debris	High permeability	Soils, cribs, trenches, etc.	Desorption and Solubility	Top of vadose zone
Glass	Very low permeability	Vitrification	Dissolution of glass	Top of vadose zone
Saltcake or sludge	High permeability	Tank waste	Dissolution congruent to saltcake waste form	Top of vadose zone
Concrete or cement	Low/medium permeability	Misc.	Diffusion	Top of vadose zone
Reactor blocks	Other	Irradiated solids	Corrosion	Top of vadose zone
Reactor compartments	Other	Irradiated solids	Corrosion of containment with congruent release	Top of vadose zone
Special	Other	Other	Specified by table, mechanism not specified	Top of vadose zone to liquid or to solid pathway
Liquid	Instantaneous	Various	Pass-through	Top of vadose zone
Liquid to river	Instantaneous	Reactor coolant	Pass-through	River

The simplest release model in the SAC framework is a pass-through model. This model is invoked for all liquid waste streams, and the final waste destination may be either the vadose zone or the river. An example model result for liquid releases of H-3 at site 116-B-7 is provided in Figure 3.5. In this example, the liquid wastes are routed directly to the river. Liquid releases are spread out over a number of years, and the quantity disposed of (incoming inventory at the waste site) is equal to the quantity received every year. The curve labeled “cumulative release” represents cumulative release for this analyte from this waste form. The curve decreases with time because it is corrected for radioactive decay and H-3 has a half life of about 12.32 years.

A fractional release rate model has often been used to estimate releases of chlorine-36 from retired production reactor cores. In the example shown in Figure 3.6, the retired Hanford production reactor cores are assumed to all be buried at 218-W-5 in the year 2065. The fractional release rate in this example is calculated from the initial inventory and does not vary in time. In this example, the inventory is totally depleted by about calendar year 4800.

A more complex data set for a soil-debris model is illustrated in Figure 3.7. This example is based on record past disposals and projected future disposals of iodine-129 at the US Ecology site. The soil-debris model depends on the water flux passing through the waste form; thus the release rate is a function of the water flux rate. The left pane of the figure displays results from the calculations through the year 2100 and illustrates the capability to handle disposals of different quantities every year. The right pane of the figure illustrates the calculations out to the time where the inventory of iodine-129 is exhausted from the waste form. Both panes illustrate the dependence of the release rate on the water (Darcy) flux.

An example calculation for the release of technetium-99 from a buried cement waste form at the 200-E-136 waste site is provided in Figure 3.8. The cement model is a diffusion release model. It is not dependent on the water flux passing near the waste form.

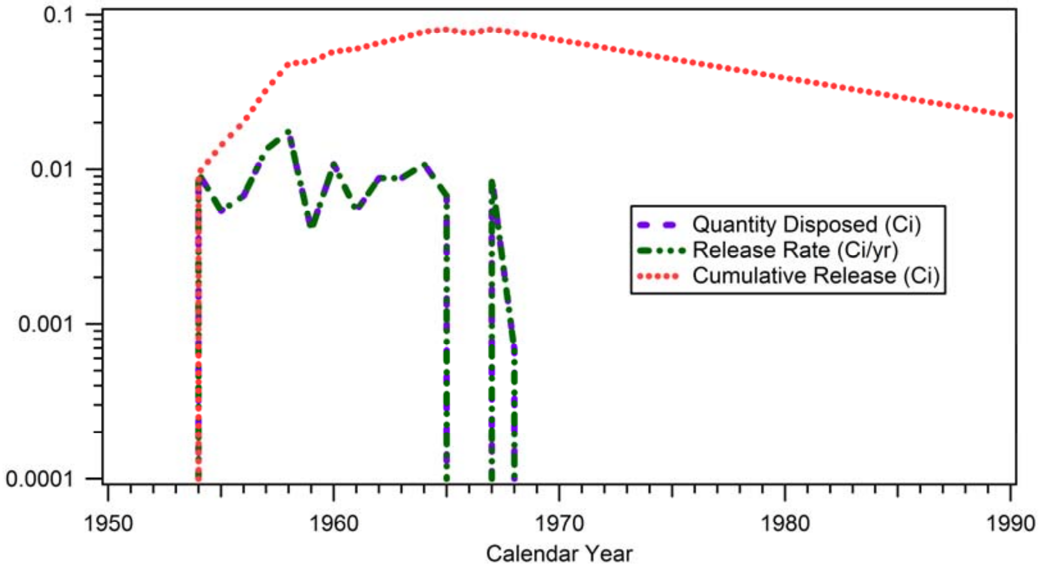


Figure 3.5. Release Model Results for H-3 in Liquids Passed Directly to the River at Site 116-B-7

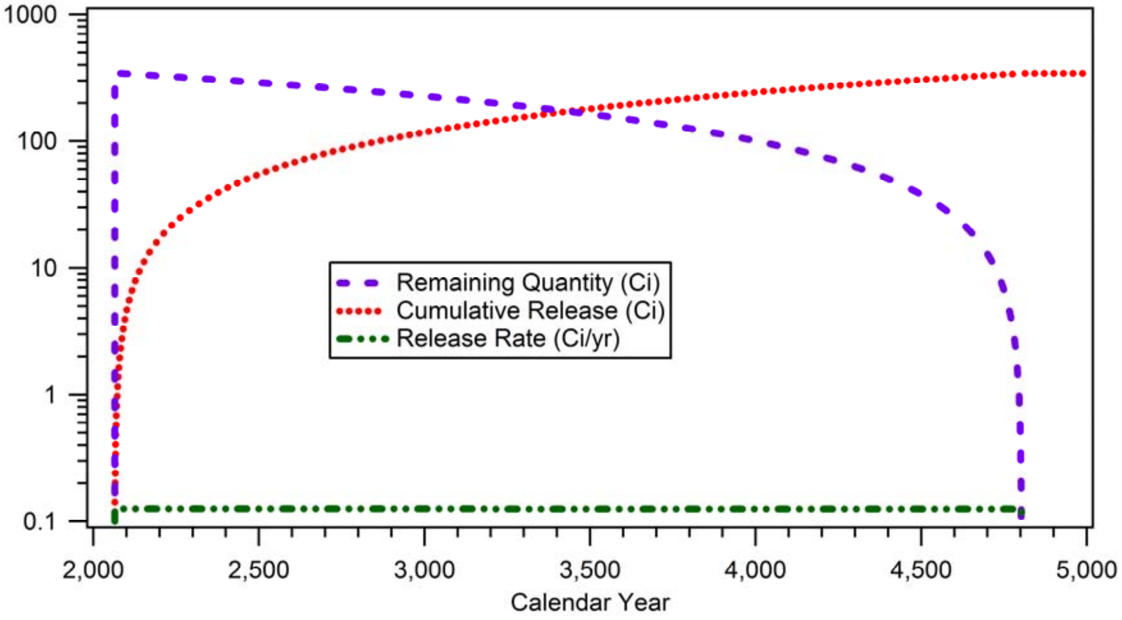


Figure 3.6. Release Model Results for Chlorine-36 in Production Reactor Cores Using a Fractional Release Rate Calculated From the Initial Inventory (Reactor Cores Buried at Site 218-W-5)

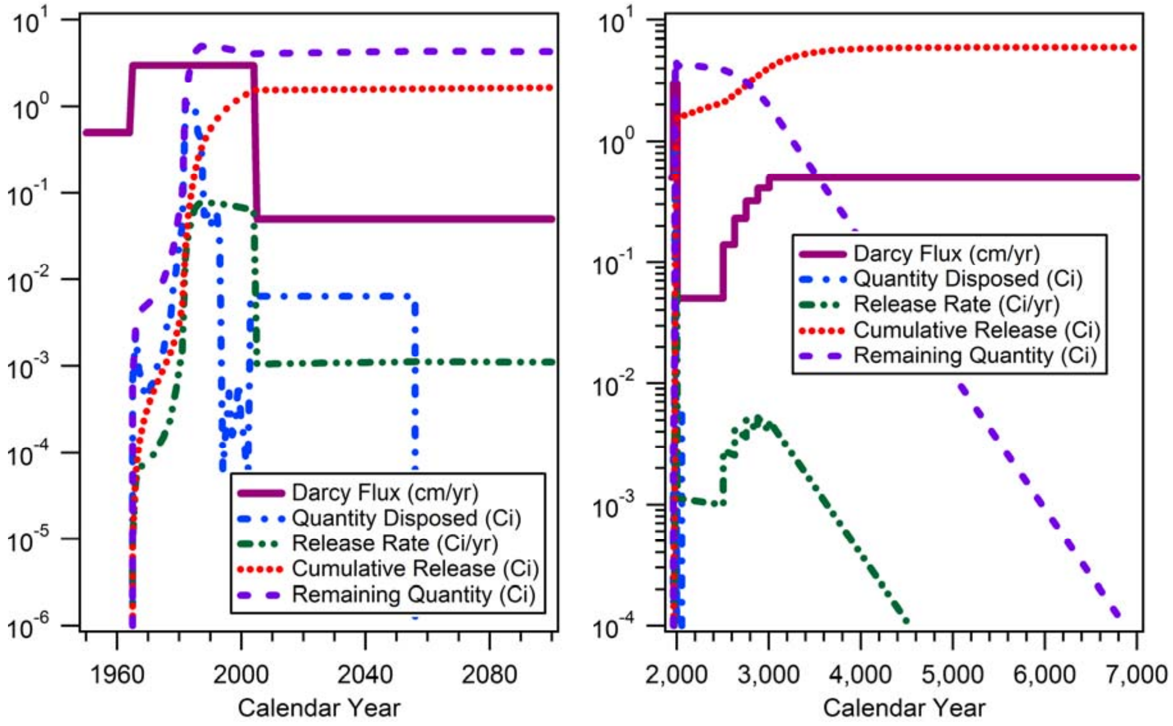


Figure 3.7. Release Model Results for Iodine-129 Buried at the US Ecology Site Using a Desorption Model for Soil-Debris Disposals (near-term results in left pane; long-term results in right pane)

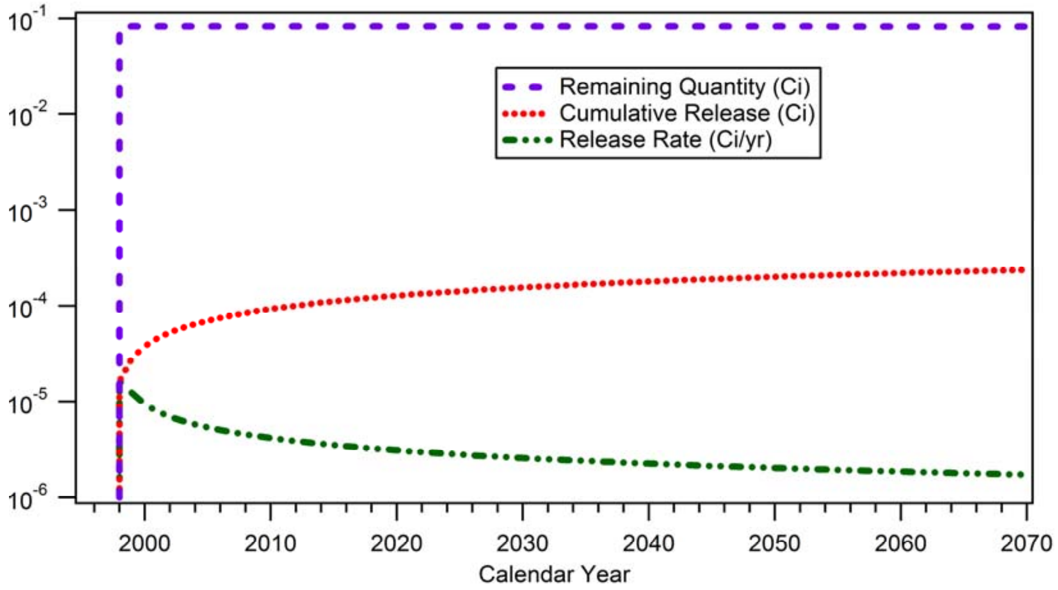


Figure 3.8. Release Model Results for Technetium-99 Encapsulated in a Cement Waste Form at 200-E-136 Using a Diffusion Model

3.3 Vadose Zone Flow and Transport Module

The vadose zone is the hydrogeologic region that extends from the land surface to the regional water table (DOE/RL 1998; Looney and Falta 2000). The goal of the Vadose Zone Flow and Transport Module is to understand the transport and fate of contaminants as they move through the vadose zone. Thus, the principal geographic focus of this module is on areas at the Hanford Site that 1) underlie liquid waste disposal sites, 2) underlie underground storage tanks or solid waste burial grounds that have the potential for leaks and leaking, and/or 3) have experienced past leaks and spills.

Vadose zone contamination is primarily the result of waste effluent releases to liquid waste disposal facilities, leakage from retention basins, and—to a lesser degree—accidental releases of contaminants through low-volume spills and dry waste burial grounds. Billions of liters of wastewater have created large contamination plumes within the vadose zone. Figure 3.9 illustrates the conceptual model from the Vadose Zone. The primary driving forces for contaminant transport are the source/release events, and recharge events. The dominant transport pathway is downward through the vadose sediments. Stratigraphic layering, variations in the hydraulic properties, and the presence of impeding features (for example, caliche layers) can locally alter and redirect the movement of contaminants laterally. Wilson et al. (1995) describes flow within the vadose zone as dynamic and characterized by periods of unsaturated flow at varying degrees of partial saturation punctuated by episodes of preferential, saturated flow in response to hydrologic events or releases of liquids.

The movement of contaminants in the vadose zone is affected by their sorption in the far field, and sometimes by complex dissolution/precipitation reactions between the waste liquids of extreme pH and the slightly alkaline sediments near the disposal location. The significance of sorption is that it delays downward movement of the contaminant and allows degradation processes (such as radioactive decay) to occur while the contaminant is chemically bound to the soil. The sorptive capacity of the vadose zone soils is fairly high; however the amount of sorption is a function of many factors, including mineral surface area and type, contaminant type (speciation) and concentration, overall solution chemistry and concentration, and reaction rates for adsorption or precipitation, dissolution, and hydrolysis reactions. Some contaminants do not sorb at all and are moved along with the bulk solution. Contaminants that can exist in the gas phase (for example, carbon tetrachloride, or carbon-14 in the form of carbon dioxide) are subject to atmospheric venting.

The vadose zone flow and transport module models water, gas, and contaminant movement in the vadose zone. Single-phase (water) or two-phase (water and air) coupled solution modes are available, depending on whether the contaminant considered is transported in gas phase in the vadose zone. The model geometry may be represented as either pseudo-2D (one-dimensional, vertically oriented with variable column area that is conditioned upon minimum conductivity and maximum annual discharge volume) or in explicit multiple dimensions (2D or 3D). For explicit multiple-dimension geometry, the use of symmetry boundaries is supported. Releases of contaminant mass or activity to the atmosphere and aquifer are passed to other models for further transport. The module may also be operated to produce releases of liquid discharge to the aquifer for use in calibration of the groundwater flow module.

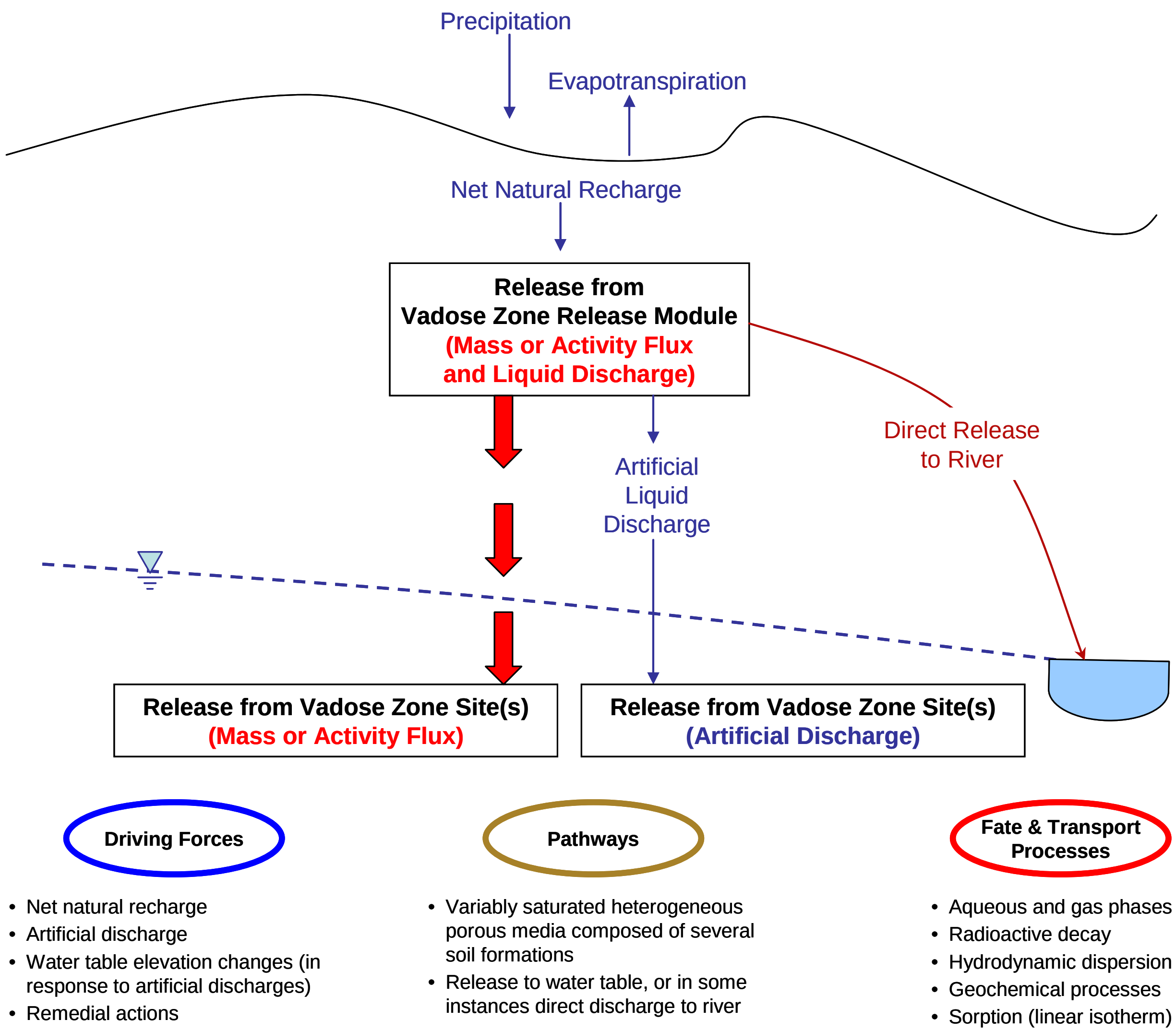


Figure 3.9. Conceptual Model for the Vadose Zone Module

Fluid flow in the vadose zone is solved using a water mass conservation equation for a variably saturated media together with constitutive equations that predict the nonlinear relationships between aqueous pressure, saturation, and hydraulic conductivity. Water transport occurs by advection through the aqueous and gas phases and by diffusion-dispersion through the gas phase. The flow of fluid phases is computed from Darcy's law. Sources of fluid in the domain are the natural recharge, applied as a Neumann type boundary condition on the upper boundary, and artificial discharges received from VADER, applied as sources to a specified portion of the model domain (White and Oostrom 1996).

If the contaminant is transported in the gas phase, then air flow is solved using an air mass conservation equation. Phase partitioning of air mass is computed assuming equilibrium conditions. Air transport occurs by advection and diffusion-dispersion through the aqueous and gas phases. Again, flow of fluid phases (including air) is computed from Darcy's law. Transport of phase components is computed from a modified form of Fick's law, where a combined diffusion-dispersion coefficient replaces the classical Fickian diffusion coefficient (White and Oostrom 1996).

Contaminant transport is solved using a solute mass conservation equation. Solute mass is partitioned among the fluid and solid phases assuming thermodynamic and geochemical equilibrium conditions. Radioactive decay is computed using first-order decay kinetics (White and Oostrom 1996).

In contrast to preceding SAC modules that operate in calendar-year time increments, STOMP operates in variable time steps because of the need to solve the nonlinear equations describing fluid flow to meet convergence criteria. Consequently, the releases predicted by STOMP are in variable time steps that range from seconds to tens or hundreds of years.

The location of the vadose zone flow and transport calculations in the SAC processing sequence for contaminant transport is provided in Figure 3.10. This is a most complicated point in the SAC processing sequence because numerous (usually hundreds) of vadose zone sites are simultaneously processed in this part of the sequence by both VADER and STOMP acting in tandem. Figure 3.10 provides a simplified illustration of the handling of multiple sites by showing only two sites—Sites A and B in which contaminant is remediated from Site A to Site B at some point in time. The processing is similar with additional sites. SAC can accommodate any number of sites (within computer storage limitations) with or without the movement of remediated mass to or from any site so long as no site receives remediated mass that was at that same site previously. For each site, contaminant mass and liquid discharges released to the vadose zone are passed from VADER to STOMP. VADER may also pass contaminant mass to the VADER simulation of another site in response to a remediation event. At each site, STOMP simulates fully coupled fluid flow and solute transport (and, if applicable, gas phase fluid flow) and produces for a single vadose zone site the transient release of contaminant to the aquifer and, if applicable, transient release of contaminant to the atmosphere. Further, if a remediation event (or events) is declared, then the contaminant mass or activity extracted as part of remediation activity is passed back to VADER at the receiving site where it may again release to the STOMP model for that site.

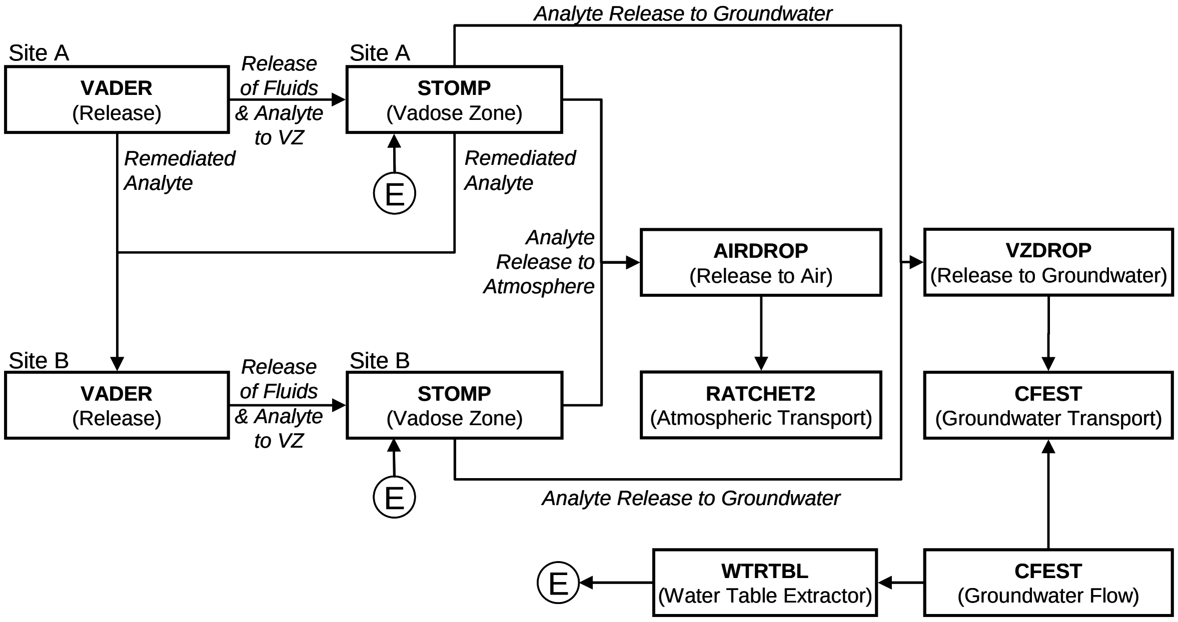


Figure 3.10. Location of Vadose Zone Flow and Transport Module in the SAC Processing Sequence for Contaminant Transport (Example for two sites: Site A subject to movement of remediated contaminant mass to Site B)

One other input required by STOMP in SAC that is evident in Figure 3.10 is the water table elevation at each vadose zone site. The water table history at every location in the groundwater flow model is passed from a previous groundwater flow solution (extracted for rapid access using the WTRTBL utility code). This information is used to direct STOMP to control aqueous phase pressure at the lower boundary to maintain the water table to the elevation predicted by the CFEST flow solution, and to cause contaminant mass to be extracted at the current water table elevation throughout the simulation.

Integration of the releases predicted by multiple independent vadose zone simulations of all waste sites into a unified input for the atmospheric transport and groundwater transport codes require additional processing. AIRDROP integrates the transient release to the atmosphere from vadose zone simulations of multiple sites for simulation with the atmospheric transport code RATCHET2. Similarly, VZDROP integrates the transient release to the aquifer from vadose zone simulations of multiple sites for input to the groundwater transport code CFEST. These "DROP" modules are currently configured for the identified transport models (RATCHET2 and CFEST) but could be reconfigured to support the same function for different transport codes.

The vadose zone flow and transport module also can process liquid discharges to support calibration of a groundwater model by producing a simulation of artificial discharge to the aquifer that includes vadose zone attenuation. Figure 3.11 illustrates the use of the vadose zone flow and transport module for this purpose for two sites (A and B); note that the same process applies for any number of sites, but for simplicity only two are shown in this example. For this purpose, VADER and STOMP are run as for a contaminant transport simulation (typically for a single contaminant for speed) with output options enabled that direct the output of liquid discharges from the vadose zone to the aquifer. Because variably saturated media response to water movement is nonlinear, it is not possible to realistically simulate the artificial discharge separate from the natural infiltration of precipitation at a site. To address this, the vadose zone must be modeled twice at each site: once with only natural infiltration to obtain the *natural signal* using an "empty" inventory results file, and again with the natural infiltration and the artificial discharges using the normal, complete inventory results file (that includes artificial discharges) to obtain the *total signal*. This is illustrated in Figure 3.11, where sites A and B are simulated twice each. Simulation of the vadose zone sites culminates in two runs of the VZDROP utility to integrate the release of fluids to the vadose zone for the *natural* and *total* signals. The resulting input files for the groundwater model are both submitted to the utility code L3IDIFF, which differences the *natural* and *total* signals to produce a final groundwater flow input file containing fully transient, vadose-zone-attenuated artificial discharge to the aquifer for use in the groundwater model calibration process. This process constitutes a "feedback loop" in that the final calibrated groundwater model predicts the water table elevations that are then provided back to the vadose zone simulation. Ideally, this loop would be repeated until there is no difference in the water table elevation at any vadose zone site between successive iterations. However, in practice the groundwater calibration process is so demanding that the loop is only performed once for each planned groundwater calibration exercise.

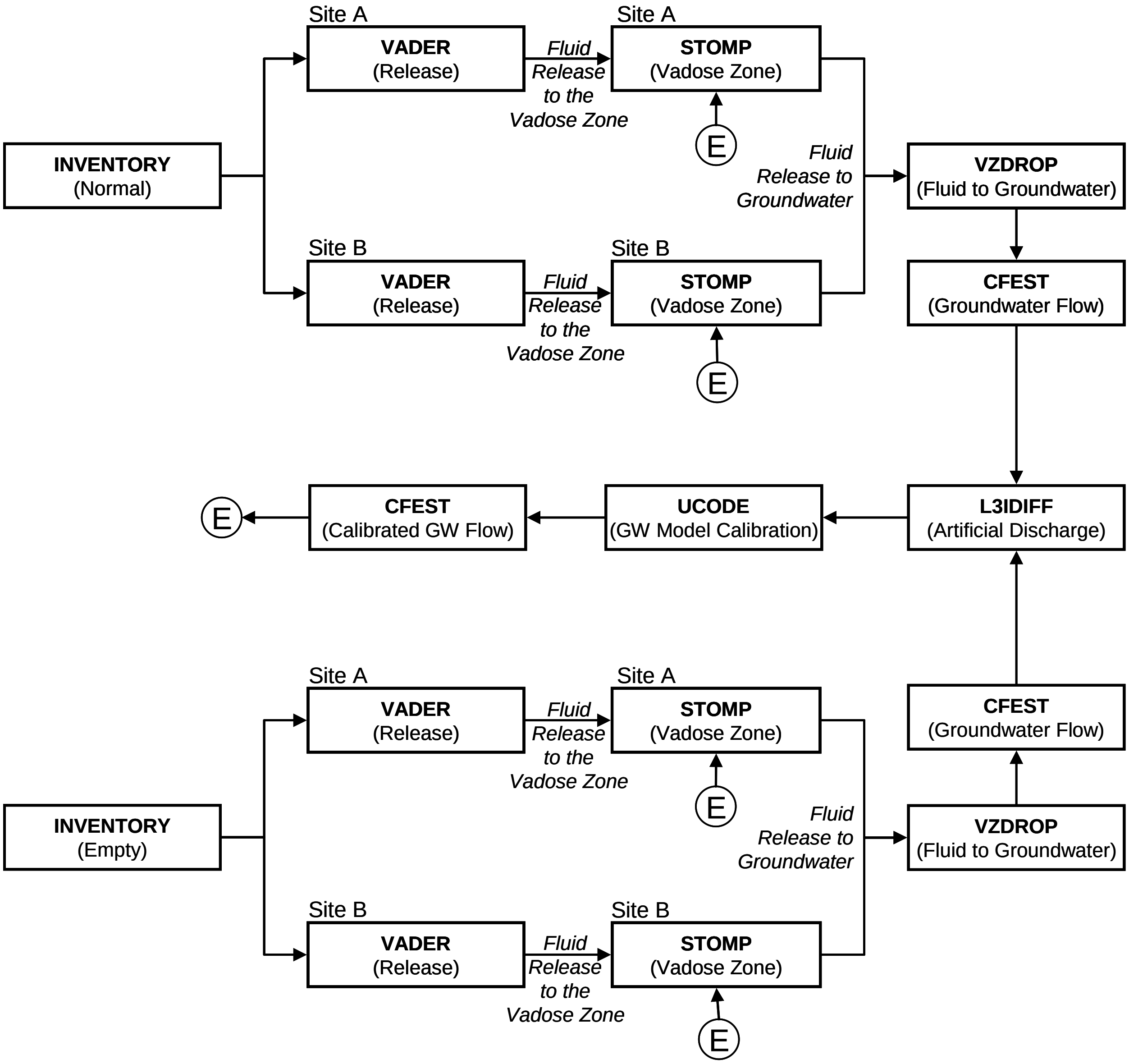


Figure 3.11. Location of Vadose Zone Flow and Transport Module in the SAC Processing Sequence for Artificial Discharge (Example for two sites: Entire system is modeled twice, once with normal inventory and once with an empty INVENTORY results file to obtain total liquid discharge to aquifer from vadose zone sites with and without artificial discharges)

The approach to using the SAC capability to obtain vadose-zone-attenuated artificial discharge for inclusion in groundwater model calibration is outlined above and summarized in Figure 3.11. In this example, SAC Rev. 1 is used for this purpose to process 1,052 sites to produce a vadose–zone-attenuated artificial discharge record for use in subsequent groundwater model calibration. Of the 1052 sites, 70 are "clean water" sites that contain no contaminant discharge, but these are included because they are sites where water was discharged to ground or pumped from the aquifer. This includes septic disposal systems, water supply wells, and water injection wells. The remaining 982 sites are identified as having potentially the disposal of at least one contaminant considered in the development of the inventory.

Tritium is the only contaminant simulated for this application, although the purpose of the simulation is to obtain water discharges, not tritium (SAC requires at least one analyte be processed).

The results for the final vadose–zone-attenuated artificial discharge record for the sum of all nodes in the groundwater model is shown in Figure 3.12, where it is contrasted with the un-attenuated artificial discharge. The time steps shown in this figure represent the predefined time steps used in the groundwater model. The difference in the two records shown in this figure represents the influence of the vadose zone that would *not* be included in subsequent contaminant transport simulations if the fluid discharges to ground were included *without* accounting for the vadose zone.

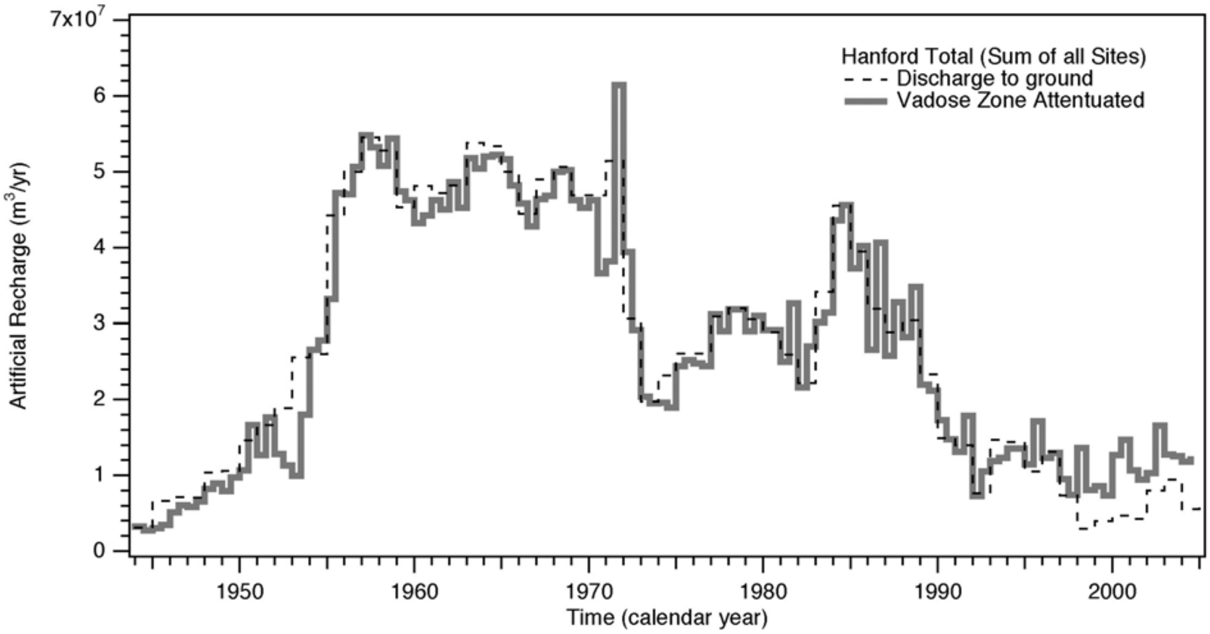


Figure 3.12. Vadose–Zone-Attenuated Artificial Discharge to the Aquifer Produced Using SAC Rev. 1 Contrasted With the Artificial Discharge to Ground (un-attenuated by the vadose zone)

An example of the application of the vadose zone flow and transport module to individual sites best illustrates the use of this module at the site scale that calculations are performed. Results from the CA_Action and CA_NoAction Assessments at site 218-W-5 are used in this illustration. These deterministic assessments contrast a scenario in which currently planned remedial activities at Hanford are carried out to another scenario in which they are not. Site 218-W-5 is a solid waste burial ground that is planned for use in disposal of decommissioned reactor cores from the production reactors in 2065 under the action scenario (but not in the no-action scenario). Because these cores arrive in the modeling sequence through a remedial action and because other waste are also assigned to this site through the inventory modeling, the Vadose Zone Release module passes to STOMP a variety of releases integrated into one source term for simulation. The site is modeled using the quasi-two-dimensional approach in which a variable wetted area is calculated to achieve a unit gradient condition at the time of maximum annual artificial liquid discharge in the least permeable soil layer at the site. The buried reactor cores will release a quantity of chlorine-36 over a long period of time.

The water flux at the upper and lower boundaries is shown for site 218-W-5 in Figure 3.13. These boundary water fluxes show the response of the vadose zone model for this location to the high operational recharge rate (44 mm/yr). For the action scenario, the infiltration rate is reduced to 0.5 mm/yr when an infiltration barrier is installed. Following the 500-year barrier design life, the infiltration rate degrades (over a period of time equal to the barrier design life) to a long-term infiltration rate of 1.0 mm/yr where it remains thereafter. In the no-action scenario, the infiltration rate is increased to a higher long-term rate after the operational period. The fluid discharge to the aquifer under each scenario reflects both infiltration and the liquid disposal.

The arrival of weakly sorbed chlorine-36 from the reactors into the vadose zone (as modeled in the Vadose Zone Release module with the VADER code) and subsequent release to the aquifer (modeled in the Vadose Zone Flow and Transport module by the STOMP code) is shown in Figure 3.14. The impact of the different infiltration histories under each scenario is very pronounced in this figure. Under the no-action scenario, chlorine-36 emerges into the aquifer, starting about one thousand years after site closure, and the release continues another one thousand years afterward. In contrast, under the action scenario, the infiltration barrier delays the onset of release by about 1,500 years compared to the no-action scenario. Overall, the infiltration barrier results in a later, longer release of chlorine-36 at this site with a reduced peak. Comparison of the releases under these scenarios demonstrates how the impact of the planned infiltration barrier can be quantified.

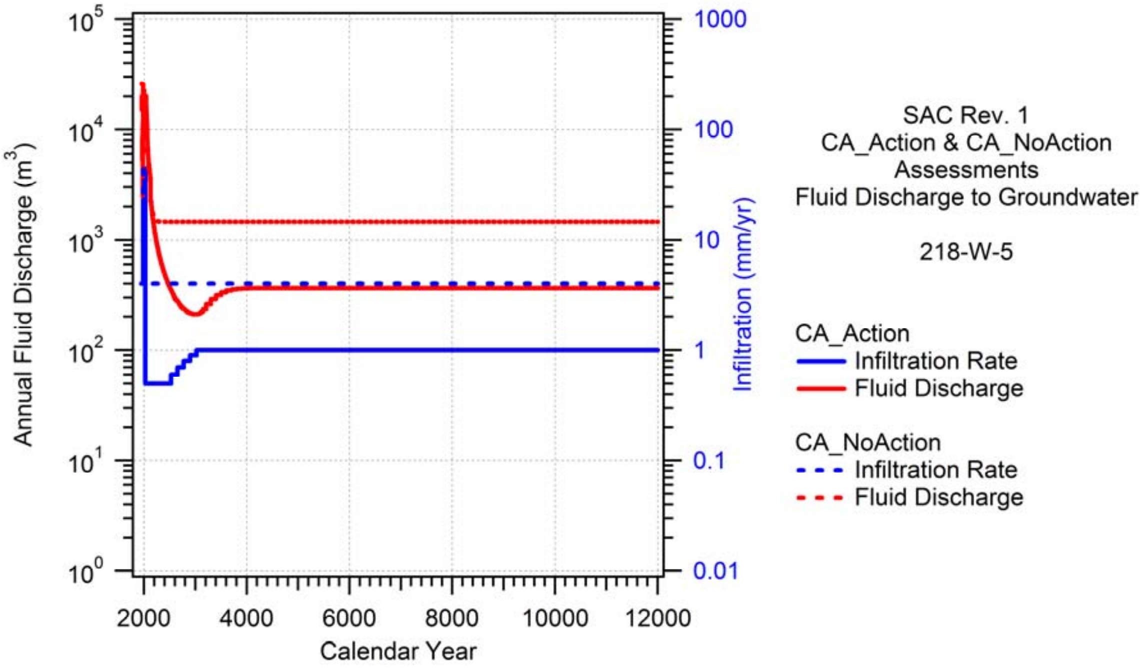


Figure 3.13. Vadose Zone Flow and Transport Module Natural Infiltration and Fluid Discharge to the Aquifer for Site 218-W-5 for Action and No-Action Scenarios

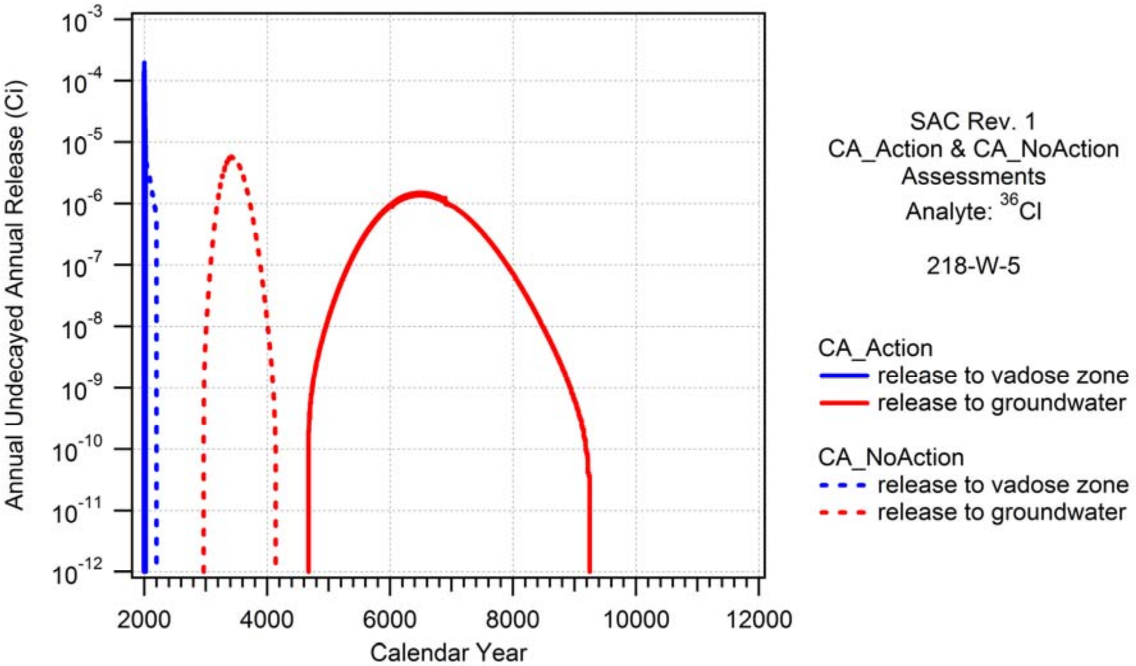


Figure 3.14. Vadose Zone Flow and Transport Module Influx and Release of Chlorine-36 for Site 218-W-5 for Action and No-Action Scenarios

The purpose of embedding the STOMP simulator inside the SAC software framework, of course, is to efficiently manage a large number of individual waste sites in the simulation. For example, waste sites are grouped into Operable Units for management of remedial activities at the Hanford Site, and so examination of the aggregated releases from the vadose zone sites that are assigned to an Operable Unit is imperative to support work in these areas. The ZP-1 Operable Unit includes 130 vadose zone waste sites. For purposes of examining aggregate releases from this Operable Unit, these sites are sorted into five main groups: facilities (3 sites), liquid disposal (56 sites), solid waste burial (11 sites), tanks and associated unplanned releases (42 sites), and other (18 sites).

Results for this example are from the CA_Ref_A, (Reference Case A) Assessment in which the vadose zone sites were modeled using the quasi-two-dimensional approach with infiltration rates set at reference case (rather than best estimate) values.

The aggregated arrival of technetium-99 in the vadose zone for the liquid disposal sites, and the aggregated release of this contaminant to the aquifer are shown in Figure 3.15. Liquid disposals are usually associated with a pulse of water that provides a significant driving force at these sites, and so the arrival at the aquifer is relatively rapid compared to dry disposal sites. This is shown by comparison with Figure 3.16, which shows the arrival of technetium-99 in the vadose zone for the solid waste disposal sites, and the aggregated release of this contaminant to the aquifer from those sites. For the solid waste sites, substantial arrival does not begin until more than two thousand years after disposal, whereas the liquid disposal sites have completely released all technetium-99 by that time.

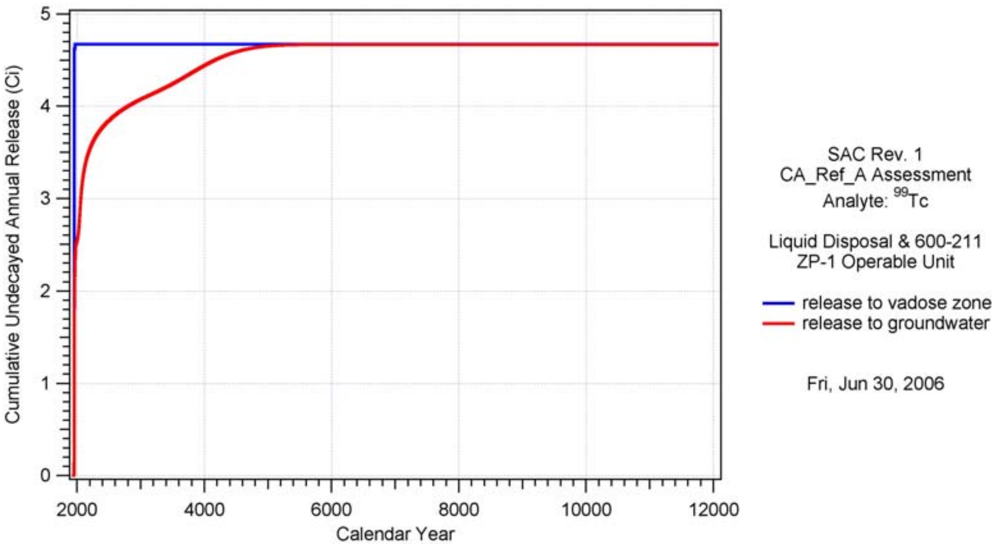


Figure 3.15. Vadose Zone Flow and Transport Module Influx and Modeled Release of Technetium-99 for Sum of Liquid Disposal Sites in the 200-ZP-1 Operable Unit

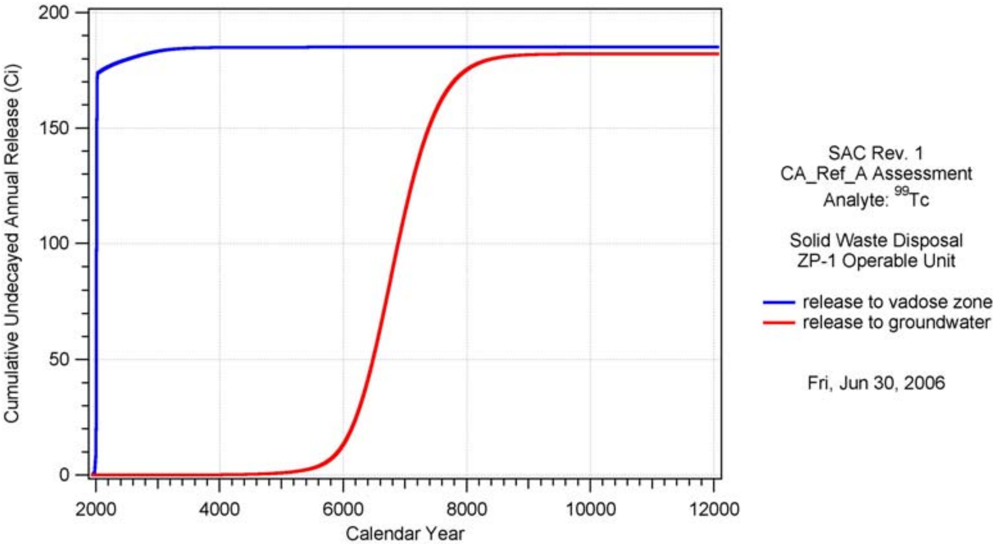


Figure 3.16. Vadose Zone Flow and Transport Module Influx and Modeled Release of Technetium-99 for Sum of Solid Waste Disposal Sites in the 200-ZP-1 Operable Unit

The quasi-two-dimensional option is the most efficient means to simulate the vadose zone in SAC Rev. 1 in terms of computational time, but at the expense of greater computer time and data requirements the vadose zone can be represented as explicitly two- or three-dimensional. This can be done either to calibrate the quasi-two-dimensional model for later use or as a part of the primary vadose zone calculations in an assessment.

The MultiDim-VZ Assessment was conducted in 2006 to test and demonstrate the multidimensional capability of SAC Rev. 1, simulating all of the BC Cribs and Trenches using idealized two-dimensional representations. The cribs, as square features, were idealized as right cylinders with an area equal to the actual area of each crib, while the trenches, as long narrow features, were represented as rectangular features. Symmetry boundaries were used at the diameter of the right cylinder for cribs and the centerline of the rectangle for trenches. SAC Rev. 1 manages scaling of releases going to and from a STOMP model to accommodate the symmetry boundary (in each case here, one half of the domain was represented in the model). The one-dimensional templates were extended to represent uniform two-dimensional soil stratigraphy for this example run. The soils are known to be anisotropic, with 10:1 anisotropy considered typical. SAC Rev. 1 has the capability to accept this ratio from input for use in the assignment of hydraulic conductivities. All other parameters, including infiltration rates, were assigned to match the "Reference Case A" values.

The results of this test for BC Crib 216-B-14 and contrast to two quasi-2D simulations for this same site are shown in Figure 3.17. One aspect this test demonstrates is that the explicit multidimensional simulation requires testing to ensure that the horizontal resolution is sufficient to simulate the release accurately; coarser resolution under predicts arrival time. Resolution should be reduced further in this test until two subsequent resolutions show no appreciable difference in contaminant arrival in the aquifer, but in this case the next step (192 radial nodes) created a vadose simulation too large to simulate with available memory on the compute nodes of the project computer (see Section 2.4). Computer simulation time also increases markedly as the nodes increase in these simulations. The release curves from explicit two-dimensional modeling are contrasted in this figure with quasi-two-dimensional results for two previous calibrations of the BC Cribs and Trenches. The current vadose zone properties and infiltration rates do not appear to allow a calibration as adequate as that accomplished in prior work, but the prior work only fitted the first thousand years of the early portion of the arrival using lower infiltration rates (best estimate rather than Reference Case A values). These newer results capture the full release by simulating to 10,000 years post-closure, and they demonstrate that explicit multidimensional modeling may be necessary for simulation of the vadose zone releases in this time range.

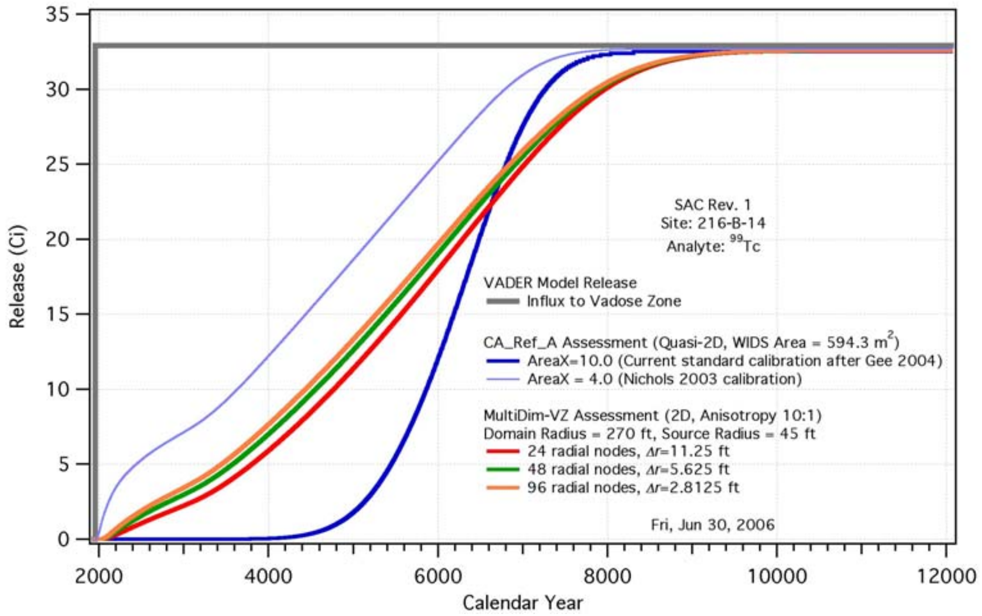


Figure 3.17. Vadose Zone Flow and Transport Module Influx and Release of Technetium-99 for BC Crib 216-B-14 Modeled as Quasi-2D With Different Calibrations and Modeled as Explicitly 2D at Various Horizontal Resolutions

3.4 Atmospheric Transport Module

Contaminants that have a gaseous phase can be released to the atmosphere where they are transported, depending on atmospheric conditions, and can result in exposure to animals and humans. The Atmospheric Transport Module calculates contaminant concentrations in the air and contaminant deposition rates for use in the SAC Soil Module and the SAC impacts modules.

Contaminants can be released to the atmosphere as either ground-level area sources (volatilized vadose zone or inventory releases from Hanford subsurface disposal sites) or elevated point sources (releases from Hanford processing plant stacks) as shown in Figure 3.18. These contaminants are then transported through the model domain using a Lagrangian trajectory, Gaussian puff atmospheric dispersion model that includes deposition and depletion. The RATCHET2 code, used to model atmospheric transport within the SAC, is described by Ramsdell and Rishel (2006). Spatial and temporal variations in meteorological data, which reflect the effects of topography, are accounted for and affect the transport of the contaminant plume from the time it is released to the atmosphere to the time it leaves the model domain. The distance and direction of contaminant transport through the atmosphere is affected by the wind speed and direction (at the surface and at the release height). Ambient air temperature as well as the temperature of the effluent affects how the plume rises as it is transported through the atmosphere. Dispersion of the contaminant plume is determined by the atmospheric mixing-layer thickness and the atmospheric stability class. The deposition of contaminants within the model domain is controlled by atmospheric conditions and surface roughness. Contaminants are considered to be one of three types for the evaluation of deposition: noble gas, iodine, or particle. Noble gases are not deposited on the ground; iodine and particles are both deposited on the ground but have different deposition characteristics. Precipitation is the mechanism for wet deposition of contaminants. Any portion of contaminant that is deposited on the ground is removed from the atmospheric plume to maintain a mass balance.

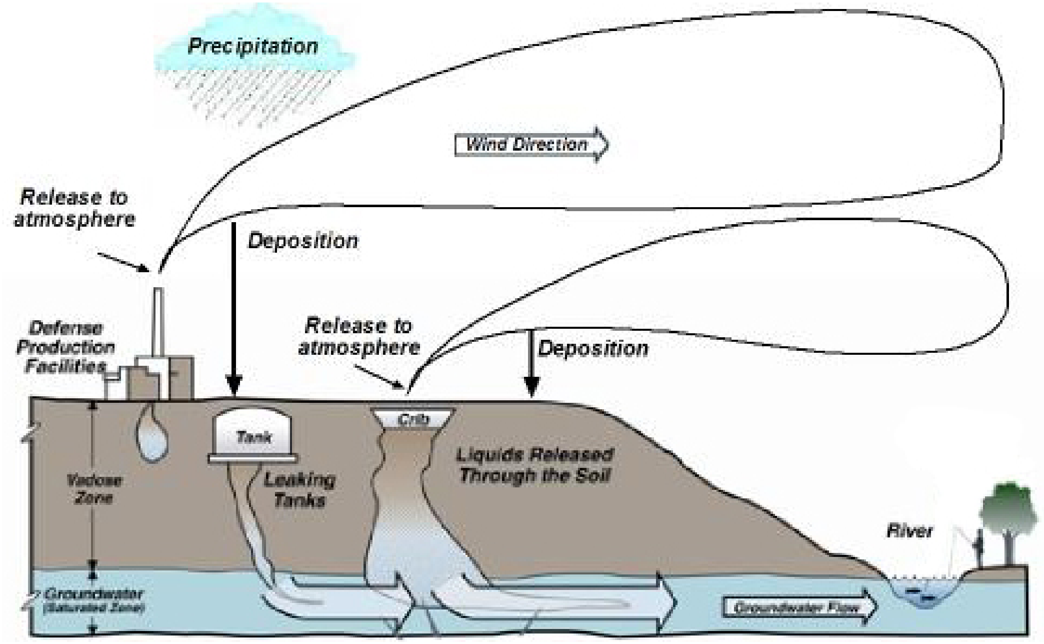


Figure 3.18. Conceptual Model of Atmospheric Transport Module and Its Relationship to Other Environmental Pathways

The Atmospheric Transport receives releases from the Vadose Zone Module and the Inventory Module. Within the SAC, atmospheric transport is modeled by simulating a unit release of an analyte for one year at all possible source locations, resulting in gridded fields of time-integrated air concentration and deposition values. These results are then scaled by the release values and interpolated to impact locations. This approach minimizes the number of simulations that need to be performed to model the many releases to the atmosphere. For stochastic simulations, a year that is within the historical meteorological record is randomly sampled for each realization, and that year's meteorological data are used for each year simulated in that realization. The model domain of the numerical atmospheric transport model used in the SAC extends beyond the boundaries of the Hanford Site.

The location of the atmospheric transport calculations in the SAC processing sequence is illustrated in Figure 3.19. As described above, the RATCHET2 code simulates the annual transport of a released source, and the AIRDROP data translator scales the results based on releases from the inventory and vadose zone modules.

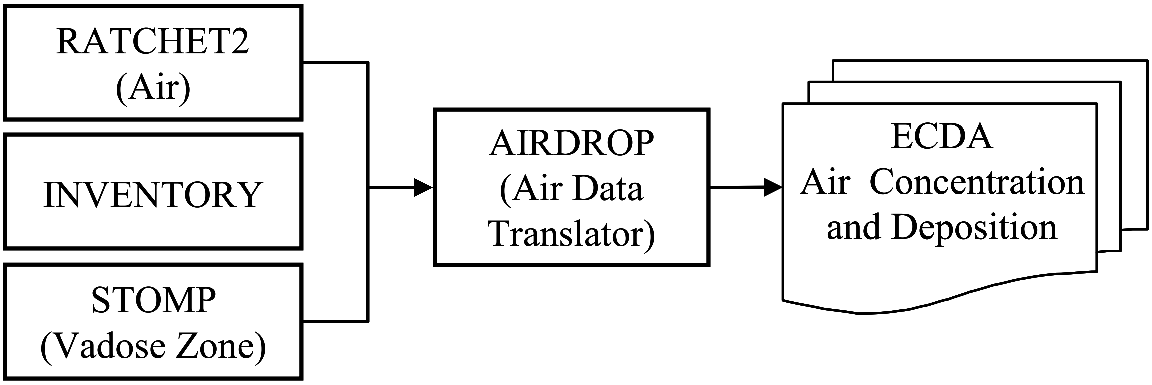


Figure 3.19. Location of Atmospheric Transport Module in the SAC Processing Sequence

Results from an example air transport simulation are provided in Figure 3.20. The left pane in the figure shows the time-integrated relative concentration (χ/Q) field (units of Ci-s/m^3) resulting from 1 Ci of iodine-129 being released at a single location. The release is assumed to occur uniformly in time across an entire calendar year, and the point of release is the physical location of the PUREX facility. The right pane of the figure shows hypothetical annual average air concentrations (pCi/m^3) in the atmosphere resulting from modeling a 0.3 Ci release of iodine-129. Meteorological data from the year 1993 were used in this example calculation. This calculation is not intended to mimic an actual historical release.

3.5 Groundwater Flow and Transport Module

Groundwater is generally defined as all subsurface water that is part of the saturated zone of an aquifer system. The Groundwater Flow and Transport Modules focus on groundwater that is in the uppermost saturated zone on the Hanford Site, commonly referred to as the unconfined aquifer, which constitutes a pathway for contaminants released through the vadose zone from past, present, and future site activities to reach the accessible environment both through well water extractions and through discharge to surface water (the Columbia River). Radioactive and hazardous chemicals have been released on the Hanford

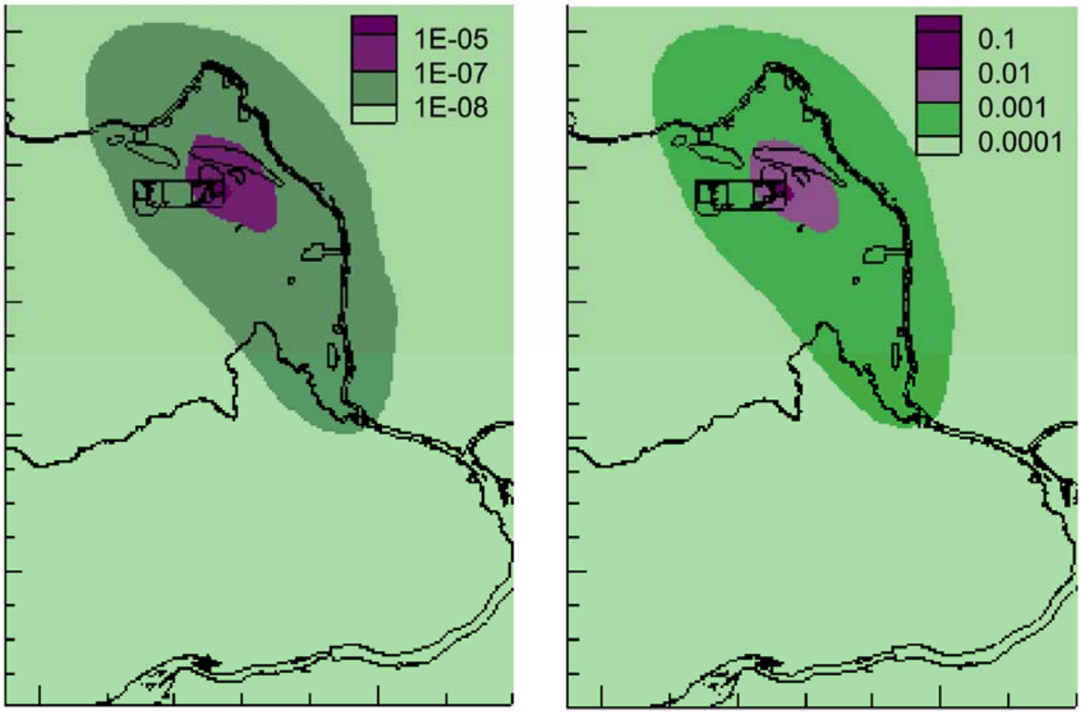


Figure 3.20. Time Integrated χ/Q (Ci-s/m^3) from a Unit Release of Iodine-129 (left pane) and Air Concentrations (pCi/m^3) of Iodine-129 (right pane)

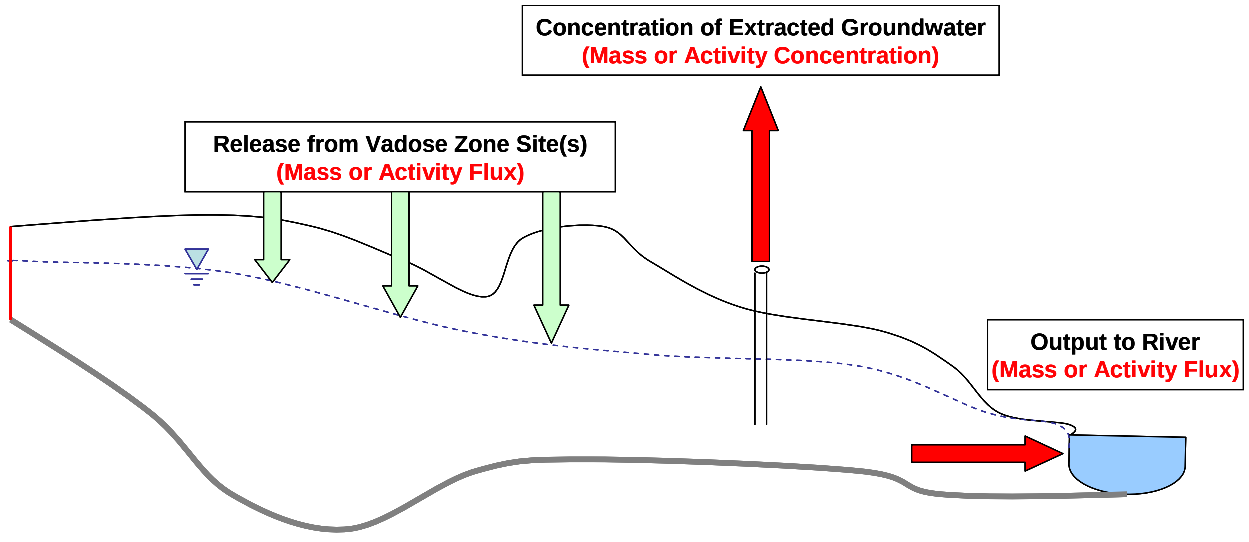
Site from a variety of sources including ponds, cribs, ditches, injection wells (referred to as reverse wells), surface spills, and tank leaks. Many of these sources have already affected the groundwater, and some will continue to affect it in the future. Once in groundwater, contaminants move through advection and diffusion along the pathways of least resistance, from higher to lower elevations where some contaminants may ultimately discharge into the Columbia River. The goal of the Groundwater Transport Module is to evaluate the transport of contaminants released from the vadose zone to points of regional discharge of groundwater along the Columbia River as well as to estimate well water concentrations at selected locations during the assessment period. Contaminants released to the groundwater form plumes, some of which extend from their source areas to the Columbia River.

Fluid flow in the saturated zone is described by a governing partial differential equation for isothermal hydrologic responses of an aquifer. One governing partial differential equation is for conservation of total liquid mass, while the other is for conservation of the mass of a dissolved contaminant. Water transport occurs by advection and dispersion through the aqueous phase. The current groundwater code used for the Groundwater Flow and Transport Modules in SAC is CFEST (Freedman et al. 2006), which solves the governing partial differential equations using the finite element method.

Groundwater Flow and Groundwater Transport are different modules in SAC because the current implementation of the inverse calibrated groundwater flow model does not provide a basis for stochastic treatment of flow parameters. The solution of the groundwater flow model also requires considerable computational effort. To increase efficiency, the flow solution is pre-computed and stored for repeated use with stochastically sampled transport properties by the Groundwater Transport Module. At such time as the information becomes available to treat groundwater flow stochastically, this approach may be changed to solve flow and transport in a fully coupled fashion.

The Groundwater Transport Module integrates releases from one or more vadose zone sites and transports them through the aquifer within a transient or steady-state flow field provided by the Groundwater Flow Module. Water flux and contaminant flux are passed to the River Flow and Transport Module, and analyte concentrations in the groundwater are passed to the impacts modules.

The Groundwater Flow Module is capable of simulating three-dimensional flow through the groundwater system in either steady-state or transient mode. The Groundwater Transport module is capable of simulating three-dimensional transport of mass released from the vadose zone through the groundwater domain and delivery to the Columbia River. Figure 3.21 shows a general conceptual model of the groundwater system. The groundwater model used in the SAC is based on a subsurface system characterized by nine hydrogeologic units, seven of which are found below the water table. The Columbia River constitutes the regional discharge for the unconfined aquifer system. The aquifer is recharged naturally from surface precipitation and from groundwater entering the model domain from the western upstream boundaries (primarily Dry and Cold Creek discharges and the Rattlesnake Springs). The spatial distribution of natural recharge across the model domain is based on climate, soil type, and vegetation (Fayer and Walters 1995). Artificial recharge to the aquifer from wastewater disposal is calculated in the Vadose Zone Module in order to account for the attenuating effects of travel through the vadose zone (see Section 3.3 for more information on this calculation).



Driving Forces

- Natural recharge at regional water table
- Flow at upstream boundaries
- Artificial discharge (into aquifer)
- River stage at downstream boundaries
- Pumping well water extractions
- Water leakage to and from the underlying basalt-confined aquifer

Pathways

- Fully saturated heterogeneous porous media composed of several geologic formations
- Extraction through pumping wells
- Release to seeps and surface water along the Columbia River

Fate & Transport Processes

- Aqueous phase
- Radioactive decay
- Geochemical processes
- Sorption (linear isotherm)

Figure 3.21. General Conceptual Model of the Groundwater System

The Groundwater Module relies on input from the Vadose Zone Flow and Transport Module and passes information to the River Flow and Transport Module and the impacts modules. Figure 3.22 shows the location of the Groundwater Transport Module within the SAC. Releases of fluid and mass from the vadose zone to the groundwater are temporally and spatially translated from the Vadose Zone Flow and Transport Module time steps to the Groundwater Module time-stepping scheme. Fluid and mass sources are also spatially interpolated between the Vadose Zone Module grid and the Groundwater Module grid. Note that the STOMP solutions indicated in this figure may represent one or several vadose zone sites. The Groundwater Transport Module is designed to handle multiple sources that can be distributed both spatially and temporally. For stochastic simulations, the retardation factor for a particular analyte may be varied.

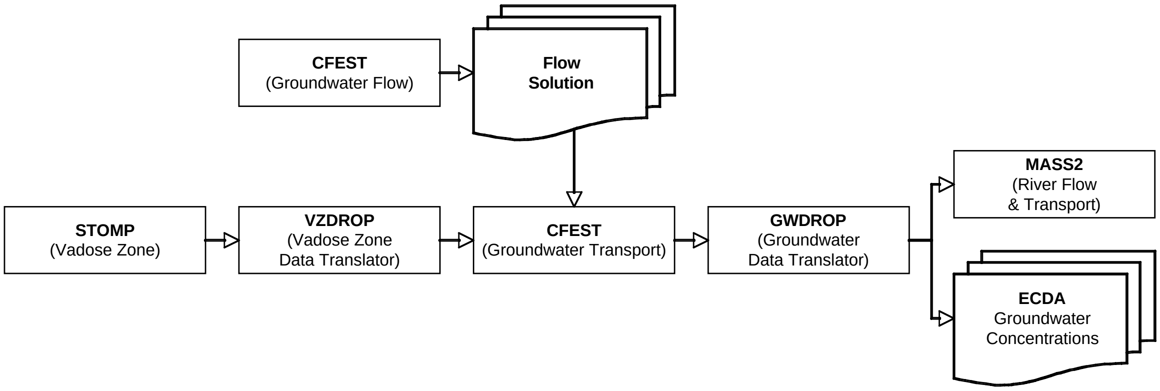


Figure 3.22. Location of Groundwater Module in the SAC Processing Sequence

An example application illustrates the capability of the Groundwater Module to model several contaminant sources that are introduced to the aquifer at different times and locations. Figure 3.23 shows the tritium distribution resulting from contaminant discharge to the aquifer from a number of waste sites. Four major source locations can be seen, expressed as individual contaminant plumes that commingle as they are transported through the groundwater system. The white cutout in the upper right portion of the figure represents locations where basalt outcrops intrude into the aquifer. This plot is intended to illustrate the capability to integrate sources from multiple locations and is not intended to portray historical concentrations of tritium in the Hanford aquifer.

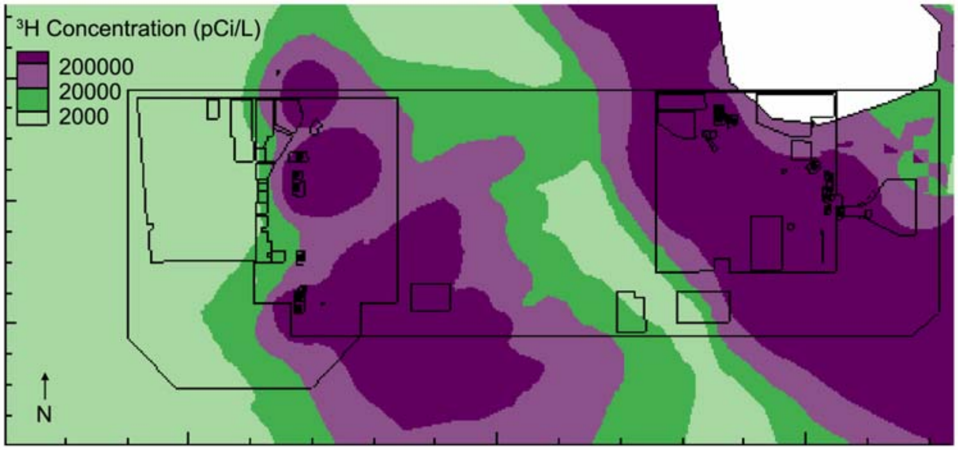


Figure 3.23. Modeled Concentration of Tritium in Groundwater from Multiple Waste Sites

3.6 River Flow and Transport Module

The River Module is used to simulate water movement and contaminant transport in the Columbia River. Contaminants modeled in the river come from three sources:

- contaminants already in the river when water reaches the Vernita Bridge from upstream sources and atmospheric fallout
- contaminant influx from Hanford waste sites entering the river through groundwater
- direct discharge to the river from Hanford facilities.

Transported sediments and river bottom sediments are also modeled in addition to the flow of water and transport of contaminants in the river. The River Module receives fluid and mass flux from the groundwater and passes contaminant concentrations in river water and sediment to the impacts modules.

The River Module uses a two-dimensional, depth-averaged hydrodynamics model capable of simulating the lateral (bank-to-bank) variation of flow and transport of sediments and contaminants. Figure 3.24 illustrates the processes modeled within the River Module. River hydraulics (velocity and river depth), influx of water and contaminants from groundwater and point sources, and contaminant and sediment transport are simulated. The capability to simulate interaction between river sediments and contaminants (adsorption/desorption) is also included. The River Module accounts for background contamination, or contaminants that have been introduced to the river upstream of the model domain used within the SAC. Direct discharges to the river from facilities can also be modeled.

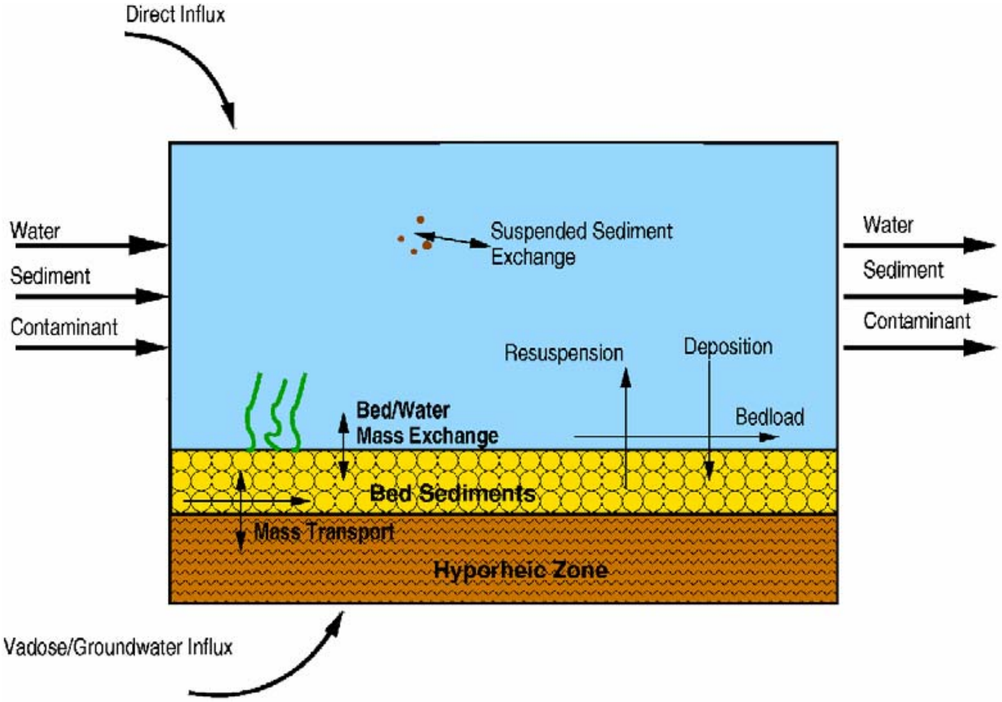


Figure 3.24. Conceptual Model of the River Module

The River Module accepts input from the Release Module and the Groundwater Module and passes information to the Soil Module and impacts modules. Figure 3.25 shows the location of the River Module within the SAC sequence of computer codes. Fluid and mass are spatially interpolated between the computational grids for the groundwater model and the river model.

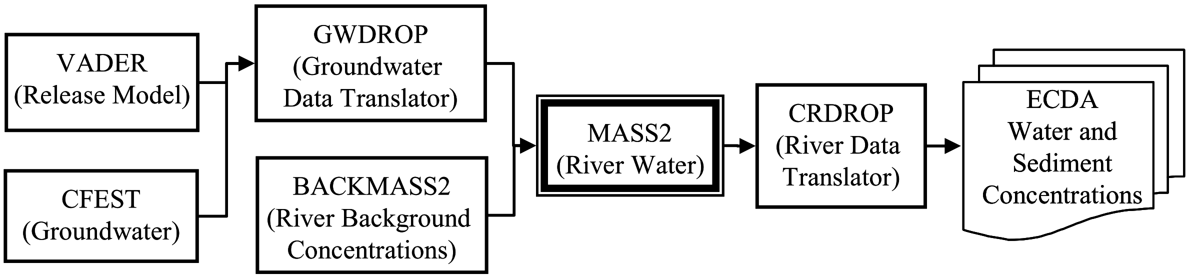


Figure 3.25. Location of the River Module within the SAC Framework

The Modular Aquatic Simulation System 2D, called MASS2 (Perkins and Richmond 2004a, 2004b), includes the capability to simulate sediment transport, sediment-contaminant partitioning (using K_d), sediment-sorbed contaminant transport, and contaminant transport within the riverbed sediment layer. The model is an unsteady finite-volume code formulated using the general principles described by Patankar (1980). The model uses a structured multiblock scheme on an orthogonal curvilinear grid system. The momentum and mass conservation (continuity) equations are coupled using a variation of Patankar's (1980) SIMPLE algorithm extended to shallow-water flows by Zhou (1995). Spasojevic and Holly (1990) give an example of a two-dimensional model of this type. The governing equations are formulated in a conservation form using a full transformation in the curvilinear system (Richmond et al. 1986). The governing equation for the transport of a contaminant is obtained by applying the principle of conservation of mass to a fluid element. The model is coded in standard FORTRAN 95 and runs under the Windows, Unix, and Linux operating systems.

To numerically solve the system of governing equations, initial and boundary conditions must be specified. Initial conditions for each dependent variable (velocity, depth, and species) are assigned at the start of each simulation, either as approximate values or using the results of a previous simulation (that is, a hotstart or restart file). At the upstream boundary, the incoming velocity or discharge is specified as a function of time for each cell, and depth is extrapolated from the nearest interior cell. At the downstream boundary, the depth for each cell is specified as a function of time, and zero-gradient conditions are assigned for the velocity. Along the shoreline, a zero gradient or slip condition is applied to the longitudinal velocity component, and the normal velocity to the shore is set to zero. The depth is extrapolated from the nearest interior cell to the shore.

A substantial amount of river bed survey and river discharge data are required to run a realistic river simulation. Data have been collected for the Columbia River and are documented by Rakowski et al. (2006). An example application shown in Figure 3.26 illustrates the capability of the River Module to accept contaminant concentrations from the Groundwater Module and transport them through the river while simultaneously considering upstream sources of contamination. Figure 3.26 shows modeled concentrations (Ci/m^3) of technetium-99 in river bottom pore water (left pane) and in the free-flowing river water (right pane) in the year 1995 for a stretch of the river where contamination from groundwater is released into the river bottom pore water. Contaminated groundwater intersects the river on the shore nearest the bottom of the figure. The direction of river flow is from left to right in this figure.

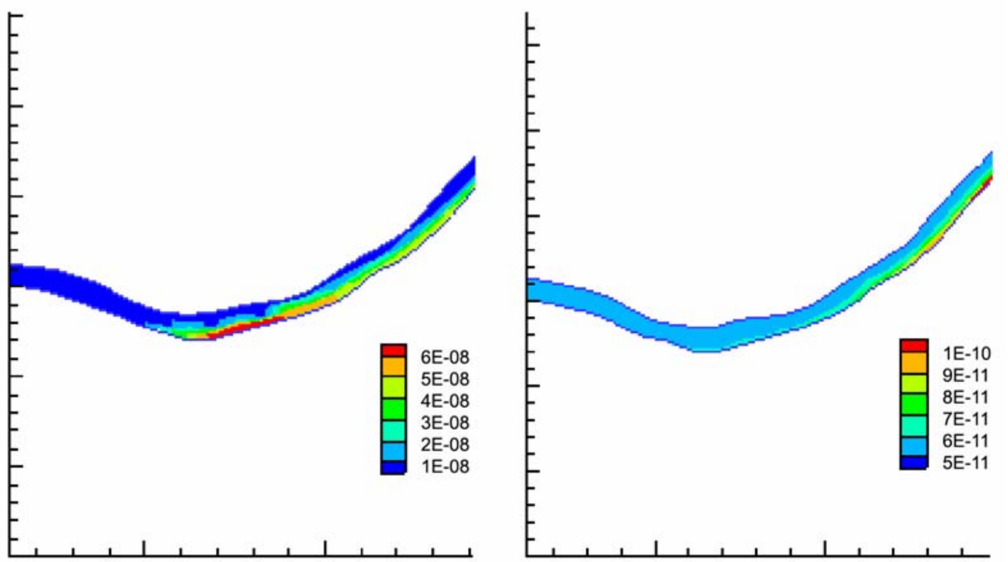


Figure 3.26. Concentration (Ci/m3) of Technetium-99 in River Bottom Pore Water (left pane) and in the River Water (right pane) in the Year 1995 for a Stretch of the River Where Contamination from Groundwater is Released into the River Bottom Pore Water

3.7 Riparian Zone Module

The riparian zone is the vegetated corridor of land adjacent to the river where there is significant interaction between groundwater and river water. It is an area of transition between aquatic and upland ecosystems. The Columbia River is the primary discharge area for the unconfined aquifer underlying the Hanford Site (Hartman et al. 2006), and the contaminated groundwater discharges into the Columbia River through surface and subsurface seeps. The purpose of the Riparian Zone Module is to calculate the concentrations in riparian zone seep water and the associated wetted soil.

The water levels in the Columbia River in the vicinity of Hanford vary seasonally (because of precipitation and runoff) and daily (because of the operation of the Priest Rapids Dam upstream of the Hanford Site). Water flows into the river bank as the river stage rises, and water flows from the bank into the river when the river stage declines. Water discharged from the seeps following a river stage decline consists of a mix of river water and groundwater. The percentage of groundwater in the seep water increases over time following a drop in river stage. Figure 3.27 shows a diagram of the riparian zone conceptual model.

The location of the Riparian Zone Module in the SAC sequence of calculations is shown in Figure 3.28. The Riparian Zone Module runs only after completion of the Groundwater and River Modules.

The Riparian Zone Module relies on input from the Groundwater and River Modules. The input includes the spatial and temporal distribution of contaminant concentrations in the groundwater and surface water. These input data are annual, time-averaged, concentrations, so seasonal and daily changes in river stage are not reflected in the seep and riverbank soil concentrations calculated by Riparian Zone Module.

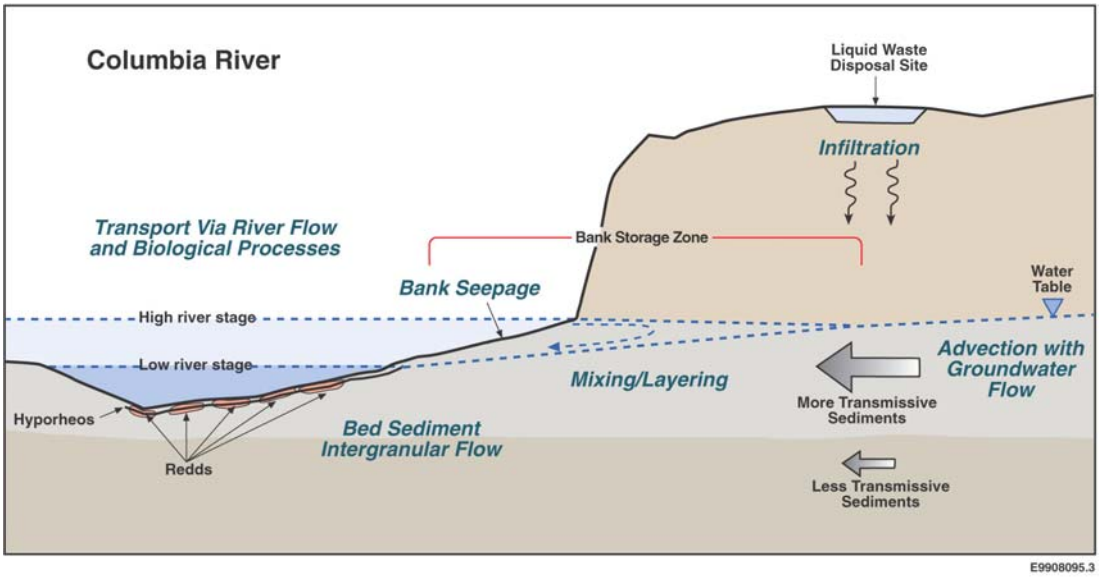


Figure 3.27. Conceptual Model for the Riparian Zone Module

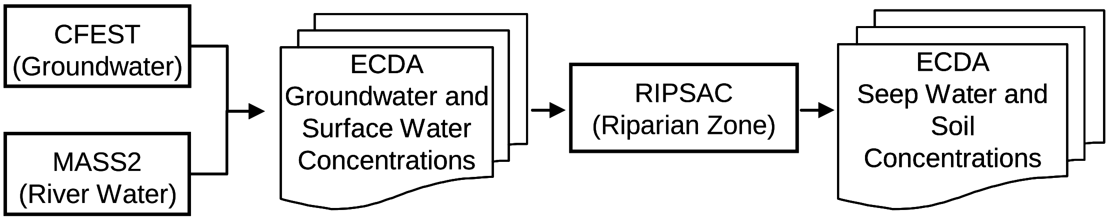


Figure 3.28. Location of the Riparian Module in the SAC Processing Sequence

The numerical model RIPSAC was written for the SAC to model the groundwater/river water interface. Seep water and riparian soil calculations are calculated only for user-specified locations and times. The model uses a stochastic mixing fraction to calculate the seep water concentration as a linear combination of groundwater and river water. The riparian soil concentration is the product of the calculated seep concentration and a stochastic K_d value.

Background information related to development of a dilution factor for seeps near the river edge is provided by Peterson and Connelly (2001). Stochastic descriptions of the K_d values for a number of major Hanford contaminants are provided by Last et al. (2006a).

The Riparian Zone module is structured to provide stochastic solutions as well as deterministic solutions. A stochastic simulation case described in Section 4.2 addresses the issue of uncertainty propagation in the context of both sampling data and field measurements of contaminant movement. Stochastic estimates of seep concentrations for technetium-99 along the Columbia River from above the 100 B/C Area to just below the 100 D area are shown in Figure 3.29 for a for a 100-realization simulation. Although the seep concentrations in the figure are calculated for the year 2000, they are not intended to represent a definitive assessment of the historical expression of technetium-99 in seep water in this region of the Hanford Site. The uncertainty in the seep concentrations results from the propagation through the modeling system of the uncertainty in inventory, vadose zone hydraulic and transport models, sorption in the groundwater model and riparian zone, and dilution with river water.

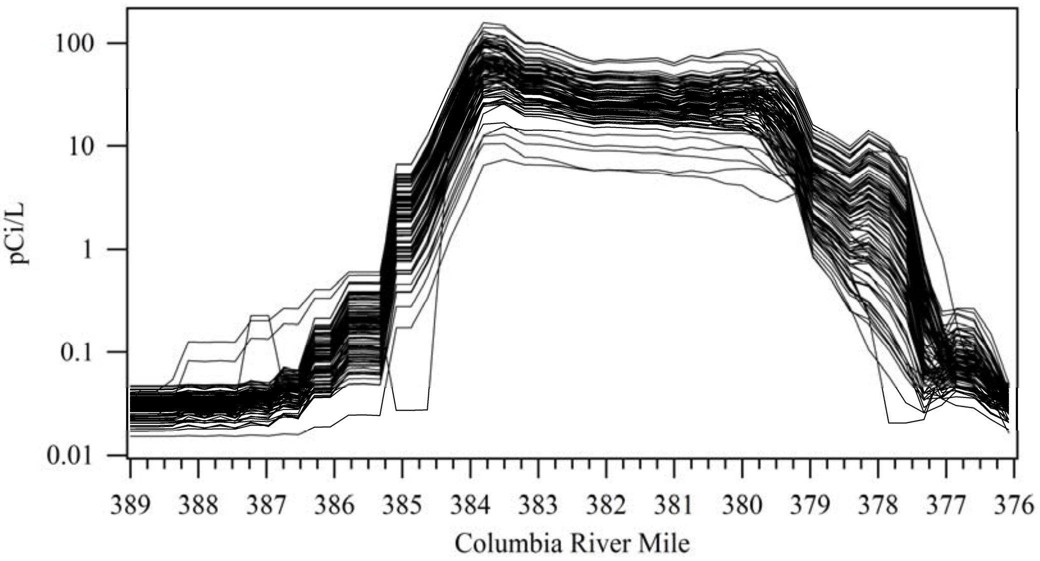


Figure 3.29. Estimated Concentrations (100 Stochastic Realizations) of Technetium-99 in Seep Water in the Riparian Zone from Above the 100 B/C Area to Below the 100 D Area in 2000

The seep concentrations shown in Figure 3.29 are based on a run that represents background concentrations of technetium-99 in the river in a stochastic manner. The band of curves in the plot near the left axis illustrates the effect of the uncertainty in the background concentration estimate for technetium-99 as river water interacts with clean groundwater to form seep water. Further downstream a low concentration plume of technetium-99 intersects the river in this model run, resulting in elevated concentrations of technetium-99 in the seep water. The drinking water standard for technetium-99 is 900 pCi/L (EPA 1976).

Estimates of the concentration of technetium-99 in the wetted soil near a seep for a modeled location near Columbia River mile 383 are shown in Figure 3.30. The data in this plot are from the same code runs as the data in Figure 3.29. Although it is not apparent from the plot, the stochastic distribution for the technetium-99 K_d value is assigned a value of zero in 50% of the runs, resulting in a soil concentration estimate of zero in 50% of the runs.

3.8 Soil Concentration Module

Contamination of the upper soil layer can result in direct exposures of animals and humans as well as serve as the basis for contaminants entering the food chain. The Soil Module calculates time-dependent concentrations in the surface soil layer for use in the SAC impacts modules.

The soil concentration model considers the effects of air deposition, precipitation and evaporation, irrigation with contaminated water, leaching from soil due to precipitation and irrigation, and radioactive decay. These processes are incorporated into a differential equation that tracks the change of the amount of the contaminant entrained in the upper soil layer as a function of time. The governing equations are provided by Eslinger et al. (2006a) and further details of the model are documented by Miley et al. (2006). The conceptual processes for the soil model are shown in Figure 3.31.

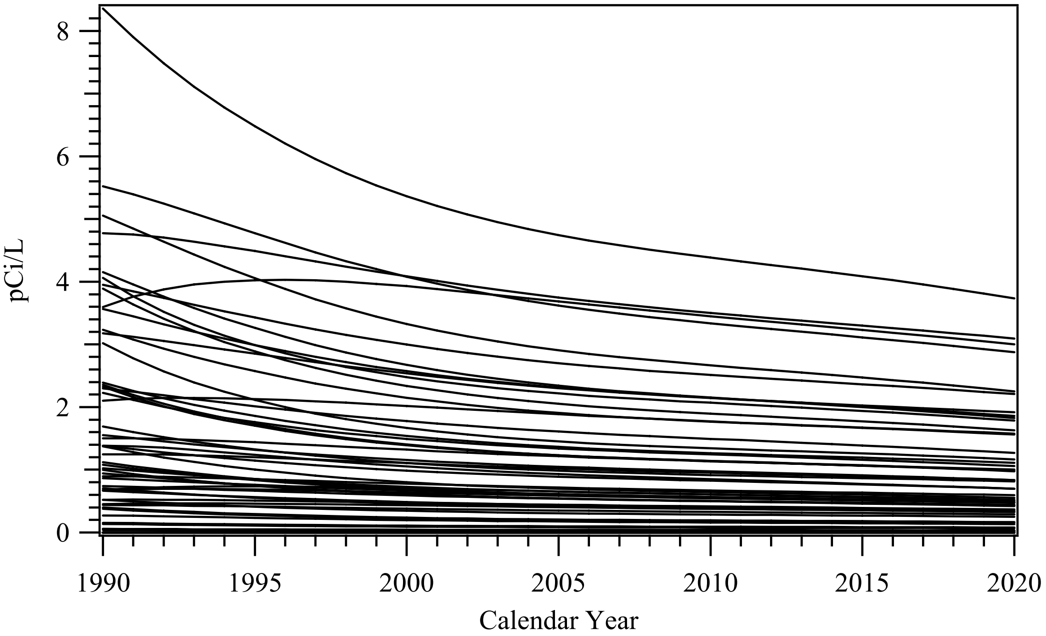


Figure 3.30. Modeled Concentrations (100 Stochastic Realizations) of Technetium-99 in Riparian Soil Near Columbia River Mile 383

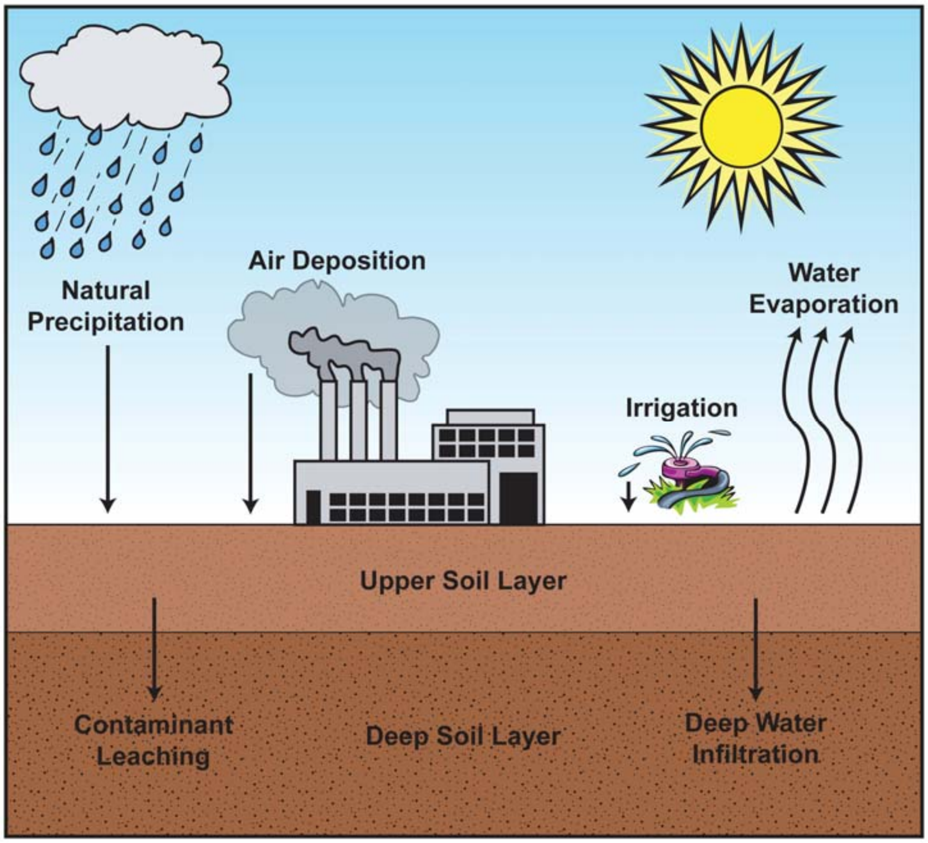


Figure 3.31. Conceptual Model for the Upper Soil Layer Module

The numerical model calculates soil concentrations under three different water use assumptions. One assumption is that no irrigation occurs and the source of water for vegetation growth, if any, is only natural precipitation. In this case, the only contamination source is airborne contaminants that deposit directly on the ground. In addition, there are two irrigation use patterns. The irrigation water source can be contaminated local groundwater, or the irrigation water source can be surface water from the Columbia River, which may contain contaminants, depending on the location of the contamination sources and the type of transport assumptions made in the modeling scenario.

The Soil Module relies on input from the Air, Groundwater, and River Modules. The input includes the spatial and temporal distribution of contaminant concentrations in the groundwater and surface water, and concentrations and deposition rates for airborne contaminants. These input data are annual, time-averaged values, and the output soil concentrations represent the end of the growing season. Figure 3.32 shows the location of the Soil Module in the SAC sequence of calculations. The Soil Module is the final environmental transport module, and it runs only after completion of the Air, Groundwater, and River Modules. Following the completion of the run(s) for the implementation model, the media concentration data set for the Impact Modules is complete.

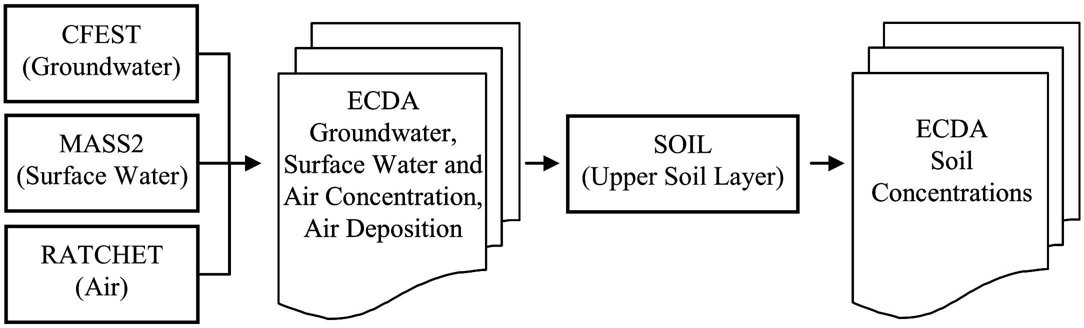


Figure 3.32. Location of the Soil Module in the SAC Processing Sequence

Much of the water applied as irrigation evaporates from the soil or is lost through transpiration from the plant cover. The net fraction of irrigation water that is available to infiltrate deeper than 15 cm in the soil profile is needed in the model. A net infiltration fraction of 0.2 (unitless) is suggested, although this value can vary substantially, depending on soil type, the amount of vegetation cover, and the amount of irrigation water that is applied. The upper soil layer is assumed to be 15 cm in depth. This depth matches with the presumed depth of the contaminated area in the “infinite slab” model (EPA 1993) used for external radiation exposure calculations.

The irrigation model uses one description of irrigation and applies it to all locations and all crops within a single run of the code. The soil code can be run on a subset of locations; thus multiple runs of the code can be used to build varying assumptions for irrigation and crop types for different physical locations. The growing season is assumed to coincide with the irrigation season and is defined by a start day and an end day. A nominal starting day for the irrigation season is the first day of May, and a nominal ending day for the irrigation season is the middle of September, yielding an irrigation season of 136 days. The same irrigation water application rate is applied to all crops. A value of 76 cm/yr is recommended by the Washington Department of Health (WDOH 1997), and this value corresponds to EPA guidance for Region 10 (EPA 1991).

Net surface soil infiltration rates are defined by Last et al. (2006, Sec 4.5.1) for a number of soil and vegetation types. Each location in the soil model must have a soil type associated with it, thereby defining a link to the appropriate net infiltration rate from natural sources.

An example run was made to illustrate the solutions that can be obtained from the soil model. This example used three radioactive analytes with long half-lives that were assigned soil-to-water distribution coefficients (K_d) of 0, 0.2, and 0.8 mL/g. The run started with a clean site in the year 2000, and irrigation with contaminated water was assumed to begin at the site in the year 2001. All water concentrations were set to 1 pCi/L for the entire time period, and the air deposition of all three analytes was set to zero. The buildup of the contaminants in the soil after the onset of irrigation is shown in Figure 3.33. As expected, higher K_d values lead to higher soil concentrations, and they also require longer times before an equilibrium concentration is reached in the soil layer.

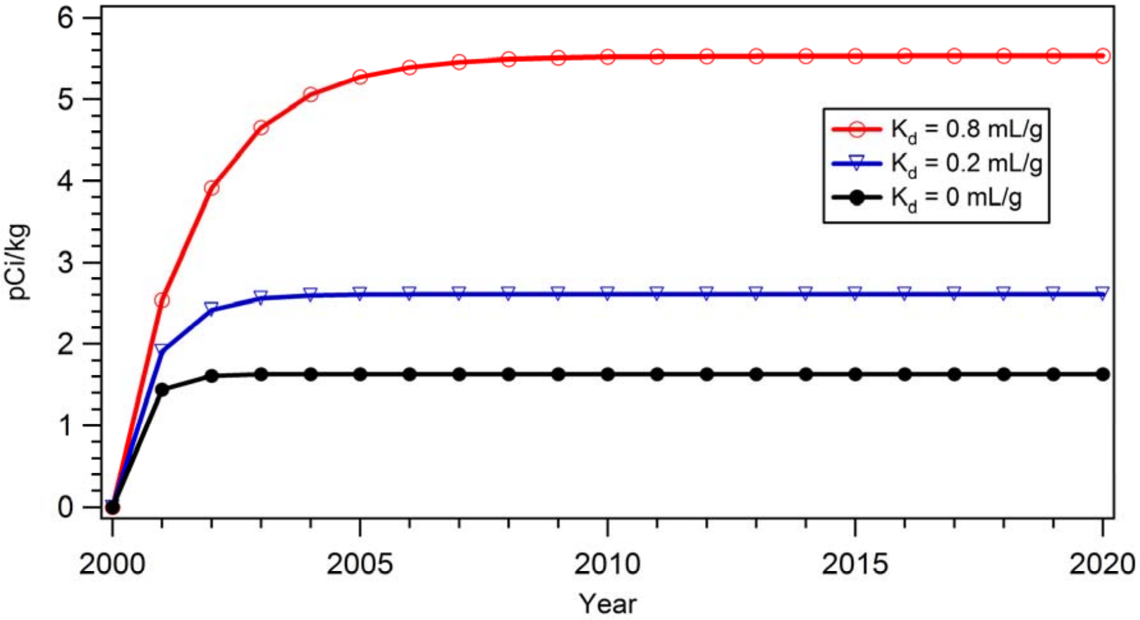


Figure 3.33. Modeled Soil Concentrations After the Onset of Irrigation as a Function of Sorption

A similar example run was made to illustrate the effect of air deposition on soil concentrations using only the analyte with a K_d of 0.2 mL/g. The air deposition rate was set to 0.1 pCi/m^2/yr, and the water concentration was again set to 1 pCi/L. These values remained unchanged for the entire time period. The run started with a clean site, and air deposition started in the year 2000, while the start of irrigation was delayed until 2001. One of the three curves in Figure 3.34 illustrates the soil concentration achieved when the air deposition term is the only contaminant and no irrigation is performed. The other two curves show the soil concentrations when the irrigation occurs with and without air deposition. As expected, the 760 pCi/ m^2/yr of contaminant applied in the irrigation water source dominates the 0.1 pCi/m^2/yr of contaminant applied from the air deposition term.

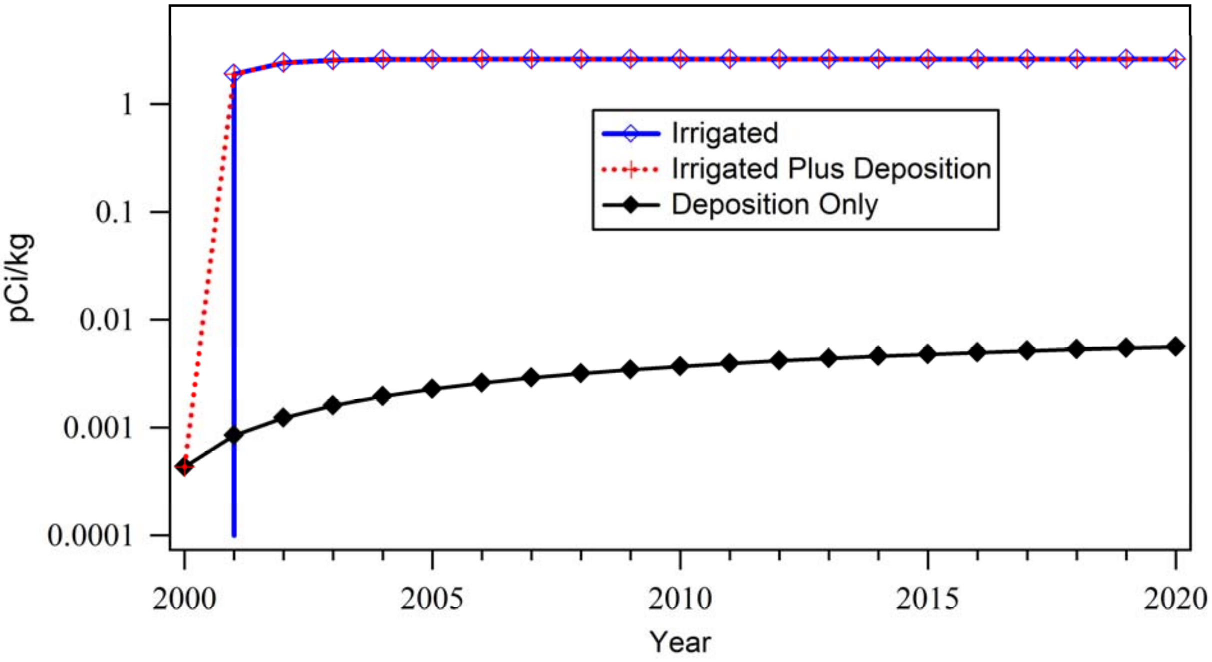


Figure 3.34. Effect of Air Deposition on Modeled Soil Concentrations After the Onset of Irrigation Using a Sorption of 0.2 mL/g

The soil module is structured to provide stochastic solutions as well as deterministic solutions. A stochastic simulation case described more fully in Section 4.2 addressed the issue of uncertainty propagation in the context of both sampling data and field measurements of contaminant movement. The soil concentration values of iodine-129 for a 100-realization simulation shown in Figure 3.35 were calculated under a hypothetical assumption that irrigation using contaminated groundwater started in the year 2005. The physical location associated with these hypothetical soil results is just outside the southeast corner of the core zone near the centerline of an iodine-129 groundwater plume originating in the 200 E area. The uncertainty in the output results from the propagation of the uncertainty in inventory, vadose zone hydraulic and transport models, and sorption in the groundwater system through the modeling system.

One of the powerful features of the SAC is the ability to analyze the effect of uncertainties in model parameters on the output performance measures. For example, one may inquire about the major sources of uncertainty in the soil concentration curves shown in Figure 3.35. A scatter plot of soil concentration against surface soil K_d value for the year 2010 is shown in Figure 3.36. The variability in the K_d values explains more than 70% of the variability in the soil concentration.

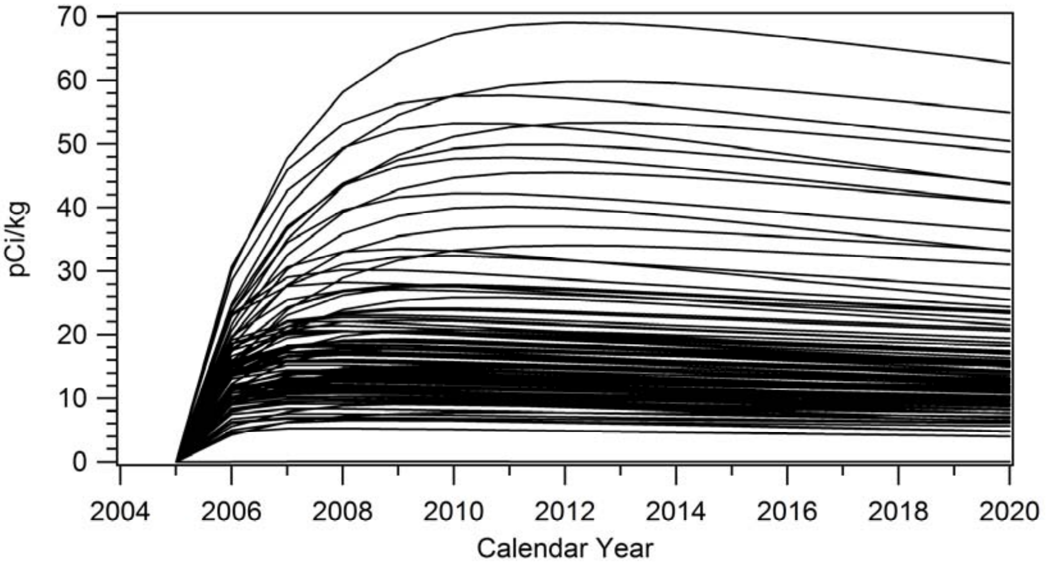


Figure 3.35. Stochastic Predictions (100 Realizations) of Soil Concentrations of Iodine-129 Near the Core Zone Boundary for a Hypothetical Irrigation Scenario Assuming Irrigation Using Groundwater Started in 2005

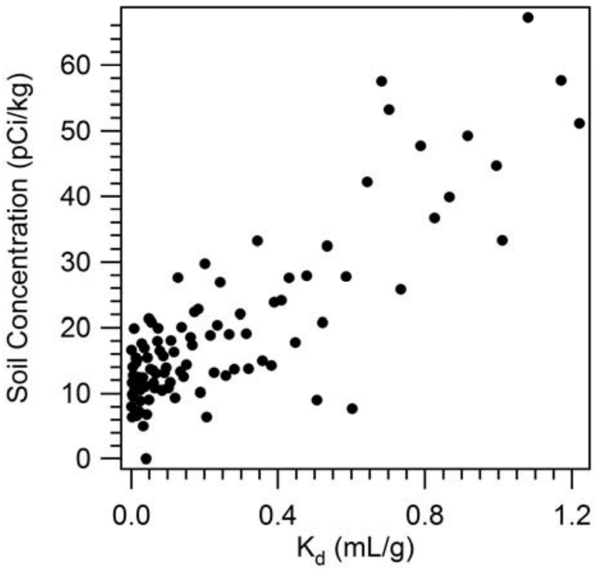


Figure 3.36. Scatterplot of Soil Concentrations of Iodine-129 in 2010 Versus Surface Soil Sorption Values for a Hypothetical Irrigation Scenario

3.9 Ecological Impacts Module

The conceptual model for assessing ecological risk or impacts has two parts: quantifying exposure to contaminants and translating exposure into effect. Organisms in the Columbia River may be exposed to contaminants in surface water, sediment, or pore water, and organisms on the adjacent land may be exposed to contaminants in the air, soil, and groundwater used for irrigation or discharging in seeps along the river shore (Figure 3.37).

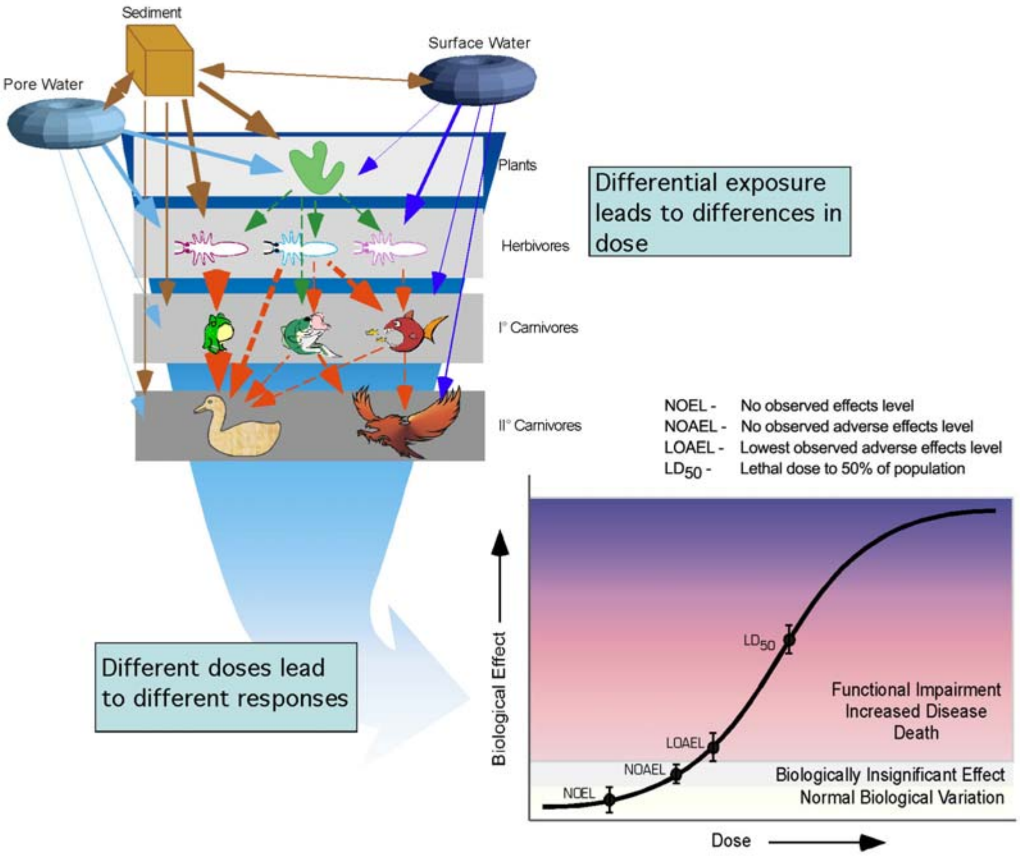


Figure 3.37. General Conceptual Model for Biological Exposure and Effects in the Columbia River System (Arrows show contaminant movement between compartments in the exposure model.)

Estimates of contaminant concentrations in these media are generated by other components of the SAC. Organisms can include plants, herbivores, and carnivores or omnivores (consumers of both flesh and plant material) in terrestrial and aquatic environments. An organism's exposure to contaminated food, sediment, surface water, or pore water is a function of its way of life. Exposures for all animals are estimated at either upland locations or at paired aquatic and riparian locations. All the species in the model for a given type of location (aquatic, riparian or upland) are assumed to occur at each location. This allows a complete estimation of risk from contaminant exposure for all species, even though some species may actually avoid contaminated areas. The model assumes equilibrium conditions (exposures are estimated based on the assumption that the organism spends enough time in a given location that the concentration of contaminants in its tissue is in equilibrium with the environment: there is no net gain or loss of contaminant from the organism).

The numerical implementation of the ecological risk model is called the Ecological Contaminant Exposure Model or ECEM. The numerical model runs after all environmental media concentrations have been calculated (see Figure 2.1). The location of the ECEM model in the SAC processing sequence is also illustrated in Figure 3.38. The detailed algorithms in the model and user instructions are documented by Eslinger et al. (2006b). Hanford-specific data for a species food web with 134 species and 15 radioactive analytes has been collected and is documented in a Hanford risk data package (Miley et al. 2006).

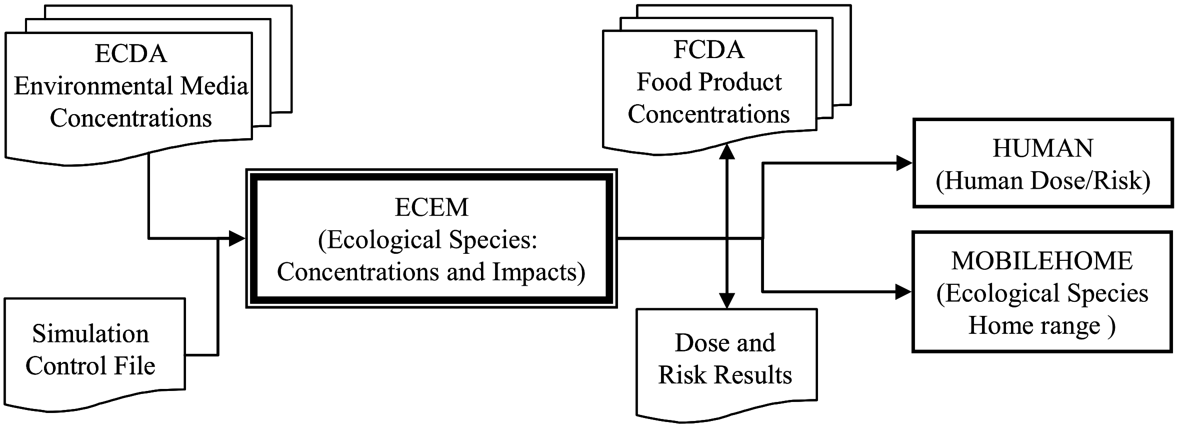


Figure 3.38. Location of the ECEM Module in the SAC Processing Sequence

Given a complete set of contaminant concentrations in air, surface water, and groundwater, along with the associated contaminant concentrations in seeps, soils, and sediments, the ECEM code calculates contaminant concentrations in aquatic foods, terrestrial crops, and animal products. Exposure pathways explicitly modeled include external irradiation, dermal contact, inhalation, and ingestion via a food web. Organisms include plants, herbivores, and carnivores or omnivores (consumers of both flesh and plant material) in upland, terrestrial, and aquatic environments. An organism's exposure to contaminated food, sediment, surface water, or pore water is a function of its life style. Algorithms in the ecological risk model are provided for radioactive contaminants; non-radioactive but carcinogenic contaminants; and non-radioactive, non-carcinogenic but still hazardous contaminants. Radiation impact to an individual organism is calculated as radiation dose; carcinogenic chemical impact is calculated as body burden or dose; risks from hazardous chemicals are provided as body burden, dose, or Environmental Hazard Quotient—the dimensionless ratio of the estimated intake to a standard Reference Dose.

The ECEM code is designed to accept multiple realizations of concentration of contaminants in the environment. It allows the definition of stochastic exposure parameters, which combine with the uncertainty in the input media concentrations to provide a full range of uncertainty on the final dose or risk to the hypothetical exposed individual organism. The code also accepts constant input data for all stochastic parameters, so a single deterministic run of the code can be made. The contaminant concentrations in food crops and animals can also be passed to the human impacts module for human consumption.

An example application of the ecological model is based on the concentrations of technetium-99 in different environmental media shown in Table 3.3. Although these concentrations are illustrative, they were obtained from a transport run modeling a low-concentration groundwater plume of technetium-99 intersecting the Columbia River in the 100 Area. The concentrations of technetium-99 in the groundwater for this example are roughly 10% of the drinking water standard for technetium-99 (EPA 1976) in 2006. The concentrations in Table 3.3 are modeled results rather than sampled results. The water-to-soil partition coefficient (K_d) was set to zero for the riparian soil, but it was set to a small positive value for the river bottom sediment because the MASS2 code cannot implement a zero value for this parameter.

Table 3.3. Example Concentrations of Technetium-99 in 2006 in Environmental Media in the Columbia River and Along the Nearby Shoreline

Location	Media	Value	Units
River (Aquatic)	Surface water	0.05910	pCi/L
River (Aquatic)	River bottom sediment	2.654E-07	pCi/kg
River (Aquatic)	River bottom pore water	0.05896	pCi/L
Riparian (Shore)	Groundwater	96.70	pCi/L
Riparian (Shore)	Seep	45.48	pCi/L
Riparian (Shore)	Riparian soil (near seep)	0	pCi/kg
Upland	Groundwater	84.13	pCi/L
Upland	Soil (irrigated with groundwater)	121.4	pCi/kg

The food web used in this example run for the ECEM code uses 134 species that are subdivided into aquatic species, terrestrial species living in the riparian zone along the river bank, and upland species that do not visit the river shore. The data provided in Table 3.4, Table 3.5, and Table 3.6 represent the highest tissue concentration of the ecological species by each division of habitat location. The grain concentration for the riparian habitat is zero because the plant is not irrigated and the roots are assumed to be too shallow to reach to the water table. The tissue concentrations in the upland plants and animals are substantially higher than plant concentrations in the riparian zone because this code run used an assumption that upland plants were irrigated using contaminated groundwater. The upland animals drank contaminated groundwater as well as consuming plants that were irrigated using contaminated groundwater.

Table 3.4. Modeled Tissue Concentrations of Technetium-99 in Selected Aquatic Species for 2006

Aquatic Plants		Aquatic Animals	
Species	Concentration (pCi/kg)	Species	Concentration (pCi/kg)
Phytoplankton	1.48E+00	Mayfly	5.53E+01
Periphyton	1.47E+00	Daphnia magna	5.28E+01
Water milfoil	2.36E-01	Rainbow trout (juvenile)	4.94E+01

Table 3.5. Modeled Tissue Concentrations of Technetium-99 in Selected Riparian Species for 2006

Terrestrial Plants		Terrestrial Animals	
Species	Concentration (pCi/kg)	Species	Concentration (pCi/kg)
Tule	3.19E-06	Milk	6.77E+01
Columbia yellowcress	3.78E-07	Egg (chicken)	6.08E+01
Grain	0.00E+00	Cliff swallow	2.18E+01

Table 3.6. Modeled Tissue Concentrations of Technetium-99 in Selected Upland Species for 2006

Terrestrial Plants		Terrestrial Animals	
Species	Concentration (pCi/kg)	Species	Concentration (pCi/kg)
Thistle	4.15E+03	Egg (chicken)	1.47E+04
Shrub	2.60E+03	Vole	1.24E+04
Fungus	2.40E+03	Finch	1.09E+04

This example run also examined the time varying concentrations in plant and animal tissue concentrations for the same locations as the data provided in Table 3.3. These time-varying tissue concentrations are provided in Figure 3.39. The species definitions and some observations are as follows:

- QSAMAD: Adult salmon are a migratory species that do not consume food while they are transiting the Hanford Reach. Thus their tissue concentration of technetium-99 is essentially zero.
- QSBASS: The smallmouth bass consumes periphyton, mayfly, and water milfoil in this example food web with 60% of the intake coming from periphyton. As seen for those food sources in Table 3.4, technetium-99 will accumulates in the species, resulting in a body burden for the bass.
- QMFOL: Water milfoil accumulates a low concentration of technetium-99 (see Table 3.4), but the concentration is much lower than for some other aquatic plants.
- QCPBSN: The Columbia pebblesnail consumes mostly periphyton and a small amount of water milfoil.
- QRBTAD: The adult rainbow trout obtains about 90% of its dietary requirements from consuming mayfly, with the rest of the intake spread out almost evenly between five other plant and animal species. The rainbow trout consumes more of the higher-concentration mayfly (see Table 3.4) than does the smallmouth bass, thus the resulting tissue concentration is higher.
- RCATMK: The milk from a cow grazing in the unirrigated riparian zone accumulates technetium-99. Most of the contaminant source for this cow comes from drinking 125 L/day of contaminated seep water.
- RRTVEG: Root vegetables in the riparian zone are assumed to have a shallow root system that does not reach the water table. Therefore, there is no pathway for contamination to reach the plant.
- RLFVEG: Leafy vegetables in the riparian zone are assumed to have a shallow root system that does not reach the water table. Therefore, there is no pathway for contamination to reach the plant.
- RCHKEG: Chicken eggs in the riparian zone accumulate technetium-99. Because grain and vegetables have no technetium-99 concentration, the accumulation in the egg comes from the chicken drinking a small amount of water each day from a contaminated seep.

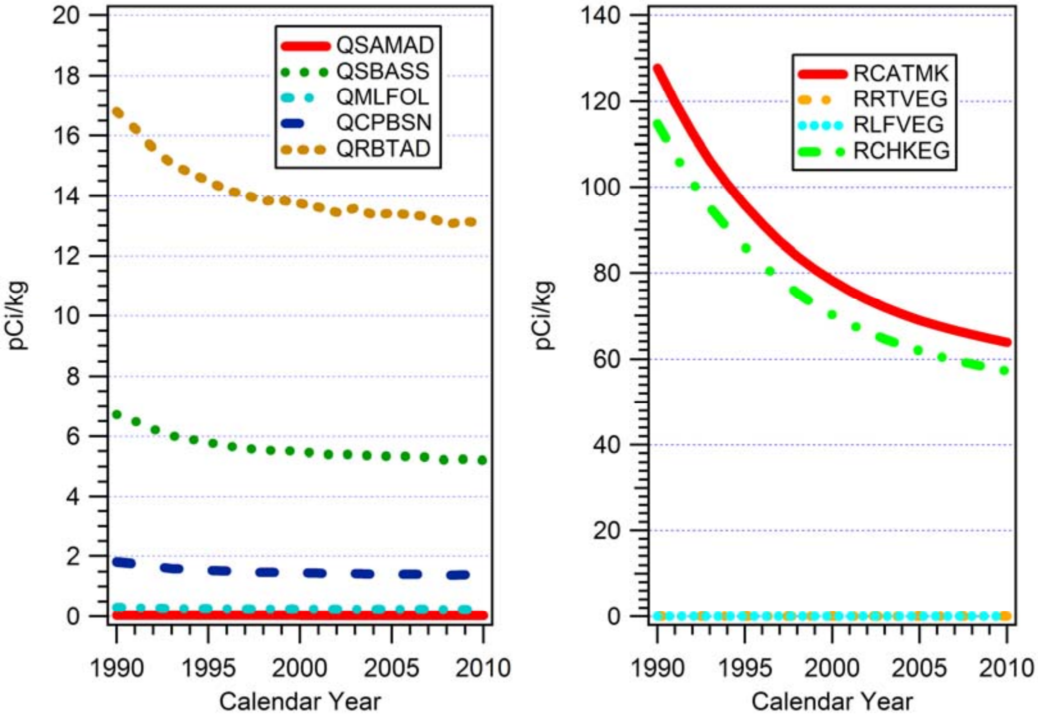


Figure 3.39. Concentrations of Technetium-99 in Selected Aquatic Organisms (left pane) and Terrestrial Organisms (right pane) for a Low-Concentration Groundwater Plume Intersecting the Columbia River

The technetium-99 results just discussed were obtained from a model run where the best estimate for every input parameter was selected. Usually, the median of the input statistical distribution was used for the best estimate. In cases where only point estimates for the parameters were available and a description of variability was added, the best estimate was selected as the point estimate. The mean value was used as the best estimate for liquid discharge inventory data where the data were provided in the form of a statistical distribution. A corresponding run of the sequence of SAC codes was made using stochastic inputs. Because these runs modeled the inventory, release, and transport of technetium-99, there were several hundred stochastic inputs. Estimated concentrations of technetium-99 in chicken eggs for an irrigated upland location and a nearby unirrigated riparian zone location are provided in Figure 3.40 for both best-estimate and stochastic (100-realization) runs. The "Best Estimate" line in the right pane of Figure 3.40 represents the same data as the "RCHKEG" line in the right pane of Figure 3.39.

Two observations are offered for data in Figure 3.40. First, stochastic parameters in the inventory, release, and transport models induce a significant variability in the groundwater and seep water concentrations of technetium-99. The seep water concentrations in the riparian zone come from mixing groundwater and surface water, and the resulting concentrations (on an annual basis) in the seep water is approximately half of the concentration in the groundwater. Second, use of contaminated groundwater to irrigate crops is the major reason that concentrations in eggs are much higher at the upland location than at the riparian location.

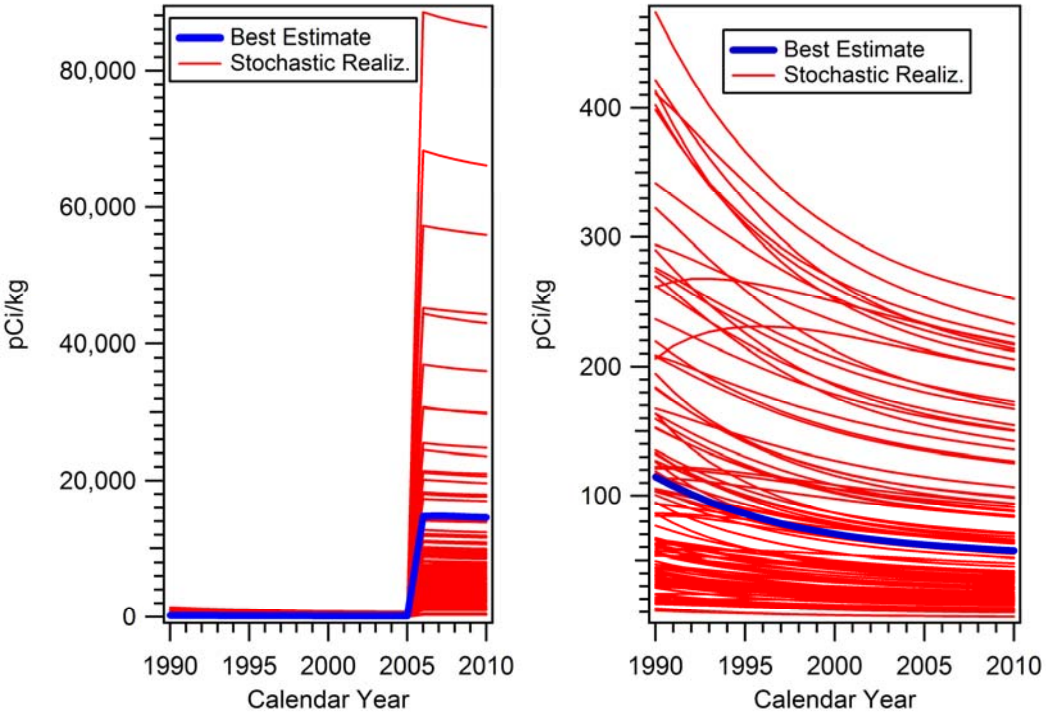


Figure 3.40. Modeled Concentrations of Technetium-99 in Chicken Eggs for an Irrigated Upland Location With Irrigation Starting in 2006 (left pane) and a Nearby Unirrigated Riparian Location (right pane)

The ECEM code supports calculations for multiple locations as well as multiple times in a single run. Thus, almost any output of the code (such as concentration, dose, or risk) can be assembled into time-varying contour animations for the entire Hanford Site. Although these animated results have been presented in other contexts, they are not amenable to presentation in a paper document and no animations are presented here.

3.10 Human Impacts Module

The conceptual human health impact model illustrated in Figure 3.41 includes exposure pathways of ingestion, inhalation, dermal contact, and direct radiation exposure from abiotic and biotic media. Relative exposures to these sources depend on individual lifestyles or exposure scenarios. Exposure scenarios are defined by groups of inputs and can represent such diverse lifestyles as a resident farmer using groundwater from upland areas or river sources, a Native American lifestyle along the river or upland on the Hanford Site, river recreational users, and Richland residents. Input parameter definitions for many of these scenarios are available in a Hanford risk data package (Miley et al. 2006).

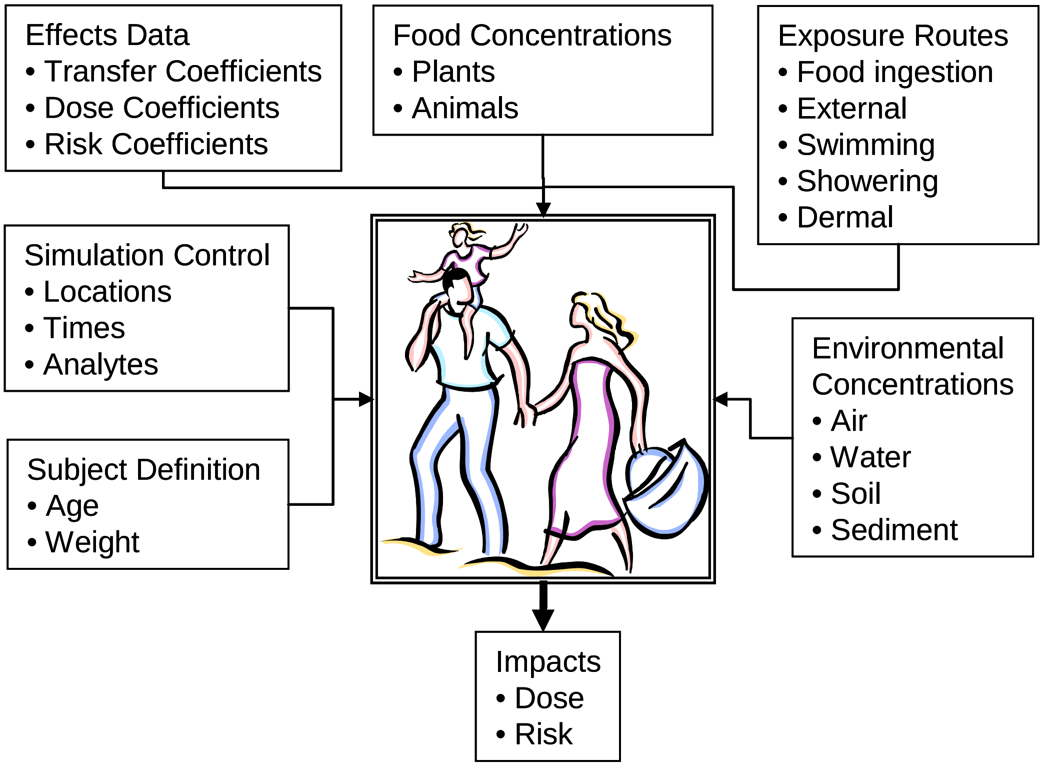


Figure 3.41. Conceptual Model of the Human Impacts Module

The human risk model is a screening-level chronic exposure model. It is intended for use in situations where the environmental contamination conditions are static or slowly varying. These conditions are met for many sources on the Hanford Site where contamination is released to the accessible environment and primarily transported via groundwater. This human risk model is not appropriate for estimating risks from short-term accidental releases or from short-term exposures to high-concentration wastes.

The human risk model is embodied in the HUMAN computer code in the SAC framework. Specific algorithms and user instructions are documented by Eslinger et al. (2006b). The location of the HUMAN code in the SAC processing sequence is identified in Figure 2.2 and also in Figure 3.42.

The human risk model addresses pathways related to long-term contamination from sources in air, surface water, and groundwater, with the associated contaminated media of seeps, springs, soils, and sediments. With these as starting points, the code estimates the exposures from contaminant concentrations in surface soil from irrigation, air from resuspension and volatilization, aquatic foods, terrestrial crops, and animal products. Both domestic animals and wild animals may be included. Exposure pathways explicitly modeled include external irradiation, dermal contact, inhalation, and ingestion of contaminants. The modeling framework distinguishes between exposures at surface water locations (such as the Columbia River), river shore locations (called riparian locations), and upland locations with no local sources of surface water.

The HUMAN code provides flexibility in combining the various pathways into exposure scenarios. Scenarios are defined through the use of representative sets of input parameters to simulate annual

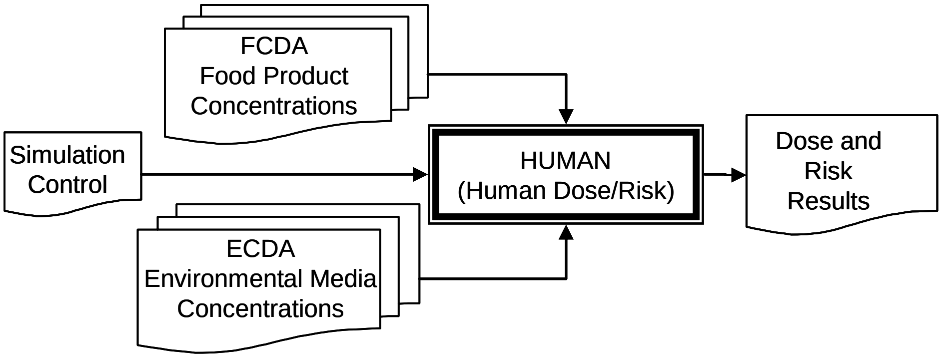


Figure 3.42. Location of the Human Module in the SAC Processing Sequence

average or lifetime average exposure conditions. The scenarios are focused on individual exposures. Individuals are assumed to spend the amounts of time specified in the scenario at the location of analysis; individual mobility throughout the analysis domain is not supported. Population risks are addressed for radionuclides.

The human impacts model uses time-dependent predictions of radionuclide and chemical contaminant concentrations at locations throughout the study area to estimate impacts. Concentrations of contaminants in food crops are obtained from the ecological exposure model ECEM (see Section 3.9). Risk or dose results are available for radioactive contaminants; non-radioactive but carcinogenic contaminants; and non-radioactive, non-carcinogenic but still hazardous contaminants. Radiation impacts to people may be calculated as either radiation dose or risk; carcinogenic chemical risks are provided in terms of risk of fatal cancer; risks from hazardous chemicals are provided as Hazard Quotients—the dimensionless ratio of the estimated intake to a standard Reference Dose.

The HUMAN code is designed to accept multiple realizations of concentration of contaminants in the environment. It allows the definition of stochastic exposure parameters, which combine with the uncertainty in the input media concentrations to provide a full range of uncertainty for the final dose or risk to the hypothetical exposed individual. It can also accept constant inputs, allowing calculation of deterministic results.

Doses from radioactive contaminants are calculated by methods consistent with Federal Guidance Report 11 (EPA 1988) and Federal Guidance Report 12 (EPA 1993). Risks from radioactive contaminants are calculated by methods consistent with Federal Guidance Report 13 (EPA 2002).

The concentrations of technetium-99 in different environmental media shown in Table 3.3 were used in a run of the HUMAN code. Although these concentrations are illustrative, they were obtained from a transport run modeling a low-concentration groundwater plume of technetium-99 intersecting the Columbia River in the 100 Area. The concentrations of technetium-99 in the groundwater for this example are roughly 10% of the drinking water standard for technetium-99 (EPA 1976) in 2006. The concentrations in Table 3.3 are modeled results rather than sampled results. The ecological model was used to estimate the concentrations in food products. These concentrations are combined with ingestion rates and exposure frequencies to provide the dose estimated by exposure pathway in Table 3.7. In this example, approximately half of the dose comes from ingestion of milk. The food and water intake for the milk cow is modeled using an assumption of groundwater use for irrigation.

Table 3.7. Example Modeled Doses from Technetium-99 by Food Type in 2006

Pathway	Dose (mrem)	Ingestion Rate		Concentration		Exposure Frequency	
Milk	0.4667	0.274	L/day	3193	pCi/L	365	day/yr
Root Vegetables	0.1513	0.2	kg/day	1418	pCi/kg	365	day/yr
Eggs	0.1465	0.01863	kg/day	14740	pCi/kg	365	day/yr
Fruit	0.07063	0.1014	kg/day	1306	pCi/kg	365	day/yr
Meat	0.008126	0.08219	kg/day	185.3	pCi/kg	365	day/yr
Leafy Vegetables	0.005543	0.007397	kg/day	1405	pCi/kg	365	day/yr
Bird	0.003447	0.01644	kg/day	393.1	pCi/kg	365	day/yr
Soil	6.477E-06	0.0001151	kg/day	121.4	pCi/kg	317	day/yr
Groundwater	0.05847	1.5	L/day	84.13	pCi/L	317	day/yr

The information in Table 3.7 was combined with other code outputs to summarize the dose by exposure pathways, which is shown in Table 3.8. The doses for soil ingestion and external radiation are not identically zero, but together they account for less than 0.01% of the total dose.

Table 3.8. Example Modeled Doses from Technetium-99 by Exposure Pathway in 2006

Dose (mrem)	Percent of Dose by Pathway	Pathway
8.522E-01	93.54%	Food ingestion
6.478E-06	0.00%	Soil/sediment ingestion
5.847E-02	6.42%	Water ingestion
9.107E-01	99.96%	All ingestion
3.873E-04	0.04%	Inhalation
7.035E-06	0.00%	External
9.111E-01		Total dose

The data presented in Table 3.7 and Table 3.8 were calculated at the year 2006. They were extracted from a longer simulation that calculated results between 1944 and 2070. Dose results for this simulation are presented in Figure 3.43 for years between 1990 and 2010. The results in the figure are obtained from two separate model runs. One model run uses the best estimate for every input parameter. Usually, the median of the input statistical distribution was used for the best estimate. In cases where only point estimates for the parameters were available and a description of variability was added, the best estimate was selected as the point estimate. The mean value was used as the best estimate for liquid discharge inventory data where the data were provided in the form of a statistical distribution. A corresponding run of the sequence of SAC codes was made using stochastic inputs. Because these runs modeled the inventory, release, transport, and effects of technetium-99, there were several hundred stochastic inputs. The jump in the magnitude of the dose in 2006 results from the onset of irrigation and the impact on concentrations in food products for both animals and humans. In this case, milk cows and chickens (and associated eggs) took in contaminated groundwater before and after 2006 but no plants were irrigated with groundwater until the year 2006.

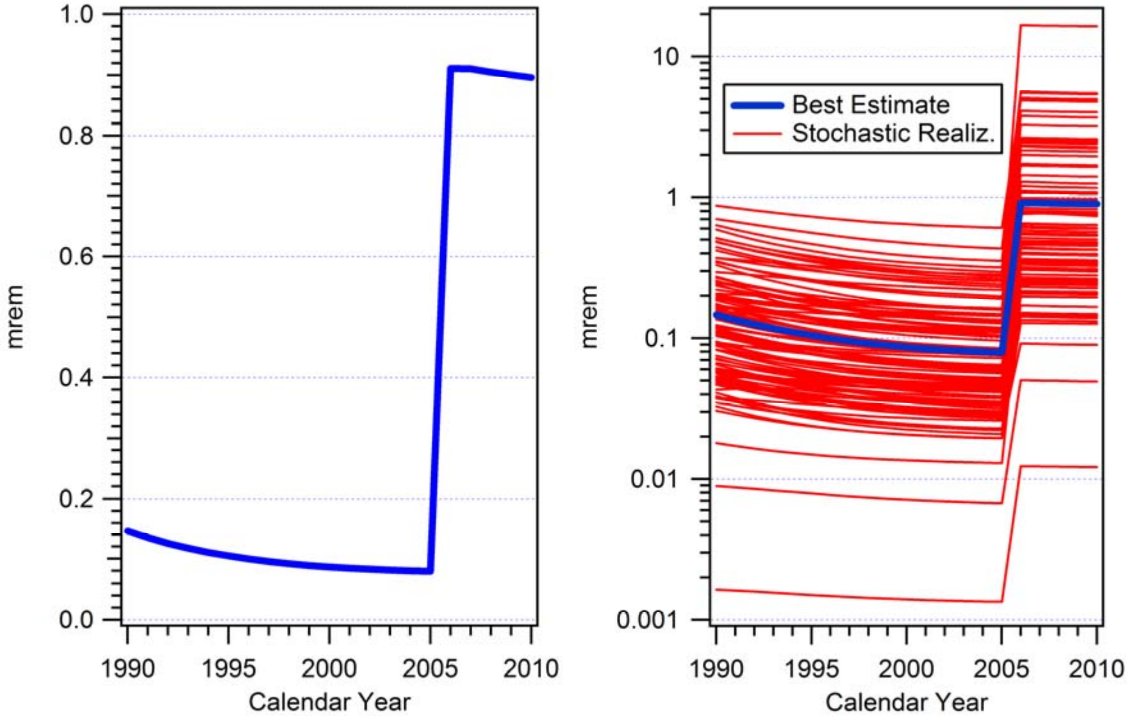


Figure 3.43. Modeled Doses from Technetium-99 for a Residential Farmer Scenario at a Representative Upland Location With Irrigation Starting in 2006 Using Best-Estimate Inputs (left pane) and Using Stochastic Inputs (right pane)

The HUMAN code supports calculations for multiple locations as well as multiple times in a single run. Thus, almost any output of the code (such as concentration, dose, or risk) can be assembled into time-varying contour animations for the entire Hanford Site. Although these animated results have been presented in other contexts, they are not amendable to presentation in a paper document and no animations are presented here.

3.11 Other Modules

The first conceptualization of the SAC (Kincaid et al. 2000) was intended to include a module that evaluated cultural impacts. However, defining a metric of true cultural impact that could be evaluated numerically has proven impossible so far. Thus, although a cultural impacts module is included in the SAC framework (see Figure 2.2), it is largely a sophisticated postprocessor that allows examination of results produced by other modules. Three types of solutions are available in this module. The first solution is the proximity of a site to contaminated water. Proximity is evaluated as the distance between a user-specified site and the closest location where water (groundwater or surface water) concentrations exceed a specified threshold value. The second solution is the total area where contamination exceeds a specified threshold. Although typical use of this solution has been to determine the area of a groundwater plume above the drinking water standard, this approach could be used to evaluate the area associated with other concentration solutions such as high soil or air concentrations. A final solution is the probability that concentrations in one or more environmental media exceed a user-specified threshold on a user-specified set of locations.

A 100-realization run was used to model all the waste sites within the Hanford core zone (see Figure 1.1) that are known to have released uranium and iodine-129. Stochastic data were used for the inventory, release, and vadose zone transport models as well as the retardation of uranium and iodine-129 in groundwater. A stochastic example of a proximity calculation resulting from this run is provided in Figure 3.44. The curve in the figure provides the minimum distance from a user-specified location on the core zone boundary to the closest location inside the core zone where the uranium concentration in groundwater exceeds 3 pCi/L in the year 2006. The discrete jumps in the distance result from using a relatively coarse grid of saved groundwater concentration values in this example run. Although this run is simply designed to illustrate the solution metric, one could interpret these data to imply that there is a low probability that the groundwater plume actually reaches this particular location in the 1990-2010 times period at a level above 3 pCi/L in concentration. The illustrative value of 3 pCi/L was chosen because it is approximately 10% of the drinking water standard for uranium (EPA, 2001).

Of course, there is only one past history of the movement of this contaminant in the groundwater, so calculating probabilities of past plume movements in one sense is merely a computational exercise. However, matching high probabilities of modeled plume movement with historical plume movements is one way to build confidence in the model that future projections may be representative enough of actual future plume movements to support decision making.

The probability that groundwater concentrations exceed a user-specified threshold over a user-specified set of locations is illustrated in Figure 3.45 for the same set of runs supporting the data shown in Figure 3.44. The curve in Figure 3.45 represents the probability that the maximum uranium concentration in groundwater along the boundary of the Hanford core zone exceeds 10% of the drinking water standard as a function of time. The values shown in Figure 3.45 are illustrative of the available solutions and are not definitive results that can be used in support of any particular project analysis.

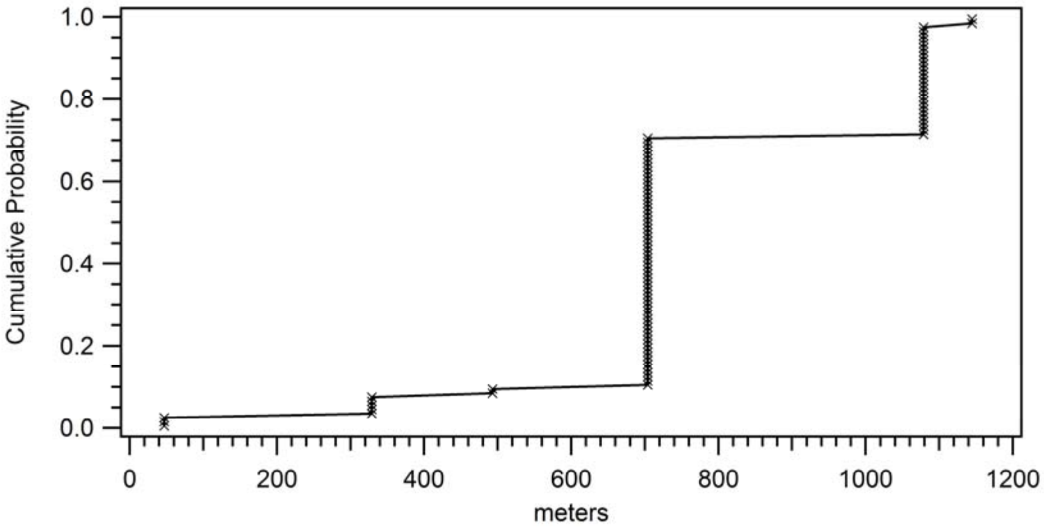


Figure 3.44. Stochastic Representation of the Minimum Distance from a User-Specified Location on the Core Zone Boundary to the Closest Location Where the Uranium Concentration in Groundwater Exceeds 3 pCi/L in 2006

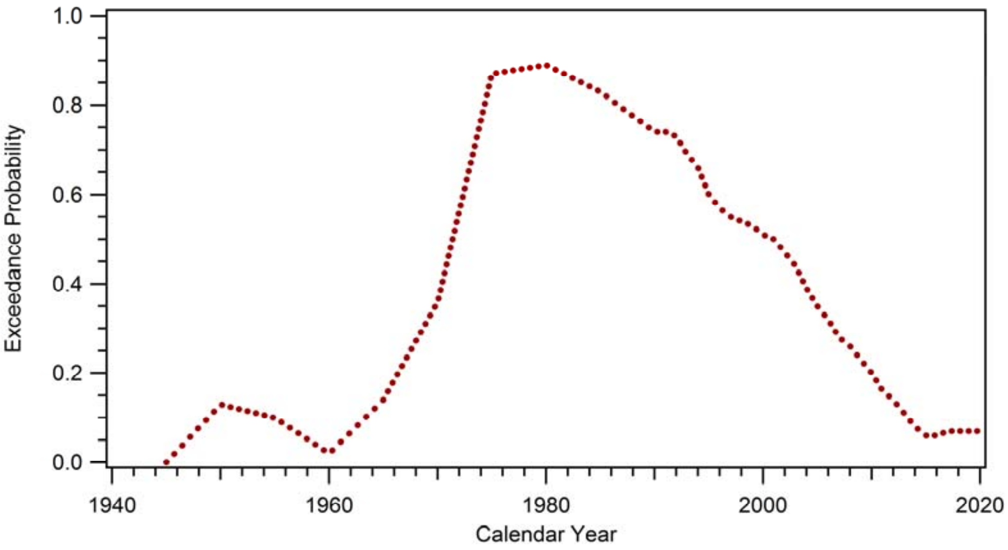


Figure 3.45. Modeled Probability That the Maximum Uranium Concentration in Groundwater Around the Boundary of the Hanford Core Zone Exceeds 10% of the Drinking Water Standard

The size of groundwater plume with concentrations that exceed a user-specified threshold is illustrated in Figure 3.46 for the same set of runs supporting the data shown in Figure 3.44. The figure shows the size of the groundwater plume with concentrations above 10% of the drinking water standard for iodine-129 (left pane) and uranium (right pane) based on releases from waste sites located in the core zone. The plume size curves are shown for all 100 realizations in the run. The relatively larger variability in the size of the uranium plume is driven largely by the stochastic distribution for uranium sorption. Although these example code runs are not intended to support any project-specific analysis, this figure provides a graphical indication of the large-scale implications of a wide distribution on the values for the uranium sorption coefficient.

The first conceptualization of the SAC (Kincaid et al. 2000) also included an economic impacts module that calculates incremental impacts to the local economy due to changes in agricultural, lifestyle, and recreational practices resulting from excessive levels of contamination. These impacts are described by the two following metrics:

- changes in the economic output due to voluntary avoidance of crops or activities because of the presence of contamination
- changes in the economic output due to proscriptions on the use of crops or participation in activities based on regulatory requirements.

Although the economic impacts module was upgraded to work in the SAC Rev. 1 framework, the module has not been applied to a problem since the Rev. 0 results were documented (Bryce et al. 2002). Therefore, no examples or applications of this module were included in this report.

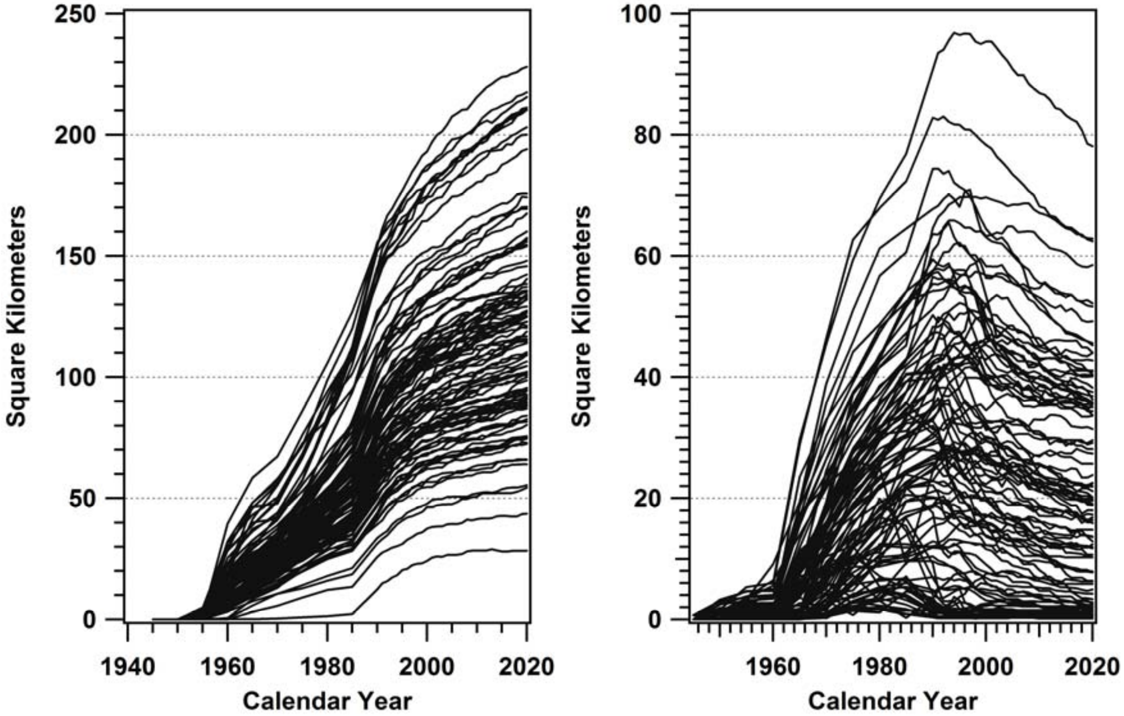


Figure 3.46. Stochastic Representation of the Size of the Groundwater Plume with Concentrations Above 10% of the Drinking Water Standard for Iodine-129 (left pane) and Uranium (right pane) Based on Releases from Waste Sites Located in the Core Zone

4.0 Example Applications

4.1 Inventory

The Hanford Remediation Assessment Project has developed an inventory data set to support remedial investigations. This inventory contains data on a suite of 23 radionuclides (^{14}C, ^{36}Cl, ^{137}Cs, ^{152}Eu, ^{3}H, ^{129}I, ^{237}Np, ^{231}Pa, ^{226}Ra, ^{79}Se, ^{90}Sr, ^{99}Tc, ^{233}U, ^{234}U, ^{235}U, ^{238}U, ^{238}Pu, ^{239}Pu, ^{240}Pu, ^{241}Pu, ^{242}Pu, ^{241}Am and ^{243}Am) and has been documented by Kincaid et al. (2006). That report also documents the primary data sources used to estimate annual and cumulative inventories. The inventory includes data produced by the Hanford Tank Waste Operation Simulator (HTWOS) (Kirkbride et al. 2005). The HTWOS simulation incorporated into this inventory is known as the Development Run executed on January 5, 2005, and documented in Kirkbride et al. (2005). The inventory also includes data from the Hanford Soil Inventory Model (SIM) that is documented by Corbin et al. (2005). The SIM model is used to estimate the more substantial past liquid discharges and unplanned liquid releases at the Hanford Site. It is based on 1) process knowledge of chemical separations plants and waste transfer records from those plants to single-shell tanks, double-shell tanks, and liquid discharges (for example, to cribs, specific-retention trenches, French drains, and ponds), and 2) unplanned releases, including past tank leaks. The HTWOS data and the SIM data sets contain information on more contaminants than are incorporated into the inventory for the Hanford Remediation Assessment Project.

Many early Hanford records focus on inventories of strontium-90, cesium-137, uranium isotopes, and plutonium isotopes, as well as the waste volume. Therefore, the record information of isotope composition is incomplete for the purpose of conducting a risk analysis. A variety of fill-in rules for fission products, activation products, and uranium isotopes are used to complete missing inventory values. In addition to fill-in rules, inventories for some waste sites or unplanned releases can only be estimated using similarity or surrogate estimates. For example, waste streams of similar origin can be used to estimate the inventory that may be discharged or released by an unplanned event when volume data and separations plant waste stream of origin are documented in the record. Descriptions of the inventory fill-in rules and surrogate site rules are provided by Kincaid et al. (2006).

A number of different waste forms are represented in the inventory data set. The SAC release module (see Section 3.2) has release models to account for the following waste forms:

- releases directly to the atmosphere for airborne analytes (for example, iodine-129 and carbon-14)
- liquid releases to the ground (cribs, trenches, retention basins), both intentional and accidental
- liquid releases directly to the river
- production reactor cores (before and after burial)
- naval reactor compartments
- contaminated soil or debris that is buried without treatment
- contaminated soil or items incorporated into a cement (grout) waste form
- residual material remaining in high-level waste tanks after they are cleaned out
- waste encapsulated in vitrified glass.

4.1.1 Inventory Estimation Rule Application

As an example of the application of estimation rules, consider an analysis that desires inventory for the following set of radionuclides: ^{3}H, ^{14}C, ^{36}Cl, ^{79}Se, ^{90}Sr, ^{99}Tc, ^{129}I, ^{137}Cs, ^{152}Eu, ^{226}Ra, ^{231}Pa, ^{234}U, ^{235}U, ^{237}Np, and ^{238}U. Further, consider the record data for a liquid waste stream at site 216-B-2-2 that was discharged to ground in 1970. Data for this example are summarized in Table 4.1. No inventory is assigned for the radionuclides: ^{3}H, ^{14}C, ^{36}Cl, ^{226}Ra, ^{231}Pa, ^{233}U, and ^{237}Np because no record data exist for them and no fill-in rule has been developed either. Values for ^{79}Se, ^{129}I, ^{152}Eu, and ^{235}U were developed from some fill-in rules described by Kincaid at al. (2006). In addition, for the purposes of the analysis for which these data were developed, the final data inventory of ^{238}U is actually the sum of the inventory of ^{234}U and ^{238}U (sum in Curies).

Table 4.1. Inventory Rule Demonstration for a Liquid Disposal from 216-B-2-2 in 1970

Constituent	Record Data (Ci)	Final Data (Ci)	Comment
^{79}Se		1.224E-07	Fission product estimation rule
^{90}Sr	2.390E+02	2.390E+02	From record data
^{99}Tc	2.978E-05	2.978E-05	From record data
^{129}I		1.503E-07	Fission product estimation rule
^{137}Cs	4.943E-01	4.943E-01	From record data
^{152}Eu		6.524E-06	Fission product estimation rule
^{235}U		6.597E-07	Uranium estimation rule
^{234}U	2.152E-05	NA	Combined with ^{238}U
^{238}U	1.571E-05	3.722E-05	Sum of ^{234}U and ^{238}U
Volume	24900 (m^{3})	24900 (m^{3})	From record data

In another example, the inventory for a liquid release from site 200-E-102 in 1956 was developed by a surrogate rule. In this case, the volume of the release from 200-E-102 was estimated to be 1,661 m^{3}, but no concentration data were available. Concentrations estimates are available for a liquid release at the 216-A-4 site in 1958, and both waste streams were judged to have resulted from the same type of processing. Therefore, the concentrations of contaminants in the release at the 216-A-4 site were assigned for the 200-E-102 site for the one release in 1956.

4.1.2 Uncertainty in the Inventory Data

The Inventory Module uses information from the inventory database and other assessment setup data to generate stochastic realizations of inventory, to aggregate inventories for sites identified for aggregation, and to account for radioactive decay to the end of each year. All disposal actions are described as contaminant concentrations within a disposal volume. The volumes and concentrations are both treated as stochastic quantities. It is possible to obtain a deterministic inventory by running a single realization with a fixed set of inputs.

A number of uncertainty rules for inventory data were developed and reported in Kincaid at al. (2006). The following sections illustrate some of the uncertainty rules.

4.1.2.1 Uncertainty in Soil Inventory Model (SIM) Data

The Hanford SIM (Corbin et al. 2005) data set is generated in a stochastic format. This data set provided inventory estimates for liquid releases for 377 waste sites. The majority of these sites are located on the Central Plateau of Hanford; however, four are located in the 300 Area. An example SIM data set for a liquid release from 200-E-25 in 1971 is provided in Table 4.2. The SIM data are represented by a 21-point cumulative probability function for the volume of the waste stream and also for each analyte in the waste stream. Analytes not present in the SIM analyses are not estimated using any fill-in rules because the SIM analysts checked for the presence of these analytes when the data were generated. Thus, for the SIM data, a missing inventory within a waste stream implies a zero inventory rather than lack of information.

Table 4.2. SIM Volume and Concentration Data for 200-E-25 for 1971 Liquid Releases

Probability	Volume (m^3)	Uranium-234 (Ci/m^3)	Uranium-235 (Ci/m^3)	Uranium-238 (Ci/m^3)
0	1	4.15E-14	5.90E-16	2.32E-14
0.05	1.2	1.80E-12	4.38E-14	1.18E-12
0.1	1.32	5.65E-12	1.61E-13	3.87E-12
0.15	1.41	1.12E-11	3.46E-13	7.72E-12
0.2	1.49	1.82E-11	6.00E-13	1.27E-11
0.25	1.56	2.66E-11	9.22E-13	1.89E-11
0.3	1.62	3.66E-11	1.32E-12	2.61E-11
0.35	1.68	4.83E-11	1.79E-12	3.47E-11
0.4	1.73	6.18E-11	2.34E-12	4.45E-11
0.45	1.78	7.69E-11	2.99E-12	5.58E-11
0.5	1.83	9.44E-11	3.75E-12	6.90E-11
0.55	1.87	1.14E-10	4.63E-12	8.39E-11
0.6	1.92	1.37E-10	5.66E-12	1.01E-10
0.65	1.97	1.64E-10	6.86E-12	1.22E-10
0.7	2.03	1.96E-10	8.31E-12	1.45E-10
0.75	2.09	2.33E-10	1.01E-11	1.75E-10
0.8	2.16	2.80E-10	1.22E-11	2.11E-10
0.85	2.24	3.41E-10	1.51E-11	2.58E-10
0.9	2.33	4.25E-10	1.90E-11	3.24E-10
0.95	2.45	5.64E-10	2.56E-11	4.36E-10
1	2.65	9.82E-10	4.52E-11	7.80E-10

The mean and standard deviation of the SIM data were provided in addition to the information shown in Table 4.2. The best-estimate inventory developed for some applications used the mean value of the SIM data rather than the stochastic representation illustrated here.

The A-10 crib (waste site 216-A-10) is a site that is known to have released large quantities of tritium. The description of inventory from the SIM data set for this site in the year 1965 is provided in Table 4.3.

Table 4.3. SIM Volume and Concentration Data for 216-A-10 for 1965 Liquid Releases

Cumulative Probability	Volume (m^3)	Tritium (Ci/m^3)	Technetium-99 (Ci/m^3)	Iodine-129 (Ci/m^3)	Uranium-234 (Ci/m^3)	Uranium-235 (Ci/m^3)	Uranium-238 (Ci/m^3)
0	3.01E+05	7.15E-05	5.17E-11	1.88E-10	2.16E-14	4.12E-16	3.52E-15
0.05	3.22E+05	2.12E-03	2.16E-09	7.88E-09	1.59E-11	4.32E-13	6.28E-12
0.1	3.34E+05	5.86E-03	6.67E-09	2.40E-08	1.18E-10	3.58E-12	5.84E-11
0.15	3.44E+05	1.08E-02	1.30E-08	4.62E-08	3.81E-10	1.22E-11	2.15E-10
0.2	3.52E+05	1.66E-02	2.09E-08	7.41E-08	8.75E-10	2.95E-11	5.43E-10
0.25	3.60E+05	2.33E-02	3.04E-08	1.08E-07	1.68E-09	5.83E-11	1.12E-09
0.3	3.66E+05	3.10E-02	4.15E-08	1.46E-07	2.84E-09	1.02E-10	2.02E-09
0.35	3.72E+05	3.97E-02	5.44E-08	1.91E-07	4.48E-09	1.65E-10	3.35E-09
0.4	3.78E+05	4.93E-02	6.87E-08	2.42E-07	6.66E-09	2.49E-10	5.17E-09
0.45	3.83E+05	6.02E-02	8.54E-08	2.99E-07	9.48E-09	3.62E-10	7.67E-09
0.5	3.88E+05	7.21E-02	1.04E-07	3.65E-07	1.31E-08	5.09E-10	1.09E-08
0.55	3.93E+05	8.58E-02	1.25E-07	4.37E-07	1.76E-08	6.94E-10	1.52E-08
0.6	3.98E+05	1.01E-01	1.49E-07	5.23E-07	2.34E-08	9.31E-10	2.06E-08
0.65	4.04E+05	1.19E-01	1.77E-07	6.19E-07	3.06E-08	1.23E-09	2.76E-08
0.7	4.10E+05	1.39E-01	2.10E-07	7.30E-07	3.97E-08	1.61E-09	3.67E-08
0.75	4.16E+05	1.62E-01	2.48E-07	8.63E-07	5.14E-08	2.11E-09	4.84E-08
0.8	4.24E+05	1.90E-01	2.94E-07	1.03E-06	6.67E-08	2.77E-09	6.41E-08
0.85	4.32E+05	2.26E-01	3.52E-07	1.22E-06	8.78E-08	3.68E-09	8.58E-08
0.9	4.42E+05	2.74E-01	4.30E-07	1.50E-06	1.19E-07	5.04E-09	1.19E-07
0.95	4.54E+05	3.49E-01	5.55E-07	1.93E-06	1.75E-07	7.45E-09	1.77E-07
1	4.75E+05	5.39E-01	8.65E-07	3.06E-06	3.60E-07	1.54E-08	3.78E-07

An example inventory calculation utilizing 5,000 realizations was conducted for this site. The tritium inventory ranges from 27.17 to 247,335 Ci and has a median of 28,051 Ci and a mean of 43,212 Ci. The generated inventory distribution is shown in Figure 4.1. Although the SIM data set has a reasonable amount of variability in the inventory for many sites, the inventory for this site, along with a few other sites that release large quantities of tritium, exhibits an unreasonably large variability. Further development work and data collection is required for the SIM model to produce estimates with a narrower range of variability. As a mitigating factor, when the releases from the A-10 crib are summed over the several years during which the crib operates, the resulting total is less variable than the data for each individual year.

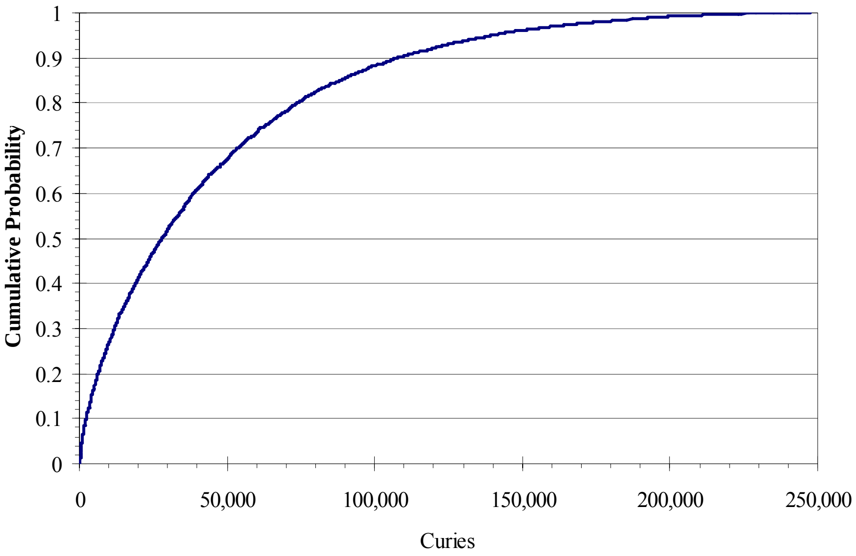


Figure 4.1. Cumulative Probability Function for the Tritium Inventory (Ci) Discharged to the 216-A-10 Crib in 1965 Based on the SIM Data Set

4.1.2.2 Uncertainty in All Other Inventory Data

For all sites not quantified by the SIM data, the Inventory Module simulates waste volumes and concentrations using a series of stochastic rules:

- All volume discharges were assigned a 20% uncertainty about the record or estimated volume of waste. This approach to uncertainty was adopted after consulting with current and retired Hanford staff familiar with inventory records. This uncertainty is implemented as a multiplier on the point estimate of volume, and the multiplier is defined as having a triangular distribution extending from 0.8 to 1.2 that is centered on 1.

- Consultations with current and retired Hanford staff familiar with inventory records indicated that individual site-specific inventories could be as much as uncertain by as much as a factor of 20 before 1970. Therefore, the analyte concentrations in waste streams released before 1970 were modeled with a lognormal (natural log) distribution, truncated at the 0.01 and 0.99 probability levels. The mean of the untruncated distribution was set to the point estimate or record value. The spread of the distribution was chosen by setting the standard deviation equal to twice the mean. This choice gave a wide variability representative of the stated 20-fold uncertainty of data during this operational era.

- Consultations with current and retired Hanford staff familiar with inventory records also indicated that individual site-specific inventory estimates were much more accurate after more detailed record keeping was initiated in 1970, and the variability in inventory records was more in the range of one-half the recorded value to twice the recorded value. The analyte concentrations in waste streams released from 1970 on were also modeled with a lognormal (natural log) distribution, truncated at the 0.01 and 0.99 probability levels where the mean of the untruncated distribution was set to the point estimate or record value. However, the spread of the distribution was chosen by setting the standard deviation equal to 0.25 times the mean. This choice gave a much narrower variability representative of the stated one-half to two fold uncertainty of data during this operational era.

- An exception to the inventory rule is that all production reactor core inventories are assumed to be relatively well known irrespective of the date they enter the inventory record. Therefore, the post-1970 rule (smaller variability) was assigned to all Hanford Site production reactor cores.
- In a few cases, a zero inventory was assigned to an analyte in a waste stream to preclude the application of a fill-in rule to that waste form. For these cases, the stochastic inventory was defined as a constant value of zero and did not implement any variability.

An example illustration of the variability rule applications is provided for the chlorine-36 inventory in D Reactor for a stochastic calculation utilizing 5,000 realizations. The record data set has a point estimate of 33.998 Ci of chlorine-36 and a disposal volume of 2,000 m^3 for this reactor. Using the above rules, the stochastic volume distribution would be triangular on 1,600 to 2,400 m^3. The sampled volume values range from 1,605.03 to 2,394.44 m^3 with a mean value of 2,000.00 and a median value of 2,000.02. The stochastic concentration distribution using the variability assumptions for pre-1970 data would use a mean value (in log space) of -4.87893 and a standard deviation (in log space) of 1.26864. The stochastic concentration distribution using the variability assumptions for post-1969 data would use the mean (in log space) of -4.104913 and a standard deviation (in log space) of 0.246221. For the pre-1970 variability assumptions, the generated inventory values (in curies) run from 0.726 to 319.2 with a mean of 29.66. For the post-1969 variability assumptions, the generated inventory values (in curies) run from 15.88 to 64.50 with a mean of 33.87. The cumulative distribution functions for these two sampled inventories are shown graphically in Figure 4.2.

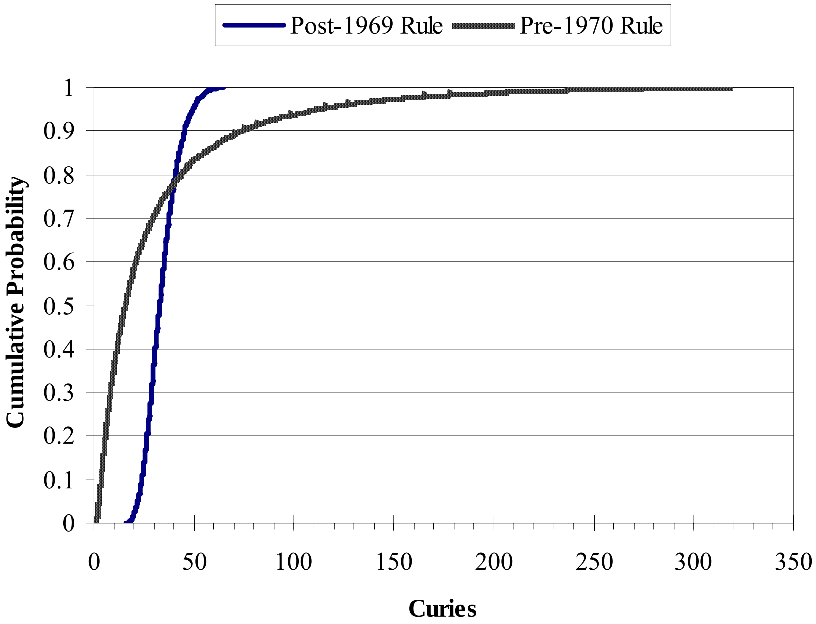


Figure 4.2. Example Cumulative Probability Functions for Chlorine-36 Inventory in D Reactor Using Pre-1970 and Post-1969 Uncertainty Rules

An example application of the uncertainty rule developed for production reactor cores is provided in Table 4.4. The analyte concentrations in the reactor cores were modeled with a lognormal (natural log) distribution, truncated at the 0.01 and 0.99 probability levels where the mean of the untruncated distribution was set to the point estimate or record value. The spread of the distribution was chosen by

Table 4.4. Application of an Uncertainty Rule for Cl36 Inventory (Ci) in Hanford Reactor Cores

Site	Record	Minimum	Median	Maximum	St. Dev.
100-N-66	75.0	34.3	74.0	119.5	17.5
118-B-8	42.0	23.0	41.1	70.6	10.0
118-C-3	12.0	7.2	11.6	22.9	3.0
118-D-6	34.0	19.7	32.9	54.4	7.9
118-DR-2	26.0	12.7	25.9	41.0	6.3
118-F-8	33.0	16.2	31.7	53.0	7.3
118-H-6	17.0	9.6	16.9	29.2	4.1
118-KE-1	54.0	29.7	50.9	83.5	12.3
118-KW-1	52.0	26.8	50.5	80.6	12.7
Sum	345.0	282.1	346.6	412.3	29.1

setting the standard deviation equal to 0.25 times the mean. This choice gave a variability representative of the one-half to two-fold uncertainty of the data developed from consultations with current and retired Hanford staff familiar with inventory records. The values for individual cores can vary over a range of approximately one half of the record value to twice the record value.

Another example illustration of the variability rule applications is provided for an iodine-129 inventory in the Integrated Disposal Facility (IDF) for a stochastic calculation using 5,000 realizations. The record data set has a point estimate of 0.33493 Ci of iodine-129 in a cement waste form with a disposal volume of 483.146 m^3 that is scheduled for disposal in the year 2011. The generated values range from 0.1583 to 0.6566 Ci and have a mean value of 0.3337 Ci. The standard deviation of the generated values was 0.0823. The cumulative distribution function for this disposal action is shown graphically in Figure 4.3.

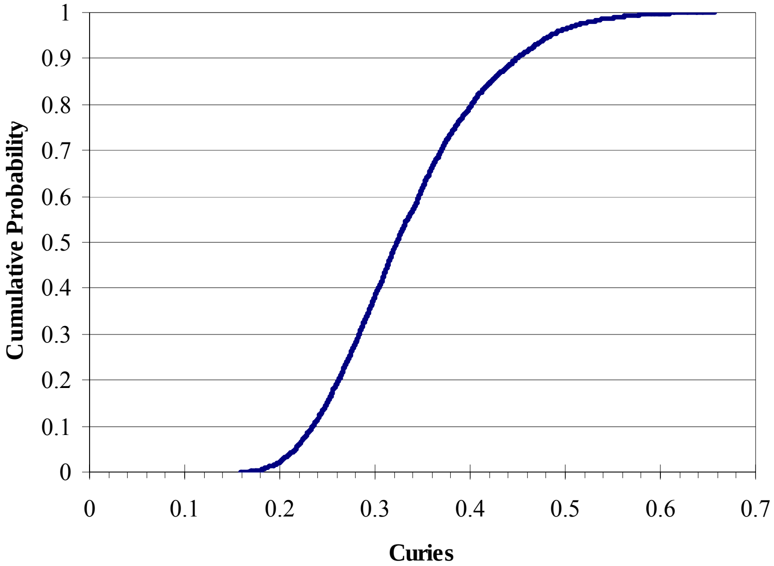


Figure 4.3. Example Cumulative Probability Functions for Iodine-129 Inventory in the IDF for a Single Cement Disposal Scheduled for 2011

A final illustration of the variability rule application is provided for disposal of some unsegregated trench waste containing uranium-238 at the 218-W-4A site in 1965. Results are provided for a stochastic calculation using 5,000 realizations. The record data set has a point estimate of 31.98 Ci of uranium-238 in a disposal volume of 2,061.48 m^3. The generated inventory values range from 0.6828 to 300.22 Ci and have a mean value of 27.89 Ci. The truncation of the lower and upper 1% of the tails of the distribution for concentration preserves the median of the generated data, but it does introduce a small downward bias on the mean value. The bias is much smaller when using the post-1969 variability rule than when using the pre-1970 variability rule.

4.1.2.3 Uncertainty in Total Onsite Inventory

A stochastic inventory run using 100 realizations was made for all onsite disposal actions. The use of 100 realizations is a modeling convenience for limiting run times. Summary statistics in the onsite inventory (curies) for four analytes is provided in Table 4.5: tritium, technetium-99, iodine-129, and uranium-234/238. Graphical representations of the variability in the inventory for these four analytes are provided in Figures 6.4 through 6.7. The best estimate for the onsite inventories is also provided to illustrate that the best-estimate inventory lies near the middle of the stochastic inventory values. Although there is some variability in the stochastic values for total onsite inventory, the variability in the total is relatively small. When acquiring inventory data and estimates, one often finds or is provided a best-estimate value without regard to its uncertainty distribution. In the case of the stochastic Hanford Soil Inventory Model (SIM) results, the best estimate is the mean. However, in all other cases the single value found or provided is simply interpreted as the best estimate.

Table 4.5. Summary Statistics for Onsite Inventory for Selected Analytes[(a)]

Analyte	Tritium (Ci)	Technetium-99 (Ci)	Iodine-129 (Ci)	Uranium-234/ Uranium-238 (Ci)
Minimum	24,640	20,560	49.3	2,730
Median	26,660	23,700	55.5	3,444
Maximum	28,700	30,100	62.0	4,349
Mean	26,680	23,860	55.5	3,433
Standard Deviation	950	1,460	2.2	265
Best Estimate	26,860	23,950	55.9	3,505
(a) Decay corrected to 2070.				

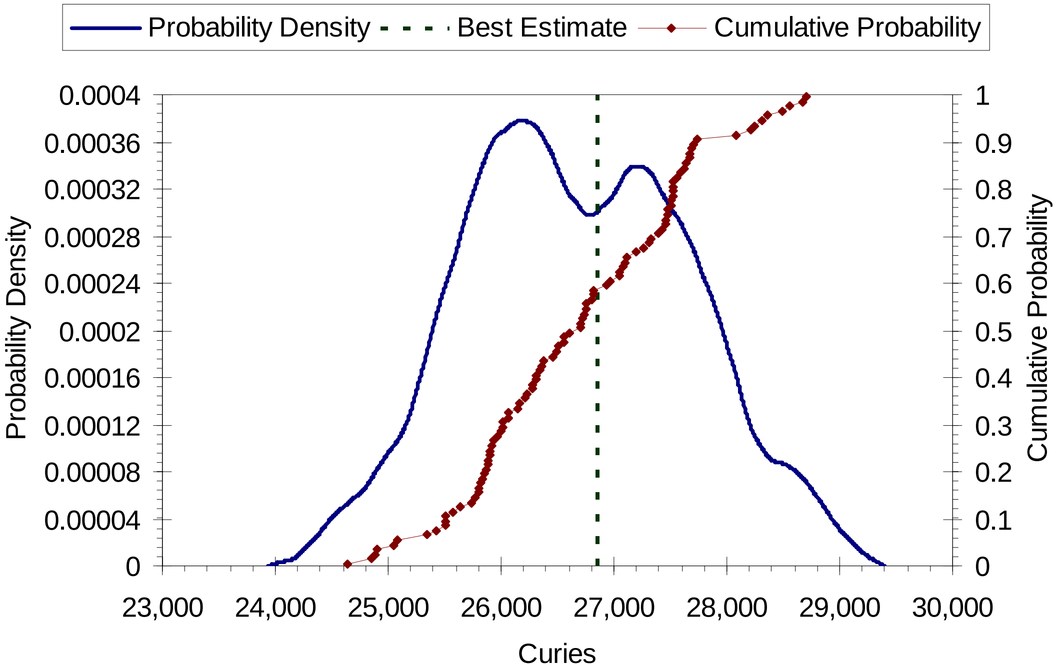


Figure 4.4. Example Stochastic Representation of Onsite Tritium Inventory (decay corrected to 2070)

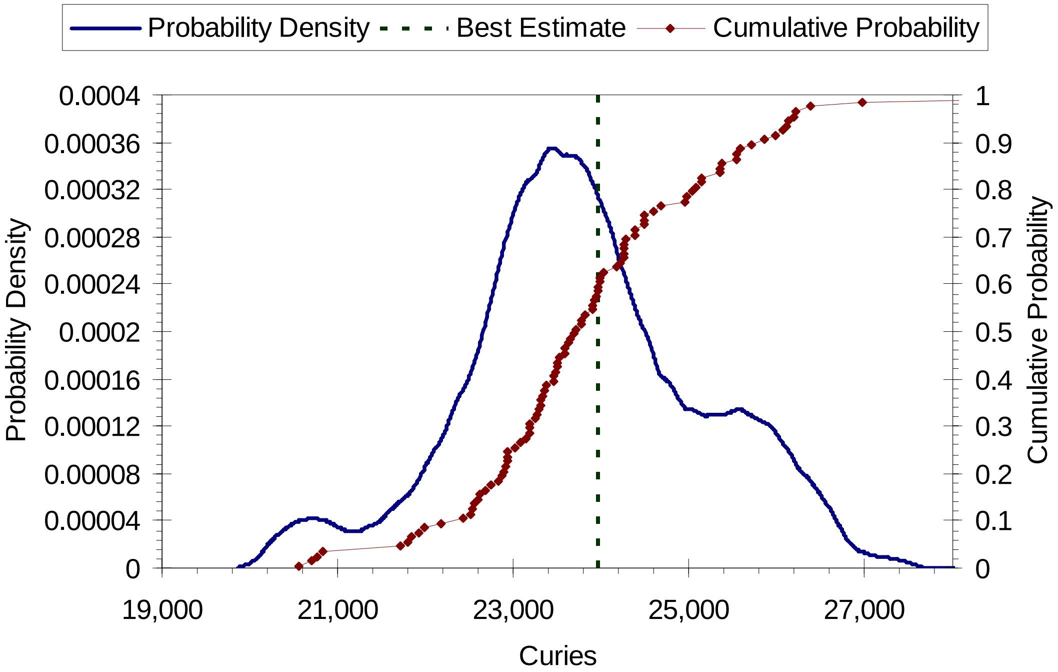


Figure 4.5. Example Stochastic Representation of Onsite Technetium-99 Inventory (decay corrected to 2070)

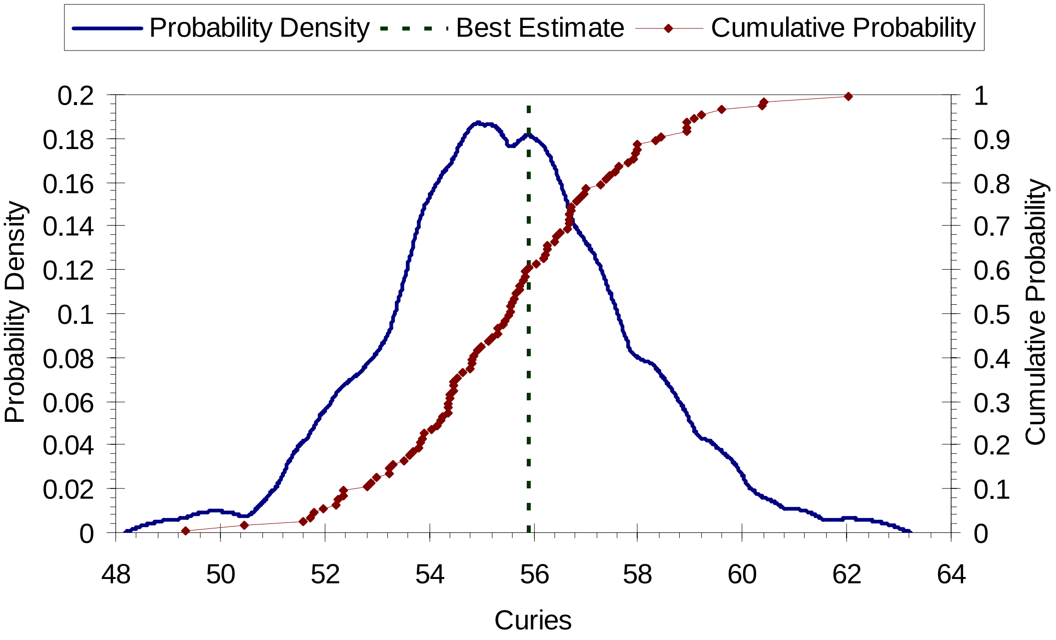


Figure 4.6. Example Stochastic Representation of Onsite Iodine-129 Inventory (decay corrected to 2070)

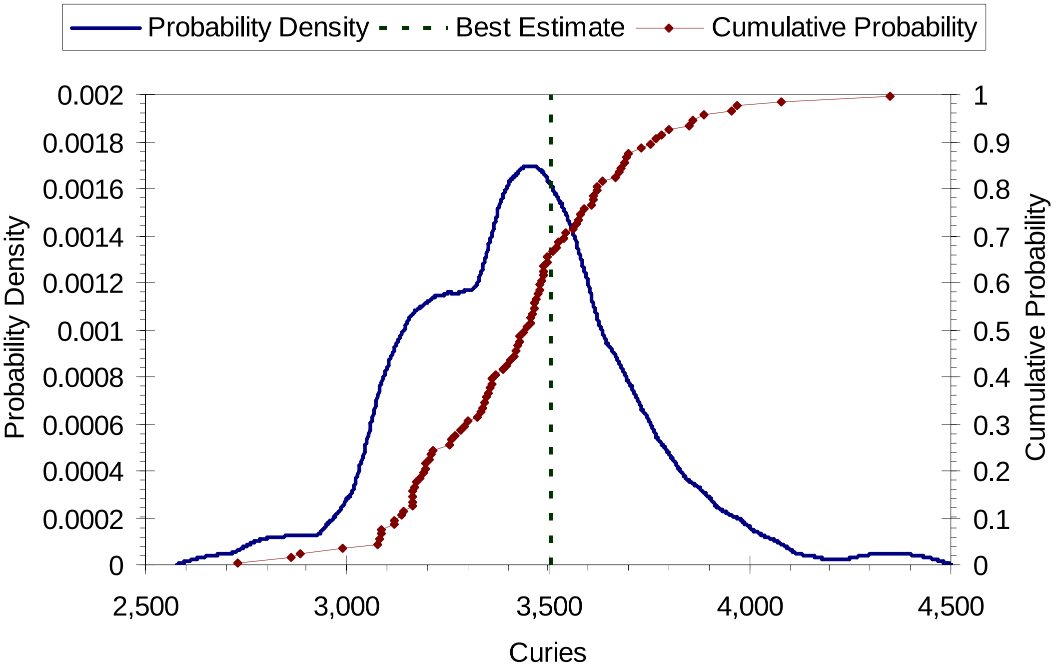


Figure 4.7. Example Stochastic Representation of Onsite Uranium-234/238 Inventory (decay corrected to 2070)

4.1.3 Example Subsets of the Inventory Data

The inventory data set contains information on waste that will be shipped offsite as well as historical or planned future releases on the Hanford Site. Example looks at the data are presented in this section to illustrate both the data and the ways that inventory can be divided for analysis or entry into a transport case. The following cases are discussed:

- Inventory for all waste streams decay corrected to the year 2070 – This inventory contains waste streams that are planned to be shipped to other locations for permanent disposal.
- Inventory for waste streams already to ground or to the river on the Hanford Site or planned for disposal on the Hanford Site – These data are decay corrected to the year 2070.
- Inventory for a single waste site – The data for the US Ecology (commercial low-level waste site) facility are used to illustrate this type of inventory summation.
- Inventory for a group of waste sites – The sites potentially releasing wastes into the groundwater for the 200-ZP-1 groundwater operable unit are used to illustrate this type of inventory summation.
- Inventory for a single waste form type summed across waste sites – The sites where liquid wastes were discharged are used to illustrate this type of inventory summation.

4.1.3.1 Inventory for all Waste Streams at the Year 2070

Summary statistics on the stochastic inventory for all waste streams in the inventory data set (decay corrected to the year 2070) are provided in Table 4.6. The summary statistics reported in the table were generated from an inventory run of 100 realizations. Also included is the best-estimate inventory. The information in Table 4.7 subdivides the best-estimate inventory by the general source of information for the inventory. Although surrogate data are provided for 52 waste streams, they represent only a very small fraction of the total inventory for any analyte.

Table 4.6. Summary Statistics on the Stochastic Inventory for All Waste Streams in the Inventory Data Set (Ci, decay corrected to the year 2070)

Analyte	Minimum	Median	Maximum	Mean	Standard Deviation	Best Estimate
^{3}H	2.53E+04	2.73E+04	2.94E+04	2.73E+04	9.51E+02	2.75E+04
^{14}C	4.71E+04	5.59E+04	6.73E+04	5.60E+04	4.41E+03	5.63E+04
^{36}Cl	2.86E+02	3.50E+02	4.15E+02	3.46E+02	2.91E+01	3.48E+02
^{79}Se	1.47E+02	1.84E+02	2.17E+02	1.84E+02	1.43E+01	1.85E+02
^{90}Sr	1.27E+07	1.53E+07	1.82E+07	1.53E+07	1.19E+06	1.54E+07
^{99}Tc	2.73E+04	3.11E+04	3.69E+04	3.12E+04	1.62E+03	3.13E+04
^{129}I	5.81E+01	6.51E+01	7.20E+01	6.51E+01	2.49E+00	6.55E+01
^{137}Cs	1.76E+07	2.29E+07	2.88E+07	2.29E+07	2.33E+06	2.30E+07
^{152}Eu	1.05E+02	1.21E+02	1.39E+02	1.22E+02	6.47E+00	1.23E+02
^{233}U	7.69E+02	9.49E+02	1.23E+03	9.55E+02	9.78E+01	9.54E+02
^{235}U	9.11E+01	1.07E+02	1.32E+02	1.08E+02	7.47E+00	1.10E+02
^{237}Np	2.30E+02	2.72E+02	3.27E+02	2.73E+02	2.05E+01	2.74E+02
^{234}U/^{238}U	4.70E+03	5.53E+03	6.64E+03	5.55E+03	3.52E+02	5.63E+03

Table 4.7. Identification of the Data Sources for the Best-Estimate Inventory Using All Waste Streams in the Inventory Data Set (Ci, decay corrected to the year 2070)

Analyte	SIM Data	HTWOS Data	Record Data	Estimated Data	Surrogate Data	Total
^{3}H	3.45E+03	2.29E+02	2.38E+04	0	2.72E+01	2.75E+04
^{14}C	1.96E+02	8.01E+02	5.53E+04	3.07E+01	4.10E+01	5.63E+04
^{36}Cl	0	0	3.48E+02	0	0	3.48E+02
^{79}Se	2.15E+00	9.23E+01	8.98E+01	6.44E-01	7.76E-03	1.85E+02
^{90}Sr	9.54E+03	8.74E+06	6.60E+06	1.84E+04	1.89E+03	1.54E+07
^{99}Tc	6.87E+02	2.67E+04	3.88E+03	2.38E+01	3.91E+00	3.13E+04
^{129}I	4.73E+00	4.29E+01	1.73E+01	6.05E-01	8.75E-03	6.55E+01
^{137}Cs	4.43E+04	9.19E+06	1.37E+07	3.84E+04	2.54E+03	2.30E+07
^{152}Eu	1.30E-01	5.01E+01	6.57E+01	1.31E+00	5.67E+00	1.23E+02
^{233}U	1.41E+02	7.24E+02	8.82E+01	0	1.36E+00	9.54E+02
^{235}U	3.10E+00	9.32E+00	9.63E+01	6.67E-01	2.60E-01	1.10E+02
^{237}Np	5.38E+01	1.27E+02	9.24E+01	0	8.20E-03	2.74E+02
^{234}U/^{238}U	1.42E+02	4.33E+02	5.04E+03	1.28E-05	1.23E+01	5.63E+03

An example representation of the difference in the inventory of onsite disposal versus the total of all waste streams for technetium-99 and uranium-234/238 is provided in Figure 4.8 for the inventory data set documented by Kincaid et al. (2006). These two isotopes have the largest difference between the two looks at the data, largely because of planned disposal of some waste streams in a national geologic repository. Information in offsite waste streams is contained in this data set only for the purpose of addressing the completeness of the onsite inventory. Thus, the data for offsite disposals may be incomplete in this data compendium.

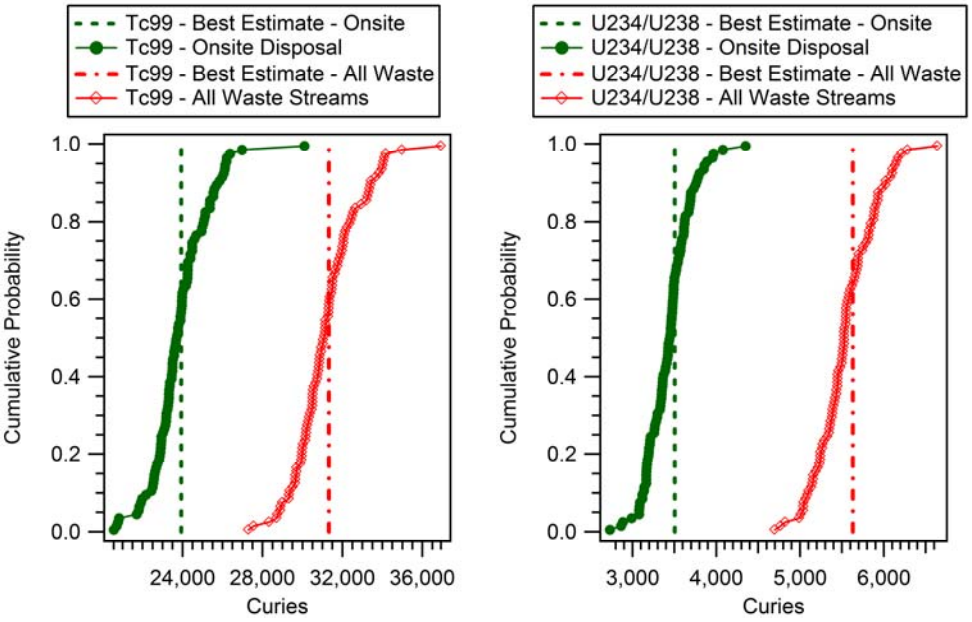


Figure 4.8. Example Representation of the Difference in the Inventory of Onsite Disposal Actions Versus the Total Inventory of All Waste Streams for Technetium-99 (left pane) and Uranium-234/238 (right pane) [Data are decay corrected to 2070.]

4.1.3.2 Inventory by Waste Stream Type

The inventory data set carries a waste form type with every disposal action, so individual data or summary values can be obtained by waste form. For example, the inventory in tritium wastes already available for release to the environment on the Hanford Site by the year 2006 are shown in Figure 4.9.

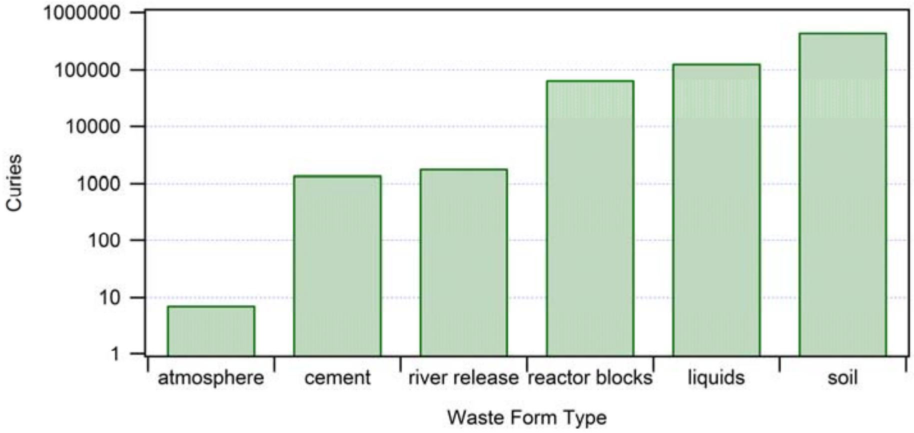


Figure 4.9. Best-Estimate Inventory of Tritium for Onsite Waste Forms Totaled by Major Waste Form Category (Data are for waste streams entering the inventory by 2006 and are decay corrected to 2006.)

4.1.3.3 Inventory for a Single Waste Site (US Ecology)

The inventory data set contains information on several hundred waste sites. Because the SAC system contains a model for each waste site, the inventory data for each site can be extracted and examined. Summary statistics on total inventory and the best estimate for total inventory are provided in Table 4.8 for the single US Ecology site.

Table 4.8. Summary Statistics and Best Estimates of the Total Inventory Sent to the US Ecology Low-Level Waste Site (Ci, decay corrected to 2070)

Analyte	Minimum	Median	Maximum	Mean	Standard Deviation	Best Estimate
^{3}H	1.46E+04	1.58E+04	1.69E+04	1.58E+04	4.87E+02	1.59E+04
^{14}C	4.56E+03	5.01E+03	5.68E+03	5.01E+03	2.46E+02	5.04E+03
^{36}Cl	2.69E+00	3.19E+00	3.78E+00	3.21E+00	2.34E-01	3.23E+00
^{79}Se	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
^{90}Sr	6.44E+03	7.37E+03	8.77E+03	7.38E+03	5.40E+02	7.43E+03
^{99}Tc	4.73E+01	5.48E+01	6.39E+01	5.47E+01	3.46E+00	5.51E+01
^{129}I	4.70E+00	5.90E+00	7.76E+00	5.94E+00	5.27E-01	5.97E+00
^{137}Cs	1.32E+04	1.62E+04	1.93E+04	1.62E+04	1.17E+03	1.63E+04
^{152}Eu	2.58E+01	3.26E+01	3.78E+01	3.26E+01	2.65E+00	3.29E+01
^{226}Ra	2.78E+02	3.09E+02	3.59E+02	3.12E+02	1.77E+01	3.13E+02
^{231}Pa	6.80E-04	1.22E-03	2.24E-03	1.31E-03	3.23E-04	1.32E-03
^{233}U	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
^{235}U	2.54E+01	2.99E+01	3.63E+01	3.04E+01	2.43E+00	3.06E+01
^{237}Np	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
^{234}U/^{238}U	1.48E+03	1.77E+03	2.13E+03	1.78E+03	1.41E+02	1.79E+03

An example representation of the total inventory of iodine-129 and carbon-14 sent to the US Ecology low-level waste site is provided in Figure 4.10. The cumulative inventory is obtained from the decay-corrected sum of annual inventory disposals. The data for annual disposals of iodine-129 at the US Ecology low-level waste site is illustrated in Figure 4.11. The plot in Figure 4.11 illustrates the combination of historical data and future disposal assumptions for the inventory at this waste site.

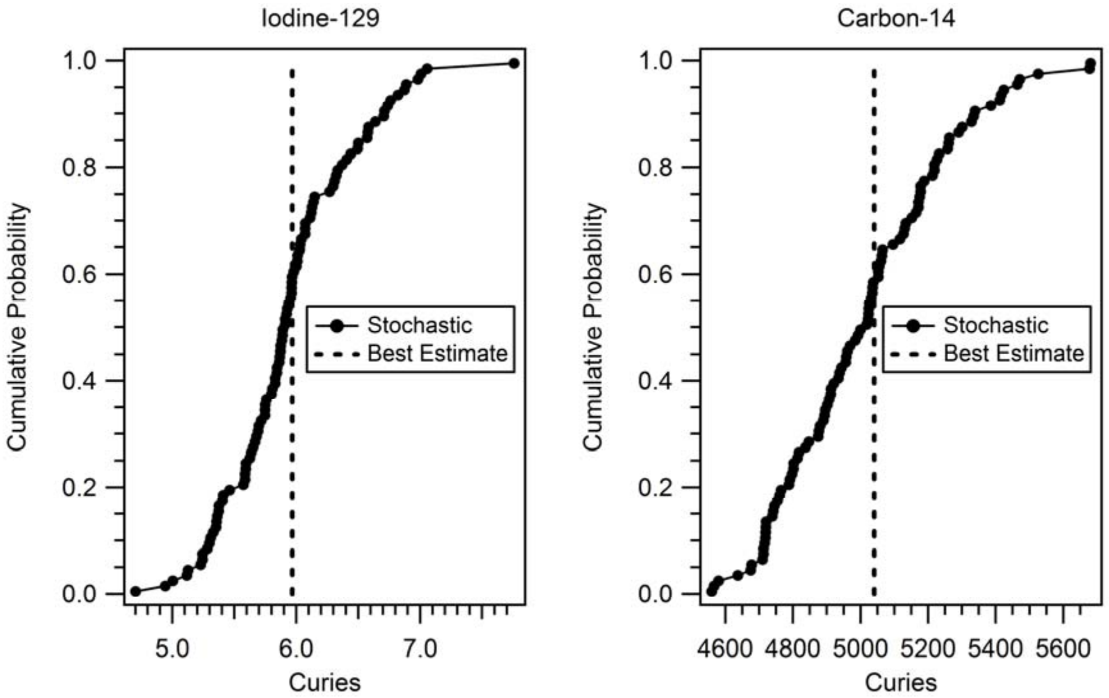


Figure 4.10. Example Representation of the Total Inventory Sent to the US Ecology Low-Level Waste Site for Iodine-129 (left pane) and Carbon-14 (right pane) [Data are decay corrected to 2070.]

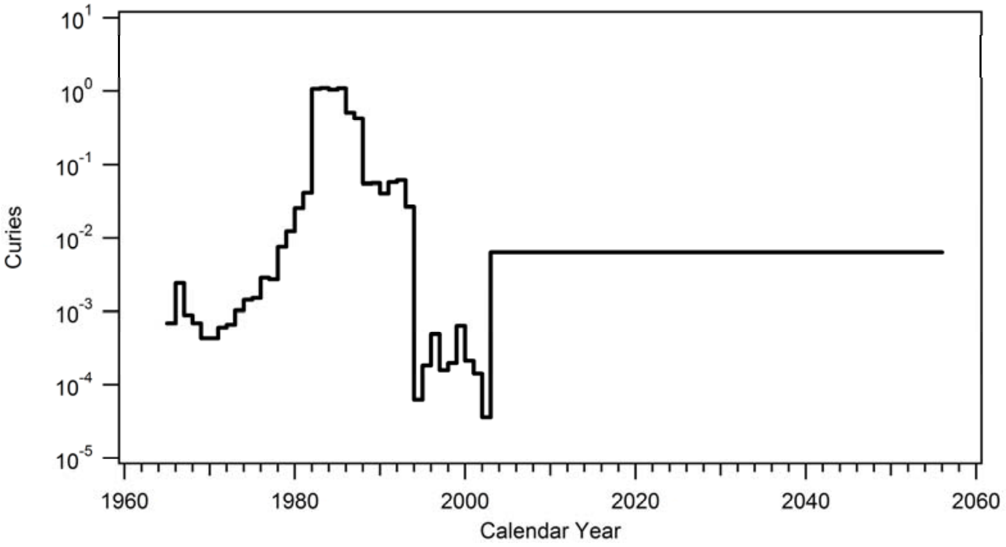


Figure 4.11. Time History of the Annual Inventory of Iodine-129 Discharged to Ground at the US Ecology Low-Level Waste Site

4.1.3.4 Inventory for a Group of Waste Sites (200-ZP-1 Groundwater Operable Unit)

The SAC inventory model operates on a user-specified set of sites, so it is simple to generate an inventory for a functional group of waste sites. As an example, summary statistics for the cumulative inventory in 2070 for selected analytes at waste sites associated with the 200-ZP-1 Groundwater Operable Unit are shown in Table 4.9.

Table 4.9. Cumulative Inventory (Ci) for Selected Analytes at Waste Sites Associated with the 200-ZP-1 Groundwater Operable Unit by 2070 (decay corrected to 2070)

Analyte	Minimum	Median	Maximum	Mean	Standard Deviation	Best Estimate
^{3}H	3.74E+03	4.85E+03	6.41E+03	4.88E+03	5.39E+02	4.99E+03
^{99}Tc	2.13E+02	3.00E+02	4.95E+02	3.09E+02	5.49E+01	3.10E+02
^{129}I	4.58E-01	7.81E-01	4.87E+00	1.00E+00	6.65E-01	1.10E+00
^{233}U	2.45E+00	6.60E+00	1.76E+01	7.23E+00	3.10E+00	7.14E+00
^{235}U	2.24E+01	2.99E+01	4.68E+01	3.06E+01	5.43E+00	3.18E+01
^{234}U/^{238}U	8.08E+02	1.27E+03	2.09E+03	1.29E+03	2.30E+02	1.34E+03

4.1.3.5 Inventory for a Single Waste Form Summed Across Sites

The SAC inventory model allows the user to selects a specific waste for further analysis. As an example, summary statistics for the cumulative inventory in 2070 for all liquid releases to the vadose zone are shown in Table 4.10.

Table 4.10. Cumulative Inventory (Ci) for All Liquid Releases to the Vadose Zone on the Hanford Site by 2070 (decay corrected to 2070)

Analyte	Minimum	Median	Maximum	Mean	Standard Deviation	Best Estimate
^{3}H	2.75E+03	3.67E+03	4.84E+03	3.70E+03	4.57E+02	3.67E+03
^{14}C	8.29E+01	1.76E+02	5.33E+02	1.94E+02	7.80E+01	1.91E+02
^{36}Cl	0	0	0	0	0	0
^{79}Se	1.51E+00	1.83E+00	2.12E+00	1.84E+00	1.21E-01	1.83E+00
^{90}Sr	4.63E+03	6.90E+03	1.06E+04	7.07E+03	1.44E+03	7.02E+03
^{99}Tc	4.77E+02	5.79E+02	6.69E+02	5.80E+02	3.93E+01	5.80E+02
^{129}I	3.23E+00	4.52E+00	6.43E+00	4.58E+00	6.14E-01	4.55E+00
^{137}Cs	5.89E+03	9.98E+03	2.95E+04	1.09E+04	3.80E+03	1.08E+04
^{152}Eu	7.74E-02	1.03E-01	1.37E-01	1.05E-01	1.22E-02	1.04E-01
^{233}U	5.67E+00	1.88E+01	8.01E+01	2.29E+01	1.52E+01	2.25E+01
^{235}U	1.35E+00	2.16E+00	4.70E+00	2.27E+00	6.27E-01	2.24E+00
^{237}Np	3.36E+01	5.23E+01	9.15E+01	5.38E+01	1.17E+01	5.34E+01
^{234}U/^{238}U	6.78E+01	1.03E+02	1.92E+02	1.06E+02	2.33E+01	1.02E+02

A graphical representation of the best estimate and stochastic inventory of technetium-99 and iodine-129 for the data in Table 4.10 is provided in Figure 4.12. The data are cumulative through the year 2070 and are decay corrected to 2070.

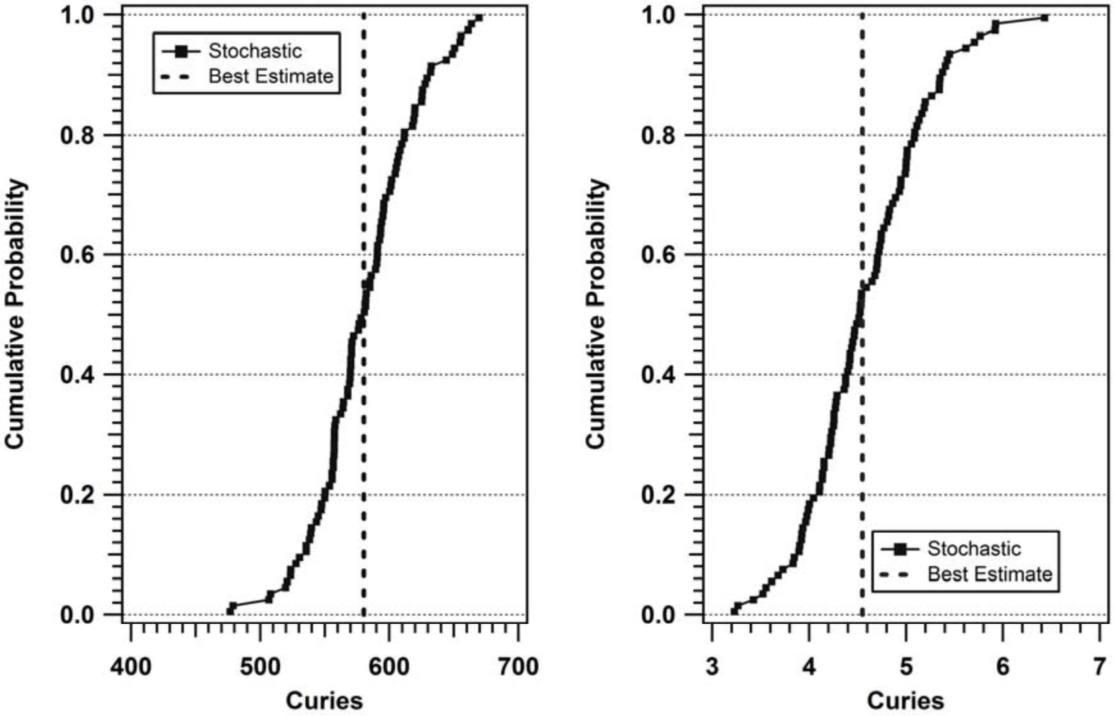


Figure 4.12. Example Representation of the Total Inventory in Liquid Waste Forms Sent to the Vadose Zone on the Hanford Site for Technetium-99 (left pane) and Iodine-129 (right pane) [Data are cumulative through 2070 and are decay corrected to 2070.]

4.2 Stochastic Properties Selection for the Vadose Zone Hydrologic Model

The groundwater plume sizes for tritium, technetium-99, iodine-129, and uranium-238 in the preliminary stochastic analysis (Bryce et al. 2002) showed unreasonably large variability when compared to historical measurements of plume movement. Initially, this variability was attributed to large variations in the vadose zone hydrology parameters for all analytes and large variation in the groundwater retardation values for technetium-99, iodine-129, and uranium-238. Additional data have been collected since 2002, and the SAC conceptual models and computer codes have been upgraded. Therefore, an analysis was conducted to determine whether the variability in groundwater plume size had improved since 2002.

4.2.1 Performance Metric Selection

The first step in the analysis was to conduct best-estimate and stochastic runs of the inventory, release, vadose zone, and groundwater modules (see Figure 2.2). Measured parameters for vadose zone hydraulic values are quite variable, and they are measured on a laboratory scale rather than a model application scale. Although some techniques have been advanced in the literature for adjusting laboratory-scale parameter values to field scale, they have not proven universally effective. Therefore, a best-estimate parameter selection was implemented where the technique for choosing the best estimate for each of several hundred vadose zone hydraulic model parameters was to assign the median value of the input distribution. Specific data values for these parameters are given in the vadose zone hydrology data package (Last et al. 2006a). The modeled sizes of tritium groundwater plumes where concentrations exceed the drinking water standard (20,000 pCi/L) are provided in Figure 4.13. Tritium was chosen for the first analysis because it moves with the water and does not confound uncertainty in hydrologic model parameters with uncertainty in sorption values.

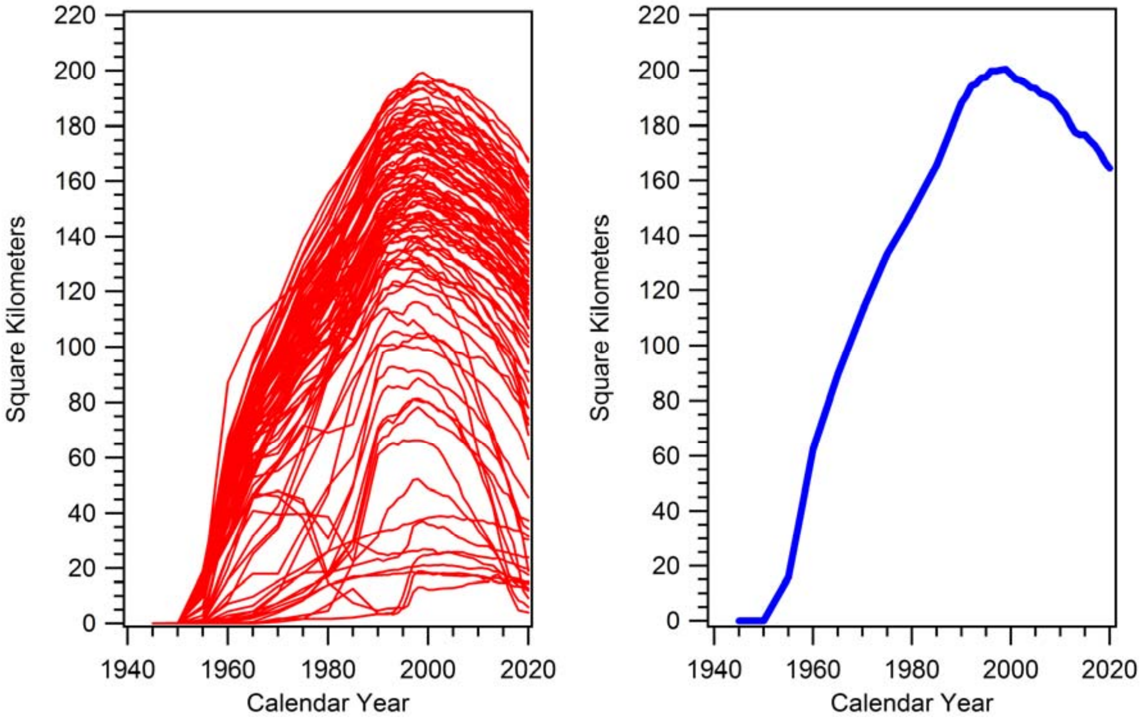


Figure 4.13. Modeled Size of the Tritium Groundwater Plume Where Concentrations Exceed the Drinking Water Standard (Left pane: plume sizes from a 100-realization stochastic run. Right pane: plume size from the best-estimate run.)

The groundwater plume sizes shown in the left pane of Figure 4.13 include a significant number of realizations with small groundwater plumes when compared with historical data. For example, the Hanford environmental report for 2001 reports an estimated plume size of 151 km^2 for tritium concentrations above the drinking water standard (Poston et al. 2002, Table 6.2.1). Figure 6.2.1 in the Hanford environmental report for 2001 also provides a plume map for tritium in 2001 based on sampling measurements. Examination of the best-estimate groundwater simulation shows that the simulation sends more tritium northward from the 200 Areas and less tritium southeast toward the river than is shown in the environmental report. Thus, simulated plume sizes are expected to be slightly larger than indicated by historical field measurements.

This flow and transport problem is ill-defined in the sense that no absolute performance metric exists that can be used to constrain the input parameter distributions such that simulated results all lie close to historical measurements. The following three performance metrics were considered, with varying degrees of success:

- Metric 1: Cumulative activity of tritium in the groundwater in 2001
- Metric 2: Size of the groundwater plume of tritium (water concentrations above the drinking water standard) in 2001
- Metric 3: Time of first significant release of tritium to the groundwater ("significant" in this sense is defined as enough tritium enters the groundwater to force concentrations above the drinking water standard).

A geostatistical analysis of historical data was performed (Murray et al. 2004) to examine the level of accuracy that could be obtained for metrics 1 and 2 above while focusing the analysis on the year 2001. Uncertainties in their geostatistical analysis include uncertainties in individual concentration values, lack of knowledge as to how the concentration varies with depth in the groundwater, uncertainty in the geologic strata, and uncertainty in the porosity in geologic units. Total tritium activity in the aquifer in 2001 was estimated to lie between about 10,000 Ci (using a 5-meter plume depth assumption) and 306,000 Ci (using a 20-meter plume depth assumption). Their results had a range of median estimate estimates of 30,900 Ci (5-meter plume depth assumption) to 92,500 Ci (20-meter plume depth assumption). The SAC modeling results yield a best-estimate value of 148,900 Ci delivered to the aquifer by 2001. The analysis also indicated that approximately 10% of that amount was delivered to the river by 2001 and left the groundwater domain. The stochastic estimates of the amount of tritium reaching the water table for the simulation shown in Figure 4.13 range from 8,700 to 190,000 Ci with a median value of 96,600 Ci. Even though there are large numbers of field measurements of tritium concentrations in groundwater, the conclusion reached from this comparison is that field measurements of tritium in the groundwater do not provide enough information to use the estimated mass in the aquifer as a metric for constraining the choice of hydrologic parameter values in the vadose zone.

The time when tritium enters the aquifer in large enough quantities to induce a significant tritium plume is not amenable to easy analysis from historical data. However, large quantities of tritium were discharged to ground starting in the early 1950s. Many of these releases were liquid releases associated with large volumes of water. Enough water was discharged that the water table elevation in the 200 Areas started to rise in the 1950s. This information is sufficient to infer that a groundwater plume should have started developing in the early 1950s. Some of the stochastic results shown in Figure 4.13 display this characteristic, but many do not. Because all of the simulations use the same groundwater flow field, one could infer that faster vadose zone transport should have occurred in some of the realizations.

The geostatistical analysis (Murray et al. 2004) also examined the size of the tritium plume in groundwater that was above the drinking water standard. Some postprocessing of their data is needed to combine results from their different grids, but once that is done the size of the tritium plume is estimated to range from 86.4 to 164.7 km^2, but only one of the their stochastic values is over 148 km^2. As noted earlier, the environmental report for 2001 reports an estimated plume size of 151 km^2 for tritium concentrations above the drinking water standard (Poston et al. 2002, Table 6.2.1). Tentatively, one could infer that the size of the tritium plume for the best-estimate run is sufficient information to use as a metric when examining the effect from different selections of vadose zone flow model parameters.

4.2.2 Application of Metrics to Model Parameters

The vadose zone flow model (STOMP code) uses the van Genuchten formulation (van Genuchten 1980) for the effective aqueous-phase saturation in conjunction with the Mualem formulation (Mualem 1976) for the aqueous-phase relative permeability. A total of five empirically derived parameters are needed to fully define the resulting curves for effective aqueous-phase saturation and effective aqueous-phase permeability. The effective aqueous-phase saturation s_ℓ^* is defined as

$$s_\ell^* = \frac{s_\ell - s_{\ell r}}{1 - s_{\ell r}}$$

where s_ℓ is the aqueous-phase saturation and $s_{\ell r}$ the residual aqueous-phase saturation. The van Genuchten formulation for the aqueous-phase saturation is defined as follows:

$$s_\ell^* = \left[1 + (\alpha\psi)^n\right]^{-(1-\frac{1}{n})} \quad \text{for } \psi < 0 \text{ (unsaturated conditions)}$$

$$s_\ell^* = 1 \quad \text{for } \psi \geq 0 \text{ (saturated conditions)}$$

where ψ is the tension head and α and n are the empirically determined fitting parameters. In practice, this curve is multiplied by the porosity of the hydrologic unit. The aqueous-phase relative permeability ($k_{r\ell}$) calculations use the Mualem formulation

$$k_{r\ell} = \sqrt{s_\ell^*}\left[1 - \left(1 - \left(s_\ell^*\right)^{\frac{1}{m}}\right)^m\right]^2$$

The effective permeability (k_ℓ) is computed as $k_\ell = K_s k_{r\ell}$ where K_s is the saturated aqueous-phase permeability. The parameter m is defined using the expression $m = 1 - 1/n$. The five sampled parameters are α, n, porosity, K_s and $s_{\ell r}$.

Examination of the sampled data for the initial stochastic run indicates that although each simulated value for these five parameters lies within the range of the field samples for the particular soil type, the individual sampled parameters can form combinations where the derived curves for saturation and permeability are substantially different than the set of curves obtained from the individual soil samples. Although multiple samples from some soil types exhibit a correlation between these input parameters, the extensive field data for the Hanford site do not exhibit any discernable correlations between these parameters.

Using the best-estimate values for each of these parameters (for 29 different soil types) does yield reasonable water flow and tritium transport results across the Hanford Site. Therefore, a sample-and-reject technique that was devised to ensure sampled combinations of sets of five parameters yielded curves for saturation and permeability that were reasonably close to the curves obtained using the best-estimate parameters. The sampling technique consisted of the following steps:

1. Obtain a sample of all five parameters from the parameter distributions derived from field sampling data, where the distributions are for each parameter individually.
2. Compute the tension head versus aqueous-phase curve using the sampled parameters.
3. Compute the integrated distance between the curve in step 2 and a similar curve calculated using the best-estimate parameter values.
4. Compute the effective permeability versus saturation curve using the sampled parameters.
5. Compute the integrated distance between the curve in step 4 and a similar curve calculated using the best-estimate parameter values.

6. Reject the set of five sampled values if either of the distances from steps 3 and 5 were larger than user-specified limits.

Although this sampling algorithm is conceptually easy to implement, no advance knowledge was available with which to determine reasonable constraints on the integrated differences for each of the 29 soil types. Therefore, two exploratory cases (called the slightly constrained and tightly constrained cases) were conducted where the rejection distances were chosen subject to two criteria:

1. The integrated curve distances were chosen subject to the following approach:
   a. Slightly constrained case: about 20% of the sampled sets of parameters would be retained.
   b. Tightly constrained case: about 3% of the sampled sets of parameters would be retained.
2. The integrated curve distances were chosen so about equal numbers of rejections were due to the aqueous-phase saturation curve and the effective permeability curve.

The effect that these constraints have on the computed curves for effective aqueous-phase saturation and effective permeability are illustrated in Figure 4.14 using data for the Ringold Formation. For this formation, the parameter causing the most rejections of the effective aqueous-phase saturation curve is the porosity. Both large and small values of the porosity were rejected. The unconstrained distribution for porosity runs from about 0.06 to 0.42, and in the slightly constrained case the retained values for porosity range from about 0.1 to 0.3. For this formation, the parameter causing the most rejections of the effective permeability curve is the saturated permeability. Large and small values of the saturated permeability were rejected. The unconstrained distribution for saturated permeability runs from about 6E-6 to 0.5, and in the slightly constrained case the retained values range from about 3E-5 to 1E-3.

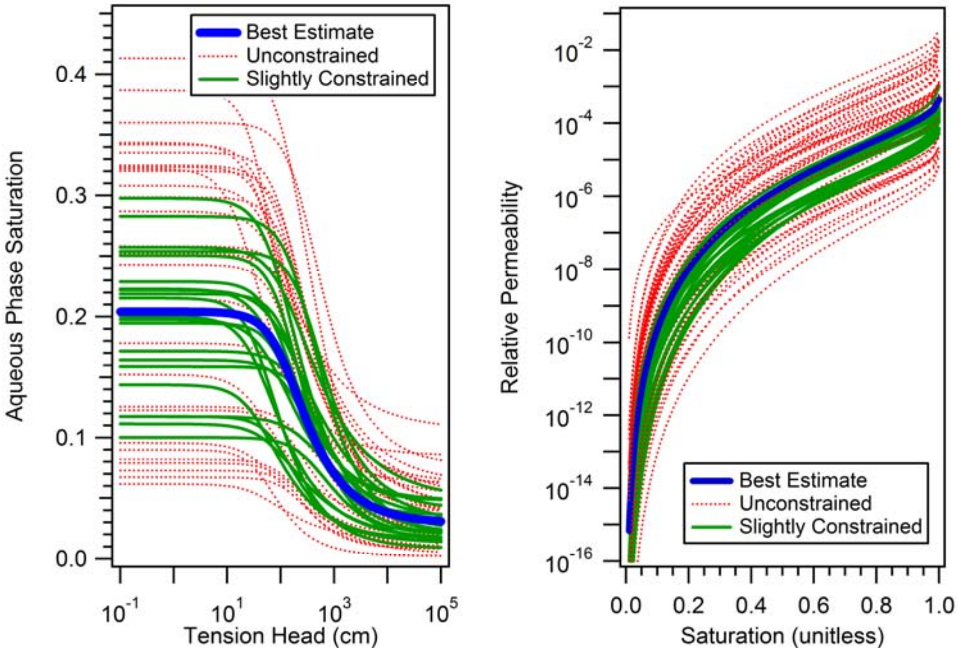


Figure 4.14. Illustration of the Effect of Constraining the Vadose Zone Hydraulic Model Parameters Using Data for the Ringold Formation (The aqueous-phase saturation shown in the left pane has been multiplied by the porosity of the Ringold Formation.)

The effect that the constrained sampling has on the decay-corrected cumulative release of tritium from the vadose zone to the groundwater by the year 2001 is shown in Figure 4.15. The original sampling distributions and the slightly constrained sampling distributions yield similar releases, but the smallest releases for the slightly constrained sampling case are larger than for the unconstrained case.

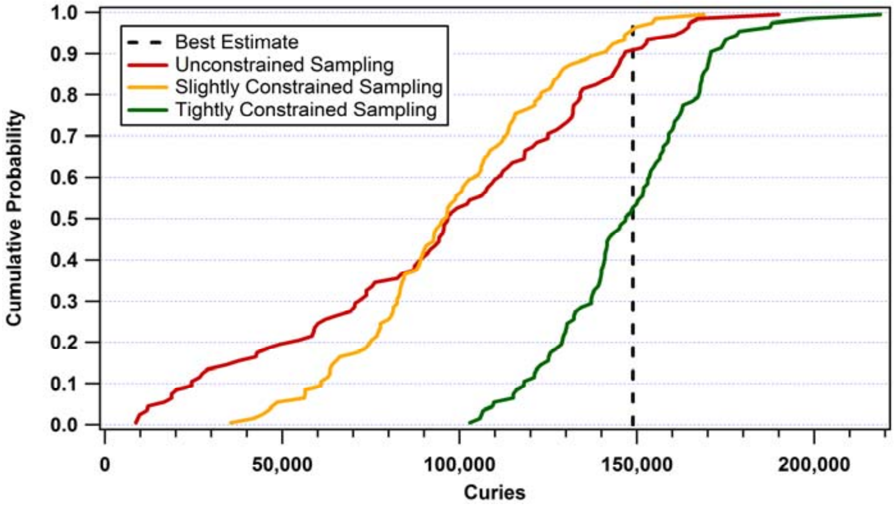


Figure 4.15. Decay-Corrected Cumulative Release of Tritium from the Vadose Zone to the Groundwater by the Year 2001 for Four Difference Sampling Cases

The effect that the constrained sampling cases have on the size of tritium groundwater plumes exceeding the drinking water standard is shown in Figure 4.16.

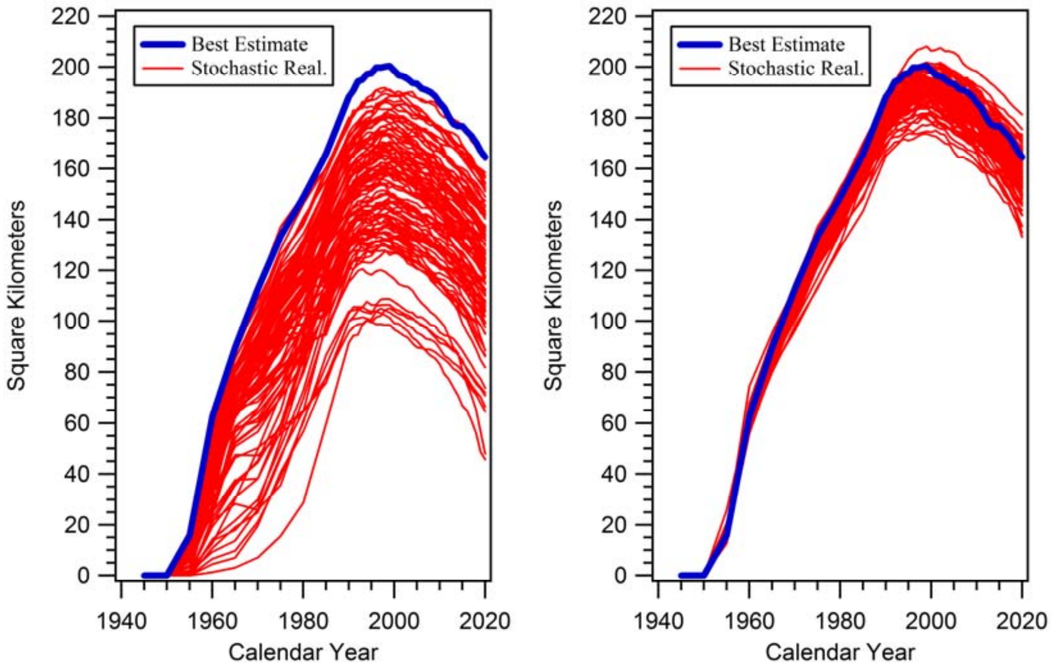


Figure 4.16. Modeled Size of the Tritium Groundwater Plume Where Concentrations Exceed the Drinking Water Standard (Left pane: plume sizes from a stochastic run using slightly constrained vadose zone parameters. Right pane: plume size using tightly constrained vadose zone parameters.)

Modeled sizes of the tritium groundwater plume where concentrations exceed the drinking water standard are plotted against cumulative release of tritium to the groundwater for the year 2001 in Figure 4.17. These plots show that the dominant factor in groundwater plume size variation is the release of tritium from the vadose zone to the groundwater.

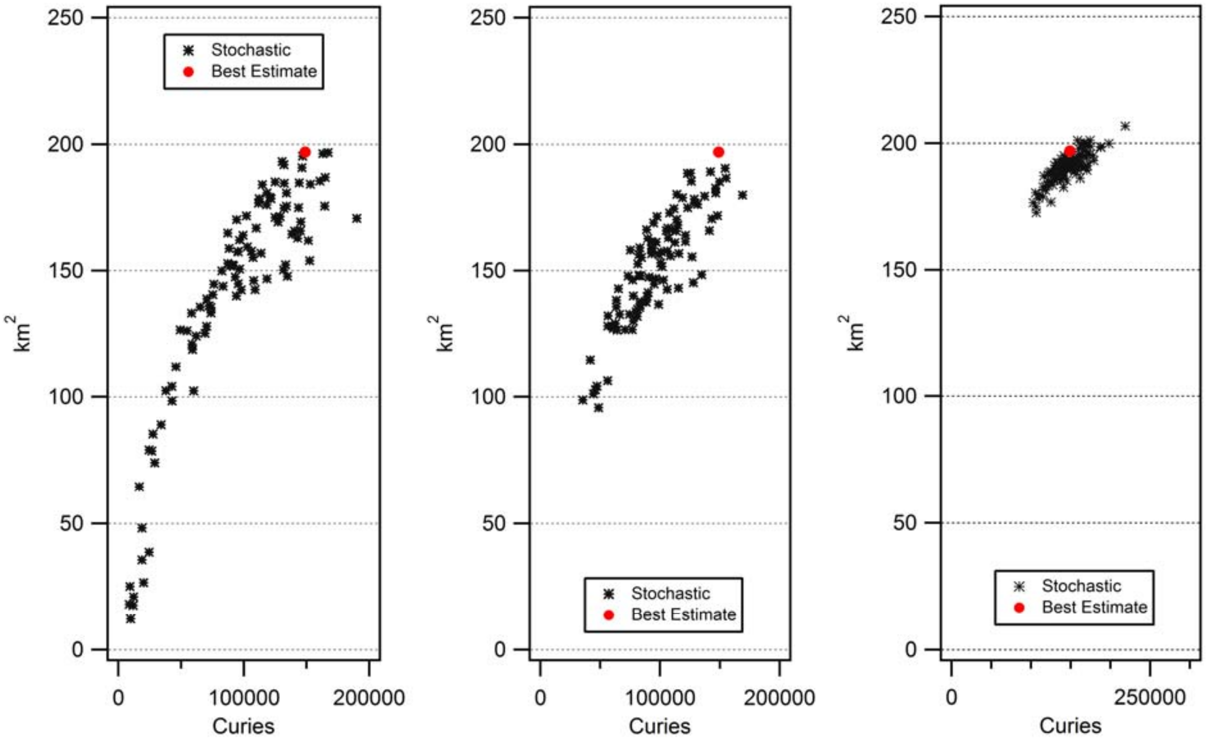


Figure 4.17. Modeled Sizes of the Tritium Groundwater Plume Where Concentrations Exceed the Drinking Water Standard Plotted Against Cumulative Release of Tritium to the Groundwater for the Year 2001 (Panes from left to right: data from the unconstrained, slightly constrained, and tightly constrained cases)

Consideration of the available field data, the modeling results, and the three performance metrics leads to the recommendation that tightly constrained sampling be implemented for vadose zone model parameters for all stochastic runs. Groundwater plume size for tritium is highly dependent on the prior cumulative release of tritium from the vadose zone to the groundwater. The unconstrained parameter case has a large number of realizations where the groundwater plume either does not become large enough or takes too long to develop initially. The slightly constrained parameter case yields better results but still has 6 or 7 of the 100 realizations where the groundwater plume size is still unreasonable, either from total size or initial time of plume development.

4.3 Uncertainty Analysis Examples

The SAC suite of codes utilizes a Monte Carlo technique to allow examination of the uncertainty in the output performance measures. This approach allows examination of the effect of parameter uncertainty within any single run of the codes. Examination of conceptual model uncertainty can partly be addressed by performing multiple runs of the code. Several examples of the types of uncertainty measures available are presented in this section.

4.3.1 Examples of Uncertainty in the Release to the Vadose Zone

The SAC suite of codes was designed to facilitate modeling of the release of one or more analytes to the vadose zone, or in some instances, direct discharge to the Columbia River (such as releases associated with operation of single-pass reactors) at a large number of simultaneous sites in the presence of uncertainty in the inventory and release model parameters for several concurrent release models (for example, salt cake dissolution, soil-debris, and liquid release models).

To illustrate the uncertainty in the release component of the SAC suite of codes, an analysis was performed on the cumulative release of technetium-99 to the vadose zone by the year 2040 from all solid waste forms. The annual release histories for the 100 realizations are shown in Figure 4.18. The cumulative probability of cumulative release of technetium-99 through calendar year 2040 is shown in Figure 4.19.

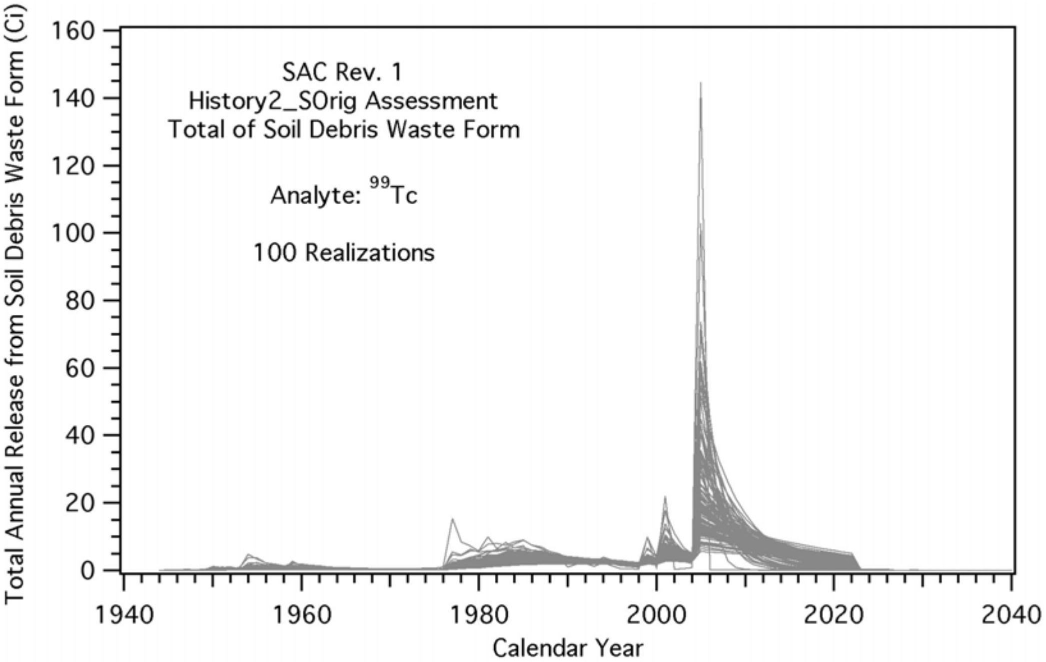


Figure 4.18. Sample Annual Releases of Technetium-99 to the Vadose Zone for All Solid Waste Forms Simulated in 100 Realizations

A further analysis of the modeled cumulative release of technetium-99 to the vadose zone by the year 2040 considered only sites with an inventory of technetium-99 assigned to a soil-debris waste form (a total of 26 sites). The dominant source of variability was the variability in the quantity of inventory assigned to soil-debris waste forms. Specifically, 50.0% of the variability in the release to the vadose zone from soil-debris waste forms was due to the variability in the inventory. Another 12.9% of the variability of the release to the vadose zone was due to the variability in the soil-debris model sampled parameter TW-2, 7.1% was due to uncertainty in infiltration, and 0.5 % was due to variability in soil-debris sampled parameter TW-4. The TW parameters are the volumetric water content fraction for the numbered class (in this case, values sampled for classes 2 and 4 were contributors to the output variability). The remainder (29.7%) of the variability was unaccounted for by a linear regression model.

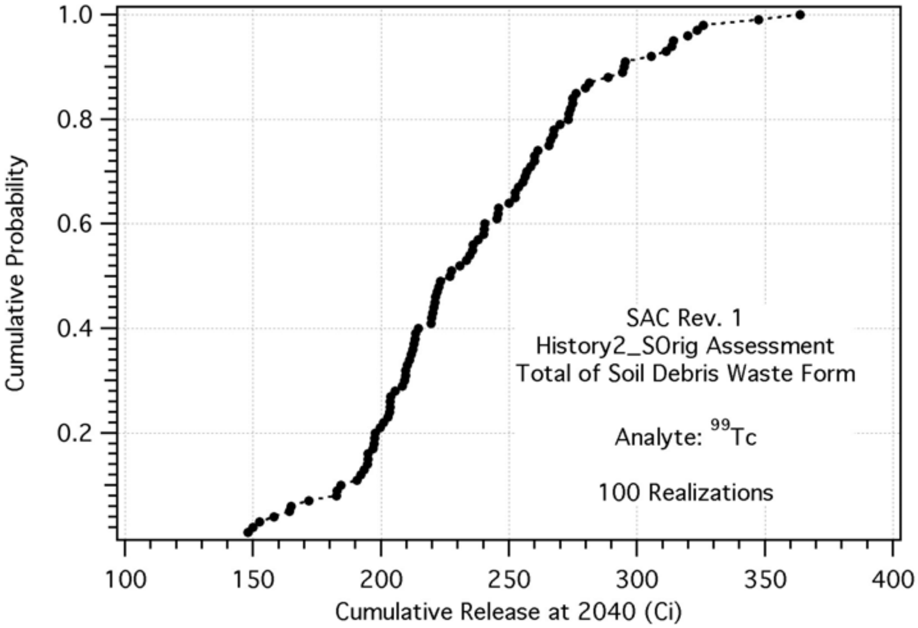


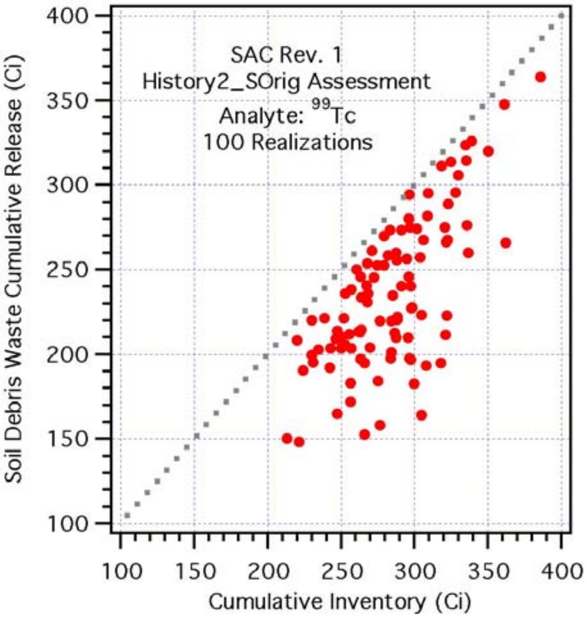
Figure 4.19. Sample Cumulative Distribution of Technetium-99 Release to the Vadose Zone by Year 2040 from All Solid Waste Forms

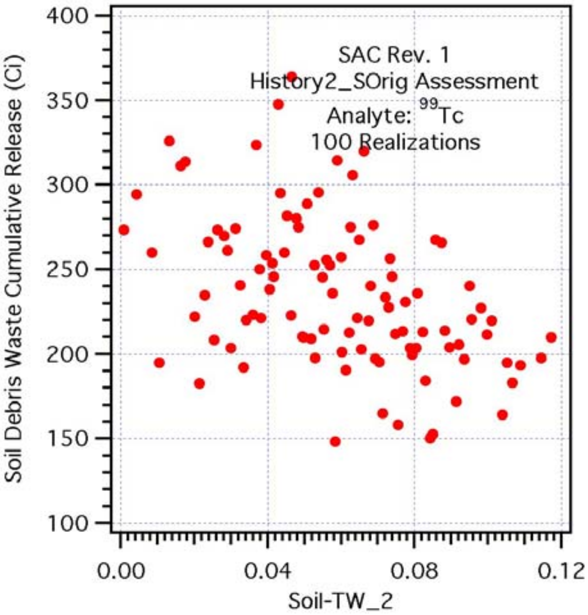
The dependence of cumulative release of technetium-99 from soil-debris waste forms to the vadose zone on the inventory, infiltration, and soil-debris model parameters is shown graphically in Figure 4.20. As expected, higher inventory values lead to higher releases from the vadose zone.

4.3.2 Examples of Uncertainty in Vadose Zone Transport

The SAC suite of codes was designed to facilitate modeling the release of one or more analytes to the groundwater in the unconfined aquifer at a large number of simultaneous sites in the presence of uncertainty in the inventory, release models, and vadose zone hydraulic model parameters. As an example, an analysis was performed on the modeled cumulative release of iodine-129 from the vadose zone to the groundwater by the year 2020. This analysis considered 402 sites with an inventory of iodine-129. Examination of the results showed the dominant sources of variability were the sorption of iodine-129 in the vadose zone and the amount of the inventory that was in liquid form. More specifically, 40.6% of the variability in the release to groundwater was due to the variability in the sorption value for low-organic, low-salt, near-neutral conditions (sorption class 4I1). The sampled sorption values are provided in Figure 4.21. Another 39.8% of the variability of release to groundwater was due to the variability in the inventory of iodine-129 in liquid waste forms. Further information on the sorption data may be obtained from the vadose zone data package (Last et al. 2006a).

The dependence of cumulative release of iodine-129 from the vadose zone to the groundwater on the dominant sorption class and liquid inventory is shown graphically in Figure 4.22. As expected, higher sorption values lead to lower releases from the vadose zone. In addition, more iodine-129 inventory in liquid waste forms leads to more release to the groundwater because the liquid wastes are released immediately and the associated water causes a higher hydraulic gradient in the unsaturated zone. In addition, large-volume liquid releases result in a higher water table, thereby shortening the unsaturated travel path length to the groundwater.


SAC Rev. 1
History2_SOrig Assessment
Analyte: ^{99}Tc
100 Realizations
Soil Debris Waste Cumulative Release (Ci)
Cumulative Inventory (Ci)
100
150
200
250
300
350
400


SAC Rev. 1
History2_SOrig Assessment
Analyte: ^{99}Tc
100 Realizations
Soil Debris Waste Cumulative Release (Ci)
Soil-TW_2
100
150
200
250
300
350
400
0.00
0.04
0.08
0.12

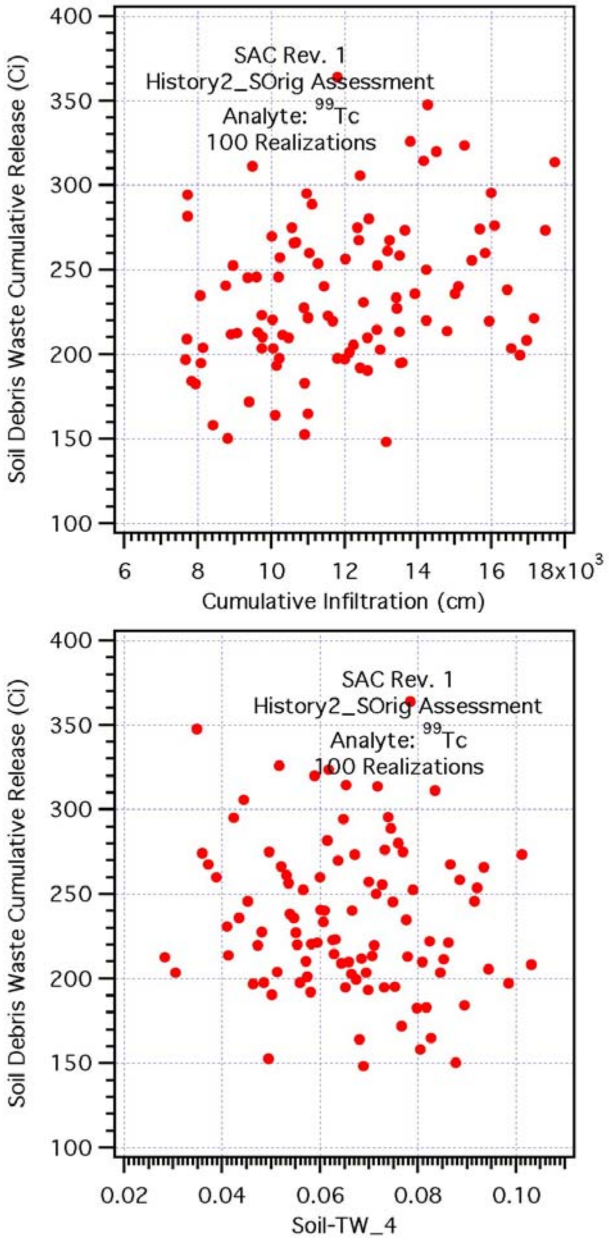


Figure 4.20. Dependence of Cumulative Release of Technetium-99 Through Year 2040 to the Vadose Zone on the Inventory Variability (upper left), Soil-Debris Release Model Parameter TW-2 (upper right), Infiltration (lower left), and Soil-Debris Release Model Parameter TW-4 (lower right)

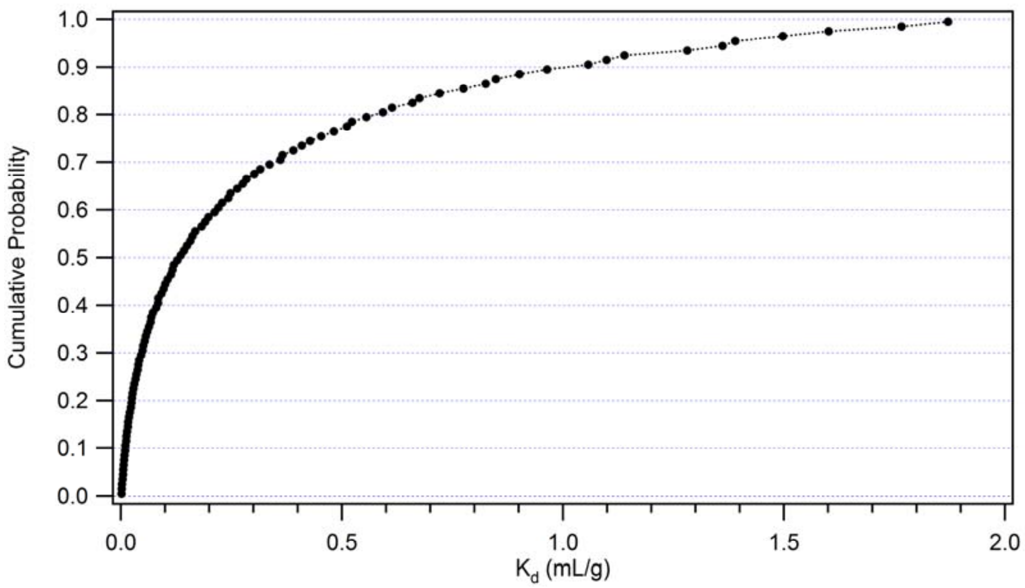


Figure 4.21. Sample Cumulative Distribution of Iodine-129 Sorption Values for Sorption Class 4I1 (low-organic, low-salt, near-neutral conditions)

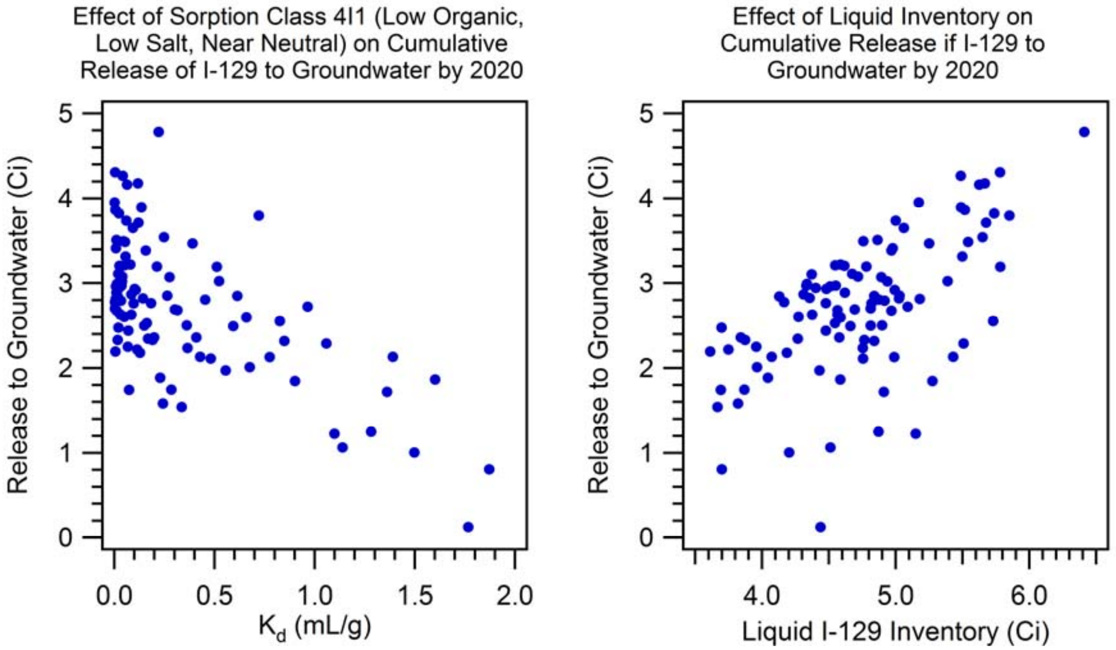


Figure 4.22. Dependence of Cumulative Release of Iodine-129 from the Vadose Zone to the Groundwater on the Dominant Sorption Class (left pane) and the Liquid Inventory (right pane)

The sources of variability in the release of technetium-99 were examined for the same analysis. In this case, several parameters contributed significantly to the variability in the release of technetium-99 to the groundwater. The dominant values were all sorption or hydraulic model parameters for a number of different geologic units. The top variables explaining the variability in release to the groundwater were the following:

- SOR_CT008: Sorption Class 3I1 (Chelates, High Salts)
- SOR_CT007: Sorption Class 3H (Chelates, High Salts)
- SAT_RT011_RES: Residual Saturation Hcs (Hanford coarse sand)
- SAT_RT017_RES: Residual Saturation Hcs_Z (Hanford coarse sand – 200-ZP-1)
- SAT_RT011_EXP: Van Genuchten Exponent Hcs (Hanford coarse sand)
- POR_RT005_DIF: Diffusive Porosity Hss_Z (Hanford silty fine sand – 200-ZP-1)
- SAT_RT002_RES: Residual saturation Hss (Hanford silty fine sand)
- HYD_RT020: Hydraulic Conductivity Hg (Hanford sandy gravel)
- SOR_CT017: Sorption Class 6I1 (IDF LLW cement waste)
- SOR_CT002: Sorption Class 1I1 (Very acidic).

Further information on the sorption data and vadose zone hydraulic model data may be obtained from the vadose zone data package (Last et al. 2006a).

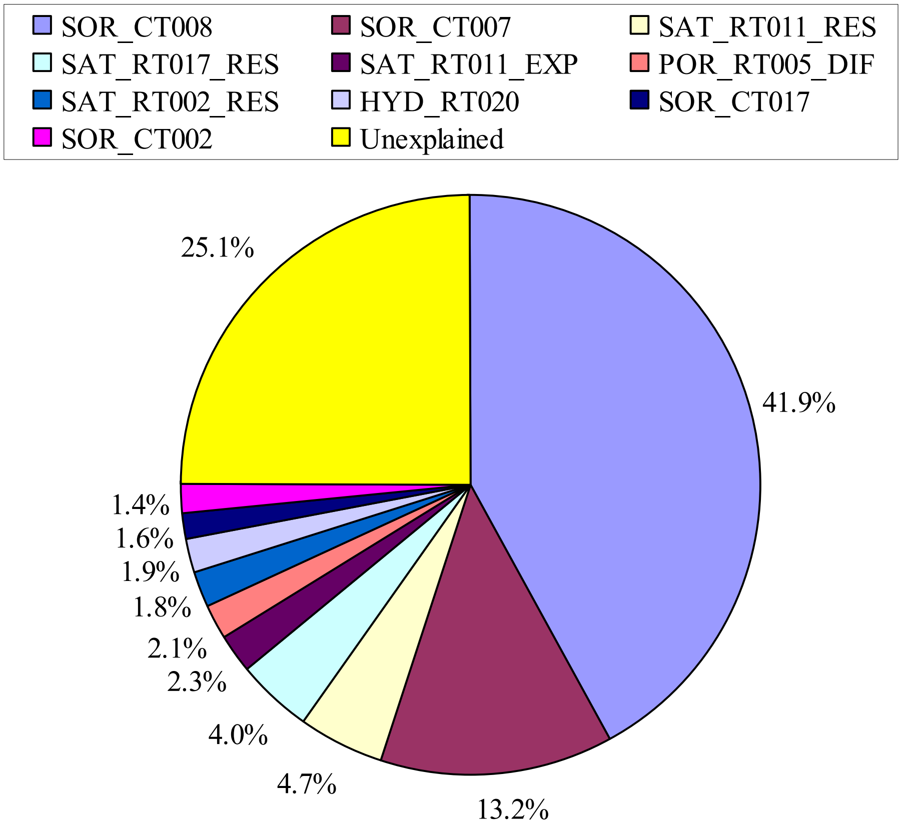


Figure 4.23. Top Variables Explaining the Cumulative Release of Technetium-99 from the Vadose Zone to the Groundwater by 2020

4.3.3 Examples of Uncertainty in Impacts for Ecological Species

The example application of the ecological model provided in Section 3.9 were obtained from a set of transport calculations modeling a low-concentration groundwater plume of technetium-99 intersecting the Columbia River in the 100 Area. Most of the results in that section were based on a single deterministic run. However, calculations were also made for a 100-realization run using stochastic inputs. This section presents some example uncertainty results for an aquatic species (juvenile rainbow trout), a terrestrial species living in the riparian zone along the river bank (egg from a chicken), and an upland species (finch) living in a groundwater-irrigated region for the year 2006.

The juvenile trout eats a small amount of sediment and is assumed to achieve 100% of its dietary needs by eating Daphnia magna (2%), Hyallela (2%), mayfly (94%) and periphyton (2%). The calculated dose to the juvenile trout ranges from 1.3E-7 to 5.3E-7 rad/day and has a median of 2.6E-7 rad/day. The variation in the technetium-99 concentration in mayfly explains 28% of the variation in the trout dose, and the variation in the chemical assimilation efficiency of consumed prey by the rainbow trout explains 30% of the variation in the trout dose. No other single variable explain any significant fraction of the variation in the rainbow trout dose. A representation of the variation in the dose to the juvenile rainbow trout and the dependence of the dose to the trout on the mayfly body burden of technetium-99 are provided in Figure 4.24.

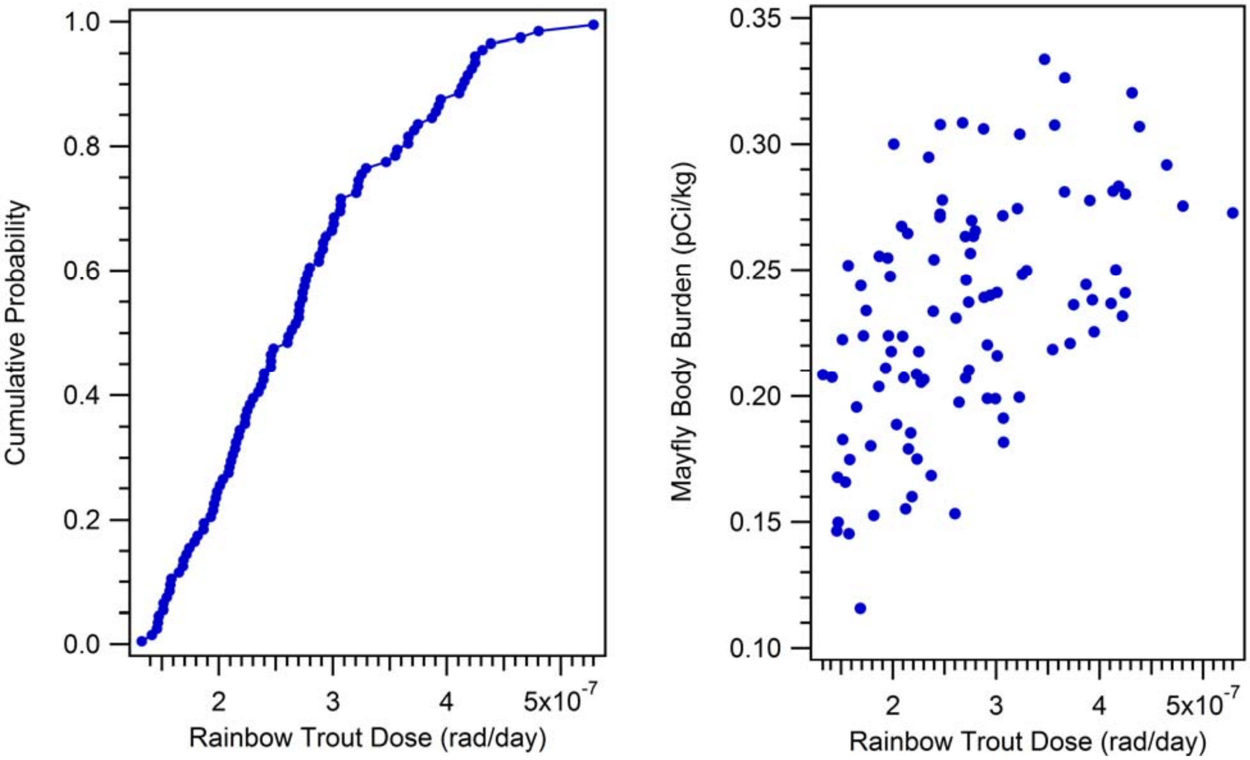


Figure 4.24. Variability in the Technetium-99 Dose to a Juvenile Rainbow Trout (left pane) and a Scatter Plot of Dose to the Trout Against the Variable (Mayfly Body Burden) Explaining the Most of the Variability of the Dose to the Trout (right pane).

A chicken living along the riparian zone that lays eggs is assumed to achieve 100% of its dietary needs by eating earthworms (10%), grain (20%), grapes (5%), grass (10%), leafy vegetables (5%), mulberry (5%), root vegetables (5%), shrubs (5%), terrestrial arthropods (30%), and tree fruit (5%). Also consumed is some soil, and the chicken obtains its water from seeps along the river shore. The calculated concentration of technetium-99 in an egg laid by the chicken ranges from 2.8E-8 to 1.2E-6 pCi/kg and has a median of 1.9E-7 pCi/kg. The variables explaining the most of the variation in the concentration of technetium-99 in the egg are shown in Figure 4.25. In this case, the variability in the transfer of technetium-99 from the chicken to the egg explains more of the total variability in the concentration of technetium-99 in the egg than any other variable.

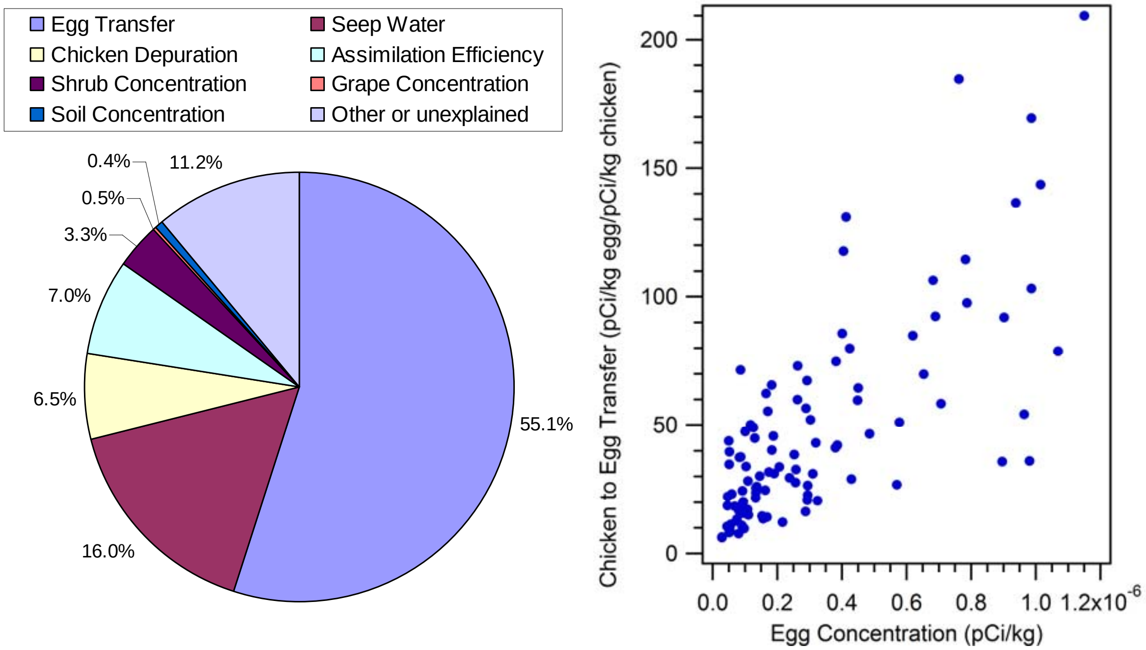


Figure 4.25. Variables Explaining the Variability in the Concentration of Technetium-99 in an Egg from a Chicken Living in the Riparian Zone (left pane) and a Scatter Plot of Concentration of Technetium-99 in an Egg Against the Variable (Chicken-to-Egg Transfer Factor) Explaining the Most of the Variability of the Egg Concentration (right pane).

A finch living among upland fields irrigated with contaminated groundwater is assumed to achieve 100% of its dietary needs by eating grain (20%), grass (25%), mulberry (5%), thistles (15%), shrubs (30%), and tree fruit (5%). Also consumed is some soil, and the finch obtains its water from irrigation sources. The calculated dose from technetium-99 to the finch ranges from 2.2E-6 to 1.2E-4 rad/day and has a median of 1.9E-5 rad/day. The variables explaining the most of the variation in the dose to the finch from technetium-99 are the following: chemical assimilation efficiency for contaminants ingested with food (23%), concentration in groundwater (17%), depuration rate (10%), and concentration of technetium-99 in grain (5%).

These uncertainty results are example results for a single contaminant. Depending on the habitat and the species, the variables dominating the variability in the calculated tissue concentration or dose to the ecological species can be either environmental concentrations or ecological model parameters. Other examples not presented show that the most important variables for explaining variability also depend on the particular analyte being modeled.

4.3.4 Examples of Uncertainty in Human Dose

An example examination of the sources of variability in the total dose to a residential farmer as a function of time is provided in Figure 4.26 for a model run using 100 realizations. The dose is based on a residential farmer scenario at a location where plumes of tritium (^{3}H), iodine-129 and technetium-99 intersect. The farm uses clean sources of irrigation water through 2005 and then switches to using groundwater for irrigation in 2006. Before the onset of irrigation, the variability in the farmer dose was dominated by the variability in the tritium dose. After the onset of irrigation, about 80% of the variation in farmer dose is explained by the variation in the dose from iodine-129. Although not displayed in this plot, the dose increases about an order of magnitude after the onset of irrigation.

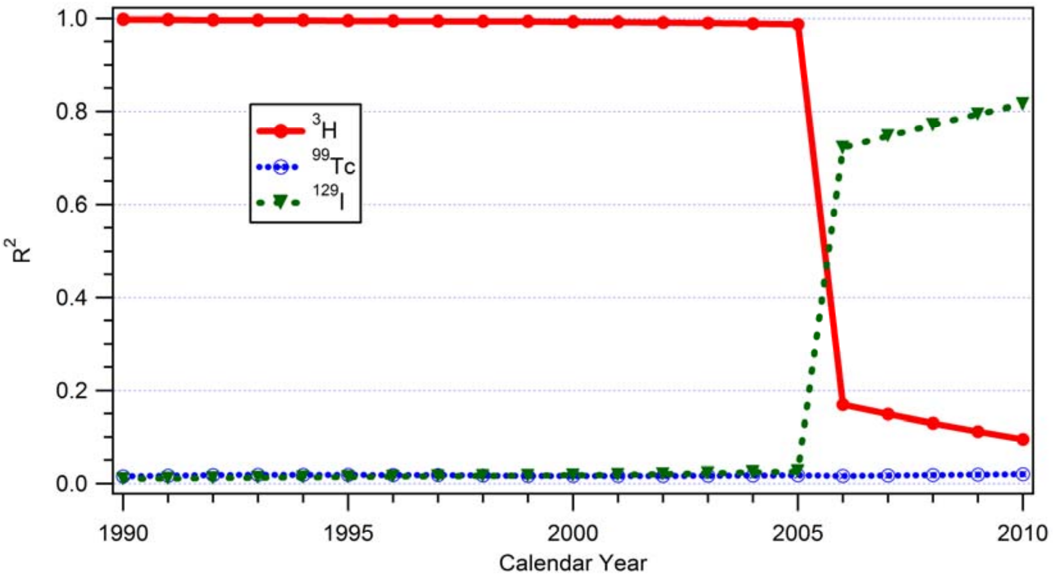


Figure 4.26. Fraction of Variability (R^2) in the Total Dose to a Residential Farmer at an Upland Location Attributable to the Dose from Individual Nuclides

In a second-tier analysis using this same set of data but focusing on the year 2009, approximately 99.9% of the variability of the dose from iodine-129 is attributable to the variability in the ingestion dose. In a third-tier examination, about 24.4% of the variability in ingestion dose is attributable to the dose factor for ingestion. Another 19% of the variability in ingestion dose is due to the variability of the concentration of iodine-129 in eggs. and 5.4% is due to the variability in the ingestion rate of eggs.

As a final example, consider a scenario where an individual lives near the river shore and obtains most of the individual's food, including fish and milk, from locations along the river shoreline. This scenario does not match with any standard Hanford scenario but supports consideration of the variability in human dose when all ingestion and external pathways are simultaneously invoked. A stochastic representation of the technetium-99 dose for this scenario is provided in Figure 4.27. The groundwater concentrations are calculated at the same locations as the example applications shown in Sections 3.9 and 3.10. Also shown in Figure 4.27 is the effect of the ingestion dose factor for technetium-99 on the total dose. The top contributors to the variability in the dose are the following:

- 62.7% – Dose factor for ingestion
- 18.5% – Concentration of technetium-99 in seep water

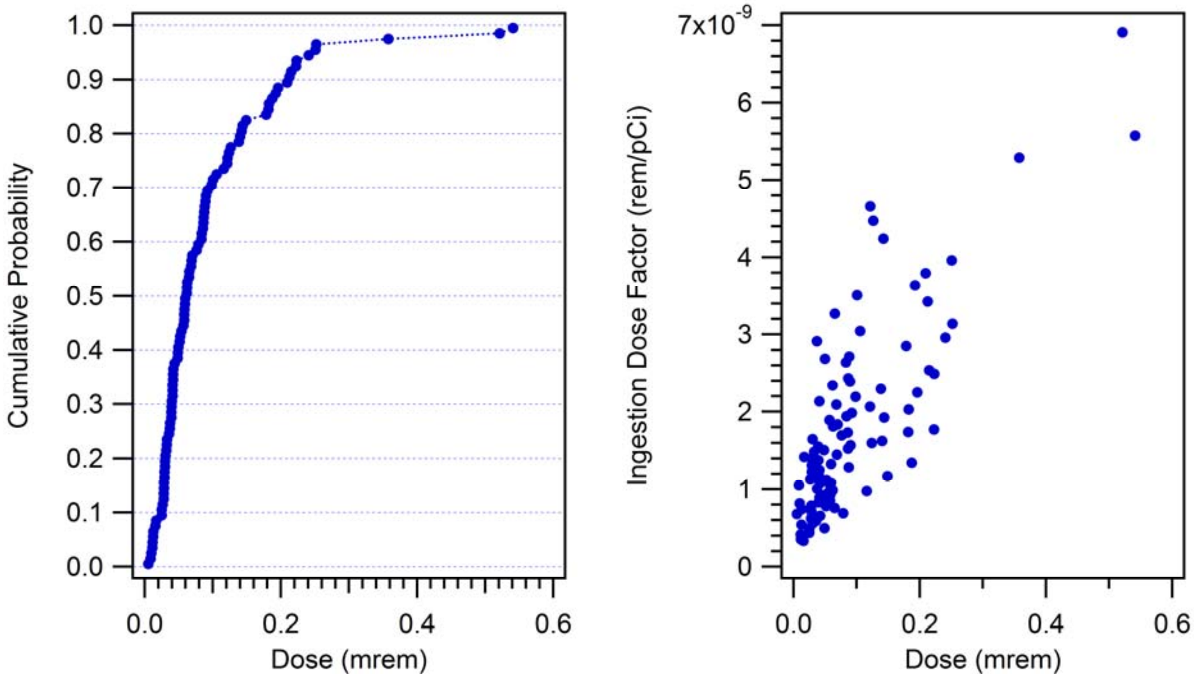


Figure 4.27. Left pane: Stochastic Representation of the Technetium-99 dose to a Human Obtaining All Food Products from a Contaminated River Shore Location. Right pane: Effect of the Ingestion Dose Factor for Technetium-99 on the Total Dose

- 4.4% – Concentration of technetium-99 in milk
- 1.9% – Ingestion rate for seep water
- 1.3% – Ingestion rate of milk
- 0.7% – Exposure frequency to seep water
- 10.6% – Other sources.

4.4 Dose and Risk Example Using the HUMAN Code in Standalone Mode

One benefit of the modular design of the SAC is the ability to exercise a subset of the codes in a limited analysis. This section presents an example case where the desired end result is the dose and risk to a human from a number of different contaminants for two different exposure scenarios. This type of analysis has already been applied to determining the risk to humans relative to cleanup actions required under CERCLA.

For this example, solutions are desired for six contaminants. The contaminants and specific solutions desired are the following:

- technetium-99 (CAS 14133-76-7) – dose (mrem) and risk of cancer incidence
- iodine-129 (CAS 7553-56-2) – dose (mrem) and risk of cancer incidence
- uranium-235 (CAS 15117-96-1) – dose (mrem) and risk of cancer incidence
- uranium (CAS 15117-96-1) – hazard quotient

- trichloroethene (TCE) (CAS 79-01-6) – risk of cancer incidence and hazard quotient
- hexavalent chromium or Cr(VI) (CAS 7440-47-3) – risk of cancer incidence and hazard quotient.

The sequence of calculations in this example uses several computer codes identified in Figure 2.2. In addition, two utility codes not referenced in that figure are used to set up and initialize data files needed by the other codes. The sequence of code runs is the following:

- ECDA: Utility code to define the ECDA files for each contaminant
- FILLECDA: Utility code to initialize the contents of the ECDA files to specified values. In this example, water concentrations are set to 1 pCi/L for the radionuclides and 1 µg/L for the other contaminants.
- SOIL: Model to calculate soil concentrations assuming irrigation for growing food crops
- ECEM: Model to calculate concentrations in food products to be consumed in the human risk scenario
- HUMAN: Model to calculate dose and risk to humans for the different exposure scenarios.

This example does not require the computational power of the entire computer cluster described in Section 2.4. All of the calculations in this example were completed in less than five seconds on a desktop computer.

4.4.1 Scenario Definitions

The first scenario is a drinking water scenario. The operational definition has an adult drinking 2 L/day of contaminated water.

The second scenario is the residential farmer scenario defined in Miley et al. (2006). Specific parameters in the scenario definition are provided in Table 4.11. The selected parameters differ slightly from, but are consistent with, those of Rittmann (2004) and Thatcher (2003). They are also of the same magnitude as those recommended for an agricultural screening scenario developed by the National Council on Radiation Protection and Measurements (NCRP 1999). Final exposure parameter selections were made after also considering the recommendations and guidance in the following (and additional) documents:

- The Agricultural Resident Scenario defined in the HSRAM (DOE 1995)
- The Washington State Department of Health (WDOH 1997) input parameters for the Agricultural Resident Scenario. The advantage of the WDOH set of parameters is that the environmental parameters are all related to the Hanford Site.
- The Hanford Reach National Monument Agricultural Resident Scenario defined in Napier and Snyder (2002).

Table 4.11. Key Exposure Parameters for the Residential Farmer Scenario

Pathway Parameters	Value
Soil Exposure	
External exposure, hr/d	16
Shielding	0.8
Soil ingestion, g/d	0.12
Soil exposure, d/yr	317
Shore External Exposure	
Swimming, hr/d	None
Boating, hr/d	None
Sediment exposure, hr/d	None
Inhalation	
Breathing rate, m^3/d	20
Soil mass loading, g/m^3	0.00005
Volatiles from water, hr/d	0.17
Breathing exposure, d/yr	317
Air exposure, hr/d	18
Water Ingestion	
Groundwater ingestion, L/d	1.5
Drinking exposure, d/yr	317
Food Ingestion	
Food exposure, d/yr	365
Leafy vegetable, kg/yr(a)	2.7
Other vegetables, kg/yr(b)	73
Fruit, kg/yr(b)	37
Grain, kg/yr(b)	0
Milk, L/yr	100
Meat, kg/yr	30
Fowl, kg/yr	6
Eggs, kg/yr	6.8
Fish, kg/yr	None

(a) As suggested by WDOH (1997)
(b) Value of 110 kg/year from WDOH (1997) apportioned 2/3 to other vegetables, 1/3 to fruits, and 0 to grains, as suggested by Rittmann (2004).

4.4.2 Soil Calculations

The concentrations of analytes in soil are based on a time-dependent model (Eslinger et al. 2006a, Volume 1) of soil accumulation that was briefly described in Section 3.8. The model used 76 cm of irrigation water applied over a six-month growing period. A net fraction of 20% of the irrigation water is assumed to infiltrate deeper that the root zone. The calculated soil concentrations provided in Table 4.12 are representative of the concentrations after 50 years of crop irrigation. They are also representative of steady-state conditions because steady-state conditions are achieved within five years of the start of irrigation for the sorption values of all analytes in this example.

Table 4.12. Soil Concentrations in Irrigated Soil

Contaminant	K_d (mL/g)	Soil Concentration	Soil Units
^{99}Tc	0.0	1.631	pCi/kg
^{129}I	0.2	2.608	pCi/kg
^{235}U	0.8	5.532	pCi/kg
U	0.8	5.532	μg/kg
TCE	1.82	2.520	μg/kg
Cr(VI)	0.0	1.631	μg/kg

4.4.3 Food Product Calculations

Concentrations of analytes in food products were calculated using the ECEM (Ecological Contaminant Exposure Model) (Eslinger et al. 2006b, Volume 2) that was briefly described in Section 3.9. The model uses a food web approach based on chronic exposures to estimate the movement of analytes into foods consumed by humans. In this case, a total of sixteen species were modeled to represent both the primary foods for human consumption and the secondary species needed to fill out a food web for the primary foods. The species are the following: terrestrial arthropods, cattle (milk), cattle (meat), chickens (adults), chickens (eggs), earthworms, fungi, grain, grapes, grasses, leafy vegetables, mulberry, root vegetables, shrubs, Russian thistle, and tree fruit. Calculated concentrations for human foods are provided in Table 4.13.

Table 4.13. Calculated Contaminant Concentrations in Food Products

Radioactive Contaminants				
Food	^{99}Tc	^{129}I	^{235}U	Units
Bird (Chicken)	4.732	15.15	16.44	pCi/kg
Eggs (Chicken)	177.5	4544	32.87	pCi/kg
Fruit	15.59	15.03	15.05	pCi/kg
Meat	2.287	15.52	5.305	pCi/kg
Milk	38.28	8.173	5.937	pCi/L
Leafy Vegetables	17.09	13.78	13.71	pCi/kg
Root Vegetables	16.90	16.54	16.55	pCi/kg
Chemical Contaminants				
Food	Cr(VI)	TCE	U	Units
Bird (Chicken)	0.1732	6.841	16.435	μg/kg
Eggs (Chicken)	0.7795	2.736	32.87	μg/kg
Fruit	15.05	15.80	15.047	μg/kg
Meat	0.05821	2.463	5.305	μg/kg
Milk	0.000149	0.6183	5.937	μg/L
Leafy Vegetables	13.70	14.45	13.71	μg/kg
Root Vegetables	16.54	17.30	16.55	μg/kg

Although all six of these contaminants bioaccumulate at very similar rates in plants, they act very differently in animal species. For example, iodine-129 concentrates the most in eggs while technetium-99 concentrates the most in milk. In contrast, little of the Cr(VI) accumulates in meat or milk.

4.4.4 Dose and Risk Factors

Dose factors for ingestion and inhalation of radionuclides were obtained from Federal Guidance Report 11 (EPA 1988). Dose factors from external exposures from radionuclides were obtained from Federal Guidance Report 12 (EPA 1993). Risk factors for radionuclides for both dietary and drinking water intake were obtained from Federal Guidance Report 13 (EPA 2002).

Data for reference doses were obtained from the references in Table 4.14.

Table 4.14. Reference Doses for Inhalation and Ingestion of Hazardous Materials

Analyte	Modifier	Description	Value[(a)]	Reference
U	RFDINH	U reference dose for inhalation	4.43E-5	ACGIH[(b)]
U	RFDING	U reference dose for ingestion	6.00E-4	RAIS[(c)]
TCE	RFDING	reference dose for ingestion	3.00E-4	RAIS
TCE	RFDINH	reference dose for inhalation	1.14E-2	RAIS
Cr(VI)	RFDING	reference dose for ingestion	2.86E-5	RAIS
Cr(VI)	RFDINH	reference dose for inhalation	3.00E-4	RAIS

(a) The units for all values in this table are mg(intake)/(kg(bodyweight)/day).

(b) Threshold limit values from ACGIH – American Conference of Industrial Hygienists, web site http://www.acgih.org/home.htm

(c) RAIS (the Risk Assessment Information System), web site http://risk.lsd.ornl.gov/cgi-bin/tox/TOX_select?select=csf

Data for slope factors for carcinogenic materials were obtained from the references in Table 4.15.

Table 4.15. Slope Factors for Carcinogenic Materials

Analyte	Modifier	Description	Value[(a)]	Reference[(b)]
TCE	SFINH	TCE slope factor for inhalation	0.4	RAIS
TCE	SFING	TCE slope factor for ingestion	0.4	RAIS
Cr(VI)	SFINH	Cr(VI) slope factor for inhalation	42	RAIS

(a) The units for all values in this table are risk per mg(intake)/(kg(bodyweight)/day)

(b) RAIS (the Risk Assessment Information System), web site http://risk.lsd.ornl.gov/cgi-bin/tox/TOX_select?select=csf

4.4.5 Results

The dose and risk factors resulting from unit concentration of each contaminant in the water used for drinking and irrigation are provided in Table 4.16 for both the drinking water and residential farmer scenarios. The radionuclide doses are generated using the methodology provided in Federal Guidance Report 11 (EPA 1988) and Federal Guidance Report 12 (EPA 1993). The radionuclide risks are presented for cancer incidence using the methodology provided in Federal Guidance Report 13 (EPA 2002).

Table 4.16. Dose, Hazard Quotient, and Cancer Incidence Risk Results

Analyte	Type	Water Conc.	Water Units	Farmer Scenario	Drinking Water Scenario	Units for Scenario Results
^{99}Tc	Radionuclide	1	pCi/L	1.09E-02	1.07E-03	mrem
^{129}I	Radionuclide	1	pCi/L	9.54E+00	2.01E-01	mrem
^{235}U	Radionuclide	1	pCi/L	9.05E-01	1.94E-01	mrem
^{99}Tc	Radionuclide	1	pCi/L	2.98E-08	2.01E-09	risk
^{129}I	Radionuclide	1	pCi/L	6.66E-06	1.08E-07	risk
^{235}U	Radionuclide	1	pCi/L	3.20E-07	5.08E-08	risk
TCE	Carcinogen	1	µg/L	5.76E-07	1.63E-07	risk
TCE	Hazardous	1	µg/L	4.79E-03	1.36E-03	none
Cr(VI)	Carcinogen	1	µg/L	1.05E-07	0.00E+00	risk
Cr(VI)	Hazardous	1	µg/L	5.14E-04	1.36E-04	none
U	Hazardous	1	µg/L	3.18E-03	6.80E-04	none

4.5 Risk Ranking for Central Plateau Waste Sites

Risk calculations can aid the decision making process for selecting which waste sites should be cleaned up and what remedial activities should be imposed. The Hanford Remediation Assessment Project performed a suite of calculations to better understand the relative threat to the unconfined aquifer from waste sites in the vadose zone of the Central Plateau region of the Hanford Site. The inventory, contaminant release into and from the vadose zone, and hypothetical concentrations in groundwater are all used to rank selected radioactive analytes based on the threat posed to the aquifer by individual waste sites and groups of waste sites. In addition, an analysis of cumulative dose from four analytes (technetium-99, iodine-129, and uranium-234/238) is used to rank the waste sites and groups of waste sites. Details of the analysis are documented by Kincaid et al. (2006), and only summary information is provided here.

The first simulated case is a *reference or action* case that incorporates a suite of remedial actions consistent with planning done by the U.S. Department of Energy (DOE) in the summer of 2005 and key parameters defined in concert with a Hanford Site configuration management board that existed during fiscal year 2005. An example of the impact of these remedial actions and key parameter values is the application of evapotranspiration barriers (ET caps) on virtually all waste sites identified as receiving surface barriers and the assignment of an infiltration rate of 0.5 mm/yr for 500 years to describe their performance during their design life. Configuration management across several analyses under way in fiscal year 2005 also dictated the long-term post barrier infiltration rates applied in the analysis to be 1.0 mm/yr for all waste sites receiving barriers except the Integrated Disposal Facility, which applied 0.9 mm/yr. From the *reference or action* case files, waste site inventory data and vadose zone analyses were mined to produce the data necessary to rank waste sites and contaminants.

The second simulated case is a *no-action* case designed to analyze waste sites that are the responsibility of Fluor Hanford, Inc., the Project Hanford Management Contractor, and omit, where possible, waste and waste sites that are the responsibility of others including CH2M HILL Hanford Group, Inc. (the Tank Farm Contractor), and Washington Closure Hanford, LLC (the River Corridor Contractor). The *no-action* case includes past disposals and discharges to better understand the threat they pose. Included are past tank leaks, liquid discharges, solid waste disposals, and unplanned releases. In addition to past releases to the vadose zone, the analysis includes nominal inventories that are scheduled for discharge and disposal after 2007. A *no-action* case implies a cutoff of activity (no remedial actions after a certain date), and 2007 was selected as the cutoff of activity because it was the planned date for opening the integrated disposal facility (IDF). However, comparison of a *no-action* case and a *reference* case is intrinsically valuable, and the *reference* case includes some nominal discharge and disposal after 2007. Accordingly, the inventory employed in both analyses is identical and includes discharges to the State-Approved Land Disposal System (SALDS; waste site 600-211) facility and disposals to open solid waste burial grounds after 2007. Any substantial inventories after 2007 that influence the rankings will be noted.

Waste sites and inventories excluded from the *no-action* case are (a) the Environmental Restoration Disposal Facility; (b) the commercial low-level radioactive waste disposal site operated by US Ecology, Inc.; (c) the Integrated Disposal Facility, which is to receive low-activity waste from tank farms; (d) the graphite cores of the production reactors; (e) naval reactor compartments[1]; (f) buildings; (g) PUREX tunnels; (h) waste sites in the 100, 300, and 400 Areas; (i) waste sites in the 600 Area located off of the Central Plateau; and (j) all atmospheric releases[2]. Infiltration rates for the operational era are those used in the *reference* case; however, for waste sites that received a surface barrier in the *reference* case no such barrier will be constructed in the *no-action* case. Exceptions are waste sites that have a surface barrier at this time, (for example, 216-B-57 received a Hanford Barrier in 1994). Waste sites assigned a surface barrier in the *reference* case are assigned for the *no-action* case an infiltration rate in the long term consistent with vegetation on a disturbed and unremediated operational surface.

The waste sites of interest in the *no-action* case are the only waste sites of interest in this study. They are either the responsibility of Fluor Hanford, Inc., or, if the responsibility of others, they are believed to have impacted groundwater in the past or have the potential to impact groundwater in the future (past tank leaks, for example). Thus the *reference* and *no-action* cases address only the list of waste sites prepared for the *no-action* case, and they are the only waste sites ranked in this study.

An evaluation of rankings for these two cases provides insight into the robustness of rankings created by proposed remedial actions and by ceasing remedial actions in 2007. That is, the analysis will show if rankings are altered substantially by implementing the proposed remedial measures versus doing nothing. Some of the analysis results (for example, the contaminant flux from the vadose zone as a function of time) may also yield a qualitative understanding of the reduction in release rate achieved by implementing proposed remedial measures. The results include some counterintuitive information. One might expect that the *no-action* case would always show greater impacts throughout time. However, the *no-action* case cuts short the operational era for many waste sites, and the operational era has high infiltration rates at waste sites that are maintained vegetation-free. Accordingly, releases to the groundwater aquifer may be higher in the *reference* case than in the *no-action* case in the next 100–300 years because of the introduction of greater amounts of infiltration during a longer operational era. Eventually the *no-action* case will appear worse because of higher long-term infiltration rates caused by the absence of surface barriers.

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[1] Inventories for naval reactor compartments are included; however, release from this waste form is so slow that releases to groundwater are negligible. Thus naval reactor compartments are omitted effectively from the study.

[2] Atmospheric releases are included to realistically portray the depletion of inventories that split between atmospheric and vadose zone pathways; however, the impacts of atmospheric releases are not evaluated.

4.5.1 Waste Sites Identified for Analysis and Waste Site Groupings

The waste sites of interest in this study are only those Central Plateau waste sites that could threaten the Hanford Site unconfined aquifer and for which Fluor Hanford, Inc., has contractual responsibility. Waste streams and waste sites that are the responsibility of others are omitted where possible. Among those omitted are waste streams and sites of the tank farm contractor CH2M HILL Hanford Group, Inc. and the river corridor contractor, Washington Closure Hanford, LLC. The *no action* case includes past disposals and discharges to better understand the threat they pose. Included are liquid discharges, solid waste burials, and unplanned releases. Because of their presence in the deep vadose zone, past tank leaks are also included in the study. In addition to these past releases to the vadose zone, the analysis includes ongoing liquid discharges to the State-Approved Land Disposal Site (SALDS, 600-211), and some ongoing solid waste disposals. These are included to make inclusion of a *reference* case that continues after present day more meaningful.

Based mostly on the contractual responsibility, but partly on the availability of inventory information, a total of 410 waste sites were identified for inclusion in the analysis. In addition to ranking individual waste sites, it is useful to rank groups of waste sites that are located in the same geographic area. Cribs, trenches, and tanks that received similar wastes were constructed in groups, and they are managed as a group. For example, the BY cribs are 7 cribs co-located north of the BY Tank Farm; the BX trenches are 8 specific retention trenches located west of the BX Tank Farm; the BC cribs and trenches are 6 cribs and 20 specific retention trenches co-located south of the 200 East Area and east of the commercial low-level radioactive waste site operated by US Ecology, Inc. Because remedial action decisions will be made for the group, it is logical to describe remedial action and closure in terms of these groups rather than individual waste sites, and it is logical to rank groups instead of individuals. Accordingly, the waste sites included in this analysis were grouped into 32 groups to better assist decisions made at the group level. The 32 waste site groupings and the associated groundwater operable units are provided in Table 4.17.

4.5.2 Inventory

Radioactive waste inventories were assembled by Kincaid et al. (2006) for a broader list of waste sites at the Hanford Site. The inventories used in this analysis were drawn from that larger data set. Rankings on the basis of cumulative inventory in curies discharged or disposed to the vadose zone considered the following radioactive analytes: tritium, carbon-14, chlorine-36, selenium-79, strontium-90, technetium-99, iodine-129, cesium-137, europium-152, uranium-233, uranium-235, neptunium-237, and uranium-234/238. The inventory rankings do not distinguish between *reference* and *no-action* cases because the inventories are the same for both cases.

The cumulative inventory from Hanford Site startup in 1944 through the end of 2007 is employed in the ranking of waste sites. An inventory through 2007 includes all past discharges, unplanned releases, and disposals. The planned date for beginning disposal of all solid waste in an Integrated Disposal Facility (IDF) was also 2007. The vast majority of remedial action and decontamination and decommissioning activities on the Central Plateau will occur after remedial investigation and feasibility studies and associated records of decision are issued, and most occur after 2007. The recovery of tank waste, and its treatment and disposal, will occur after 2007. Accordingly, cumulative inventories at the close of 2007 represent a meaningful moment in time for ranking the waste sites or groups of waste sites

Table 4.17. Waste Site Groupings and Their Associated Groundwater Operable Units

Grouping Name	Groundwater Operable Unit
200 East Ponds Region	BP-5
All others in 200-BP-5	BP-5
B Plant cribs & trenches	BP-5
B/BX/BY tank farms	BP-5
BX trenches	BP-5
BY cribs & vicinity	BP-5
C tank farm & vicinity	BP-5
Hot semi-works	BP-5
Solid Waste – 200-BP-5	BP-5
200 East – tank farm related cribs	PO-1
A/AX tank farms	PO-1
All others in 200-PO-1	PO-1
BC cribs and trenches	PO-1
PUREX cribs and trenches	PO-1
Solid Waste – 200-PO-1	PO-1
200 West Ponds – 200-UP-1	UP-1
All others in 200-UP-1	UP-1
REDOX cribs and trenches	UP-1
S cribs	UP-1
S/SX tank farms & vicinity	UP-1
U cribs	UP-1
U Plant cribs and trenches	UP-1
U tank farm	UP-1
All others in 200-ZP-1	ZP-1
PFP cribs & trenches	ZP-1
Solid Waste – 200-ZP-1	ZP-1
T Plant cribs, trenches, swamp	ZP-1
T tank farm & vicinity	ZP-1
T trenches & vicinity	ZP-1
TX trenches	ZP-1
TX/TY tank farms	ZP-1
TY cribs & vicinity	ZP-1

Of the 410 waste sites analyzed for at least one analyte, 97 waste sites contained 99% of the respective total inventory for all 13 analytes. All 97 waste sites were needed to represent 99% of the selenium-79 inventory while 99% of the inventory of chlorine-36 was contained in only two waste sites. As an example, the ranking of the waste site groups for technetium-99 is shown in Figure 4.28.

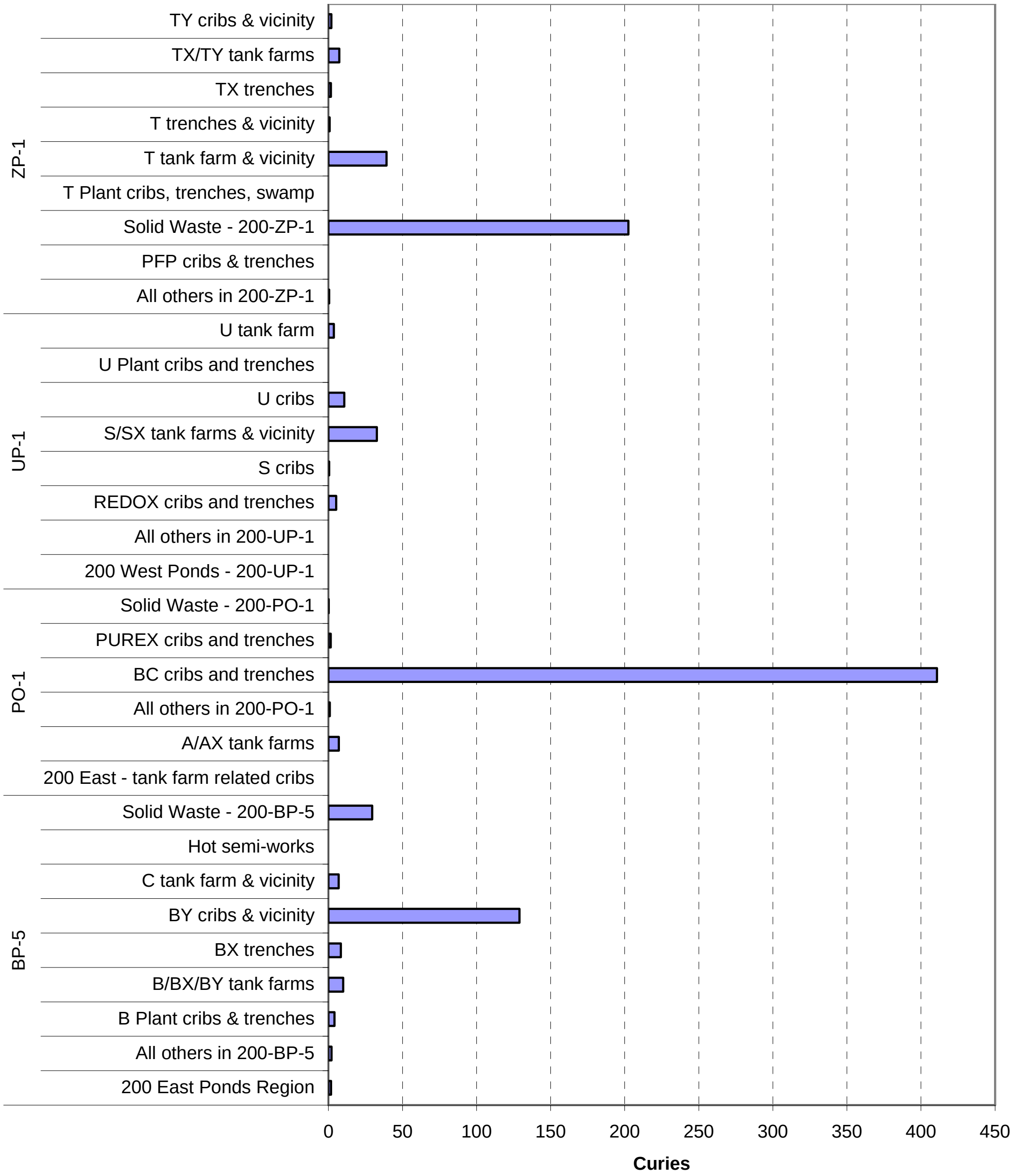

Figure 4.28. Cumulative Inventory of Technetium-99 (decay corrected to 2007) by Site Group

Consistent with the individual waste site ranking, the ranking by waste site groups reveals that solid waste burial grounds ranked first for tritium, carbon-14, chlorine-36, strontium-90, cesium-137, europium-152, and uranium isotopes 234, 235, and 238. Liquid discharge sites receiving operational discharges (such as “PUREX cribs and trenches”) and uranium recovery waste (such as “BC cribs and trenches”) ranked first for selenium-79, technetium-99, iodine-129, and uranium-233. Two unplanned releases from a waste line in the immediate vicinity of the hot semi-works facility in 200 East Area (that is, 200-E-56 and 200-E-57) are noted as having potentially substantial strontium-90, cesium-137, and europium-152 inventories. Another waste site not noted in prioritization discussions held previously is 216-A-9, which is noted as receiving the largest uranium-233 inventory (12.6 Ci).

4.5.3 Vadose Zone

The analyses of inventory, release into the vadose zone, and release from the vadose zone were conducted on an annual basis for each waste site. Inventories were assigned each year; release models were used to simulate release of contaminants into the vadose zone each year; and the vadose zone model was used to simulate the migration of contaminants and their annual release to the water table of the unconfined aquifer each year. Within the various models, each annual release into the vadose zone and into the aquifer is decay corrected to the end of that year. Within this section of the report, results will be presented on the cumulative release in curies that occurs over four time periods: a) 1944–2005, b) 2006-2100, c) 2101–3100, and d) 3101–12000. These time increments represent approximately the past, the next 100 years, the following 900 years, and the following 9,000 years. The end dates of 2100, 3100, and 12000 (AD) roughly correspond to a 100-year period of institutional control, a 1,000-year regulatory period, and a 10,000-year period of interest for radioactive wastes. Figure 4.29 shows an example result for the cumulative release of technetium-99 into the vadose zone.

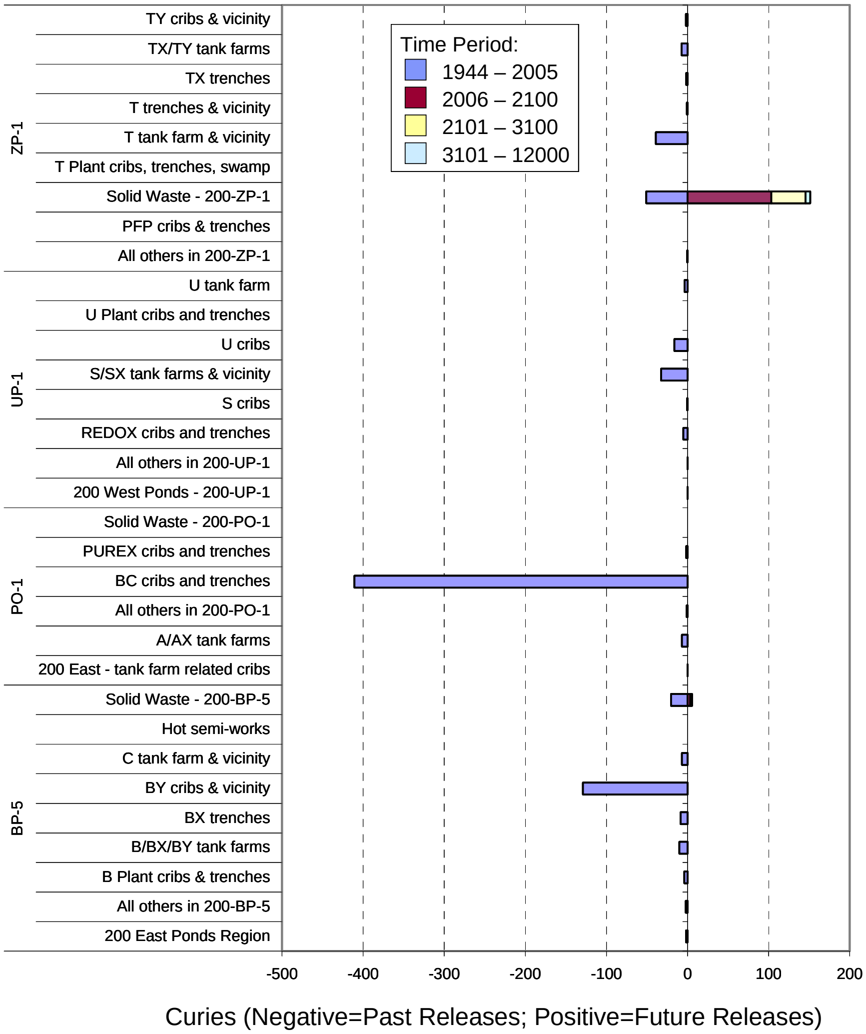

Figure 4.29. Cumulative Release (not decay-corrected) of Technetium-99 into the Vadose Zone by Site Group

For waste sites involving liquid waste discharge, unplanned releases, and past tank leaks, release to the vadose zone is immediate. Releases from solid waste deposits depend on the release model employed, and the assumed integrity of the waste form or waste package. No credit is taken for the waste package in nearly all cases in this analysis; thus, releases can begin immediately upon disposal into solid waste burial grounds. Releases from cement-based waste forms depend on diffusion of contaminant through the cement matrix and are relatively slow. Releases from soil-debris waste forms depend on the solubility, desorption distribution coefficient, and infiltration rate, and are often more rapid, especially for non-sorbed contaminants such as tritium.

Once released into the vadose zone, the waste migrates through the vadose zone and finally reaches the unconfined aquifer. A vadose zone model was developed for every one of the 410 waste sites, and the STOMP code (White and Oostrom 2000) was used to calculate the contaminant transport. Example results for the cumulative release of technetium-99 into the groundwater by site group is provided in Figure 4.30.

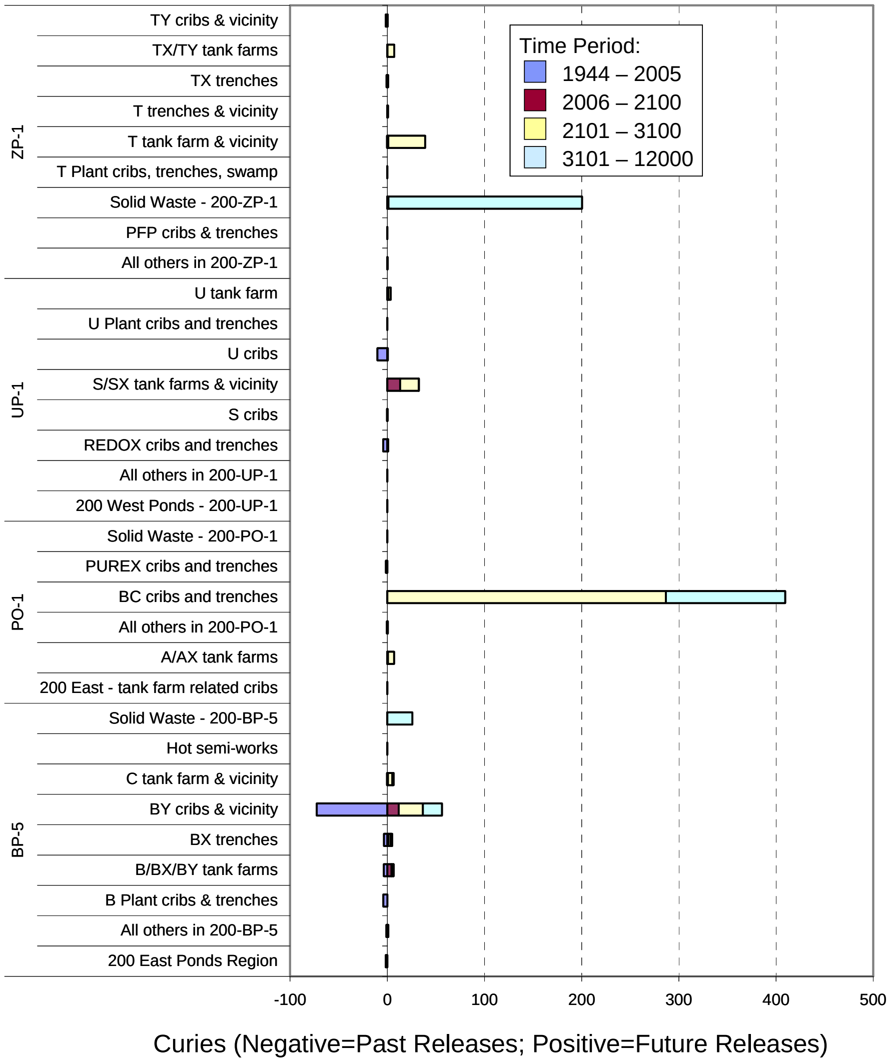

Figure 4.30. Cumulative Release (not decay-corrected) of Technetium into the Groundwater by Site Group

4.5.4 Groundwater

To support this waste site ranking activity, released activity or mass to the aquifer was converted to a hypothetical concentration in the aquifer using a “dilution factor” approach as defined in the state of Washington’s Model Toxics Control Act (WAC 173-340). Using MTCA guidance, the dilution factor is defined in the following manner.

$$\text{Dilution Factor} = (Qa + Qp)/Qp$$

where

Qa = groundwater flow (m^3/yr)
Qp = infiltration to groundwater from the vadose zone (m^3/yr)

Three sets of groundwater dilution factors were developed and used:

- The first set of values were based on groundwater velocity results from the existing site-wide groundwater model as described by Thorne et al. (2006) simulated for the year 2005.
- A second set of groundwater velocity results was derived from simulated post-Hanford water table conditions resulted from the existing groundwater model (Thorne et al. 2006) where the future groundwater flow from the Central Plateau to the Columbia River is predominantly north through the gap between Gable Butte and Gable Mountain.
- Because there is uncertainty in groundwater flow from the Central Plateau to the Columbia River in the future, a third set of dilution factors was derived based on extracted groundwater velocity results from an updated version of the site-wide groundwater model, as described in Zhang et al. (2005), where the future groundwater flow from the Central Plateau to the Columbia River was assumed to be predominantly east toward the Hanford town site.

For the calculation of the groundwater flow at each site, the estimated flow is based on the estimated groundwater velocity in the groundwater model extracted at the centroid coordinates at each waste site multiplied by an assumed groundwater flow cross-sectional area immediately below each waste site. For those sites that fall outside of the domain of the groundwater model in the Central Plateau region (that is, near internal boundary basalt subcrops in each model), the selected groundwater velocity used in the calculation was derived from the estimated groundwater velocities calculated at the element centroid nearest the waste site.

An example result for technetium-99 is shown in Figure 4.31. The peak groundwater concentration within each site group for the time period 2600–3100 is shown. The concentration calculations in the figure are based on year 2005 dilution factors.

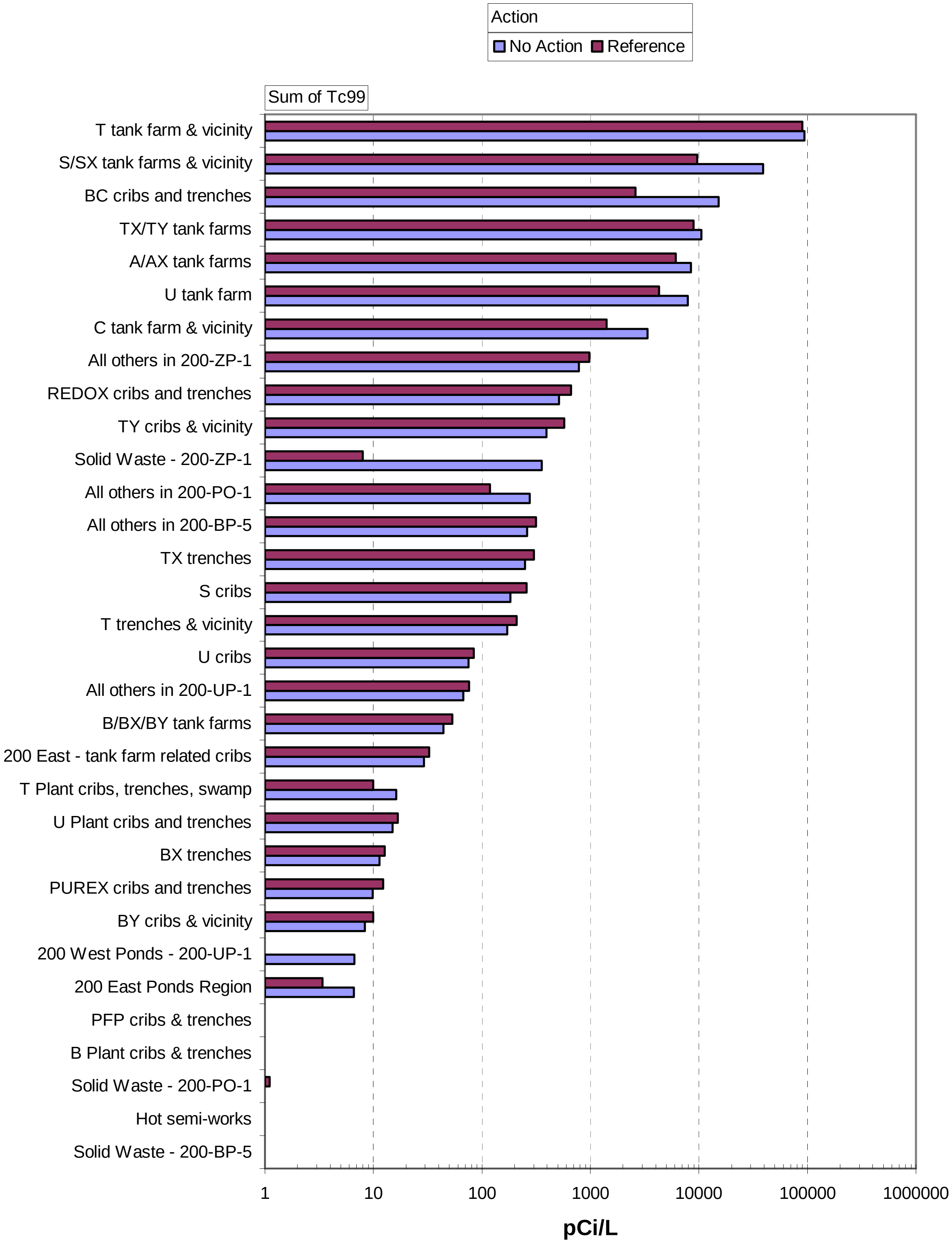

Figure 4.31. Peak Technetium-99 Groundwater Concentration Within Each Site Group for the 2600–3100 Time Period (using year 2005 dilution factors)

4.5.5 `Dose and Risk

Calculating dose and risk levels for each waste site group provides an integrated look at the combined impact of multiple analytes. Dose and risk for technetium-99, iodine-129, and uranium-234/238 were calculated using dose and risk conversion factors for a unit concentration of an analyte in the groundwater developed using the SAC codes. Dose and risk during the three time periods of analysis were estimated based on both 1) the maximum hypothetical concentrations for the three analytes simulated in groundwater for each waste site group, and 2) the maximum dose occurring at each waste site. The maximum for the waste site group based on analyte maximums is conservative because of spatial and temporal considerations. The group maximum is conservative in the spatial sense because maximum values for individual analytes are determined from all waste sites within a waste site group. Analyte maximums may be separated in space such that they do not superimpose on one another. The group maximum is conservative in the temporal sense because maximum values for individual analytes are determined over the time period of the analysis, (2006–2100, 2101–3100, 3101–12000). Because technetium, iodine, and uranium have different mobility in the subsurface, it is unlikely that analyte maximums within a time period will superimpose in time. This conservative estimate of maximum for each waste site group is used to rank groups.

While the concentrations presented in this analysis are hypothetical because of the assumptions that have been made to derive them, they and the estimates of dose derived from them provide some indication of the relative likelihood that a site will impact groundwater in the future. To further help identify those sites that have the greatest likelihood to impact groundwater, two thresholds have been applied:

- Exceedance of the following drinking water standards (EPA, 2001) for a contaminant or dose standards:
  - technetium-99, 900 pCi/L
  - iodine-129, 1 pCi/L
  - uranium-234/238, 27 pCi/L
  - drinking water dose from consumption of 2 L/day, 4 mrem/year
  - all pathways dose from a residential farmer scenario, 15 mrem/year
- Exceedance of ten times these drinking water standards or dose standards.

To gain an understanding of the number of waste sites that could be above or below a frequency count threshold—a count that indicates the strength or support for a group ranking—waste site maximum dose and risk were calculated. Maximum dose at each waste site is produced by calculating the dose from the three analytes over the entire time of analysis. In this case, the spatial and temporal discontinuities described above are eliminated, and the conservatisms are removed from the estimate. Because the conservatisms have been removed, a paradox is seen in some of the ranking and frequency count data. A waste site group ranked above the threshold using the conservative maximum of the group may not have a waste site within the group with a maximum above the threshold when the more realistic waste site maximum dose is calculated. An example result for the peak residential farmer dose (maximum within each site group) for the time period 3101–12000 using the year 2004 easterly flow dilution factors is shown in Figure 4.32.

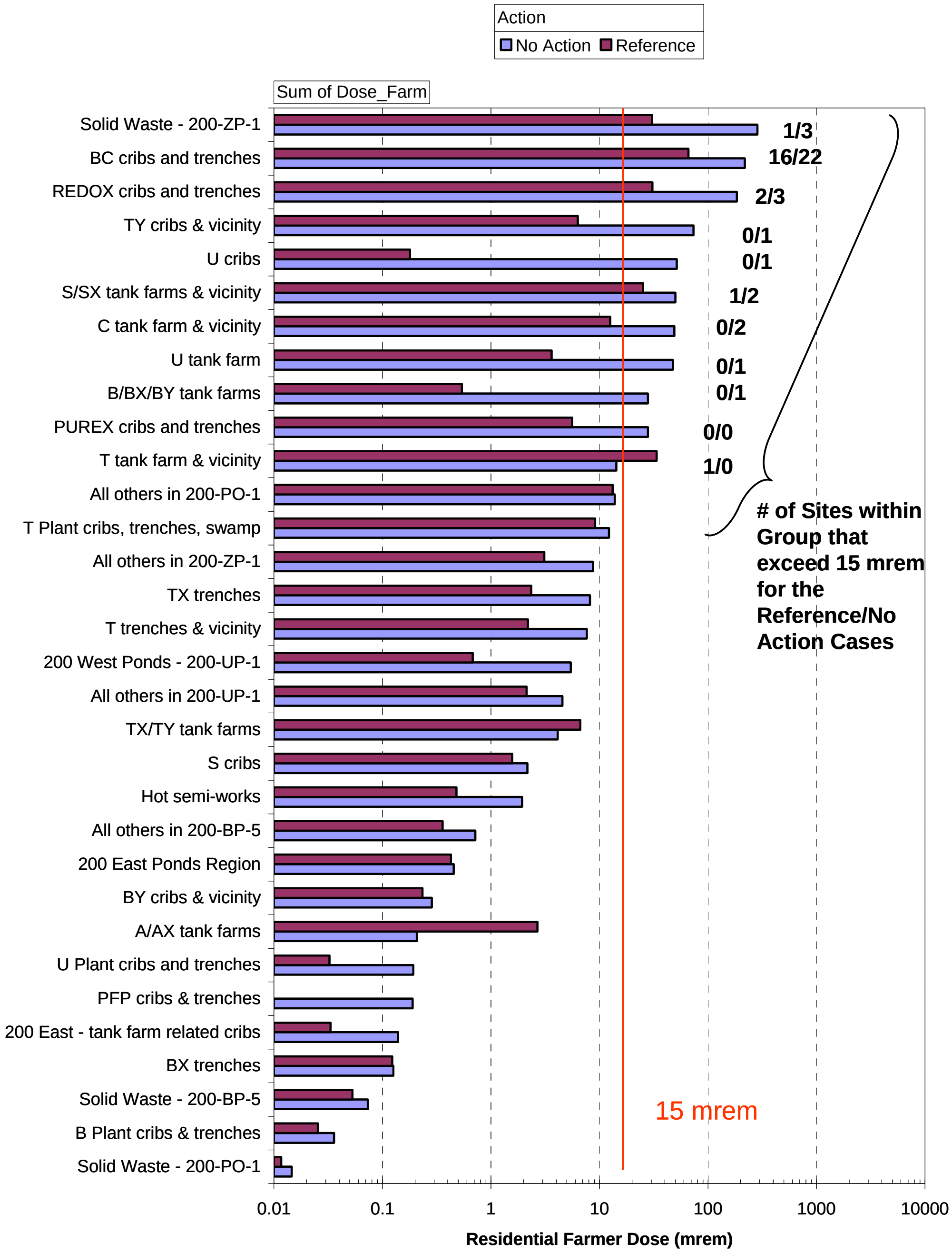

Figure 4.32. Peak Residential Farmer Dose (Maximum Within the Group) by Time Period 3101–12000 (using year 2004 easterly flow dilution factors)

The drinking water standard or dose standard is a commonly applied standard for evaluating groundwater impact. Similarly, ten times the drinking water standard has been identified as an action level in previous records of decision at Hanford (EPA 1995, 1997). Waste site groups with future potential impact levels above the action levels would require remedial action while those below the action level would not.

Seventeen of the 32 waste site groups have at least one hypothetical groundwater concentration or hypothetical dose above the threshold. Table 4.18 identifies whether these waste site groups are above

Table 4.18. Matrix of *No-Action* Case Results Exceeding 10X (√) and 1X (○) Threshold – All Metrics

Waste Site Group	Tc-99 Concentration					I-129 Concentration					U-234-238 Concentration					Drinking Water Scenario Dose					Residential Farmer Scenario Dose					Any Metric Over 10X
	2006-2100	2101-3100 N	2101-3100 E	3101-1200 N	3101-1200 E	2006-2100	2101-3100 N	2101-3100 E	3101-1200 N	3101-1200 E	2006-2100	2101-3100 N	2101-3100 E	3101-1200 N	3101-1200 E	2006-2100	2101-3100 N	2101-3100 E	3101-1200 N	3101-1200 E	2006-2100	2101-3100 N	2101-3100 E	3101-1200 N	3101-1200 E	
S/SX tank farms & vicinity	√	√	√				√	√	○	○						√	√	√			√	√	√	○	○	√
T tank farm & vicinity	√	√	√				√	√	○	○						√	√	√			√	√	√			√
U tank farm	√	○	○				○	√						○	○	○	○	○	○	○	√	√	√	○	○	√
BC crib & trenches		√	√	○	√				○	○							○	○	○	○		√	√	○	√	√
TX/TY tank farms	○	√	√				√	√								○	○	○			○	√	√			√
Solid Waste – 200-ZP-1				√	√				√	√									○	○				√	√	√
C tank farm & vicinity	○	○	○	○	○				○	○								○			○	○	○	○	○	
REDOX cribs & trenches	○					√	√	√	√	√				○	○	○	○	○	○	○	√	√	√	√	√	√
TY cribs & vicinity	○					○	○	○	○	○				○	○				○	○	○	○	○	○	○	
All others in 200-ZP-1	○															○					○					
A/AX tank farms	○	○	○				○	○									○	○				√	○			√
All others in 200-PO-01																			○					○		
All others in 200-UP-01						○	○	○													○					
PUREX cribs & trenches						○	○	○	○	○	○	○	○			○	○	○	○		○	○	○	○	○	
TX trenches						○	○	○													○	○	○			
U cribs													○	○					○	○				○	○	
B/BX/BY tank farms														○						○					○	

the threshold but less than ten times the threshold and those above ten times the threshold. Eight waste site groups are shown to exhibit impact levels above the ten times (10X) threshold value for one or more metric: “T tank farm & vicinity,” “S/SX tank farms & vicinity,” “Solid Waste – 200-ZP-1,” “REDOX cribs and trenches,” “TX/TY tank farms,” “U tank farm,” “BC cribs and trenches,” and “A/AX tank farms.” This analysis suggests that waste sites within these eight waste site groups present the greatest threat to groundwater.

The two dose metrics that integrate the impacts of technetium-99, iodine-129, and uranium-234/238 show quite different results. The hypothetical drinking water dose metric results in only two waste site groups exceeding the 10X threshold value: “T tank farm & vicinity” and “S/SX tank farms & vicinity.” The hypothetical residential farmer dose metric includes all eight of those exceeding the 10X threshold value.

This analysis generated a number of rankings using metrics on inventory, release into the vadose zone, release from the vadose zone, hypothetical groundwater concentration, and dose and risk. The last two of these, dose and risk based on the hypothetical groundwater concentrations, are the only metrics that integrate over multiple analytes. There is not a one-to-one correspondence among rankings based on analyte inventory, release to the aquifer, and hypothetical groundwater concentration. The rankings examined three time periods, two cases (no-action and reference), using groundwater flow fields from present day and two possible long-term representations (northerly flow and easterly flow). Hypothetical groundwater concentrations using the alternative long-term flow fields are quite similar. The rankings are insensitive to these alternative models.

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