

## **Appendix L**

### **System Assessment Capability: A 10,000-Year Post-Closure Assessment**

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#### **L.1 Introduction**

In late 1997, the U.S. Department of Energy (DOE) established the Groundwater/Vadose Zone Integration Project with Bechtel Hanford, Inc. (BHI), the Hanford Site environmental restoration contractor, as manager. The project transitioned to Fluor Hanford, the Project Hanford management contractor, in July 2002, and has been renamed the Groundwater Protection Program. Pacific Northwest National Laboratory (PNNL) is a partner in the program. The mission of the program is to coordinate and integrate projects that characterize, monitor, and clean up contaminants in the groundwater and vadose zone (the soil between the ground surface and the groundwater) beneath the Hanford Site. The Groundwater Protection Program also incorporates other task areas that complement these projects and several areas that represent accelerated actions leading to earlier site cleanup and closure.

In 1999, under the Integration Project, DOE initiated development of an assessment tool that will enable users to model the movement of contaminants from all waste sites at Hanford through the vadose zone, the groundwater, and the Columbia River, and to estimate the impact of contaminants on human health, ecology, and the local cultures and economy. This tool is called the System Assessment Capability (SAC).

The approach taken by the SAC is consistent with the methods, characteristics, and controls associated with a composite analysis as described by the Columbia River Comprehensive Impact Assessment (CRCIA) team (DOE-RL 1998). The CRCIA was a study initiated by DOE, the Washington State Department of Ecology, and the U.S. Environmental Protection Agency (EPA) to assess the effects of Hanford-derived materials and contaminants on the Columbia River environment, river-dependent life, and users of river resources. Part I of CRCIA is a study of present-day impacts to the Columbia River from Hanford contaminants. Part II is a suite of requirements for the development of a comprehensive impact assessment for the Columbia River. The two key elements of the SAC approach are 1) ensuring that dominant risk factors are included and 2) providing an understanding of the uncertainty of the results. Dominant factors were identified through scoping studies and the development of conceptual models for each of the analysis modules used. A stochastic modeling approach was taken to estimate uncertainty in the results. Aspects of uncertainty that could not be included in the calculation were considered in the analysis of the modeling results and discussed in the document presenting initial assessment results (Bryce et al. 2002). The analysis modules included in the SAC parallel those identified by CRCIA and were developed through work group meetings that included regulator and stakeholder participation.

Several key modules were adopted directly from the CRCIA, including the module used to calculate human health impacts (the HUMAN code) (Eslinger et al. 2002b) and the module used to calculate impacts to ecological species (the ECEM code) (Eslinger et al. 2002a, 2002b).

An initial assessment recently was completed with the SAC to demonstrate its functional assessment capability. Future modifications to the tool will be driven by the requirements of specific assessments. Improvements in the results obtained from use of the SAC will be realized as the input data are refined through characterization and scientific research. Bryce et al. (2002) reported the results of that assessment, which is the basis for application of the SAC to provide a sitewide perspective of waste disposal and remedial actions in this *Hanford Site Solid (Radioactive and Hazardous) Waste Program Environmental Impact Statement* (HSW EIS). Much of the material presented in this appendix has been taken from Bryce et al. (2002).

To simplify the discussion presented in this appendix, the term “SAC” refers to the software package used for this assessment, but it should be noted that the SAC is an evolving and maturing capability.

The initial assessment in FY 2002:

- Modeled the movement of contaminants from 533 locations throughout the Hanford Site representing 890 waste sites through the vadose zone, the groundwater, and the Columbia River.
- Incorporated data on 10 radioactive and chemical contaminants—carbon tetrachloride, cesium-137, chromium, iodine-129, plutonium-239/240, tritium, strontium-90, technetium-99, total uranium (chemical), and uranium (radionuclide).
- Focused on subsurface transport, the Columbia River, risks to human and ecological health, and the economy and culture.
- Included the geographic region from Rattlesnake Mountain to the Columbia River and from Vernita Bridge to McNary Dam on the Columbia River.
- Included the cleanup actions in Hanford’s cleanup plans and agreements as of October 2000.
- Consisted of a stochastic simulation for the period from 1944 to 3050 using 25 realizations, thus providing insight into the median response and an initial look at uncertainty.
- Simulated a 1000-year post-closure period. Three waste forms known to release after that time were not included—immobilized low-activity waste (ILAW), melters, and naval reactor compartments.

For the waste sites located on the Hanford Central Plateau and their associated contaminant plumes, the findings of the initial assessment paralleled those of the composite analysis (Kincaid et al. 1998). The results also are consistent with concentrations in environmental media measured by the Hanford Environmental Surveillance Program (Poston et al. 2002). Both the monitoring results and the assessment reported here indicate that Hanford impacts to the Columbia River have peaked and are now declining.

For the purposes of the HSW EIS, the SAC is a “best available technology” and, while it remains a tool under development, the SAC Rev. 0 tool is adequate to provide valuable information through quantification of cumulative risks and impacts associated with solid waste disposal at the Hanford Site.

### **L.1.1 Context of SAC Runs**

The principal SAC simulations made in support of the HSW EIS were a series of 25 stochastic simulations run for the period from 1944 through 12,050 A.D. (that is, a 10,000-year post-closure period) for the Hanford Site Disposition Baseline (HSDB) scenario. This simulation includes a stochastic representation of inventory, release, and transport, and a deterministic representation of exposure and dose. In addition, a median-value input case, based on the median value of each input parameter represented by a distribution in the stochastic model, was simulated.

The HSDB scenario represented in the FY 2002 initial assessment are based on a number of cleanup assumptions including waste, debris, and contaminated soil will be removed from the 100 Areas, and the remaining soil will meet residential use standards. Similarly, waste, debris, and contaminated soil will be removed from the 300 Areas, but the remaining soil will meet industrial use standards. In this scenario, retrievably stored transuranic (TRU) waste will be recovered, tested to determine waste content, repackaged, and sent offsite for disposal at the Waste Isolation Pilot Plant in New Mexico. The waste in Burial Grounds 618-10 and 618-11 will be removed, and the TRU waste will be repackaged and removed from the Hanford Site, while the low-level waste (LLW) will be disposed of in solid waste disposal facilities in the Central Plateau. Ninety-nine percent of the tank waste volume will be recovered from the tanks, and a 1 percent residual volume will remain. Losses to the subsurface during waste recovery are assumed to average 30,280 L (8000 gal) per single-shell tank recovered. The recovered tank waste will be separated into low-activity and high-activity fractions. Immobilization of both waste fractions was assumed. Low-activity waste will be disposed of onsite, while the high-activity fraction will be disposed of in the national repository. All spent fuel also will be stored in a stable configuration for shipment to and disposal in the national repository.

The initial assessment and this analysis assume that the future regional and local climate will remain unchanged for the period of the analysis. Furthermore, it is assumed that major engineered structures in the region (for example, the reservoir system on the Columbia River) will remain in place. The recorded climate and environmental response (for example, Columbia River stage and discharge records) since startup of the site operations were used to simulate the period from 1944 to the present. The climate record from 1961 to 1990 was used to represent the future climate. Consequently, the Hanford Site remains a semiarid, shrub-steppe environment in the simulations. The riparian zone, Columbia River, and river ecosystem are assumed to remain essentially unchanged for the duration of the analysis. Also, the human population will be unchanged and will be based on the current socioeconomic setting. Analyses of alternate future climates (for example, global climate change or onset of an ice age and glacial flooding) and potential future events (for example, failure or removal of the Columbia River reservoir system) are not addressed.

Where the initial assessment addressed the period from 1944 to 3050 (that is, essentially a 1000-year post-closure simulation), simulations for this EIS were carried out over a 10,000-year post-closure period.

Within the SAC, a single transport pathway element, the Columbia River model, is limited to the year 10,000 A.D. in its simulation algorithm, but all other transport pathways (release, vadose zone, groundwater) can execute for the full 10,000-year post-closure period.

The stochastic simulations supporting the HSW EIS are based on the parameter distributions assembled for the initial assessment. In addition to the environmental pathway and risk/impact model parameters, the inventory and future disposal and remedial actions assembled for the initial assessment are included. Differences between the inventory used in this extended simulation of the initial assessment and that used in the HSW EIS are described in Section L.2.2.2. Principal differences lie in the methods used to forecast solid waste disposal actions until site closure, both for onsite generators (for example, Waste Treatment Plant contributions) and for offsite generators.

The potential contaminants of greatest concern include technetium-99, iodine-129, and uranium. These contaminants appear in solid waste performance assessments (Wood et al. 1995; Wood 1996) that analyze solid waste disposals in the 200 West and East Areas. Of necessity, simulation of iodine-129 will include an initial condition for iodine-129 representative of prior releases to the unconfined aquifer, simulation of future releases of iodine-129 per the initial assessment, and superposition of the ILAW contribution to iodine-129 risk and impact. This approach to the iodine-129 simulation will include events attributed to past liquid discharges (current groundwater plumes), future solid waste releases, and long-term future releases from immobilized low-activity tank waste. The inventory estimated to exist in the unconfined aquifer and the estimate of iodine-129 in low-activity tank waste to remain at Hanford will be used in this estimate of the iodine-129 contribution to risk/impact. As in the original 1000-yr initial assessment, simulation of technetium-99 and uranium will use the complete history and forecast of their disposal and begin in 1944 with a clean subsurface environment.

It is unlikely that the plumes from these three classes of release events will superimpose in time. The liquid discharge and unplanned release (for example, tank leak) sites have created groundwater plumes and likely will continue to release to groundwater during the immediate future. Releases from dry solid waste disposals have some containment (for example, boxes, drums, plastic bags) and less driving force (infiltration) and, therefore, likely will release later than the liquid releases. Finally, the substantially stable and long-term waste forms, like vitrified low-activity tank waste, will not corrode and release for thousands of years. It is unlikely that peaks from each of these types of release will superimpose in space and time.

### **L.1.2 Relationship to EIS Calculations**

The EIS calculations focus on the impacts associated with alternatives to the disposal of solid waste. The SAC represents a holistic examination of the radioactive and chemical waste legacy of the Hanford Site. For this reason, it can be used to examine the relative risk and impact associated with disposal and remedial action alternatives and the relative role of different segments of Hanford waste (for example, solid waste, past-practice liquid discharges, or tank wastes). Used in this way, the SAC provides an ability to visualize the change in impact associated with various options and wastes. This kind of impact assessment provides a larger-scale cumulative context from which to view the alternatives and influence disposal decisions.

A line of analysis approximately 1-km from an operational area or waste disposal site was used in the 1998 composite analysis (Kincaid et al. 1998), the initial assessment completed with the SAC (Bryce et al. 2002), and in the simulations supporting this HSW EIS. The travel distance between the source and the uptake location is consistent with the groundwater model grid (that is, 375 m) and the longitudinal dispersivity (that is, 95 m) used in the sitewide groundwater model. In general, the rule of thumb for selecting an appropriate longitudinal dispersivity is to use approximately 10 percent of the mean travel distance of interest. A 1-km travel distance implies a 100-m longitudinal dispersivity. To control model stability and artificial dispersivity, the model grid Peclet number (that is, grid spacing/longitudinal dispersivity = 375 m/95 m) is typically selected to be no greater than 4 for finite element models. The existing model for the cumulative impacts was not configured to produce results at the 100-m travel distance. To achieve results at a 100-m line of analysis for cumulative impacts would require development of a local-scale model based on an approximate grid size of 40 m and longitudinal dispersivity of 10 m.

The EIS calculations provide a detailed evaluation of each specific alternative. The SAC is only able, at this time, to present the single case of an extended analysis (for example, 10,000-year post closure) of the HSDB. In essence, the SAC provides an estimate of the contribution made to risk and impact from technetium-99, iodine-129, and uranium from other Hanford waste disposal and remedial actions not explicitly considered in the HSW EIS alternative groups, and contrasts that with the contribution from solid wastes.

## L.2 Methods and Approach

Historically, DOE has used various tools to assess the effects of waste management and cleanup activities on the environment. Assessments have been performed to address a range of questions. Some assessments have focused on individual waste sites or waste types—for example, the assessment performed to evaluate the future performance of the glass waste form proposed for isolating low-activity waste currently in tanks (Mann et al. 2001). Others have looked at contaminants from a variety of sources. The Hanford Environmental Dose Reconstruction Project estimated human health impacts from past releases to the atmosphere and river (Farris et al. 1994) during Hanford operations from 1944 to 1972. The CRCIA examined ecological and human health effects that might result from the 1990 to 1996 distribution of contaminants in the environment in and near the Columbia River (DOE-RL 1998). The composite analysis performed in 1997 considered the impact of selected radionuclides from approximately 280 waste sites in the 200 Areas (Kincaid et al. 1998). In 2001, Bergeron et al. (2001) issued an addendum to the composite analysis that considered about 360 additional waste sites on the Central Plateau.

The collective impact of all of the wastes that will remain at Hanford, however, had not yet been integrated to provide an understanding of the cumulative effects of Hanford activities on the Central Plateau as well as in the river corridor. The SAC was developed to fill this gap and has benefited from the lessons learned in previous assessments.

The initial assessment and this extension to a 10,000-year post-closure analysis considers solid waste disposals in the Central Plateau as occurring within aggregated solid waste disposal facilities in the

northern and southern portions of the 200 West and East Areas. Annual inventories for each disposal facility within a subregion of the site are aggregated to create an annual solid waste inventory for the subregion. The areal footprints of disposal facilities within a subregion are aggregated to create a total solid waste disposal facility areal footprint. Contaminants from the aggregated disposal facility are released to the unconfined aquifer at the centroid coordinates of the aggregated disposal facility. Thus use of an aggregated representation of solid waste disposal facilities is an approximation in a number of ways. Notably, the inventory actually placed in individual trenches within each disposal facility is represented as distributed over the entire areal footprint of the disposal facility. Hence, the aggregated inventory is distributed over the aggregated areal footprint of all solid waste disposal facilities in a subregion of the site. Because of the scale of the aggregation (that is, half an operational area), the centroid of the aggregated area and, hence, the point where contaminants are introduced into the aquifer may lie outside an actual solid waste disposal facility.

The waste form used to represent the disposal of low-activity waste is the vitrified waste form described and analyzed in the ILAW performance assessment (PA) (Mann et al. 2001). The ILAW presents a unit release analysis of the waste inventory, contaminant release, and migration in the vadose zone and groundwater. The contribution of the ILAW source to groundwater and surface water impacts can be estimated by scaling (that is, for inventory and spatial position). These results can then be superimposed onto the groundwater and surface water impacts predicted for all other Hanford waste sources to achieve a cumulative impact projection. For the initial assessment (Bryce et al. 2002), all contaminants were simulated from 1944 forward in time to estimate the distribution of contamination in the environment. For some contaminants (for example, tritium), sufficient process knowledge and data existed to complete a history match against tritium field data. For other contaminants (for example, technetium-99, iodine-129, uranium), work is under way to improve the understanding of inventory and mobility to enable improved comparisons with field observations from Hanford's groundwater.

## **L.2.1 Modular Components of SAC**

The SAC development task involved assembling software and gathering the data needed to assess the cumulative impact of radioactive and chemical waste at Hanford. Computer codes that were well tested at the Hanford Site were used when possible and new software was written when necessary to simulate the features and processes that affect the release of contaminants into the environment, transport of contaminants through the environment, and the impact those contaminants have on living systems, cultures, and the local economy. The components were organized to simulate the transport and fate of contaminants from their presence in Hanford waste sites—through their release to the vadose zone, to their movement in the groundwater, and into the Columbia River. Components such as the groundwater model, the ecological impact component, and the human health component originally were developed and tested for previous Hanford assessments.

The elements of the SAC computational tool include:

**Inventory Module**—develops an inventory of specific waste disposal and storage locations for the period from 1944 to December 2050 based on disposal records, process knowledge, and the results of tank and field samples. December 2050 was used because it had been identified as the date of site

closure. However, for the purposes of this EIS, the Hanford closure date is considered to be 2046. Future analyses will use the current closure date. This module identifies the material scheduled for disposal in offsite repositories including high-level waste (HLW), TRU waste, and spent fuel.

**Release Module**—simulates the annual release of contaminants to the vadose zone from the variety of waste types in the modeled waste sites. Waste types explicitly modeled include soil-debris wastes as solubility limited desorption, cemented waste as diffusion limited, salt cake tank residuals as nitrate salt dissolution, and graphite cores of production reactors as an empirically defined release. Because they release after the 1000-year analysis period, waste types not included in the original SAC design included ILAW, melters, and naval reactor compartments. This module also simulates Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) (42 USC 9601) remedial actions that move waste to the Environmental Restoration Disposal Facility (ERDF) trench.

**Vadose Zone Module**—simulates the flow and transport of contaminants in the vadose zone, which is the unsaturated sediment between the land surface and the unconfined aquifer. Vadose zone simulations use a one-dimensional version of the well-established and documented Subsurface Transport Over Multiple Phases (STOMP) code.

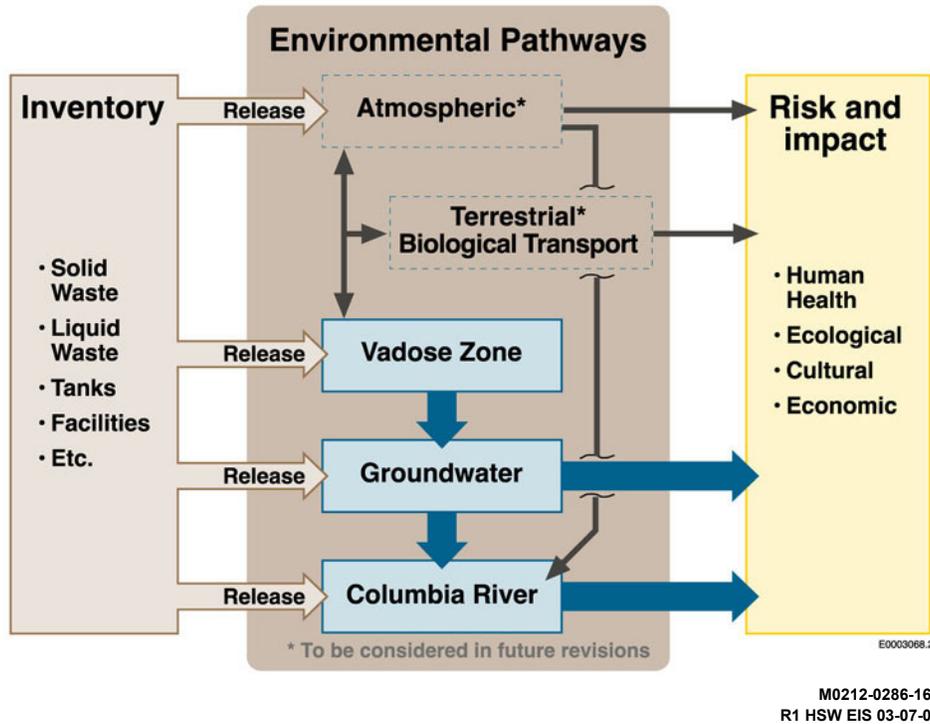
**Groundwater Module**—simulates the flow of water and the transport of contaminants in the unconfined aquifer that underlies Hanford using the three-dimensional, sitewide groundwater model. Groundwater simulations use the Coupled Fluid, Energy, and Solute Transport (CFEST) code.

**River Module**—simulates river flow and contaminant/sediment transport in the Hanford Reach from Vernita Bridge downstream to McNary Dam. This model simulates background concentrations and background plus the Hanford Site contribution to enable an assessment of the Hanford Site incremental impact to the Columbia River and its ecosystem. The river model is an extension of the Modular Aquatic Simulation System 2D (MASS2) code developed and applied to support studies of the Snake and Columbia rivers.

**Riparian Zone Module**—uses river and groundwater information to simulate the concentration of contaminants in seep or spring water and in the wet soil near the shoreline of the river.

**Risk/Impact Module**—performs risk/impact analysis in four topical areas: human health, ecological health, economic impact, and cultural impact, with economic and cultural impacts being two new impact metrics for Hanford assessments.

The conceptual illustration of SAC (Figure L.1) portrays a linear flow of information. In general, data flows in the initial assessment in the following manner: the Inventory Module provides input to the Release Module, which provides input to the Vadose Zone, Groundwater, and River Modules. The Vadose Zone Module provides input to the Groundwater Module. Finally, both the Groundwater and River Modules provide input to the Risk/Impact Modules. This version of the SAC conceptual model does not allow feedback among modules and does not include either atmospheric or terrestrial ecological pathways and, hence, receptors.



**Figure L.1.** Conceptual Model of the System Assessment Capability

The data used in the initial assessment came from a variety of sources, including environmental monitoring activities on the Hanford Site, Hanford historical records, a waste site information database, and other geohydrologic and physical property databases. The remediation actions included in the assessment are based on the collection of disposal and remedial actions identified in the Tri-Party Agreement (Ecology et al. 1989) that are planned to occur as the Hanford Site moves toward closure.

## L.2.2 Inventory

Inventory consists of the quantity of radiological and chemical constituents used and created at the Hanford Site, and their distribution in individual facilities and waste disposal sites. For the initial assessment, inventory was defined as the volume and concentration of contamination introduced annually to waste disposal sites (for example, the solid waste disposal facilities), facilities (for example, the canyon building), and the environment (for example, the vadose zone via liquid discharge sites, the Columbia River via reactor cooling water retention basins). In the initial assessment, export of contaminants to offsite locations was accounted for by collecting exports at the conclusion of the analysis. The movement of onsite waste from one location to another is included in the Release Module but is limited to the movement of excavated CERCLA wastes to ERDF trench. Finally, tank waste moves into the Inventory Module of the initial assessment only after it leaks to the environment, is defined as a tank residual, or is recovered from tanks and processed into waste forms that are disposed of onsite or shipped offsite.

The initial assessment included 533 waste site locations throughout the Hanford Site, representing 890 waste sites that were identified for consideration. Each of the 890 sites had a likelihood of containing one or more of the contaminants of interest. Some sites were combined, or aggregated, thus reducing the total to 722 sites for analysis. However, of the 722 sites chosen for analysis, only 533 sites were assigned inventories because some waste disposal and unplanned release inventories were further aggregated. For example, individual disposal ditches and ponds were all identified in the list of 722 sites, but the ditch inventories were assigned to the receptor pond. Accordingly, the inventories for the ditches leading to Gable Mountain Pond, B Pond, and U Pond were assigned zero inventories. The Inventory Module of the SAC generates annual inventories for the selected contaminants at 533 sites for the period from 1944 through 2050, and each of 25 realizations for the stochastic analysis. For the initial assessment, this represented in excess of 782,000 pieces of non-zero inventory data.

Regarding chemicals in solid waste disposals, as in the case of radionuclides, it is unlikely that chemical impacts from liquid discharges and solid waste will superimpose in time. It is believed that the majority of chemicals were either discharged to cribs and trenches or stored in tanks, as opposed to being disposed of as solid waste. When the Hanford Site moved away from liquid discharge of chemicals in the late 1960s and early 1970s, substantial chemical waste streams were routed to tanks. Mixed low-level radioactive waste currently is being stored and will be treated prior to disposal under the Resource and Conservation Recovery Act (RCRA) (42 USC 6901) and past-practice CERCLA guidelines to ensure long-term safety.

For example, the presence of carbon tetrachloride in the aquifer underlying the 200 West Area is a direct result of the disposal of liquid waste streams containing carbon tetrachloride. The mean value inventory of carbon tetrachloride in the initial assessment shows approximately 813,000 kg being released to liquid discharge sites in the 200 West Area. In comparison, all of the carbon tetrachloride in HSW is reported to be in “stored” solid waste; none is reported in “buried” solid waste, and the total inventory reported to be stored through 1997 was approximately 5000 kg. Storage is occurring in radioactive mixed waste storage facilities (primarily CWC) and the 218-W-3A, 218-W-4B, and 218-W-4C LLBGs. While there is no record of past disposals, some carbon tetrachloride might have been disposed of in HSW disposal facilities. However, it is likely that the amount, its rate of release, and its potential impact on groundwater would not be substantial compared to that of past releases to liquid discharge facilities.

An analysis of chemical inventories in solid waste disposals on the same scale as the initial assessment cannot be supported on the basis of current data and information. However, based on available information, chemicals in solid waste do not appear to be as important in terms of human health impacts as the key radionuclides—technetium-99, iodine-129, and uranium.

Carbon tetrachloride and chromium in Hanford solid waste are not expected to add substantially to impacts of those substances from other Hanford sources. For further discussion of the potential impacts from hazardous chemical constituents in Hanford solid waste see Volume I, Sections 5.3.2 and 5.3.5.

### L.2.2.1 Initial Assessment Inventory

Methods used to assemble the annual inventory database for all waste sites are described in Appendix A of Bergeron et al. (2001) issued in September 2001. Additional detail on the methods used to merge record data and estimates for the Hanford Site inventory were provided by Cooney (2002). The addendum to the composite analysis includes a summary of the inventory in each waste site at the close of 2000 and at the assumed time of Hanford Site closure in 2050 (Bergeron et al. 2001). The inventory shown in the initial assessment inventory differs from the summary inventory presented in the addendum; however, the data in the addendum provides a representative picture of the site inventory.

### L.2.2.2 Comparison of HSW EIS and Initial Assessment Inventories

The initial assessment inventory was developed over a period of time, beginning in FY 2000 with final entries completed during the spring of 2002. Some of the data entries date from September 1999, which was the close of FY 1999. The HSW EIS inventory was developed over a similar time period, but it reflects changes as recent as the summer of 2002. Table L.1 shows a comparison of the initial assessment (SAC) and the EIS (Alternative Group D<sub>1</sub>, preferred alternative) as their respective inventories existed in September 2002. The inventories addressed within the scope of the HSW EIS include wastes in the LLBGs and future disposal facilities, and, therefore, while being more current for solid waste, they do not reflect all potential sources that were evaluated for the SAC initial assessment of cumulative impacts. The HSW EIS inventories shown do not reflect inventories in other waste forms that will remain at Hanford including graphite cores of production reactors, liquid discharge and unplanned release sites, and past tank leaks and future tank residuals. Table L.1 and the discussion of inventory differences provide a review of the inventories in the two assessments and indicate the relative inventories treated by a soil-debris, cement, or liquid release models.

**Table L.1.** Comparison of Initial Assessment and HSW EIS Inventories

Summary of Technetium-99, Iodine-129, and Uranium Inventories at the Time of Hanford Site Closure							
		Initial Assessment <sup>(a)</sup>			HSW EIS <sup>(b)</sup>		
		Tc-99	I-129	U	Tc-99	I-129	U
Waste Stream	Type	Ci	Ci	Ci	Ci	Ci	Ci
200 East	Solid waste as soil debris	25.3 <sup>(c)</sup>	0.39 <sup>(c)</sup>	0.12	12 <sup>(c)</sup>	0.065 <sup>(c)</sup>	22.9 <sup>(d,e)</sup>
200 East	Solid waste as cement	0.08	0	0	3700 <sup>(d,f)</sup>	5.1 <sup>(f)</sup>	2000 <sup>(f)</sup>
200 East	Tank leaks/residuals	259	0.35	24.8			
200 East	Liquid/UPR <sup>(g)</sup>	791	0.40	66.2			
200 East	Total Activity	1075	1.14	91.3			
200 West	Solid waste as soil debris	343 <sup>(c)</sup>	0.41 <sup>(c)</sup>	209 <sup>(h)</sup>	3.1 <sup>(c)</sup>	0.045 <sup>(c)</sup>	150 <sup>(h)</sup>
200 West	Solid waste as cement	1291 <sup>(c,i)</sup>	64.2 <sup>(c,i)</sup>	1837 <sup>(h,i)</sup>	130	0.008	1000 <sup>(h)</sup>
200 West	Tank leaks/residuals	327	0.61	13.2			
200 West	Liquid/UPR	40.9	0.10	24.7			
200 West	Total Activity	1712	64.9	1803			

Table L.1. (contd)

Summary of Technetium-99, Iodine-129, and Uranium Inventories at the Time of Hanford Site Closure							
		Initial Assessment <sup>(a)</sup>			HSW EIS <sup>(b)</sup>		
		Tc-99	I-129	U	Tc-99	I-129	U
Waste Stream	Type	Ci	Ci	Ci	Ci	Ci	Ci
ERDF <sup>(j)</sup> (600-148)		2.6	0.0017	54.0			
SALDS <sup>(k)</sup> (600-211)	“soil”	0.310	2.17	0.00133			
Graphite cores (100 Areas)	“core”	0.012	.000089	0			
ILAW (200 East)	“glass”	25,550 <sup>(l)</sup>	0 <sup>(i)</sup>	52.97 <sup>(i)</sup>	25,550 <sup>(l)</sup>	22 <sup>(l)</sup>	230 <sup>(e,l)</sup>
Melters (200 East)	“glass”	37.8	0	1.70	38.9	0	1.8
Naval reactors (200 East)	“rxcomp”	5.18	1.3E-5	0			
US Ecology, Inc. (600 Area)	“soil”	60.7	5.45	11390			
Other 200 Area remaining onsite <sup>(m)</sup>		730 <sup>(n,o)</sup>	0.065 <sup>(n)</sup>	8.6 <sup>(n)</sup>			
Other Areas remaining onsite <sup>(m)</sup>		13.8	0.0044	33.4			

(a) Initial assessment inventory values are median values from a stochastic simulation of the inventory.  
(b) Alternate Group D<sub>1</sub> Upper Bound waste volume.  
(c) The initial assessment includes technetium-99 and iodine-129 inventories estimated using a fuel-ratio method for fission product inventories not reported on original records or prior estimates. The HSW EIS inventories of technetium-99 and iodine-129 include only reported or record values.  
(d) The HSW EIS includes inventories of mixed low-level waste (MLLW) that are included elsewhere in the initial assessment inventory for the SAC (see note “m” below).  
(e) The HSW EIS includes an inventory of uranium-233 not included in the initial assessment conducted using the SAC.  
(f) The HSW EIS includes inventory forecasts obtained from the Solid Waste Information Forecast Tool (SWIFT) that includes a life-cycle forecast of the composition of secondary waste streams from tank waste; in Alternative Group D<sub>1</sub>, ILAW, melter, and future solid waste inventories are disposed of in an integrated disposal facility near PUREX.  
(g) UPR = unplanned release.  
(h) The initial assessment includes uranium inventories estimated using somewhat different uranium isotopic ratios and estimation methods than used in the HSW EIS.  
(i) The initial assessment includes inventory forecasts obtained from a Hanford Tank Waste Operating System (HTWOS) (Kirkbride et al. 2002) simulation that used potentially out-of-date factors for secondary waste streams; the technetium-99 inventory is a current estimate to be routed to low-activity disposal.  
(j) ERDF = Environmental Restoration Disposal Facility.  
(k) SALDS = State-Approved Land Disposal Site.  
(l) The HSW EIS includes inventory forecasts obtained from the ILAW performance assessment (Mann et al. 2001) for isotopes, and from a current estimate of technetium-99 that will be routed to low-activity waste disposal. These inventories, that are somewhat higher than initial assessment inventories for iodine-129 and uranium, are applied in simulations that superimpose ILAW contaminants and dose onto the contaminant and dose from all other waste site releases. Results of these superimpositions appear near the end of this Appendix.  
(m) Does not include waste listed above.  
(n) The initial assessment includes inventories of MLLW at the Hanford Site that will be routed though the Radioactive Mixed Waste Storage Facility prior to disposal onsite.  
(o) Of the 730 Ci of technetium-99 shown, approximately 660 Ci are designated for offsite disposal.

The differences in the initial assessment and HSW EIS inventories highlight an issue that exists whenever knowledge evolves as fast as, or faster than, the ability to perform assessments; that is, more recent assessments have available to them more current knowledge. Thus estimates of inventory vary. Since the summer of 2002 when the HSW EIS inventories were assembled for analysis, simulations have been published creating a new baseline estimate of radionuclides created in Hanford production reactors

(Wootan and Finfrock 2002). Previous inventory estimates relied on the earlier publication by Watrous and Wootan (1997). Some substantial changes have occurred in production estimates based on Wootan and Finfrock (2002), for example, the earlier estimate of 64.1 Ci of iodine-129 has been superseded with an estimate of 49.4 Ci. The annual revision of the best basis (tank) inventory has been issued in the *Tank Farm Contractor Operation and Utilization Plan* (Kirkbride et al. 2002). An online version of this inventory is updated quarterly to reflect any further refinement of this information.

Another issue with inventories is the conservatism adopted by various projects and programs that have been responsible for compilation and publication of an inventory. Often, it requires fewer resources to generate a conservative estimate, and such an estimate may provide all a project or program needs. For example, designing and sizing a treatment process may rely on the largest inventory that may be processed and may not require knowledge of a best estimate or median inventory or the possible range of the inventory. Accordingly, when inventories from a variety of projects and programs are merged to create a total Hanford Site inventory (based on the summation of inventory estimates for waste discharge, disposal sites, and for stored tank waste), that total inventory may differ from the total inventory estimated based on production reactor operation. Conservative estimates of individual inventories tend to overestimate the actual amount of material in existence. Accordingly, the sum of these inventories tends to exceed independent estimates concerning the total amount of waste generated. Thus while use of conservative estimates for individual inventories may be used to bound environmental impacts, they may not necessarily be summed to arrive at the best estimate for a total inventory. As the activities related to Hanford waste site characterization, facility decommissioning, and tank waste disposal proceed, the conservatism found in inventory estimates may be reduced.

The SAC was applied in the HSW EIS to generate both a stochastic simulation and a median-inputs deterministic simulation. The inventory values reported for the initial assessment in Table L.1 are median values of the stochastic distribution. Thus, a varied inventory is analyzed, and each of the 25 realizations is based on a Latin hypercube selection procedure. For sites not modeled using process knowledge and a stochastic simulator (Simpson et al. 2001), site-specific inventories prior to 1970 are modeled as twenty-fold uncertain; that is, the maximum is approximately 20 times the inventory database value, and the minimum is approximately one-twentieth of the inventory database value. After 1970, the inventories for these sites are modeled as twofold uncertain; that is, the maximum is approximately twice the inventory database value, and the minimum is approximately half the database value.

The inventory analyzed by the sitewide groundwater model and the unit release approach in the HSW EIS was provided by Fluor Hanford. The inventory analyzed using the SAC tool is based on available records and was augmented with estimated inventories for fission products (for example, technetium-99 and iodine-129) and uranium isotopes where they are absent from the record. The augmented values are only estimates and should not be considered record values.

There are differences in the compilations shown in Table L.1. Solid waste deposits in the 200 East and 200 West Areas differ primarily as follows: 1) the initial assessment technetium-99 and iodine-129 inventories include fuel-ratio estimates of this fission product, 2) the initial assessment uranium inventories include estimates based on uranium-isotopic ratio methods of estimation that differ from those of the EIS, 3) the HSW EIS uranium inventories include MLLW inventories that are accounted for elsewhere in

the initial assessment, 4) HSW EIS solid waste disposal facility uranium inventories include uranium-233, which was omitted from the initial assessment, and 5) large inventories in the HSW EIS–200 East Area, solid waste as cement type and in the initial assessment–200 West Area, solid waste as cement type reflect different assumptions regarding the disposal location.

### **Inventory Assumptions**

#### **Initial Assessment SAC Inventory Assumptions**

- Past solid waste inventories were drawn from the Solid Waste Information Tracking System (SWITS).
- Future solid waste inventories were estimated using data from SWITS and the assumption that past disposals were indicative of future waste receipts.
- The vitrification process will drive off volatile gases including iodine, and, therefore, the iodine-129 inventory in ILAW essentially will be zero.
- The Hanford Tank Waste Operations Simulator and associated secondary waste stream factors provided by the Tank Farm contractor are appropriate for estimating the amounts of key contaminants.
- An amount of iodine-129 was lost to air emissions during the chemical separation, and, therefore, most of the iodine-129 produced in the reactors was processed and put in the single- and double-shell tanks.
- Commercial low-level radioactive waste inventories are based on WDOH and Ecology (2000) and the closure plan published by US Ecology, Inc. (1996).

#### **HSW-EIS Inventory Assumptions**

- Past solid waste inventories were drawn from SWITS.
- Future solid waste inventories were estimated using SWIFT and waste generator estimates of future disposals.
- The baseline inventories of iodine were held in ILAW.
- More current (summer 2002) estimates from the tank farm contractor provided the best estimate of secondary waste stream inventories.
- In addition to the 27.1 Ci of iodine-129 accounted for in the HSW EIS inventory under ILAW and the 200 East Area solid waste, some iodine-129 was emitted to the atmosphere during chemical separation activities, and there are plans to capture the majority of the iodine-129 in tank waste in the high-level waste glass.

A major difference in inventories in the 200 East and 200 West Area solid waste disposal facility “as cement” deposits and in ILAW deposits lies in the use of different resources to estimate future disposals and secondary wastes from the processing and solidification of tank wastes at Hanford. The initial assessment relied on the Hanford Tank Waste Operation System (HTWOS) model that relied on a suite of potentially out-of-date factors to estimate secondary waste stream composition. This resulted in nearly 1300 Ci of technetium-99 and 65 Ci of iodine-129 being disposed of in the 200 West Area as solid waste in cement. The initial assessment inventory also relied on an earlier estimate of ILAW inventory that assumed no iodine-129 would be retained in the glass waste form. The HSW EIS relies on more current ILAW and secondary waste inventory estimates. Accordingly, the HSW EIS shows 3700 Ci of technetium-99 and 5 Ci of iodine-129 being disposed of in the 200 East Area as solid waste in cement, and 22 Ci of iodine-129 being disposed of in the ILAW glass. Inventories with the greatest differences either are simulated as cement waste forms that release relatively slowly (for example, the 200 East and West Areas solid waste cement) or are not simulated by the initial assessment (for example, ILAW and

melter waste). A difference of approximately 2000 Ci in technetium-99 exists between the two estimates of secondary technetium-99 wastes. Similarly, a difference of approximately 60 Ci in iodine-129 exists. These differences will be reconciled as projections are updated; however, all of this waste would be disposed of in cement to minimize the hazard. In the analyses undertaken for both the initial assessment and the HSW EIS, the majority of the future uranium inventory is disposed of in cement to minimize the hazard.

Finally, because of the original design objectives of the SAC (that is, a 1000-year analysis), the initial assessment does not include the release model(s) necessary to forecast the long-term release of the ILAW and melter wastes. Hence, the influence of ILAW and melter inventories is not included in the initial assessment results or in the extended (10,000-year) initial assessment presented here. Naval reactor compartments also are omitted from SAC analyses at this time. However, for the greatest of these inventories (ILAW), their influence is introduced to the cumulative assessment by superimposing the results of the ILAW PA (Mann et al. 2001) onto the initial assessment result. Thus the influence of ILAW PA inventories shown in Table L.1 under the HSW EIS is superimposed on the initial assessment.

There is uncertainty with respect to the total inventory of iodine-129 in spent fuel irradiated at Hanford. The inventory data and information assembled for the initial assessment (Bryce et al. 2002) revealed that approximately 75 Ci of iodine-129 were generated during the irradiation of nuclear fuel in Hanford reactors. Most of the spent fuel was processed in facilities on the Central Plateau; however, some spent fuel remains onsite and is being moved to a central location on the Central Plateau prior to shipment to a national repository. Some of the iodine-129 inventory is conservatively counted in individual waste site inventories. When summed, the inventories disposed of at waste sites, released to the environment (for example, from cribs, into the atmosphere, and into the Columbia River), and stored for future disposal at offsite locations equals approximately 100 Ci, which exceeds the 75-Ci production estimate.

Iodine is found in all three phases (solid, liquid, and gas) and has been identified in each of these waste types. Accordingly, some iodine-129 is found in solid waste, some in liquid discharges, and some in atmospheric releases. There is considerable uncertainty in the amount of iodine-129 that appears in each. In prior inventory compilations and in the initial assessment, it was assumed that most of the iodine-129 resides in single-shell and double-shell tanks in the Central Plateau. Furthermore, it was assumed that all of the iodine-129 would be captured in secondary waste streams from waste separation and solidification processes, and that these wastes would be treated and the iodine primarily disposed of in solid waste disposal facilities. Of the 100 Ci in the initial assessment and in this cumulative impact analysis estimated to be present at Hanford at the time of site closure, approximately 65 Ci reside in solid waste; 19 Ci may have been released to the atmosphere; 7 Ci reside in spent fuel; 5.5 Ci reside in commercial low-level radioactive waste disposal; 3 Ci were discharged to cribs and trenches; and 1 Ci is associated with the past leaks, estimated future losses, and residuals of tanks. None of the 65 Ci of iodine-129 associated with solid waste in the initial assessment is assigned to ILAW because the early assumption was that iodine was too volatile to remain in the solidified low-activity tank waste. However, this inventory of 65 Ci is almost entirely from byproduct streams from waste separation and vitrification processing (that is, spent resins and ILAW and HLW secondary waste streams—not glass).

As a result of recent estimates of iodine retention in immobilized tank waste, about 22 Ci of the iodine-129 in the tank waste was assumed, for impact modeling purposes and evaluation of alternatives in this EIS, to be disposed of as part of the ILAW waste form. The HSW EIS analysis of alternative groups assumes an additional 5 Ci are contained in the solid waste to be disposed of (see Appendix B, Table B.19). Thus the groundwater modeling performed for the alternate actions in this EIS assumes a total source term of 27 Ci of iodine-129 in the combined ILAW and solid wastes. Some iodine-129 was emitted to the atmosphere during chemical separation. The remaining inventory of iodine-129 is not shown in the HSW EIS inventory used in the alternative analyses because it is not assumed to be part of solid wastes evaluated in the alternative groups. However, for the cumulative impact analysis an additional inventory of approximately 60 Ci of iodine-129 are accounted for as solid waste in a cement waste form.

Inventories included in the initial assessment for the commercial low-level radioactive waste disposal site operated by US Ecology, Inc., at Hanford are based in part on the published State of Washington draft EIS (WDOH and Ecology 2000) and the closure plan for the site published by US Ecology, Inc. (1996). The State of Washington now is reviewing the inventory for the commercial site during its early years of operation. Hanford staff are in contact with a representative of the Washington State Department of Health, and as soon as an updated inventory is available, it will be incorporated into Hanford assessments. Certainly, uranium inventories for the commercial low-level radioactive waste disposal site appear to be relatively high in the initial assessment.

### **L.2.3 Release**

Release is the rate at which radioactive and chemical contaminants find their way into the environment. The SAC Release Module handles liquid releases and releases from solid waste forms. It is important to note that because the initial assessment was originally designed as a 1000-year analysis; several waste forms that will not be released in this period were not analyzed and were not analyzed in this extended 10,000-year post-closure analysis even though they may be released in the 10,000-year time frame. These waste forms include naval reactor compartments, immobilized low-activity waste, and components of melter systems. Liquid discharges, liquid unplanned releases including tank leaks, and future tank losses are handled as a simple pass-through to the vadose zone or the Columbia River. The solid waste forms are primarily in solid waste disposal facilities including past-practice sites (pre-1988), active sites (post-1988), and at ERDF. Other solid waste includes residual waste in the single-shell tanks, the graphite cores of the retired production reactors, and concrete and cement waste forms associated with caissons, canyon buildings, and grouted waste.

The Release Module applies release models to waste inventory from the Inventory Module and also accounts for site remediation activities (for example, waste movement) 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. Radioactive decay is accounted for in all inputs and outputs of the Release Module. The Release Module is implemented as the VADose zone Environmental Release (VADER) computer code (Eslinger et al. 2002a).

### **L.2.3.1 Conceptual Model**

Waste containment facilities have a number of features that influence the rate at which contaminants can be released from waste. The waste may be placed in a trench or may reside in a tank. The trench, tank, or other engineered structure may have features that serve as barriers to prevent infiltrating water from making contact with and transporting contaminants from the waste to the vadose zone. Waste inside an engineered structure (for example, a trench) may also be contained in a waste package (for example, a metal drum or high-integrity concrete container). The drum or concrete container acts as an additional barrier that prevents 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 concrete materials change over time, and, likewise, time affects the features that dominate 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 of a system 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 these features. However, none of the waste sites in the initial assessment had 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, thus 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 (for example, 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 the 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.

### **L.2.3.2 Implementation Model**

The Release Module accounts for releases that occurred in the early years of Hanford operations, releases that may be expected while the site is being cleaned up over the next several decades, and future releases that may continue until the entire inventory is released. The Release Module relies on several sources for input. Input from the Inventory Module includes contaminant mass (for chemicals) and activity deposits (for radionuclides). Some of the release models (that is, soil-debris, cement) require site or waste feature information (that is, site cross-sectional area, site volume, or waste surface area or volume). Recharge rate is an important parameter for the salt cake and soil-debris models. Key process parameters are distribution coefficient (soil-debris model), solubility (soil-debris,  $C_{sol}$ , and salt cake models), diffusion coefficient (cement model), and fractional release rate (reactor block model).

To capture uncertainty in the SAC simulations, contaminant inventories and numerical model parameters are expressed in terms of statistical distributions. Each realization of the initial assessment used sample parameter values for randomly distributed variables such as bulk soil density, soil moisture content, sorption or distribution coefficient, salt cake density, and cement diffusion coefficient. Other model parameters were held to constant values over all realizations.

### **L.2.3.3 Numerical Models Relevant to HSW EIS**

#### **L.2.3.3.1 Soil-Debris Model**

The soil-debris model is used to model contaminant release from unconsolidated wastes mixed with soil. Source zones composed of this waste-form type are permeable to percolating water; therefore, all surfaces of the waste come in contact with the percolating water as it passes through the zone in a manner similar to the way infiltrating water passes through natural vadose zone material. The soil-debris model is applied to the release of contaminants from all solid waste disposal facilities, including ERDF, and the commercial low-level radioactive waste disposal facility operated by US Ecology, Inc.

For the SAC initial assessment, the model used the high-impact values of the distribution coefficient parameter ( $K_d$ ) associated with the vadose zone nearest the disposal facility. For solid waste disposal facilities, the  $K_d$  category used by the soil-debris model is that associated with sites that have a low organic and low salt content and near-neutral pH. The  $K_d$  best-estimate values for this category were 0 mL/g, 0.5 mL/g, and 3 mL/g for technetium-99, iodine-129, and uranium, respectively.

For radionuclides for which no specific solubility values were available, the aqueous solubility was fixed at an arbitrarily high default value ( $1 \times 10^{10}$  mg/L) so that the soil-debris model automatically selected algorithms for sorption ( $K_d$ ) control in these cases (Kincaid et al. 1998). Technetium-99 solubility ( $1 \times 10^{10}$  mg/L or  $1.7 \times 10^2$  Ci/cm<sup>3</sup>) was assigned using this approach. Iodine-129 solubility ( $1 \times 10^{10}$  mg/L or  $1.77 \times 10^0$  Ci/cm<sup>3</sup>) also was assigned using this approach. Uranium solubility ( $86.9$  mg/L or  $2.95 \times 10^{-11}$  Ci/cm<sup>3</sup>) was estimated in Hanford groundwater assuming that the solid controlling uranium solubility was  $UO_2(OH)_2 \cdot H_2O$  (Wood et al. 1995).

In the simulation runs,  $K_d$ ,  $\theta_w$ , and  $\beta$  were treated as stochastic over the 25 realizations, and  $Q_w$  and  $C_{sol}$  were fixed to a constant value for all analytes except tritium. For tritium,  $K_d$  was set to zero over all realizations.

Sites with soil wastes include the “118,” “218,” and “618” sites listed in Bergeron et al. (2001).

#### **Analytical Solution for Instantaneous Release—Soil-Debris Model**

The rate of loss of contaminant for a given contaminant by the soil-debris model is given by Kincaid et al. (1998) as:

$$dM / dt = -Q_w A C_w \quad (L.1)$$

where  $C_w = C_{sol}$  when the release process is solubility controlled and  $C_w = M/(\theta RAh)$  when the release process is desorption-controlled where:

$$R = I + (\beta K_d) / \theta \quad (L.2)$$

Switching regimes is controlled by comparing the remaining mass with the maximum mass  $M_{max}$  consistent with an aqueous phase saturated with the contaminant. If  $M$ , the mass remaining in the waste form, is larger than the quantity  $M_{max}$  where:

$$M_{max} = \theta RC_{sol} Ah \quad (L.3)$$

the release process is considered to be solubility controlled. Otherwise, it is considered desorption-controlled.

Coupling the soil-debris model with an aggregated waste site representation leads to a lower calculated waste concentration, a reduced likelihood of a solubility-controlled release, and a greater likelihood of a desorption-controlled release. Because the release occurs over a larger area than really occupied by the waste deposit, the calculated release is a function of a greater amount of infiltrating water contacting the waste. Thus, all contaminants are leached and for mobile contaminants such as technetium-99 that are not solubility controlled, the release is greater for an aggregated site approach.

### **Definitions**

- $M_{max}$  is the maximum amount of contaminant possible in the source zone (in Ci or kg) without a precipitated phase.
- $M = M(t)$  is the current quantity of contaminant contained in the source zone (Ci or kg).
- $Q_w$  is the recharge rate for the site in cm/yr.  $Q_w$  can be considered to be constant, or it can be time-dependent based on site climate and remediation activities.
- $A$  is the surface area of the soil waste form exposed to the release mechanism ( $cm^2$ ).
- $h$  is the depth of the waste form in the site (cm).
- $C_w$  is a coefficient expressing the effective release of the contaminant ( $Ci/cm^3$  or  $kg/cm^3$ ).
- $C_{sol}$  expresses aqueous solubility of the contaminant ( $Ci/cm^3$  or  $kg/cm^3$ ).

- R is either a retardation factor or a soil apportionment factor (unitless) that depends on the following factors:
  - $\beta$  Soil bulk density in  $\text{g}/\text{cm}^3$
  - $K_d$  Sorption factor ( $\text{cm}^3/\text{g}$ )
  - $\theta$  Soil volumetric content of water in soil (unitless fraction).
- $dM/dt$  is the rate of loss of contaminant from the source zone (the rate the contaminant crosses the soil waste form boundary and enters the environment).
- t is the elapsed time (years) from the beginning of release from containment.

#### **L.2.3.3.2 $C_{\text{sol}}$ (Solubility) Model**

The  $C_{\text{sol}}$  model is the independently operated, solubility-controlled analytical solution component of the soil-debris model. As such, it is applied to the same types of solid wastes that are applied to the soil-debris model. The difference is that the process represented by the  $C_{\text{sol}}$  model is that of a constant concentration release. The concentration at which a contaminant is released from a waste often is at its solubility limit in some aqueous medium (for example, groundwater, or grout leachate) but is not a requirement. This is different from application of the same analytical solution within the soil-debris model in which the model determines the process (solubility controlled vs. sorption controlled) that is appropriate for application at any time within a simulation. In addition, release is always at what is considered to be the solubility limit of the contaminant in the aqueous media of interest. The analytical solution and key parameters are the same as those described in the previous section for the solubility-controlled analytical solution component of the soil-debris model.

Initial application of this release model within the SAC Release Module was undertaken to provide a comparative evaluation of uranium release from a cemented waste form using three different release models (see Section L.2.3.4).

Assume that a solubility-controlled release was prescribed for several scales of disposal from aggregated areas to individual waste trenches and that each disposal scale contained the same inventory. The larger the waste site area, the greater the infiltrating water quantity contacting waste, the greater the mass or curie flux from the waste site, and the more rapid the release.

#### **L.2.3.3.3 Cement Model**

The cement model generally is applied to cementitious waste forms. Knowledge of the total external surface area and the volume of the waste form is required. The area-to-volume ratio is assumed to be constant (that is, the waste form is assumed not to degrade in terms of shape over the duration of the contaminant release process). In the SAC initial assessment, the cement model was used to simulate release of contaminants from cementitious wastes within selected solid waste disposal facilities. Delay of contaminant release from containerized waste can be accomplished with the current capability by arbitrarily assigning a time of delay. In the SAC initial assessment, however, no credit was taken for

container integrity. Plans call for incorporating one or more models into a future revision of the SAC capability that will accommodate delay of release from contained waste based on specific processes (for example, metal corrosion).

The range in diffusion coefficient values ( $1.58 \times 10^{-4}$  cm/yr to  $1.89 \times 10^{-3}$  cm/y) used in the SAC initial assessment for technetium-99 was obtained from recent laboratory work (Mattigod et al. 2000). The diffusion coefficient for uranium ( $3.15 \times 10^{-5}$  cm/yr) was obtained from Serne et al. (1992). In the simulations, the diffusion coefficient for technetium-99 was stochastic; for uranium, it was set to a constant for all realizations.

Sites containing cementitious wastes include the “202,” “221,” “224,” and “276” sites listed in Bergeron et al. (2001).

### **Analytical Solution for Instantaneous Release—Cement Model**

The contaminant release mechanism of the cement model is diffusion in the pore water of the solidified waste material to the outer surface of the waste form. The rate-of-loss for a given contaminant is given by Kincaid et al. (1998) as:

$$dM/dt = M_0(A/V) \sqrt{D / \pi t} \quad (\text{L.4})$$

where:

$M_0$	=	the original quantity of the contaminant contained in the cement (Ci or kg)
$M$	=	current quantity of the contaminant contained in the cement (Ci or kg)
$A$	=	the surface area of the cement structure ( $\text{cm}^2$ )
$V$	=	the volume of the cement structure ( $\text{cm}^3$ )
$D$	=	the diffusion coefficient of the contaminant ( $\text{cm}^2/\text{yr}$ )
$t$	=	the elapsed time (years) from the beginning of release from containment
$dM/dt$	=	the rate of loss of contaminant from the cement waste form
$\pi$	=	3.14159.

Note, the original quantity  $M_0$  can be seen as a function of concentration ( $\text{kg}/\text{cm}^3$  or  $\text{Ci}/\text{cm}^3$ ) and volume ( $\text{cm}^3$ ).

With regard to the scale of the disposal, assuming the aggregated area of an aggregated volume is simply the exterior surface of the volume, the larger the disposal area, the smaller the ratio of area to volume ( $A/V$ ) in the equation above. Accordingly, if the contaminant mass or  $C_i$  and the diffusion coefficient are unchanged for multiple scales of waste site, then the larger aggregated site will exhibit a lower release rate.

#### **L.2.3.3.4 Containment**

The release models implemented in the current version of SAC have no provisions for specifically modeling containment of wastes, such as high-integrity steel containers. The models do have a provision

for delaying release to a specific start year (that is, the STARTREL argument in the MODELS keyword). The default start year is the year the waste begins to be deposited at the site. In the initial assessment, STARTREL was set to 1944 throughout the simulation, so for the initial assessment, the release mechanism was active as soon as wastes were deposited.

#### **L.2.3.4 Comparison of Release Model Parameters**

A comparison of key source-term release models (that is, soil-debris, solubility-controlled, and cement) and values of key parameters used in the SAC analysis, the HSW EIS analysis (described in Appendix G), and the solid waste burial ground (SWBG) PAs for the 200 West and East Areas (as described by Wood et al. [1995] and Wood [1996]) is summarized in Table L.2. The three constituents addressed are technetium-99, iodine-129, and uranium. This summary of parameter values, coupled with the release model formulations of the preceding section, allows a comparison of relative release characteristics included in the three assessments. The parameter values shown here are somewhat generic and not necessarily related to specific waste streams and, therefore, could be changed according to specific waste disposal conditions for application in specific wastes and especially for regulatory compliance simulations (that is, a performance assessment for a specific disposal).

There are several key differences in the way these different analysis approaches address selective contaminant releases from the source term. The SAC analysis differs from the other two analyses in the way that uranium is released from LLW. For non-cemented waste, the SAC analysis uses a soil-debris model coupled with uranium specific solubility-limits to simulate uranium release. For cemented wastes, the SAC analysis uses a cement (that is, diffusion-controlled) release model to simulate uranium release. In contrast, the release of uranium in HSW EIS analysis and the SWBG PAs relies on a solubility-controlled release model with uranium-specific solubility limits depending on whether the uranium inventory is contained in non-cemented wastes or in cemented wastes (for example, 64 mg/L for non-cemented wastes and 0.23 mg/L for cemented wastes).

The SAC application of the cement model to technetium-99, iodine-129, and uranium releases assumed a cemented waste and a surface A/V ratio based on a waste volume that constituted a number of aggregated burial ground sites. In contrast, the HSW EIS and SWBG PA analyses relied on a conceptualization of surface A/V ratio based on the surface area and volume of individual waste containers (for example, individual steel barrels, boxes, high integrity containers that would contain grouted wastes). As a result, the surface A/V ratio for the SAC source term was up to 10 times lower than those reported for HSW EIS and SWBG PA analyses. Lower releases of technetium-99, iodine-129, and uranium from the SAC analysis would be expected based on this difference alone. However, when the diffusion coefficient is roughly one order of magnitude higher in the SAC application, the lower A/V ratio is partially offset by the higher diffusion coefficient.

**Table L.2.** Comparison of Selected Values of Key Parameters Used in Source-Term Release Models for the System Assessment Capability Analysis Described in this Appendix, the HSW EIS Analysis Described in Appendix G, and the Solid Waste Burial Ground Performance Assessments for the 200 West and East Areas Described by Wood et al. (1995) and Wood (1996)

	System Assessment Capability (SAC)	HSW EIS	Solid Waste Performance Assessment
<b>Source-Term Release Models</b>			
<b>Soil-Debris Model</b>			
<b>Model or Zone/Parameter</b>	<b>Data/Statistical Treatment</b>		
Volumetric Moisture Content (%)	0.0594 ± 0.0310 <sup>(a)</sup> (mean/standard deviation, normal distribution)	0.05	0.05
Bulk Density (g/cm <sup>3</sup> )	1.535 ± 0.1085 <sup>(a)</sup> (mean/standard deviation, normal distribution)	1.6	1.5
Waste Thickness (m)	5.349 <sup>(b)</sup> (deterministic)	6	4.5
K <sub>d</sub> uranium (mL/g)	Low organic/low salt/near neutral, high impact: (best estimate, min and max) <sup>(c)</sup> best estimate: 3, min: 0.1, max: 500	Mobility Class (K <sub>d</sub> =0.6) <sup>(b)</sup> covering constituents with K <sub>d</sub> s between 0.6 and 0.9999	Mobility Class (K <sub>d</sub> =0.0) covering constituents with K <sub>d</sub> s between 0.0 and 0.9999
K <sub>d</sub> technetium-99 (mL/g)	Low organic/low salt/near neutral, high impact: (best estimate, min and max) <sup>(c)</sup> best estimate: 0; min: 0; max: 0.1	Mobility Class (K <sub>d</sub> =0.0) <sup>(b)</sup> covering constituents with K <sub>d</sub> s between 0.0 and 0.5999	Mobility Class (K <sub>d</sub> =0.0) covering constituents with K <sub>d</sub> s between 0.0 and 0.9999
K <sub>d</sub> iodine-129 (mL/g)	Low organic/low salt/near neutral, high impact: (triangular distribution, mode, min and max) <sup>(c)</sup> median: 0.5; min: 0; max: 15	Mobility Class (K <sub>d</sub> =0.0) <sup>(b)</sup> covering constituents with K <sub>d</sub> s between 0.0 and 0.5999	Mobility Class (K <sub>d</sub> =0.0) covering constituents with K <sub>d</sub> s between 0.0 and 0.9999
Solubility; uranium (mg/L)	86.9 (2.95 x 10 <sup>-11</sup> Ci/cm <sup>3</sup> ) <sup>(d)</sup> (deterministic) (non-cemented wastes)	NA <sup>(l)</sup>	NA
Solubility; technetium-99 (mg/L)	1 x 10 <sup>10</sup> (1.7 x 10 <sup>-2</sup> Ci/cm <sup>3</sup> ) <sup>(e)</sup> (deterministic) (non-cemented wastes)	NA	NA
Solubility; iodine-129 (mg/L)	1 x 10 <sup>10</sup> (1.77 x 10 <sup>0</sup> Ci/cm <sup>3</sup> ) <sup>(e)</sup> deterministic (non-cemented wastes)	NA	NA
<b>Solubility-Control Model</b>			
<b>Model or Zone/Parameter</b>	<b>Data/Statistical Treatment</b>		
Solubility; uranium (mg/L)	86.9 (2.95 x 10 <sup>-11</sup> Ci/cm <sup>3</sup> ) <sup>(d)</sup> (deterministic) (non-cemented wastes)	64 (non-cemented wastes); 0.23 (cemented wastes)	64 (non-cemented wastes); 0.23 (cemented wastes)
Solubility; technetium-99 (mg/L)	1 x 10 <sup>10</sup> (1.7 x 10 <sup>-2</sup> Ci/cm <sup>3</sup> ) <sup>(e)</sup> (deterministic) (non-cemented wastes)	NA	NA
Solubility; iodine-129 (mg/L)	1 x 10 <sup>10</sup> (1.77 x 10 <sup>0</sup> Ci/cm <sup>3</sup> ) <sup>(e)</sup> (deterministic) (non-cemented wastes)	NA	NA
<b>Cement Model</b>			
<b>Model or Zone/Parameter</b>	<b>Statistical Treatment</b>		
Area to Volume Ratio (m <sup>2</sup> /m <sup>3</sup> )	0.378 <sup>(k)</sup>	1.55 to 1.93	5.33 <sup>(i)</sup>
Diffusion Coefficient; uranium (cm <sup>2</sup> /yr)	3.15 x 10 <sup>-5</sup> (1 x 10 <sup>-12</sup> cm <sup>2</sup> /s) <sup>(e, f)</sup> (deterministic)	NA	NA
Diffusion Coefficient; technetium-99 (cm <sup>2</sup> /yr)	(uniform distribution, median, min, max) median: 1.02 x 10 <sup>-3</sup> , min: 1.58 x 10 <sup>-4</sup> , max: 1.89 x 10 <sup>-3</sup> <sup>(g)</sup>	3.15 x 10 <sup>-4</sup> (range - min: 1.58 x 10 <sup>-4</sup> , max: 1.89 x 10 <sup>-3</sup> ) <sup>(g)</sup>	3.15 x 10 <sup>-5</sup> to 31.5 <sup>(i)</sup>
Diffusion Coefficient (iodine-129) (cm <sup>2</sup> /yr)	3.5 x 10 <sup>-5</sup> <sup>(g)</sup>	3.15 x 10 <sup>-5</sup>	3.15 x 10 <sup>-5</sup> to 31.5 <sup>(i)</sup>
<p>(a) Values based on statistical treatment of individual data points measured or calculated over a depth ranging from 0- to 20-ft values calculated from bulk density and moisture content data from Fayer et al. (1999).</p> <p>(b) An average height calculated for burial ground sites based on available height information in the Waste Information Database System (WIDS).</p> <p>(c) Based on revision of K<sub>d</sub>s in Kincaid et al. (1998) resulting from a recent compilation and evaluation of distribution coefficient data in Hanford sediments (Cantrell et al. 2002).</p> <p>(d) Estimated solubility in Hanford groundwater assuming solid controlling solubility was UO<sub>2</sub>(OH)<sub>2</sub> • H<sub>2</sub>O (Wood et al. 1995).</p> <p>(e) Default value from Table D.2 of Kincaid et al. (1998).</p> <p>(f) Recommended value (default) for generic grout performance assessment when actual grout-specific data is lacking (Table 6, Serne et al. 1992).</p> <p>(g) Based on results obtained from Mattigod et al. (2000).</p> <p>(h) Best estimate K<sub>d</sub> values after Cantrell et al. (2002).</p> <p>(i) Values as low as 1.7 m<sup>2</sup>/m<sup>3</sup> have been used in subsequent waste stream specific analyses.</p> <p>(j) A range of values was considered for an unspecified constituent in the PA analysis (Wood et al. 1995).</p> <p>(k) Based on all cemented waste placed in aggregate area 218-W@T-6-12 (SAC Rev. 0).</p> <p>(l) NA = not applicable; the process or parameter was not used in the assessment.</p>			

From the formulations of the soil-debris model, which is the release model associated with early solid waste disposals at Hanford (that is, pre-1970 wastes), it is apparent that the use of larger aggregated areas as opposed to burial ground, trench, or caisson scales to represent waste, leads to lower initial concentrations of waste but exposes waste to greater infiltration and, hence, leaching. Use of aggregated representations and the soil-debris model tends to release waste more rapidly than would occur if simulations were conducted on the burial ground or trench scale.

#### **L.2.4 Vadose Zone Module**

The Vadose Zone Module is designed to simulate the transport and fate of contaminants as they move through the hydrogeologic region that extends from the land surface to the regional water table. Kincaid et al. (2000) identified the STOMP computer code (White and Oostrom 1996) as the code for the Vadose Zone Flow and Transport Module for SAC. Inputs to the Vadose Zone Module come primarily from the inventory and release elements, including recharge, and the mass flux and concentrations of the selected constituents. Other inputs include the effectiveness and timing of remedial actions that might either reduce the mass and/or concentration of contaminants in the vadose zone or that might reduce the flux of deep infiltrating moisture (that is, capping). These inputs include infiltration rates from both natural events (for example, precipitation) and operational activities (for example, excavation or capping). A few major hydro-stratigraphic units that are of uniform thickness and horizontal with homogeneous and isotropic properties were used to represent each site. Hydraulic and geochemical parameters for each hydro-stratigraphic unit are represented by stochastic distributions that reflect the uncertainty in measured properties. Definitions of the hydro-stratigraphy and the associated hydraulic, transport, and geochemical properties of the one-dimensional soil column were based on existing geologic, soil physics, and geochemical databases.

##### **L.2.4.1 Distribution Coefficients ( $K_d$ s) for Technetium-99, Iodine-129, and Uranium**

The SAC initial assessments used  $K_d$  values that were assigned to each hydrogeologic unit in a manner similar to that done for the Composite Analysis (Kincaid et al. 1998). The waste characteristics were assumed to dominate the near-field mobility of the contaminants in the vadose zone. After being in contact with vadose zone sediments and soil water for some distance, the waste undergoes a change in its mobility based on buffering of the contaminant solution by the vadose zone sediments. Thus, distribution coefficients were defined separately for each contaminant in the upper vadose zone (near-field or high-impact zone) and in the lower vadose zone (far-field or intermediate-impact zone) (Kincaid et al. 1998).

Distribution coefficient zones were defined as either high-impact or intermediate-impact depending on the nature of the contaminant. Zones in which the organic concentration, pH, or salt concentration in the fluids may have affected the  $K_d$  values were designated high-impact. Zones in which the acidic or basic nature of the wastes was estimated to have been neutralized by the natural soil were designated intermediate-impact. Kincaid et al. (1998) estimated the depths of this transition zone by examining the peak location of beta/gamma contamination, as presented by Fecht et al. (1977a, b), for 200 Area cribs receiving very acid or high-salt/very basic waste. In general, these transition depths ranged from 10 to 40 m (33 to 130 ft). Given the limited data available on which to base further interpretations on the depths of transition and the desire to simplify the numerical simulations, a slightly different approach was

used here. Generally, the hydrogeologic unit into which waste streams were introduced was designated as high-impact regardless of waste stream characteristics. If those hydrogeologic units were thin (for example, less than 10 m), then the hydrogeologic unit immediately below that into which the waste stream was introduced was also designated as high-impact. All other hydrogeologic units lower in the profile were designated intermediate-impact. This approach kept the numerical simulations relatively simple by using the existing number of hydrogeologic units (that is, new layers did not need to be added to make the  $K_d$  change where it might have occurred within a single hydrogeologic unit). At the same time, the depths of change, corresponding to the thickness of the hydrogeologic units, are still on the same scale (tens of meters) as those used by Kincaid et al. (1998). A summary of the  $K_d$  values used for technetium-99, iodine-129, and uranium is presented in Tables L.3, L.4, and L.5, respectively.

Carbon-14 was not simulated in this cumulative assessment but was simulated in the evaluation of alternative groups in this EIS. The composite analysis (Kincaid et al. 1998) assigned carbon in solid waste disposal facilities a distribution coefficient of 5 mL/g. Consequently, the release and migration of carbon-14 from solid waste is substantially retarded compared with those of uranium, and carbon-14 impacts to groundwater would occur after the 10,000-year post-closure period analyzed for the cumulative assessment.

**Table L.3.** Technetium-99 Distribution Coefficients (mL/g)

Waste Chemistry	Vadose Zone			Groundwater	Riparian Zone
	High-Impact (Near-Field)	Intermediate-Impact (Far-Field)			
		Sand	Gravel		
All	<u>0</u> (0-0.1)	<u>0</u> (0- 0.1)	<u>0</u> (0-0.01)	<u>0</u> (0-0.1)	<u>0</u> (0-0.1)

Values are listed as **best** (minimum–maximum).

**Table L.4.** Iodine Distribution Coefficients (mL/g)

Waste Chemistry	Vadose Zone			Groundwater	Riparian Zone
	High-Impact (Near-Field)	Intermediate-Impact (Far-Field)			
		Sand	Gravel		
High Organic/Very Acidic; Low Organic/Low Salts/Acidic	<u>4</u> (0-15)				
High Organic/Near Neutral; Very High Salt/Very Basic; Chelates/High Salts; Low Organic/Low Salt/ Near Neutral	<u>0.2</u> (0-2)	<u>0.2</u> (0-2)	<u>0.02</u> (0-0.2)	<u>0.2</u> (0-2)	<u>0.2</u> (0-2)

Values are listed as **best** (minimum–maximum).

**Table L.5.** Uranium Distribution Coefficients (mL/g)

Waste Chemistry	Vadose Zone		Groundwater	Riparian Zone	
	High-Impact (Near-Field)	Intermediate-Impact (Far-Field)			
		Sand			Gravel
High Organic/Very Acidic; Chelates/High Salts; Low Organic/Low Salts/Acidic	<u>0.2</u> (0-4)	<u>0.8</u> (0.2-4)	<u>0.08</u> (0.02-0.4)	<u>0.8</u> (0.2-4)	
High Organic/Near Neutral; Very High Salt/Very Basic; Low Organic/Low Salt/ Near Neutral	<u>0.8</u> (0.2-4)				
Values are listed as <u>best</u> (minimum–maximum).					

#### L.2.4.2 Vadose Zone Strata and Hydraulic Properties

Of the more than 2600 waste sites at Hanford cataloged in Waste Information Database System (WIDS), a subset of 533 was selected for simulation in the initial assessment. Because of the aggregation of solid waste disposal facilities, unplanned releases, and various liquid discharge sites into fewer global waste sites within operational areas or portions of operational areas, these 533 sites represent 890 waste sites.

##### L.2.4.2.1 Geologic Profiles

Each of these sites was assigned to one of 64 base templates defined on the basis of 1) the type of waste site, 2) its geographic location (that is, area/geology), and 3) the characteristics of the waste stream.

Generalized hydrostratigraphic columns were specified for each of the 13 geographic areas. These columns were assembled from existing information, including:

- logs (from drillers, geologists, and geophysicists)
- published interpretive depths to the top and bottom surfaces of hydrogeologic units
- surface elevations (to convert hydrogeologic unit depths to elevations)
- elevation of the 1944 water table (to define the bottom of the vadose zone prior to waste disposal).

The generalized hydrostratigraphic units used in this study are summarized in Table L.6.

**Table L.6.** Summary of Hydrogeologic Units Used in This Study

<b>Hydrogeologic Units</b>	<b>Facies/Subunit</b>	<b>Description</b>
Not applicable	Backfill	Poorly sorted gravel, sand, and silt derived from the Hanford formation and/or Holocene deposits.
Holocene	Eolian	Dune sand and silt.
Hanford formation	Silt-dominated	Interbedded silt and fine to coarse sand.
	Fine sand-dominated	Stratified fine sand with minor pebbles and minor laterally discontinuous silt interbeds.
	Coarse sand-dominated	Stratified coarse sand with minor pebbles and minor laterally discontinuous silt interbeds.
	Gravelly sand	Cross bedded, interstratified coarse sand with up to 30 wt% very fine pebble to cobble.
	Gravel-dominated	Cross bedded, interstratified coarse sand and gravel with greater than 30 wt% very fine pebble to boulder.
	Undifferentiated	Undifferentiated sand and gravel with minor discontinuous silt interbeds.
Plio-Pleistocene unit	Silt/sand dominated	Very fine sand to clayey silt sequence. Interstratified silt to silty very fine sand and clay deposits.
	Carbonate rich	Carbonate-rich sequence. Weathered and naturally altered sandy silt to sandy gravel, moderately to strongly cemented with secondary pedogenic calcium carbonate.
Ringold Formation	Fluvial sand (member of Taylor Flat)	Interstratified sand and silt deposits.
	Fluvial gravel (member of Wooded Island, subunit E)	Moderate to strongly cemented well-rounded gravel and sand deposits, and interstratified finer-grained deposits.
	Overbank/Lacustrine deposits (lower mud sequence)	Predominantly mud (silt and clay) with well-developed argillic to calcic paleosols.

In general, the depth and thickness of each hydrogeologic layer (strata) for each geographic area were taken from published maps and cross sections. The estimated average strata thickness was used for the generalized columns extending from the surface to the 1944 water table (Kipp and Mudd 1974). Because the sum of the average thickness did not always equal the distance from the land surface to the groundwater, small adjustments were made to the average strata thickness.

#### **L.2.4.2.2 Hydraulic Properties**

Hydraulic property data were primarily taken from Khaleel and Freeman (1995) as supplemented by Khaleel (1999) and Khaleel et al. (2000). Because this data set is rather limited in terms of the spatial location of samples and the soil types represented, individual stochastic data sets were selected to represent each hydrogeologic strata present in the 13 geographical areas. Care was taken to ensure that the soil classifications for which hydraulic property data was available could be correlated to the sediment facies within each template.

The statistical distributions of the van Genuchten model (van Genuchten 1980) parameters, saturated hydraulic conductivity, and bulk density data were taken primarily from Khaleel and Freeman (1995)

and Khaleel et al. (2000), and the distributions for longitudinal dispersivity were primarily taken from Ho et al. (1999). Values for residual saturation ( $S_r$ ) were calculated by dividing the raw residual water content ( $\theta_R$ ) by the raw saturated content ( $\theta_s$ ), as provided by Khaleel and Freeman (1995). Effective porosity is assumed to be equal to the saturated water content ( $\theta_s$ ). Note that all model nodes within a single hydrogeologic unit are assigned the same hydraulic properties for a single realization.

### L.2.4.3 Surface Covers

The SAC incorporates recharge estimates into the STOMP model to provide deterministic values that change stepwise as the surface cover changes and to represent the degradation of engineered covers following their design life. The recharge rates (actually, deep drainage rates) used for the SAC were estimated for all surface conditions under consideration for the initial assessments. These conditions included four different barrier designs, degraded barriers, the natural conditions that surround the barriers, and the unique conditions created by human activities (for example, facility construction, gravel-covered tank farms). Recharge estimates were based on the best available data (Fayer and Walters 1995; Fayer et al. 1999; Murphy et al. 1996; Prych 1998).

#### L.2.4.3.1 Barrier Recharge Estimates

Recharge through engineered surface covers was estimated based on the Focused Feasibility Study (FFS) conducted by DOE-RL (1996). The FFS was conducted to determine the barrier needs at Hanford and to identify a set of barrier designs to meet those needs. Table L.8 identifies the four barrier designs that were proposed. According to the FFS, the Modified RCRA Subtitle C Barrier design will be the predominant barrier type. DOE-RL (1996) used the HELP model<sup>(a)</sup> to simulate the recharge rate through the Hanford Barrier, modified RCRA barriers, and the standard RCRA barriers. The estimates ranged from 0.2 to 0.8 mm/yr, assuming that the annual mean precipitation remained at 160 mm/yr (6.3 in/yr). Subsequent to the FFS, additional data and model results became available. As a result, the recharge rates for the barriers were updated as reflected in Table L.7.

**Table L.7.** Barrier Design Lifetimes and Estimated Recharge Rates (actual rates are expected to be less than shown)

DOE-RL Design	Design Life (yr)	Recharge Rate (mm/yr)	Source
Hanford Barrier	1000	0.1	Based on lysimeter data and simulation results (Fayer et al. 1999).
Modified RCRA Subtitle C	500	0.1	Based on lysimeter data and simulation results (Fayer et al. 1999).
Standard RCRA Subtitle C	30	0.1	No data; recommendation is based on presence of geomembrane, 2-ft thick clay admix layer, and short design life.
Modified RCRA Subtitle D	100	0.1	Based on simulation results using parameters from Fayer et al. (1999).

(a) Hydrologic Evaluation of Landfill Performance model, after Schroeder (1997).

No guidance is available for specifying barrier performance after the design life. However, an immediate decrease in performance is not expected, and it is likely that some of these barriers will perform as designed far beyond their design life. Without data to understand and predict that long-term performance, however, an assumption was made that the performance would degrade stepwise after reaching its design life, until the recharge rate matches the rate in the surrounding environment. This approach is based on the assumption that a degraded cover eventually will return to its natural state and, at that time, will behave like the surrounding environment. A further assumption was that the period of degradation would be the same as the design life. For example, the Modified RCRA Subtitle C Cover would perform as designed for 500 years and then degrade stepwise in five equal steps over the next 500 years to the point at which recharge rates are equivalent to the rates of the natural surrounding environment.

The schedule and type of engineered cover to be applied to each site was based on the Hanford Disposition Baseline as defined by Kincaid et al. (2000).

#### **L.2.4.3.2 Natural (Non-Barrier) Recharge Rates**

Most of the waste sites at Hanford have not had a surface barrier, and it is assumed that many sites will not have a surface barrier applied prior to site closure. The effort to estimate recharge in these areas addressed four site conditions:

- undisturbed soil and shrub-steppe vegetation
- undisturbed soil with no vegetation
- disturbed soil with no vegetation
- disturbed soil with shrub-steppe vegetation.

The Hanford soil map (Hajek 1966) was examined to identify the soil types prevalent in the waste areas. Table L.8 lists the four soil types that dominate the areas being evaluated in the initial assessment and their recharge rates. It was assumed that these soils, in their undisturbed condition, support a shrub-steppe plant community.

For some Hanford activities, the shrub-steppe plant community often was removed while leaving the existing soil type relatively intact. For other activities, the sites were excavated, which removed the existing soil structure, and then backfilled with Hanford formation sand or gravel. Some activities also covered selected surface areas with a layer of gravel (for example, the tank farms). Table L.9 shows the estimated recharge rates for native soils and backfilled sediments without vegetation. Eventually, the disturbed areas may become revegetated and a shrub-steppe plant community re-established. Under these conditions, it is assumed that the estimated recharge rate will return to that equivalent to the pre-Hanford conditions after a period of 100 years.

**Table L.8.** Estimated Recharge Rates for Predominant Soil Types and Sediments with a Shrub-Steppe Plant Community

Soil Type	Recharge Rate Estimate (mm/yr)	Description
Ephrata stony loam (Eb)	1.5	No data; used estimate for E1, which is a similar soil.
Ephrata sandy loam (E1)	1.5	Average of two estimates (1.2; 1.8) from deep (> 10 m) chloride data collected from the two boreholes B17 and B18 (Prych 1998).
Burbank loamy sand (Ba)	3.0	Average of three estimates (0.66, 2.8, 5.5) from deep (> 10 m) chloride data collected from the three boreholes B10, B12, and B20 (Prych 1998).
Rupert sand (Rp) inside the 200 East Area	0.9	Average of four estimates (0.16, 0.58, 1.0, and 1.8) from deep (> 10 m) chloride data collected from the four boreholes E24-161, E24-162, B8501, B8502 (Fayer et al. 1999).
Rupert sand (Rp) outside the 200 East Area	4.0	Estimated from chloride data collected from a borehole near the Wye Barricade (Murphy et al. 1996).
Hanford formation sand	4.0	No data; used estimate for Rupert sand outside the 200 East Area.

**Table L.9.** Estimated Recharge Rates for Native Soils and Backfilled Sediments without Vegetation

Soil Type	Recharge Rate Estimate (mm/yr)	Description
Ephrata stony loam (Eb)	17.3	Simulation estimate from Fayer and Walters (1995).
Ephrata sandy loam (E1)	17.3	Simulation estimate from Fayer and Walters (1995).
Burbank loamy sand (Ba)	52.5	Simulation estimate from Fayer et al. (1999).
Rupert sand (Rp)	44.3	Simulation estimate from Fayer et al. (1999).
Hanford formation sand	55.4	8-yr lysimeter record for Hanford sand (Fayer and Walters 1995).
Graveled surface	104	8-yr lysimeter record for gravelled surface (Fayer et al. 1999).

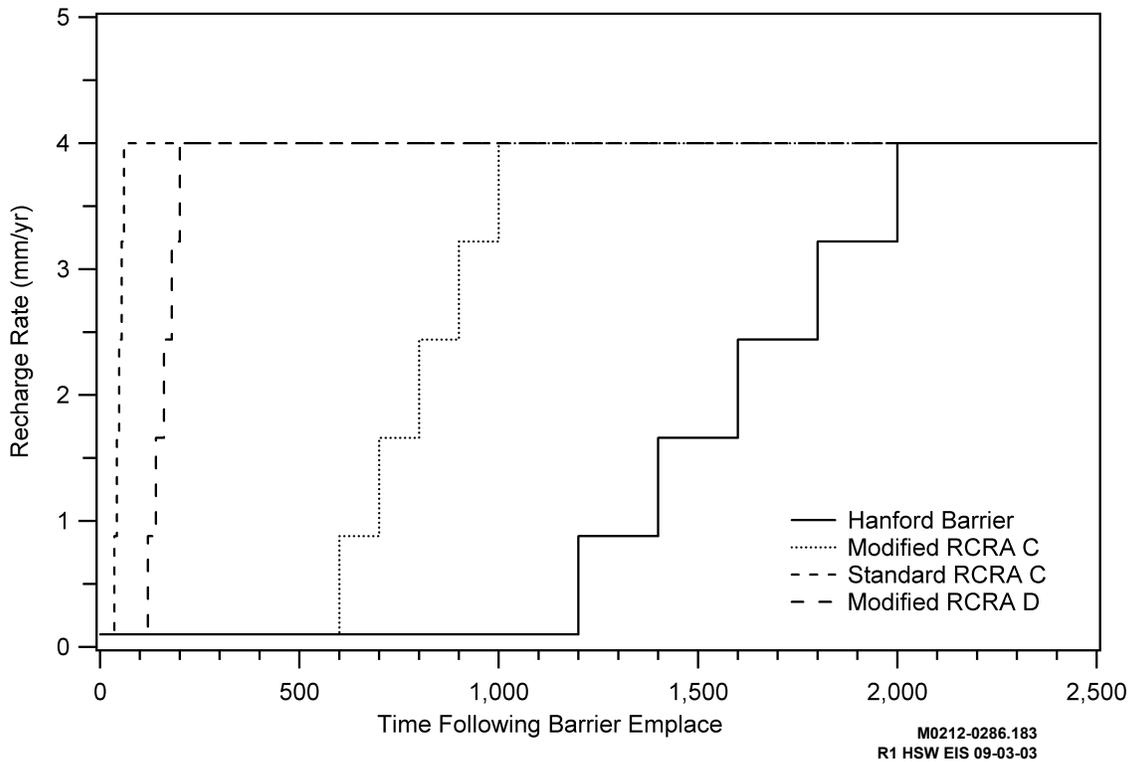
#### L.2.4.3.3 Summary of Recharge Estimates for the Initial Assessment

The estimated recharge rates for various surface conditions for each of the 13 geographic areas included in the initial assessment are provided in Table L.10. This table presents a brief description of each setting and identifies the major soil type that was identified visually for each area using the soil map developed by Hajek (1966). If a substantial secondary soil type was present, that soil type is shown in parentheses. Likewise, its recharge rate also is shown in parentheses. Figure L.2 illustrates how the recharge rates for various surface covers were assumed to change over time as performance degrades.

**Table L.10.** Recharge Estimates for the Initial Assessment (substantial secondary soil types and their associated recharge estimates are shown in parentheses)

Area Label	Brief Description	Major (secondary) Soil Type(s) <sup>(a)</sup>	Recharge Rates Used in the Initial SAC Assessment(s) (mm/yr)			
			Pre- and Post-Hanford (shrub-steppe)	Operations (soil intact, no vegetation)	Operations (soil disturbed, with/without vegetation)	Operations (gravel surface, no vegetation)
C	Reactor along river	Eb (Ba)	1.5 (3.0)	17.3 (52.5)	4.0 / 55.4	104
K	Reactor along river	Eb (El)	1.5 (1.5)	17.3 (17.3)	4.0 / 55.4	104
N	Reactor along river	Eb	1.5	17.3	4.0 / 55.4	104
D	Reactor along river	El	1.5	17.3	4.0 / 55.4	104
H	Reactor along river	Ba	3.0	52.5	4.0 / 55.4	104
F	Reactor along river	Rp (El)	4.0 (1.5)	44.3 (17.3)	4.0 / 55.4	104
R	300 Area	Rp (El)	4.0 (1.5)	44.3 (17.3)	4.0 / 55.4	104
G	200 N Area	El (Ba)	1.5 (3.0)	17.3 (52.5)	4.0 / 55.4	104
T	Northern 200 West Area	Rp (Ba)	4.0 (3.0)	44.3 (52.5)	4.0 / 55.4	104
S	Southern 200 West Area and ERDF	Rp	4.0	44.3	4.0 / 55.4	104
A	Southern 200 East Area	Rp (Ba)	0.9 (3.0)	44.3 (52.5)	4.0 / 55.4	104
B	Northwestern 200 East Area	El	1.5	17.3	4.0 / 55.4	104
E	Eastern 200 East Area	Ba (Rp)	3.0 (0.9)	52.5 (44.3)	4.0 / 55.4	104
Eb = Ephrata stony loam		El = Ephrata sandy loam		Ba = Burbank loamy sand		Rp = Rupert sand
(a) Note: Only the major soil types were used to represent each aggregate area.						

The recharge rates estimated for the initial assessment do not account for overland flow from roadways or roofs, water line leaks, or any other anthropogenic additions of water. The rates also do not account for variations within soil type, plant community succession (for example, a takeover by cheatgrass), dune sand deposition, or climate change. Finally, these rates were developed for fairly large geographic areas and may not represent the local recharge rates at specific locations.



**Figure L.2.** Recharge Through Covers as a Function of Time

### L.2.5 Groundwater Module

The Groundwater Module focuses on groundwater that is part of the upper most saturated zone on the Hanford Site. This zone, commonly referred to as the unconfined aquifer, offers a pathway for contaminants released through the vadose zone from past, present, and future site activities to reach the environment accessible to man. Radioactive and hazardous chemicals have been released on the Hanford 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 may affect it in the future. Once in the groundwater, contaminants move along the pathways of least resistance, from higher to lower potentials (for example, elevations), where some contaminants may ultimately discharge into the Columbia River.

The goal of the Groundwater Module is to evaluate the transport of contaminants released from the vadose zone to points of regional discharge of groundwater along the Columbia River within the assessment period. Contaminants released to the groundwater form plumes, some of which extend from their source areas to the Columbia River. The Groundwater Module calculates the concentrations of contaminants in the groundwater for direct use in impact and risk calculations.

Information concerning characterization, modeling, and monitoring of the groundwater system, described in DOE-RL (1999), provides the primary basis for the conceptual model and numerical imple-

mentation of the Groundwater Module supporting the initial assessment. The groundwater conceptual model is an interpretation or working description of the characteristics and dynamics of the physical hydrogeologic system, and it consolidates Hanford Site data (for example, geologic, hydraulic, transport, and contaminant data) into a set of assumptions and concepts that can be quantitatively evaluated.

The Groundwater Module takes the results of the analyses from the vadose zone technical element in the form of contaminant flux from various waste sources. In addition to the influx from the vadose zone element, the Groundwater Module requires information that defines the physical characteristics of the hydrologic system, transport parameters, and natural and artificial recharge rates. Driving forces, including natural recharge from precipitation and artificial recharge from waste disposal activities, contribute to the movement of the contaminants through the vadose zone and into the groundwater of the unconfined aquifer. Several important fate and transport processes, including advection and dispersion, first-order radioactive decay, thermal and chemical interactions with the water and sediment, and contaminant density, may control the fate and transport of the contaminants in the groundwater. For the initial assessment, the thermal and chemical processes considered in the groundwater transport element were limited to assumptions of isothermal conditions, uniform density, and adsorption using the linear sorption isotherm model and, hence, the distribution coefficient,  $K_d$ , concept.

The definition of the hydrologic system is based on previous subsurface investigations from which data on the hydrologic units, unit boundaries, hydraulic conductivity, hydraulic heads, storativity, and specific yield were assembled. Transport parameters are based on both site-specific work of previous investigations and published literature values for parameters including effective porosity, dispersivity, contaminant-specific retardation coefficients, and vertical and horizontal anisotropy. The groundwater flow and transport model also requires estimates of natural recharge rates and locations and magnitude of artificial recharge to the hydrologic system, which are available from historic records and direct measurements. Model domain boundaries are established for the flow system based on site-specific knowledge and output data requirements. Boundaries are established along the northern and eastern portion of the site corresponding to the course of the Columbia River and along the southeastern portion of the model along the course of the Yakima River. Basalt ridgelines and the Cold Creek Valley form the western model domain boundaries. Lower flow boundaries are established between the confined basalt aquifer system and the overlying unconfined aquifer. A complete description of the groundwater conceptual model is provided in Appendix D of DOE-RL (1999).

The conceptual model of the groundwater system used in this assessment is based on nine major hydrogeologic units identified in Thorne and Chamness (1992), Thorne and Newcomer (1992), and Thorne et al. (1993, 1994). Although nine hydrogeologic units were defined, only seven are found in the unconfined aquifer during the period of interest. The Hanford formation combined with the pre-Missoula gravel deposits were designated as model unit 1. Model units 2 and 3 correspond to the early Palouse soil and Plio-Pleistocene deposits, respectively. Odd-numbered Ringold model units (5, 7, and 9) are predominantly coarse-grained sediment. Even-numbered Ringold model units (4, 6, and 8) are predominantly fine-grained sediment with low permeability. The underlying basalt was designated model unit 10. However, the basalt was assigned a very low hydraulic conductivity and was essentially treated as an impermeable unit in the model.

A complete description of the sitewide groundwater flow and transport model used in the current assessment is provided in Cole et al. (2001a). The current Hanford sitewide groundwater model is implemented with the CFEST code (Gupta et al. 1987). The current model has been transient-inverse calibrated to the record of hydraulic head (that is, water-table elevation) measurements from Hanford startup in 1944 to the present.

Simulated flow conditions during the historical period of operations that provided the basis for all transport calculations are described in Cole et al. (2001b). These flow conditions incorporate the effect of large-volume discharges of wastewater to a variety of waste facilities since the inception of the Hanford Site in 1943. These operational discharges have raised the water table, created groundwater mounds, and been the source of local- and regional-scale contaminant plumes under waste management sites and facilities along the Columbia River and in the Central Plateau. Since 1988, the mission of the Hanford Site has changed from weapons material production to environmental restoration. As a result, wastewater discharges have declined substantially, which caused the water table to decline substantially over the past decade. Simulation of future water table decline indicates that the aquifer would return to more natural levels within 150 to 300 years. These results are consistent with previous work on future water table declines described in Cole et al. (1997) and Kincaid et al. (1998).

The SAC has been inverse calibrated to the hydraulic head data, and history matched to the most abundant data, that for tritium the most mobile of radioactive contaminants. Use of the hydraulic head and tritium data sets provide confidence that the underlying liquid release, vadose zone, and groundwater models duplicate the essential features of the tritium groundwater plume; extent of tritium contamination, its arrival at the Columbia River, and its decay as a function of time.

Historical field data specific to solid waste disposal facilities are not available. Solid wastes disposed in containers of either cardboard, wood, plastic, or metal construction are not believed to have released from their containers and contaminated the sediments immediately below the disposal facilities. It may be decades or centuries before contaminants in some solid waste disposal facilities reach the underlying groundwater and are available for detection. Thus history matching to solid waste releases is not feasible at this time.

Calculation of dose, risk, or impact from contaminated groundwater requires groundwater contaminant concentrations. The three-dimensional groundwater model includes nodes throughout each vertical profile of the unconfined aquifer. To define the maximum concentration of each contaminant at a land surface location above the aquifer, all values in the underlying vertical profile are considered. Thus the suite of maximum concentrations at a given location are selected regardless of their vertical position within the aquifer model, and the maximums used for different contaminants need not come from the same vertical horizon of the model. This is a conservatism in the groundwater contaminant concentrations used in all dose, risk, and impact simulations.

## L.2.6 River Transport Module

The River Transport Module simulates the Columbia River between the Vernita Bridge and McNary Dam and includes inputs from groundwater and the Yakima and the Snake Rivers. The contaminants modeled in the river come from three sources:

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

Groundwater and irrigation return discharges to the river along the shore opposite Hanford are not included in the initial assessment.

The MASS2 code provides the basis of the River Transport Module (Richmond et al. 2000). MASS2 is a two-dimensional, depth-averaged hydrodynamics model that provides the capability to simulate the lateral (bank-to-bank) variation of flow and transport of sediments and contaminants. The model incorporates river hydraulics (velocity and water depth), contaminant influx to the river through groundwater and point sources, sediment and contaminant transport, and adsorption/desorption of contaminant to sediments.

The Columbia River is the largest North American River to discharge into the Pacific Ocean. The river originates in Canada and flows south 1953 km (1212 mi) to the Pacific Ocean. The watershed drains a total of 670,000 km<sup>2</sup> (258,620 mi<sup>2</sup>) and receives water from seven states and one Canadian province. Key contributors to the flow are runoff from the Cascade Mountains in Washington and Oregon and from the western slopes of the Rocky Mountains in Idaho, Montana, and British Columbia. Average annual flows below Priest Rapids and The Dalles dams are approximately 3360 m<sup>3</sup>/s (120,000 ft<sup>3</sup>/s) and 5376 m<sup>3</sup>/s (192,000 ft<sup>3</sup>/s), respectively. Numerous dams within the United States and Canada regulate flow on the main stem of the Columbia River. Priest Rapids Dam is the nearest dam upstream of the Hanford Site, and McNary Dam is the nearest downstream. The dams on the lower Columbia River greatly increase the water travel times from the upper reaches of the river to the mouth, subsequently reducing the sediment loads discharged downstream. The increased travel times also allow for greater radionuclide deposition and decay.

The Snake, Yakima, and Walla Walla rivers all contribute suspended sediment to the Columbia River; contributions from the Snake River are the most substantial. Since completion of McNary Dam in 1953, much of the sediment load has been trapped behind the dam. However, at McNary Dam and other Columbia River dams, some of the trapped sediment is resuspended and transported downstream by seasonal high discharges. As expected, much of this material is redeposited behind dams located farther downstream. Within the domain of this model that only extends to McNary Dam, sediment accumulates faster on the Oregon shore than on the Washington shore because sediment input from the Snake and Walla Walla rivers stays near the shore on the Oregon side. Sediment-monitoring samples taken for the

Hanford Sitewide Surface Environmental Surveillance Project indicated cobble and coarse- and fine-sand bed sediments at sampling locations along the Hanford Site (Blanton et al. 1995). Silt and clay sediment was observed at the McNary Dam sampling site.

The conceptual model used in the initial assessment included the environmental pathways and transport processes that affect contaminant transport in surface water systems. The physical processes include river hydrodynamics and suspended sediment transport, deposition, and resuspension. Because of runtime constraints, suspended and bed sediments were modeled with only the silt-size fraction. The contaminant transport processes include surface water advection and dispersion, sorption and desorption to sediments, decay, and exchange between bed pore water and the overlying surface water. The initial assessment River Transport Module, which is the MASS2 model, included these key features, events, and processes in the mathematical implementation of the conceptual model.

### **L.2.7 Risk and Impact**

The SAC has implemented a suite of impact assessment modules that treat ecological, economic, cultural, and human impacts and include internal stochastic capabilities. An initial assessment of the Hanford Site using these modules is provided in Bryce et al. (2002). The HUMAN code (Eslinger et al. 2002b) was used in calculations for this EIS. The human impact model includes exposure pathways from ingestion, inhalation, skin contact, and direct radiation exposure. Relative exposures to these sources depend on individual lifestyles or exposure scenarios.

The human exposure scenarios for the EIS were limited to the ingestion of water. In addition, the ingestion dose factors were selected as deterministic rather than stochastic factors. With these assumptions, annual human dose calculations do not depend on stochastic variables internal to the human exposure model. Thus, all variability in the human doses arises from the variability in the inventory, release, and transport models. The dose factor used for ingestion of technetium-99 was  $1.5 \times 10^{-9}$  rem/pCi, uranium-238 was  $2.5 \times 10^{-7}$  rem/pCi, and iodine-129 was  $2.7 \times 10^{-7}$  rem/pCi. These values were obtained by converting the values in Table 2.2 of Eckerman et al. (1988) from Sv/Bq to rem/pCi (the values were multiplied by a conversion factor of 3700).

Intrusion events by man, vegetation, or animals and the potential for terrestrial ecological pathways to be impacted by Hanford Site wastes in shallow earth deposits is an intrusion analysis—not a long-term exposure analysis. Intrusion analyses are part of the site-specific or waste-specific analyses included in remedial investigation/feasibility studies required under CERCLA, and performance assessment required by DOE Order 435.1 (DOE 2001). Intrusion analyses contribute to our understanding of the waste concentration that can be safely disposed of (that is, at levels less than chronic and acute intruder dose limits), and of the performance necessary in a barrier system to prevent intrusion by man, vegetation, or animals. However, because intrusion exposures are not included in long-term exposure scenarios, such analyses are not included in the sitewide assessment tool (the SAC).

The version of SAC applied to the initial assessment (Bryce et al. 2002) and in the HSW EIS does not include a terrestrial ecological pathway analysis. Essentially, the SAC does not analyze intruder exposure/risk scenarios. Design of the SAC tool was predicated on the assumption that the Hanford Site

would be closed following the remediation of all sites, and the further assumption that any contaminants at substantial levels in the subsurface would be covered with a proven infiltration and intrusion barrier. A Modified RCRA Subtitle C Barrier has been proposed for waste sites receiving surface barriers on the Central Plateau. Thus, the long-term exposure scenarios do not include intrusion as a source of contamination.

### **L.2.8 Uncertainty**

The SAC was designed to provide a stochastic simulation capability able to quantify uncertainty through a Monte Carlo analysis. An uncertainty analysis can be completed for the SAC results. The goal of such an uncertainty analysis is to determine the model parameters that contribute the most variability to the performance measures. Results of the stochastic realizations can also be used to reveal the maximum–minimum range of performance measures.

The uncertainty analysis addresses the role of uncertainty as caused by the variation of parameters within the modeling systems. It does not address causes of errors between modeled and observed data. It does not address uncertainty due to the use of different models. In addition, the analysis of uncertainty does not differentiate between uncertainties due to lack of knowledge and uncertainty due to natural variability in the parameters.

The uncertainty analysis can identify controlling sources of variability in the simulation estimates of the performance measure, but not necessarily the source of the overall magnitude of the performance measure. However, the source of the overall magnitude is obtained from direct examination of model results.

The uncertainty analysis technique employed is a step-wise linear regression analysis using the output results and input parameters of an assessment. Because the SAC uses a sequential analysis structure (that is, analysis progressively treats inventory, release, vadose zone), a top-down hierarchical analysis is performed to identify first-tier quantities (for example, derived quantities like tritium concentration in groundwater), and associated second-tier parameters (for example, unsaturated hydraulic properties, distribution coefficient) responsible for variability.

The initial assessment (Bryce et al. 2002) demonstrated that a relatively small number of input parameters could determine most of the variability in calculated performance measures. It was observed that when the performance measure is human dose, variability with regard to individual behavior and exposure affects uncertainty in the estimated dose more than variability in inventory, release, or environmental transport of the contaminants.

## **L.3 Results**

Results of the initial assessment for a 10,000-year period conducted using the SAC software are presented below in three sections. Section L.3.1 details the release of contamination to the groundwater from the vadose zone. Section L.3.2 presents the drinking water dose that occurs from a 2-L/d drinking water

exposure to groundwater at various points in the environment. Section L.3.3 presents the drinking water dose from consumption of water in the Columbia River at the City of Richland pump station.

### **L.3.1 Release to Groundwater Results**

Releases to the unconfined aquifer from the vadose zone predicted using the SAC software and data are summarized in this section. Vadose zone releases to the groundwater are aggregated into the following categories for the numerous vadose zone sites simulated:

- solid waste disposal facilities (only “218” sites)
- tanks (only “241” sites)
- liquid discharge (“216” sites plus unplanned release sites and the State-Approved Land Disposal Site)
- ERDF
- commercial low-level radioactive waste disposal (referred to as the US Ecology, Inc., site)
- other sites in the 200 East or 200 West Areas not included in the above categories
- all sites not in the 200 East or 200 West Areas (that is, 100, 300, 400, and 600 Areas).

For each result, both annual releases and the cumulative of all annual releases (undecayed) are presented. Note, releases from ILAW, melters, and naval reactor compartments are omitted. The stochastic capability of the SAC was employed for these simulations, so the following results are shown in each plot:

- individual stochastic results (25 realizations)
- the median result of the 25 realizations—that is, the realization that resulted in the median cumulative release in the year 12,050 A.D. (at the end of the simulation) is emphasized
- the median-inputs simulation—that is, a separate single-realization simulation with SAC using the median value of all stochastic input variables.

The median result as defined by the cumulative release to the groundwater is highlighted in both the annual release and cumulative release plots. Each new pair of annual and cumulative plots identifies a new median case from the 25 realizations simulated.

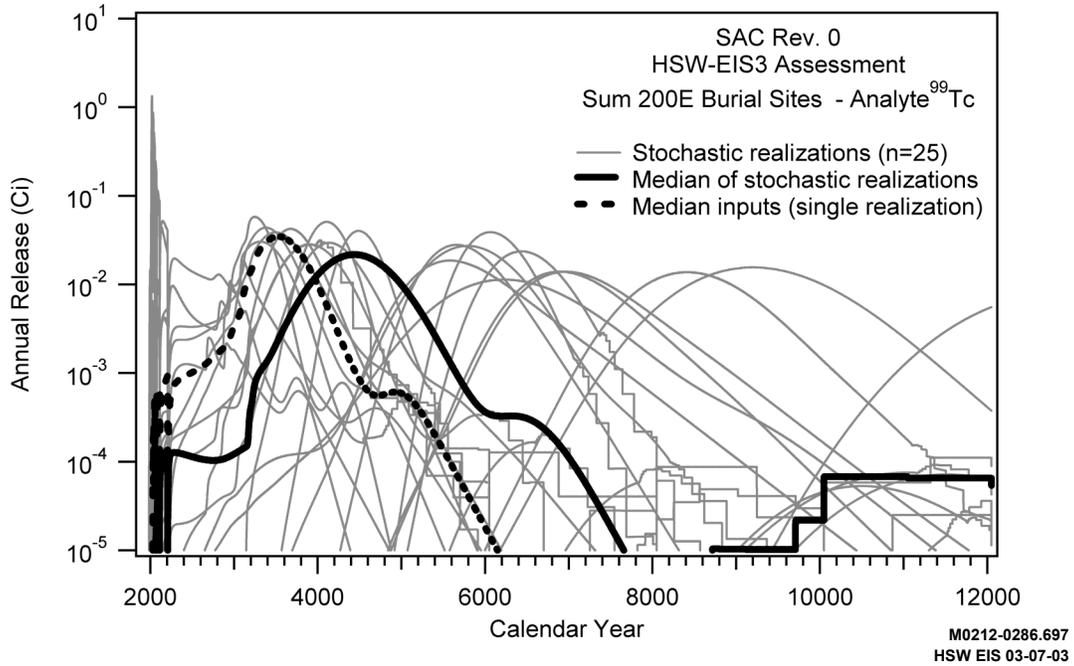
The annual release plots have the appearance of being either a series of piecewise constant (stair-step) values, a smooth continuous curve, or a variable width curve. This is a function of the temporal resolution of both the release model and the vadose zone simulation. Piecewise constant curves result

when the release rate is constant over a period of time and the vadose zone model is able to adopt relatively long time steps (for example, hundreds of years). When either the release or vadose zone model use a fine time step to forecast a more variable release, the release to groundwater appears as a smooth and continuous curve. The appearance of a variable width curve reflects a numerical artifact of the method used to calculate mass release from the vadose zone to the groundwater in the presence of a transient water table. The oscillation in annual values is most pronounced for very small mass releases over long time periods, which is the case for iodine-129. The oscillation is purely cosmetic, because the annual mass release tracks correctly to produce the cumulative mass release and the simulation exhibits mass conservation. In reality, all the annual curves are a series of piecewise constant values.

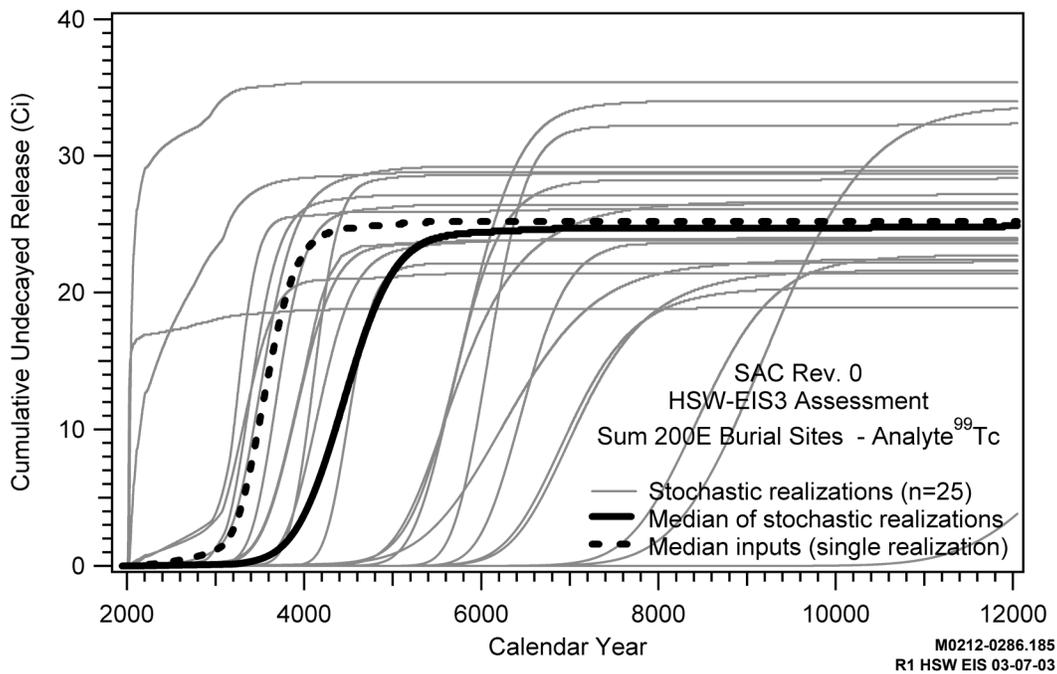
Figures L.3 through L.14 present the vadose zone release to groundwater results for the sum of all solid waste disposal facilities. Each cumulative plot showing the 25 stochastic realizations provides information on the range of cumulative response as well as the median for solid waste disposals. Cumulative releases to groundwater for solid waste disposed of in the Central Plateau range from approximately 323 to approximately 445 Ci for technetium-99 during the 10,000-year analysis period. However, for uranium the release is nil—none in any realization in the 200 East Area and only 5 of 25 realizations exhibit any release in 200 West Area. The median uranium releases for both 200 East and 200 West Areas are zero essentially. For iodine-129, the median release from 200 East Area deposits is approximately 0 Ci, while for 200 West Area it is approximately 0.1 Ci. Iodine-129 releases range from 0 to approximately 2.2 Ci.

Figures L.15 through L.26 present the results for vadose zone releases to groundwater for the sum of all tank sites. Cumulative releases to groundwater for tank waste (that is, past leaks, future losses, and residuals) in the Central Plateau range from approximately 440 to approximately 645 Ci for technetium-99 during the 10,000-year analysis period. As in the case of solid waste, uranium in tank waste does not exhibit substantial release during the 10,000-year period. Only 5 of 25 realizations show uranium releases from 200 East Area tank sites, and hence, the median release is zero. For 200 West Area tank sites, the median case predicts release of approximately 1 Ci of uranium to groundwater during the entire 10,000-year period. For iodine-129, the median releases from 200 East and West Area tank sites are approximately 0.018 and 0.065 Ci, respectively. Iodine-129 releases from tanks range from approximately 0.01 to 0.22 Ci.

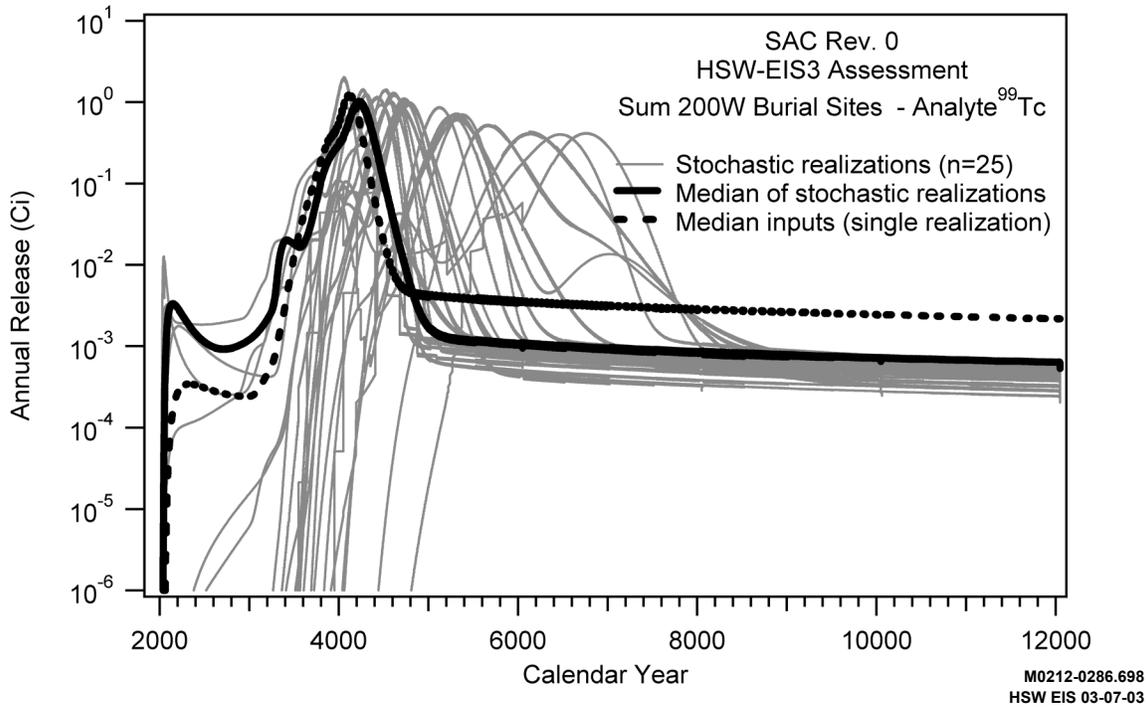
Figures L.27 through L.38 present the vadose zone release to groundwater results for the sum of all liquid discharge and unplanned release (UPR) sites and (in the case of 200 West) the SALDS facility. Cumulative releases to groundwater for liquid releases in the Central Plateau range from approximately 735 to approximately 1030 Ci for technetium-99 during the 10,000-year analysis period. The vast majority of this activity is associated with 200 East Area. The liquid release of uranium ranges between approximately 5 and approximately 100 Ci for the Central Plateau with median values of approximately 26 Ci for 200 East Area and approximately 5 Ci for 200 West Area. In addition to iodine-129 estimated to reside in the groundwater aquifer today (that is, 0.82 Ci in solution and 5.1 Ci overall in solution and sorbed), future releases range between 0 and approximately 1 Ci during the period of analysis. Median values for 200 East and 200 West Area releases to the water table are approximately 0.015 and 0.15 Ci, respectively.



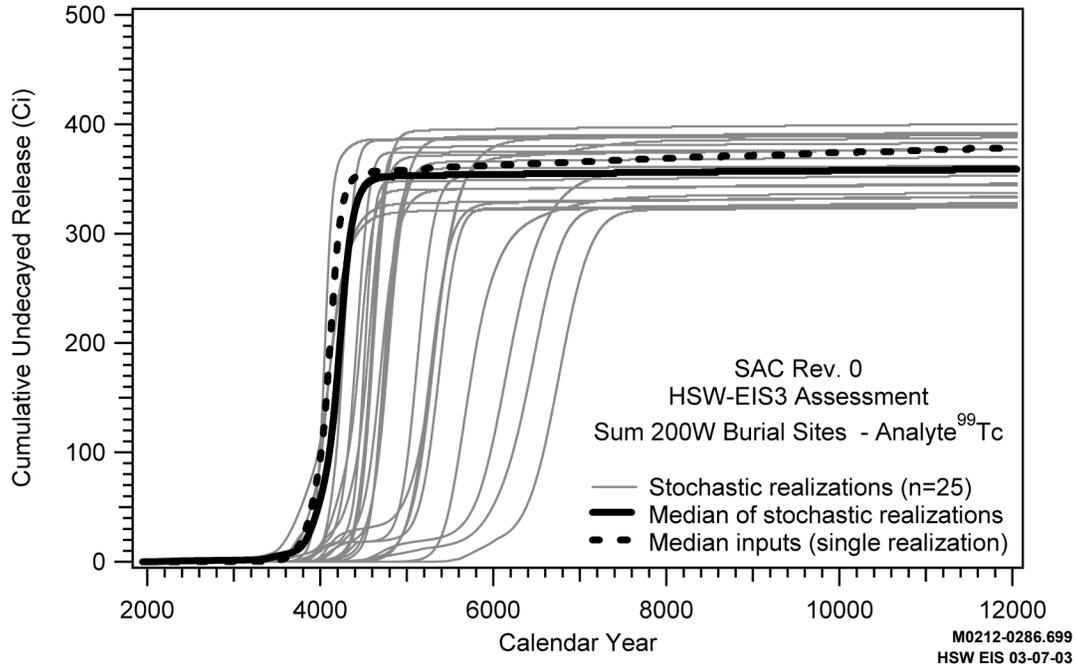
**Figure L.3.** SAC Results for Annual Vadose Zone Release of Technetium-99 from All Solid Waste Disposal Facilities Sites in the 200 East Area (including all “218” sites except 218-E-14 and 218-E-15, and excluding ILAW)



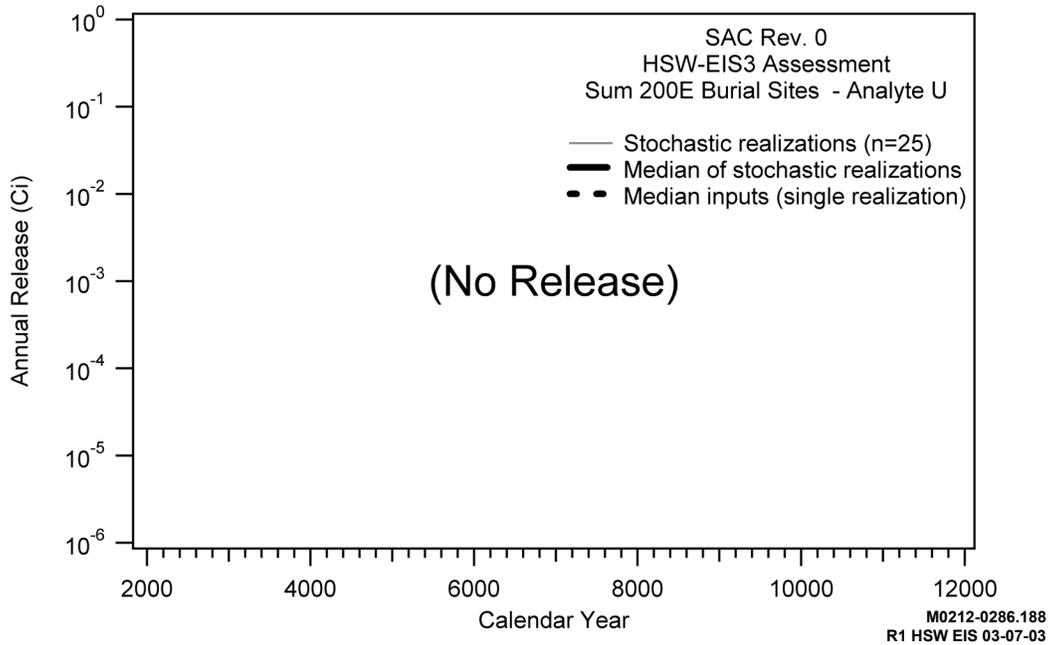
**Figure L.4.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Solid Waste Disposal Facilities Sites in the 200 East Area (including all “218” sites except 218-E-14 and 218-E-15, and excluding ILAW)



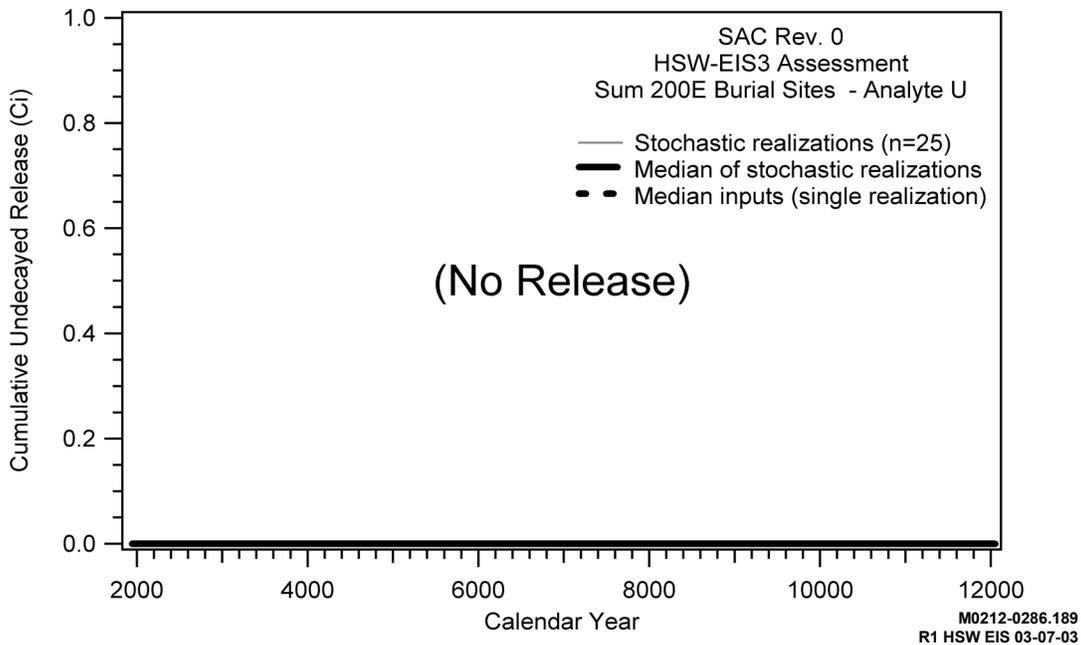
**Figure L.5.** SAC Results for Annual Vadose Zone Release of Technetium-99 from All Solid Waste Disposal Facilities Sites in the 200 West Area (including all “218” sites)



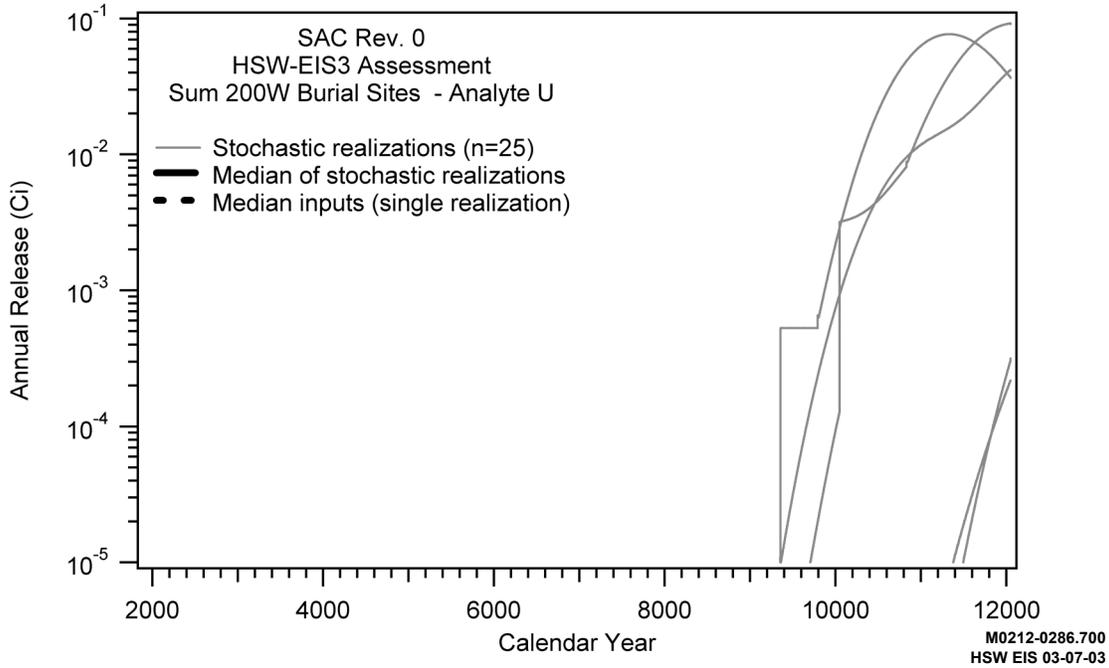
**Figure L.6.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Solid Waste Disposal Facilities Sites in the 200 West Area (including all “218” sites)



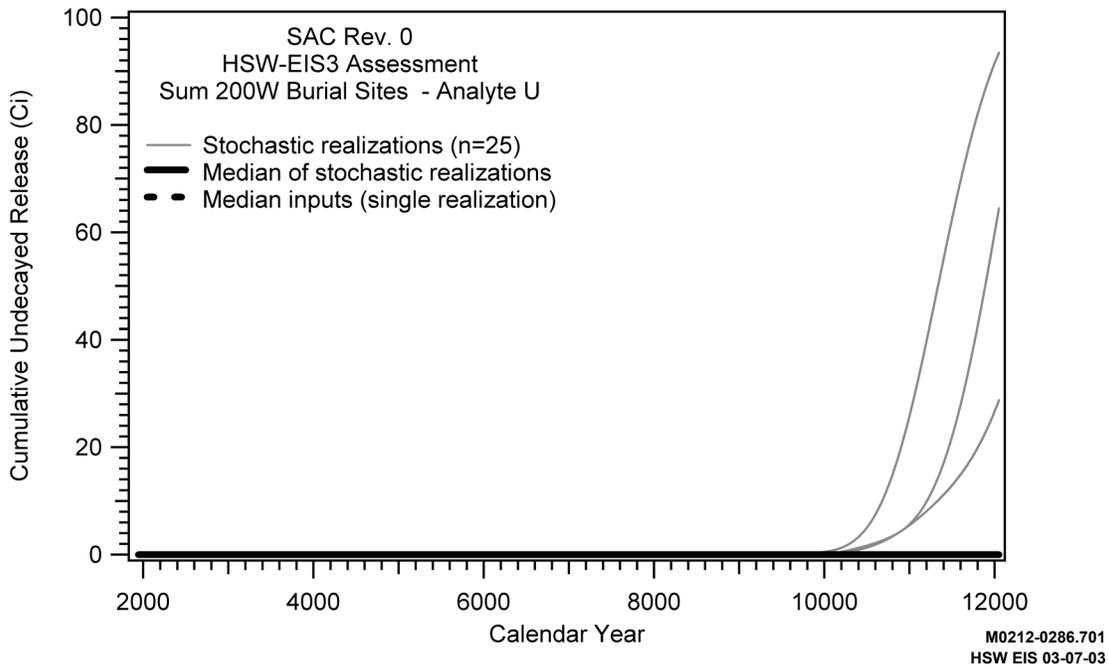
**Figure L.7.** SAC Results for Annual Vadose Zone Release of Uranium from All Solid Waste Disposal Facilities Sites in the 200 East Area (including all “218” sites except 218-E-14 and 218-E-15, and excluding ILAW)



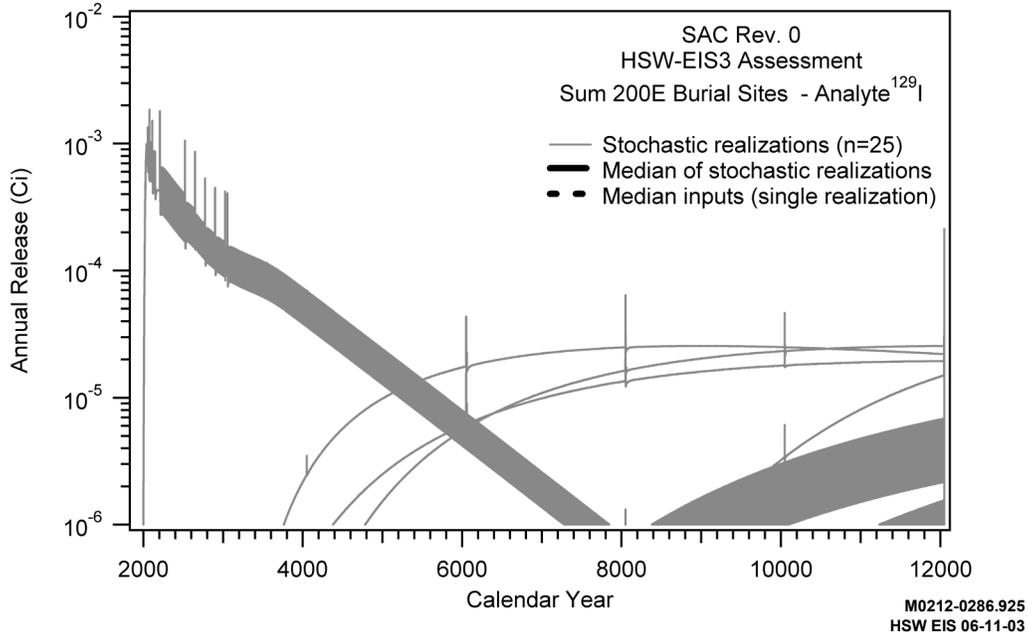
**Figure L.8.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Solid Waste Disposal Facilities Sites in the 200 East Area (including all “218” sites except 218-E-14 and 218-E-15, and excluding ILAW)



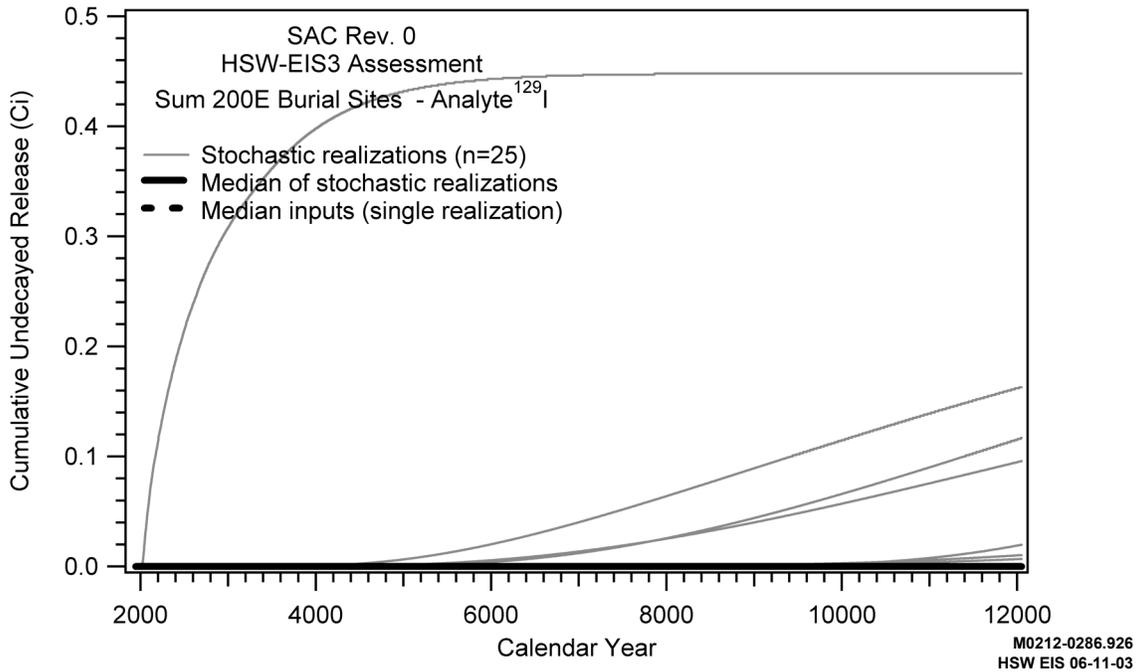
**Figure L.9.** SAC Results for Annual Vadose Zone Release of Uranium from All Solid Waste Disposal Facilities Sites in the 200 West Area (including all “218” sites)



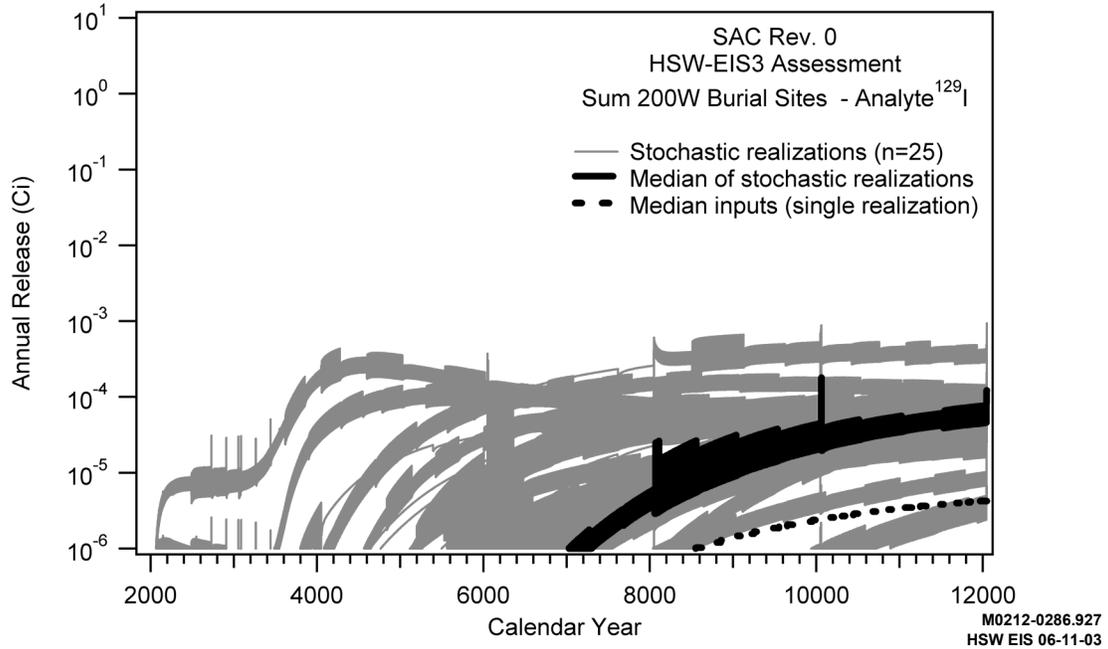
**Figure L.10.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Solid Waste Disposal Facilities Sites in the 200 West Area (including all “218” sites)



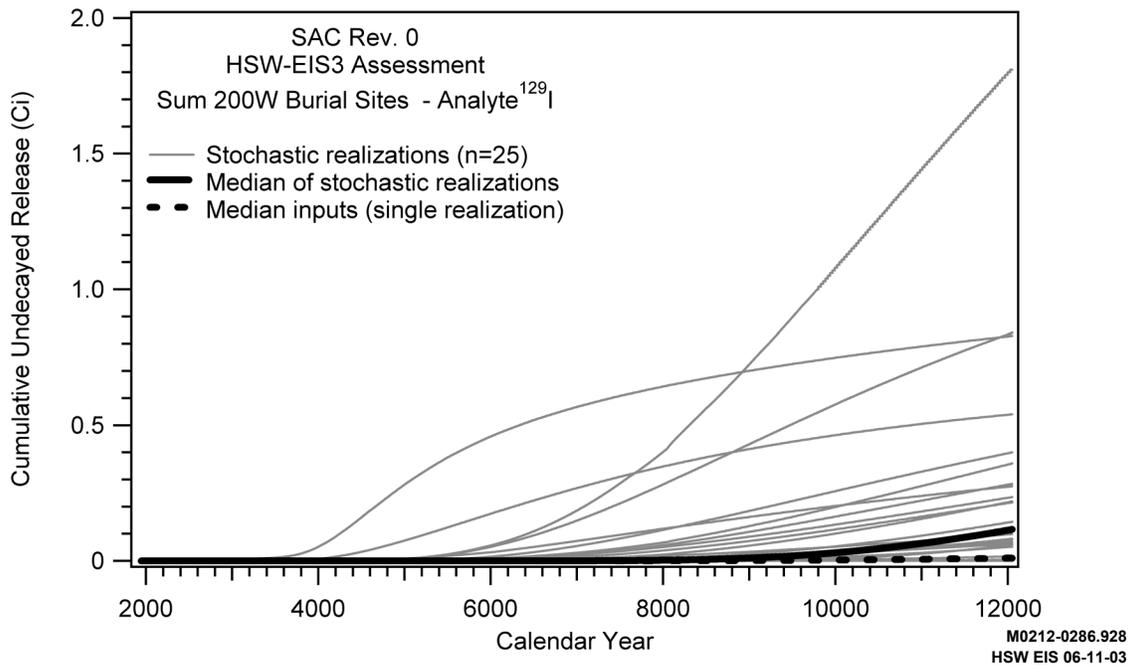
**Figure L.11.** SAC Results for Annual Vadose Zone Release of Iodine-129 from All Solid Waste Disposal Facility Sites in the 200 East Area (including all “218” sites except 218-E-14, 218-E-15, and excluding ILAW).



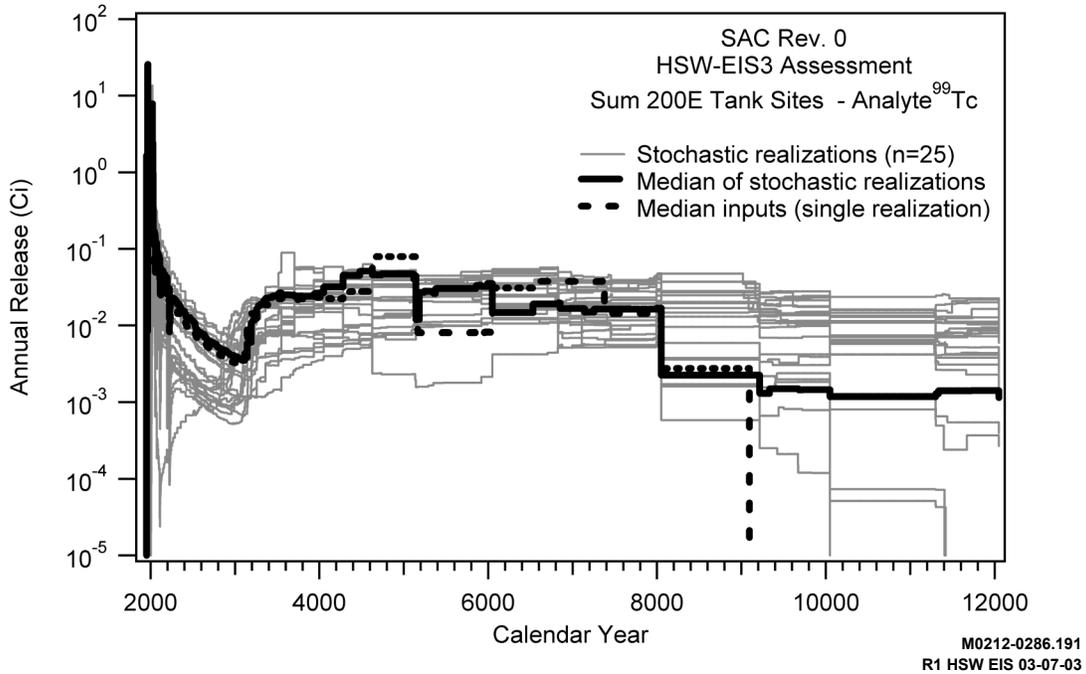
**Figure L.12.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Iodine-129 from All Solid Waste Disposal Facility Sites in the 200 East Area (including all “218” sites except 218-E-14, 218-E-15, and excluding ILAW).



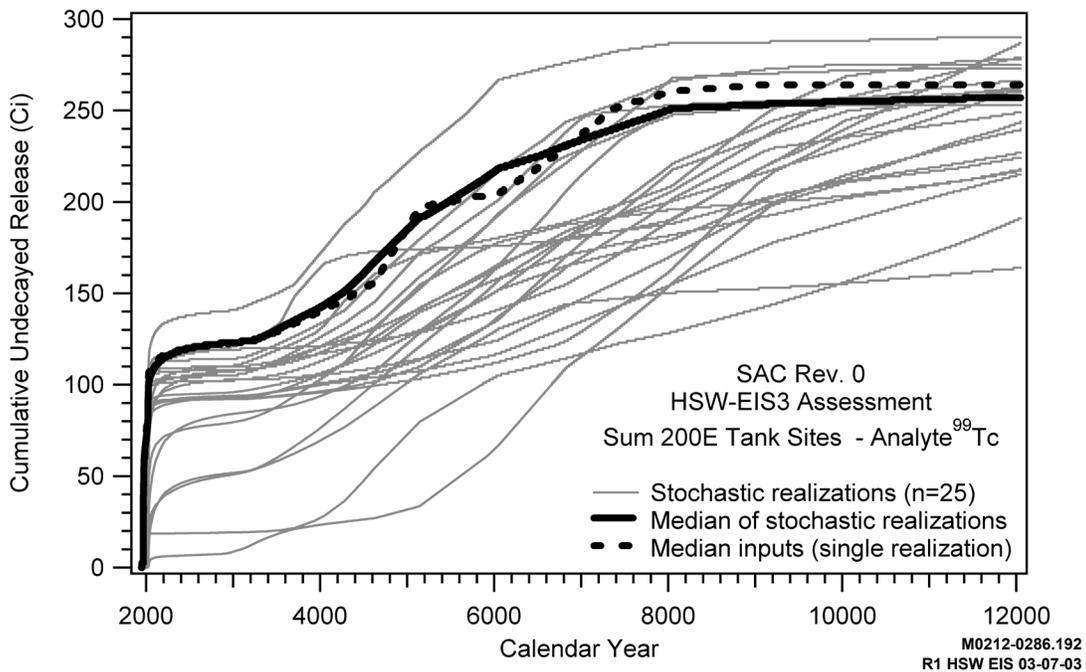
**Figure L.13.** SAC Results for Annual Vadose Zone Release of Iodine-129 from All Solid Waste Disposal Facility Sites in the 200 West Area (including all “218” sites).



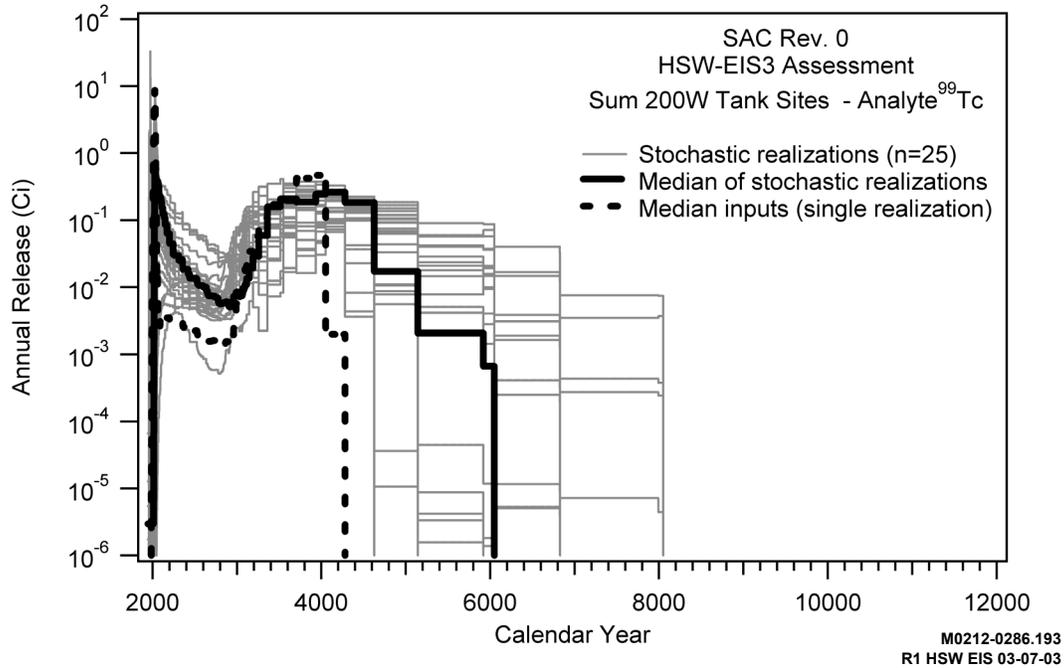
**Figure L.14.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Iodine-129 from All Solid Waste Disposal Facility Sites in the 200 West Area (including all “218” sites).



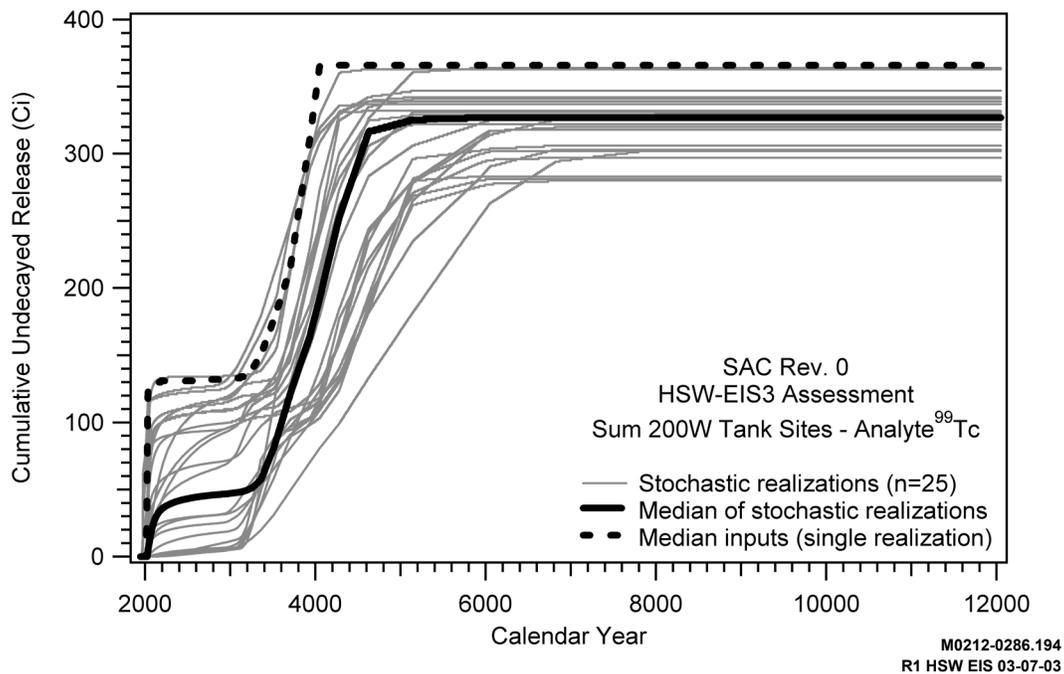
**Figure L.15.** SAC Results for Annual Vadose Zone Release of Technetium-99 from All Tank Sites in the 200 East Area



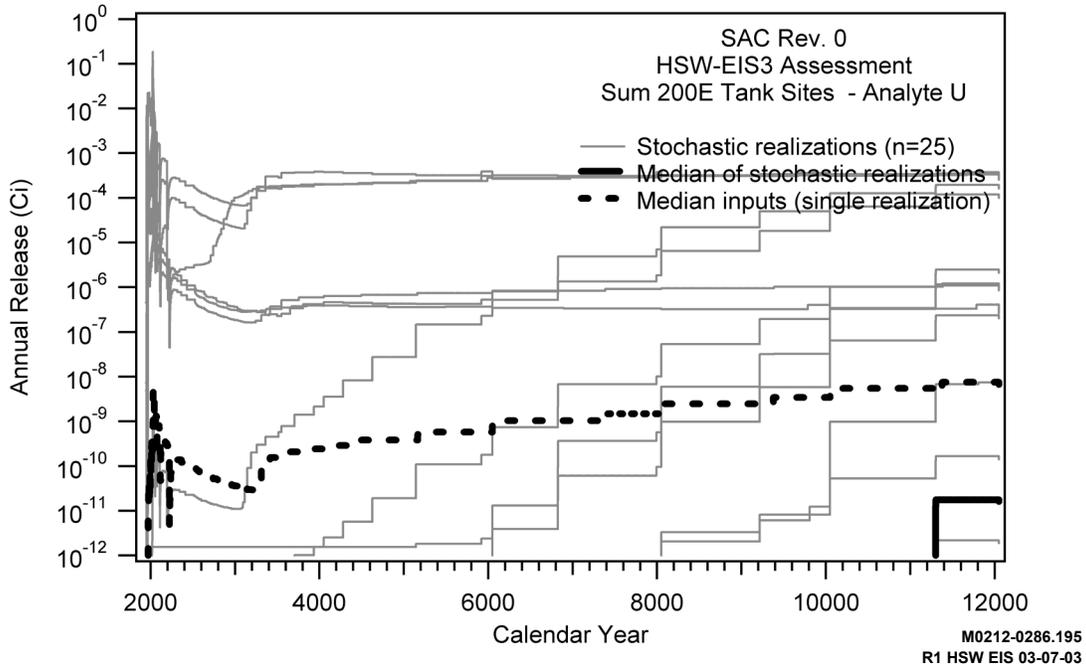
**Figure L.16.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Tank Sites in the 200 East Area



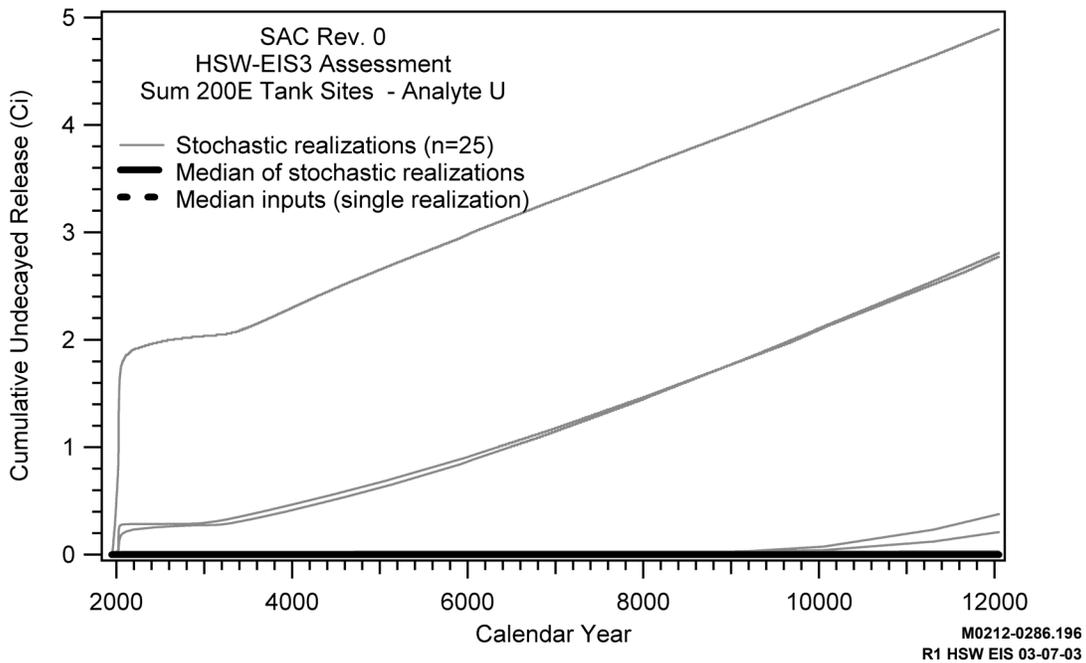
**Figure L.17.** SAC Results for Annual Vadose Zone Release of Technetium-99 from All Tank Sites in the 200 West Area



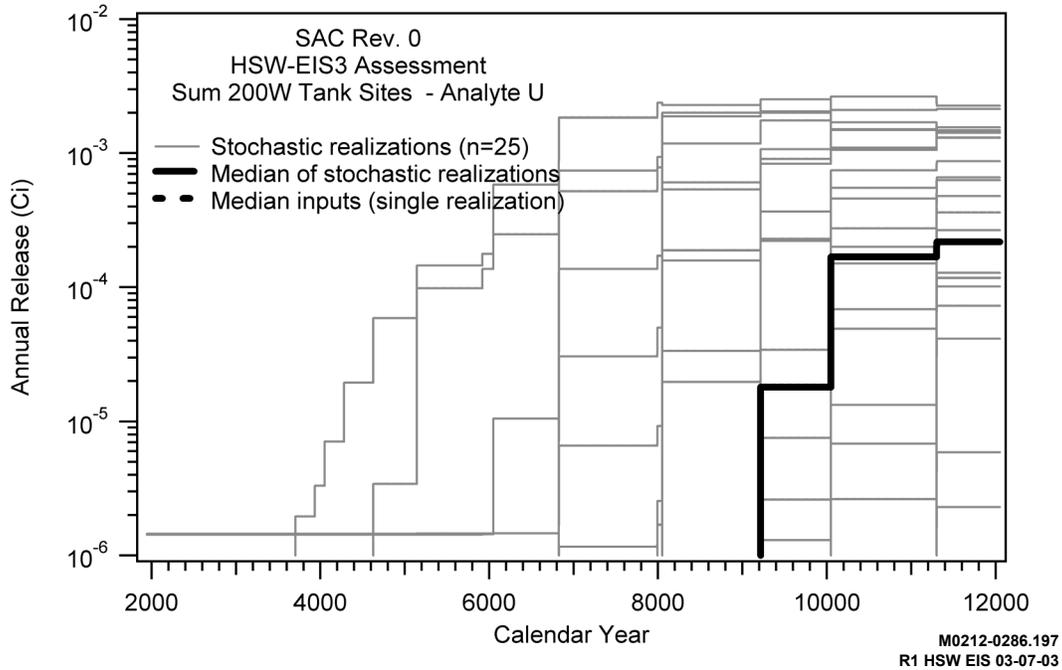
**Figure L.18.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Tank Sites in the 200 West Area



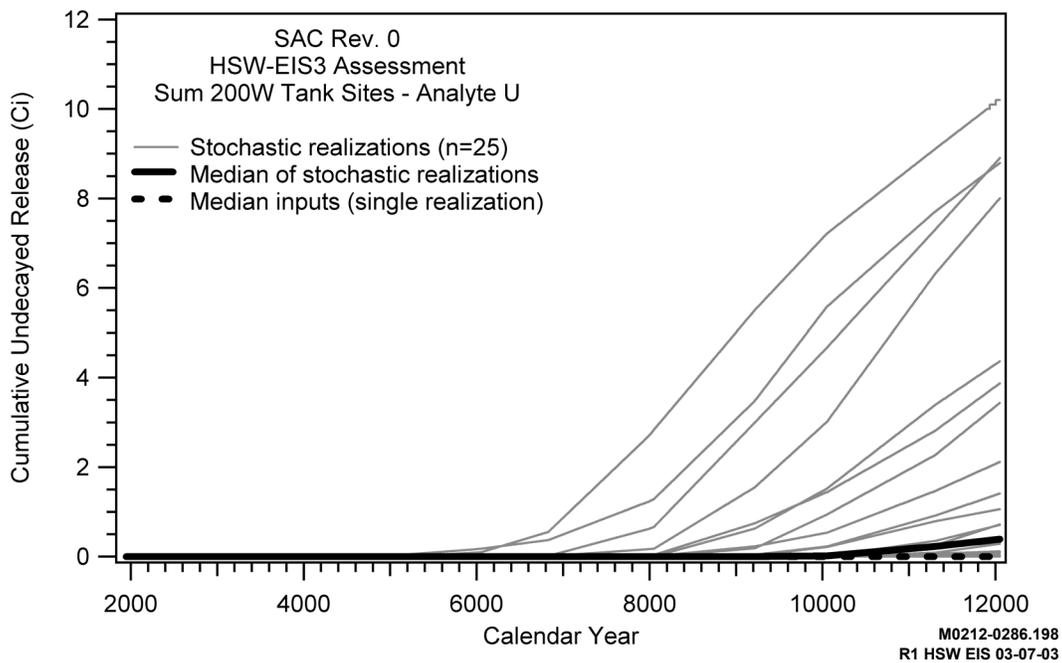
**Figure L.19.** SAC Results for Annual Vadose Zone Release of Uranium from All Tank Sites in the 200 East Area



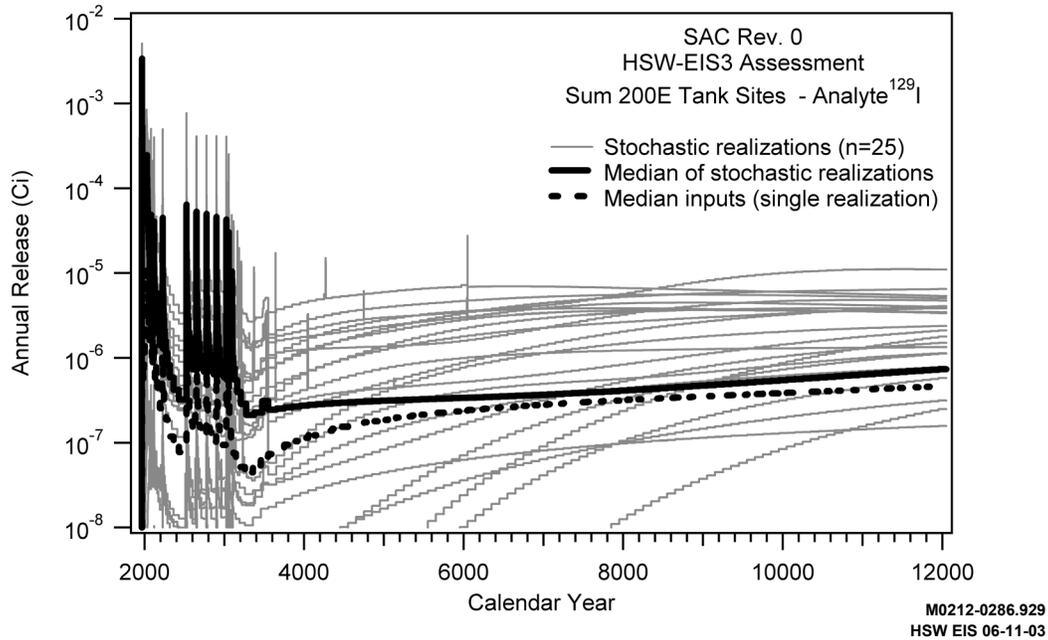
**Figure L.20.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Tank Sites in the 200 East Area



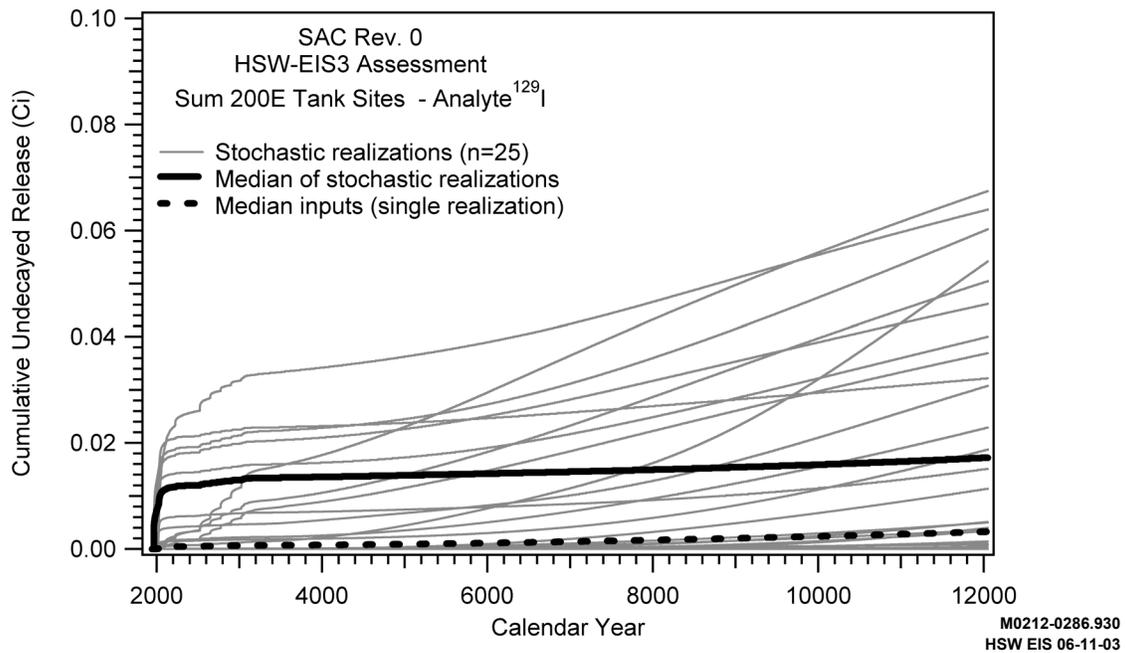
**Figure L.21.** SAC Results for Annual Vadose Zone Release of Uranium from All Tank Sites in the 200 West Area



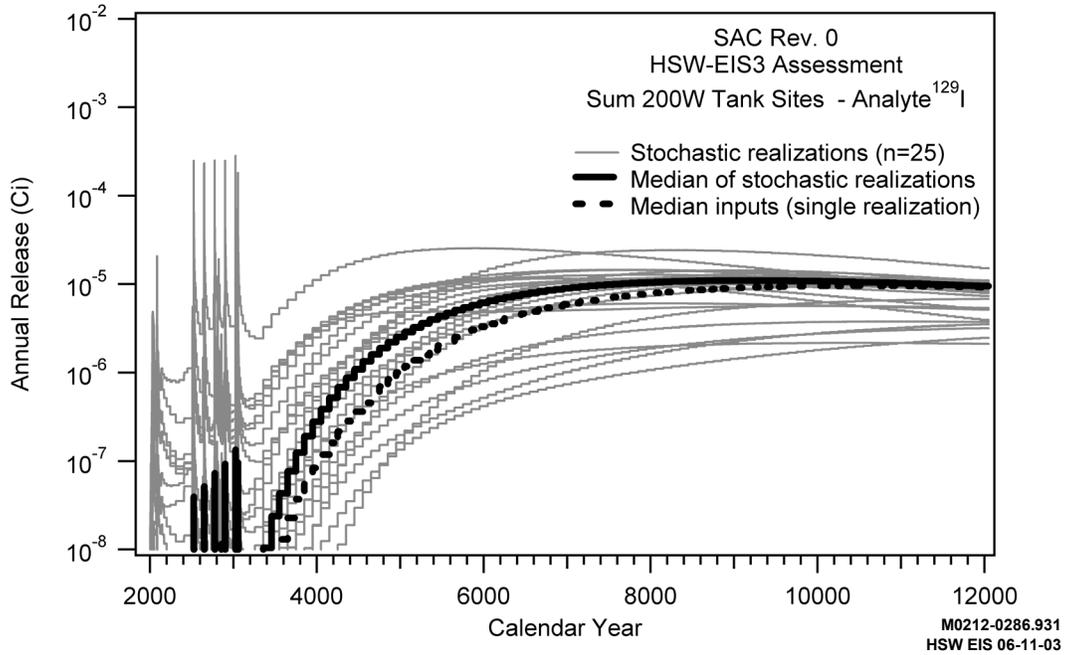
**Figure L.22.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Tank Sites in the 200 West Area



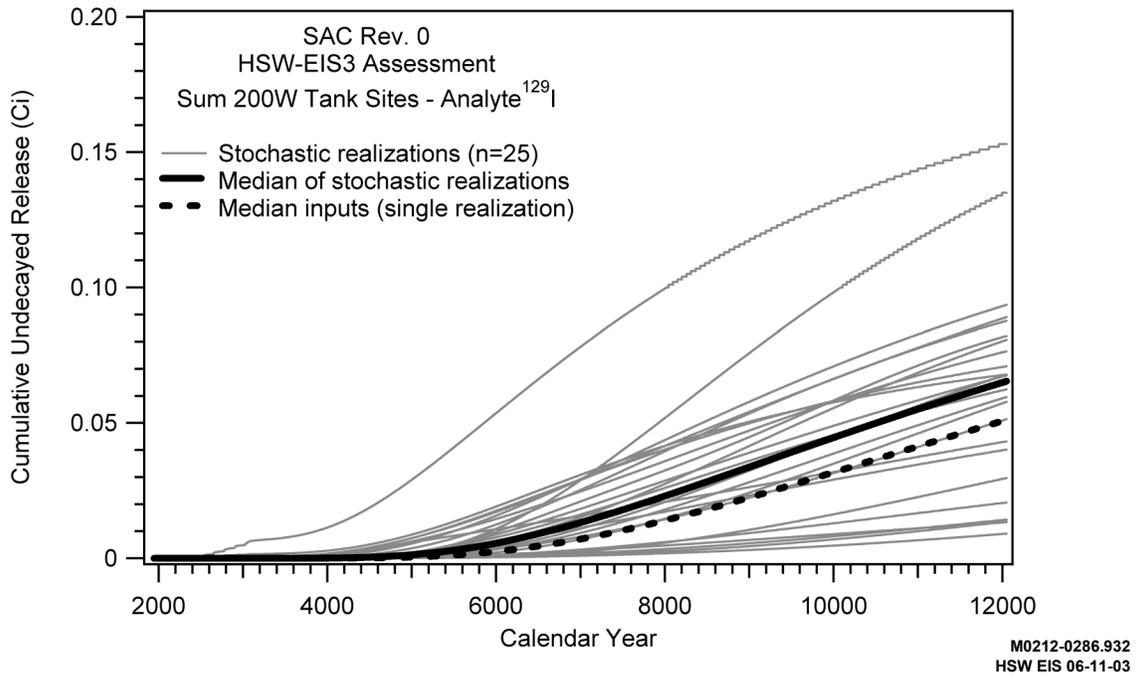
**Figure L.23.** SAC Results for Annual Vadose Zone Release of Iodine-129 from All Tank Sites in the 200 East Area



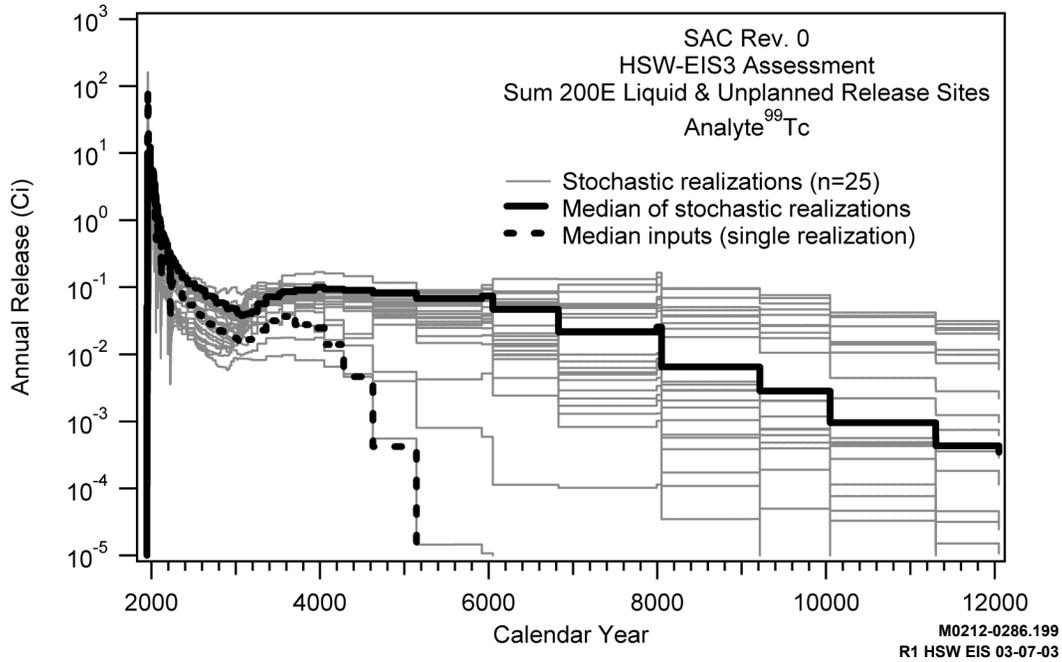
**Figure L.24.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Iodine-129 from All Tank Sites in the 200 East Area



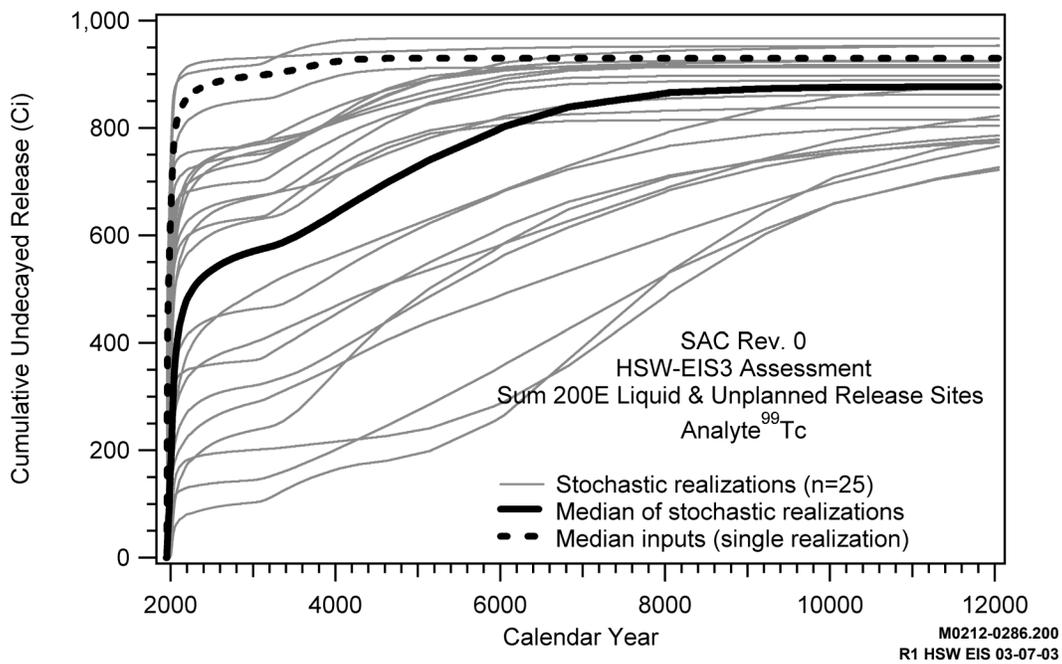
**Figure L.25.** SAC Results for Annual Vadose Zone Release of Iodine-129 from All Tank Sites in the 200 West Area



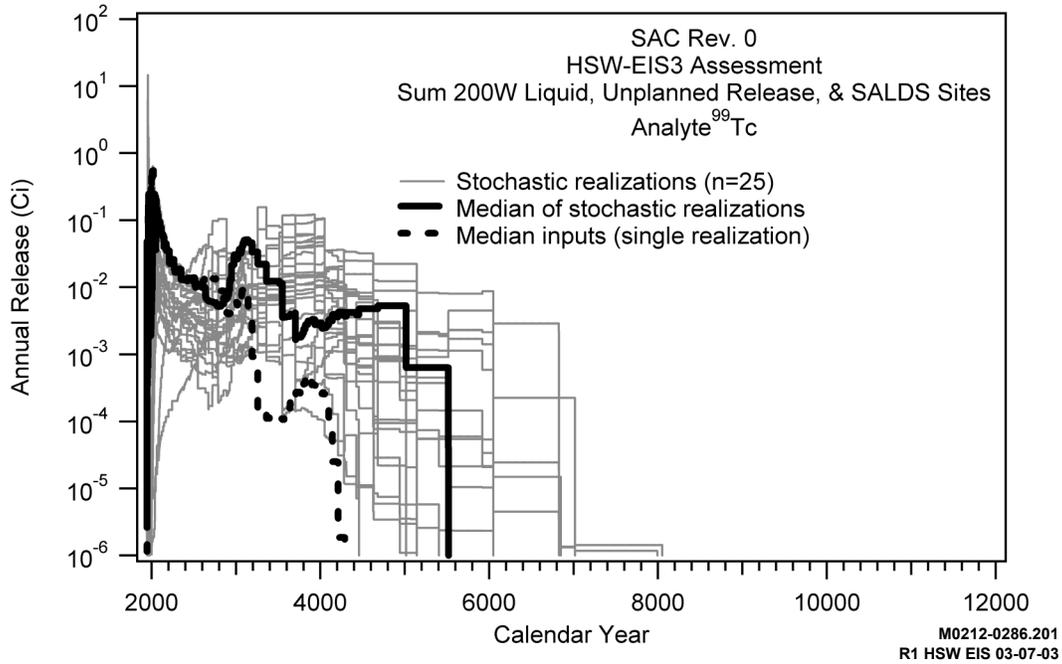
**Figure L.26.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Iodine-129 from All Tank Sites in the 200 West Area



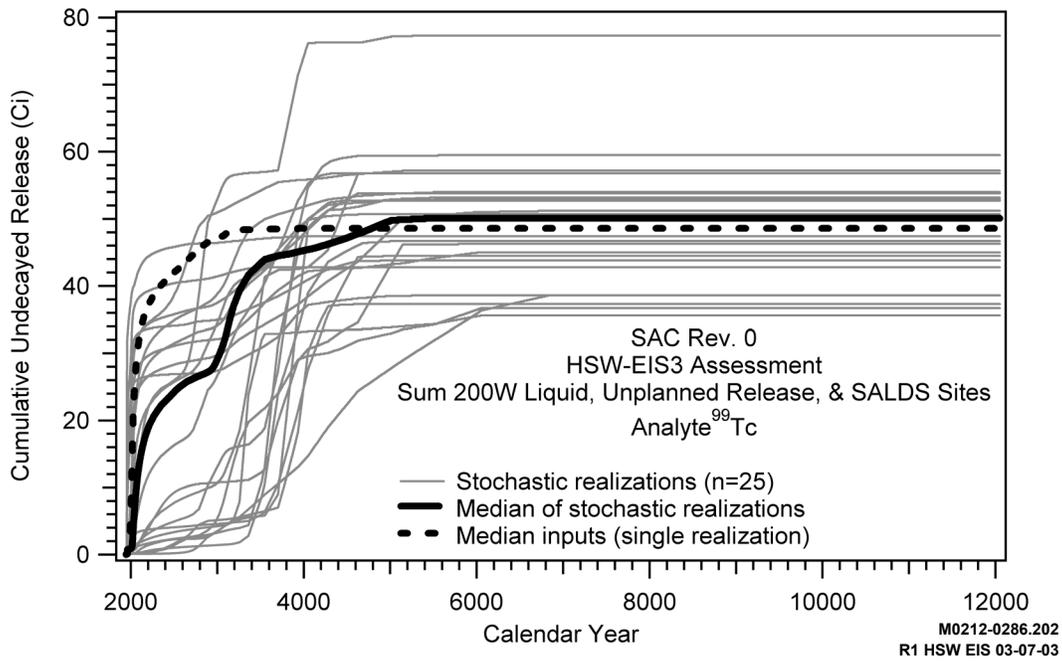
**Figure L.27.** SAC Results for Annual Vadose Zone Release of Technetium-99 from All Liquid Discharge and Unplanned Release Sites in the 200 East Area



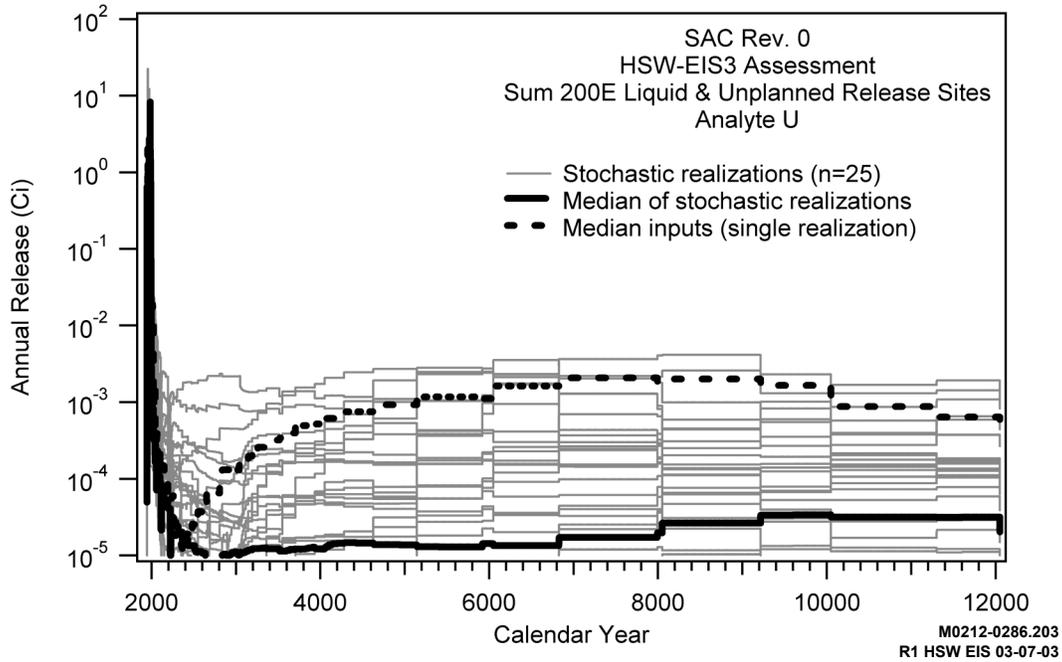
**Figure L.28.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Liquid Discharge and Unplanned Release Sites in the 200 East Area



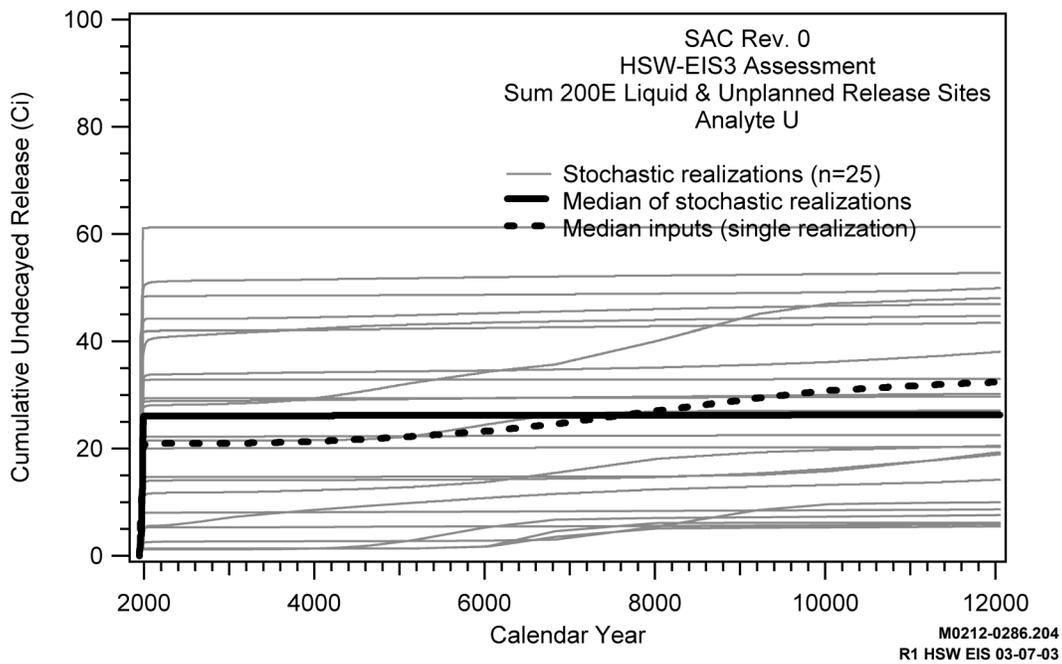
**Figure L.29.** SAC Results for Annual Vadose Zone Release of Technetium-99 from All Liquid Discharge and Unplanned Release Sites in the 200 West Area Plus SALDS



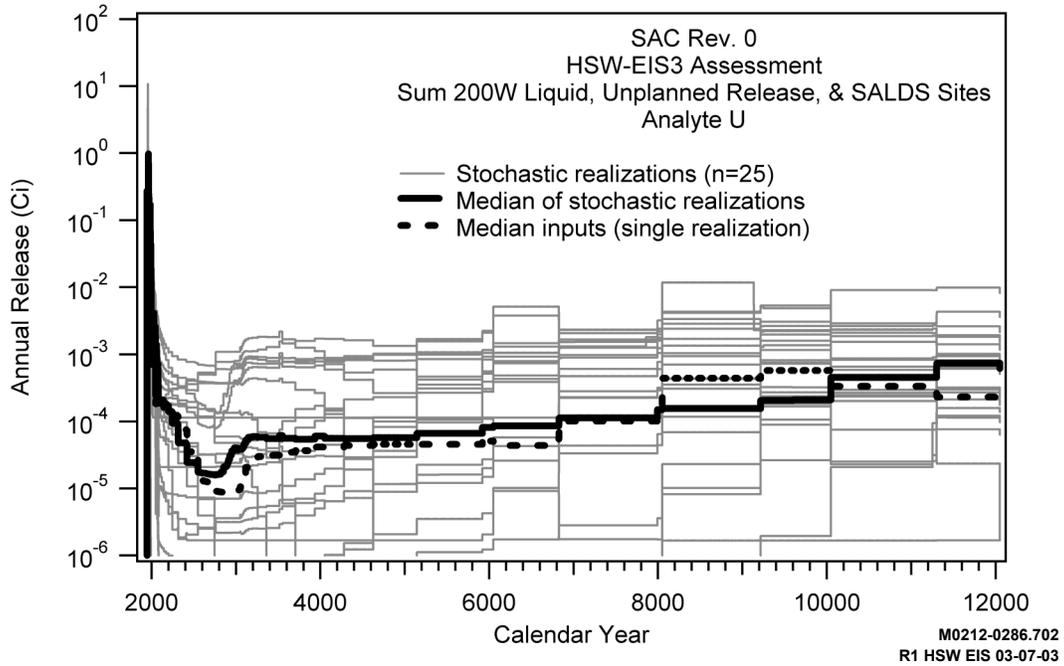
**Figure L.30.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Liquid Discharge and Unplanned Release Sites in the 200 West Area Plus SALDS



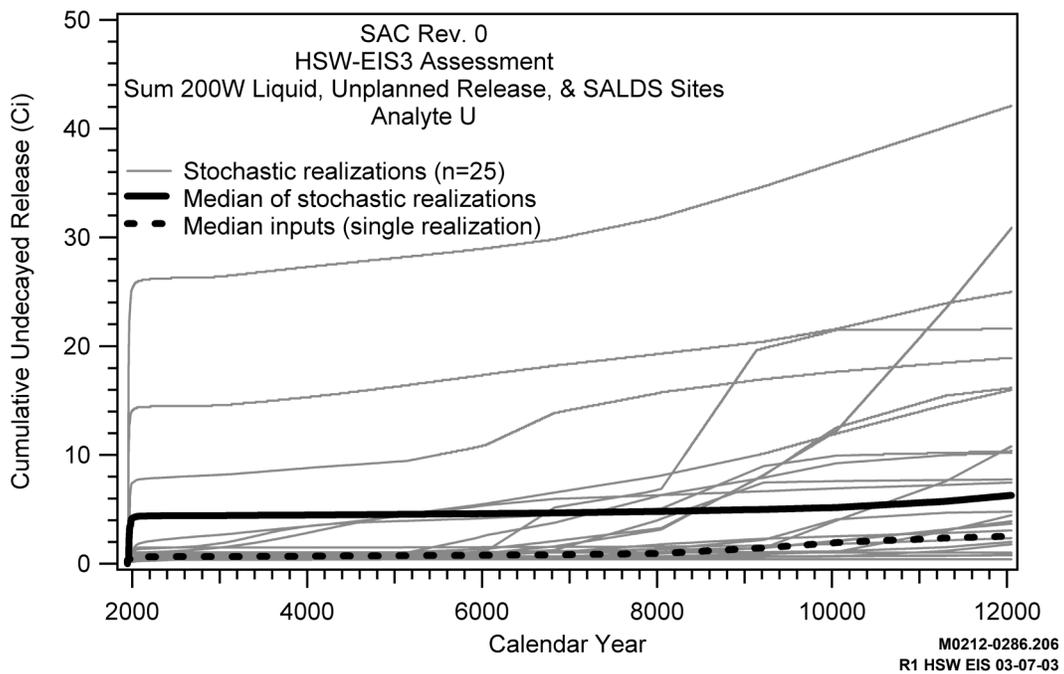
**Figure L.31.** SAC Results for Annual Vadose Zone Release of Uranium from All Liquid Discharge and Unplanned Release Sites in the 200 East Area



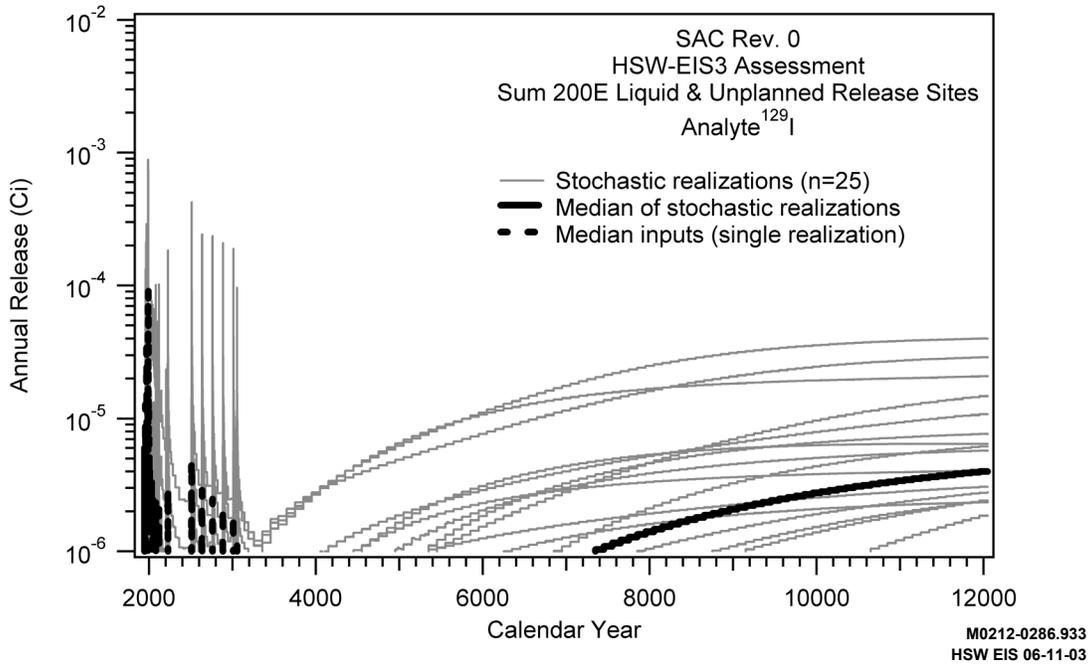
**Figure L.32.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Liquid Discharge and Unplanned Release Sites in the 200 East Area



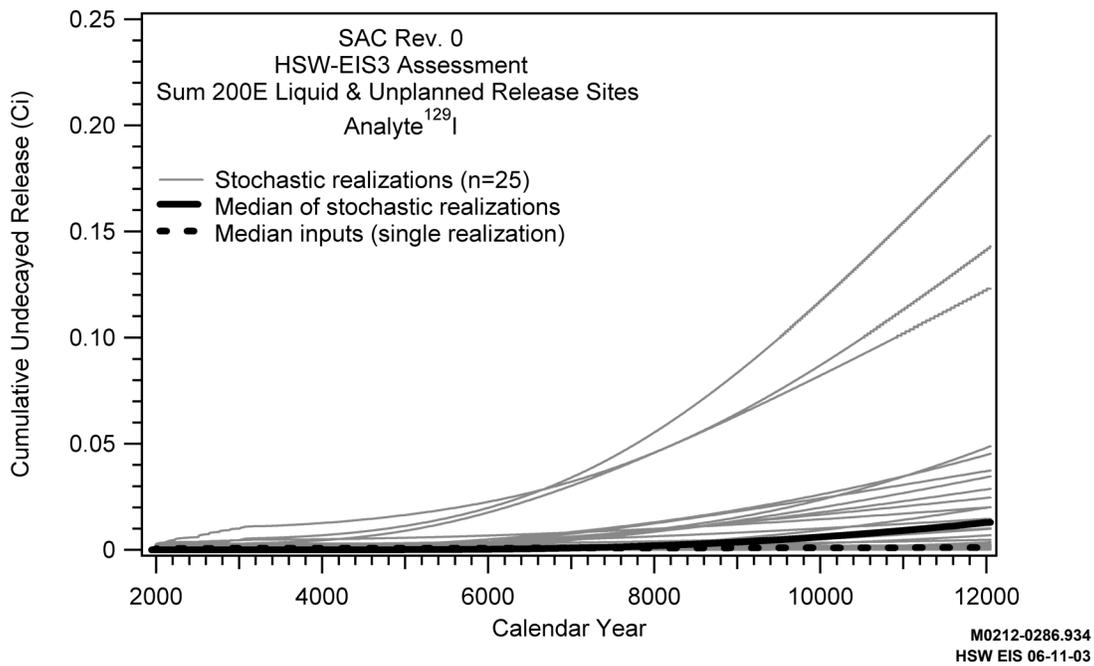
**Figure L.33.** SAC Results for Annual Vadose Zone Release of Uranium from All Liquid Discharge and Unplanned Release Sites in the 200 West Area Plus SALDS



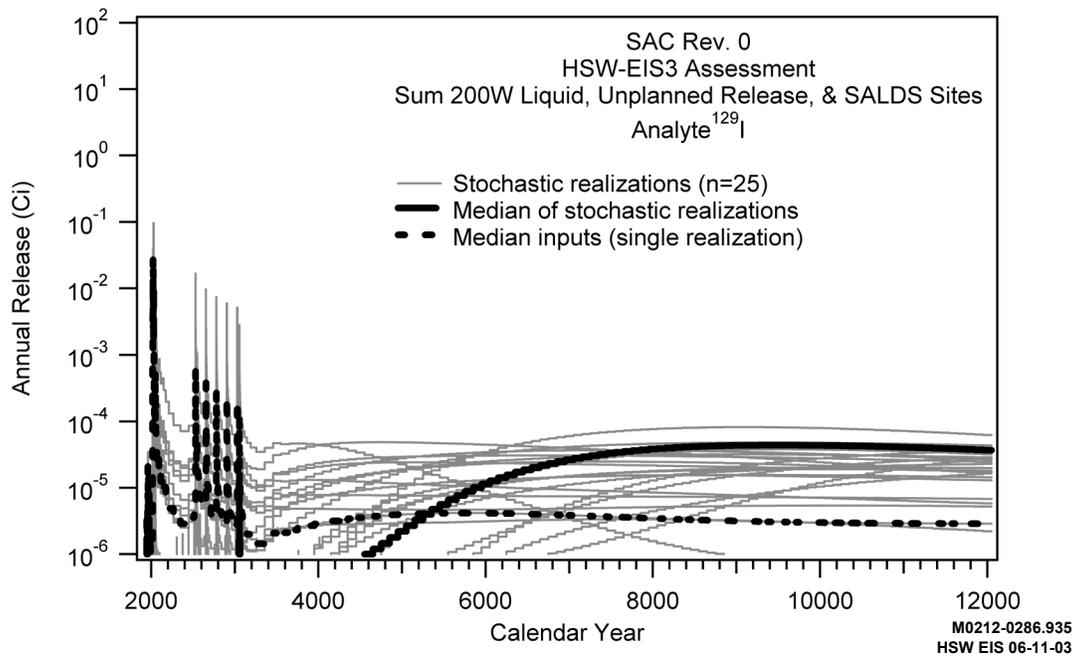
**Figure L.34.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Liquid Discharge and Unplanned Release Sites in the 200 West Area Plus SALDS



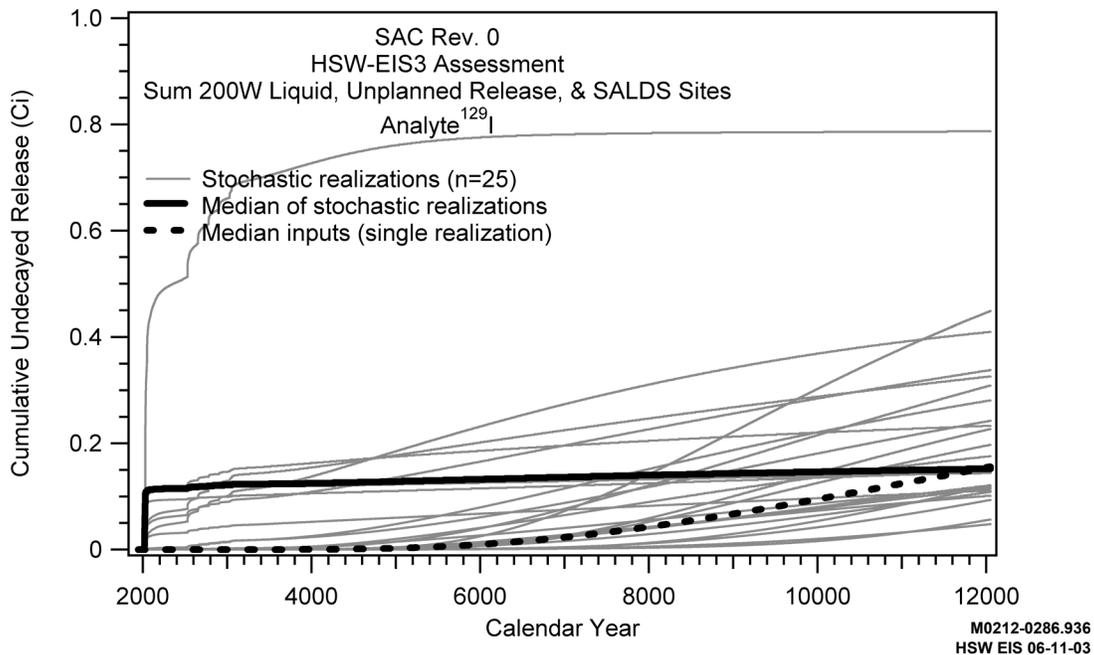
**Figure L.35.** SAC Results for Annual Vadose Zone Release of Iodine-129 from All Liquid Discharge and Unplanned Release Sites in the 200 East Area



**Figure L.36.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Iodine-129 from All Liquid Discharge and Unplanned Release Sites in the 200 East Area



**Figure L.37.** SAC Results for Annual Vadose Zone Release of Iodine-129 from All Liquid Discharge and Unplanned Release Sites in the 200 West Area plus SALDS

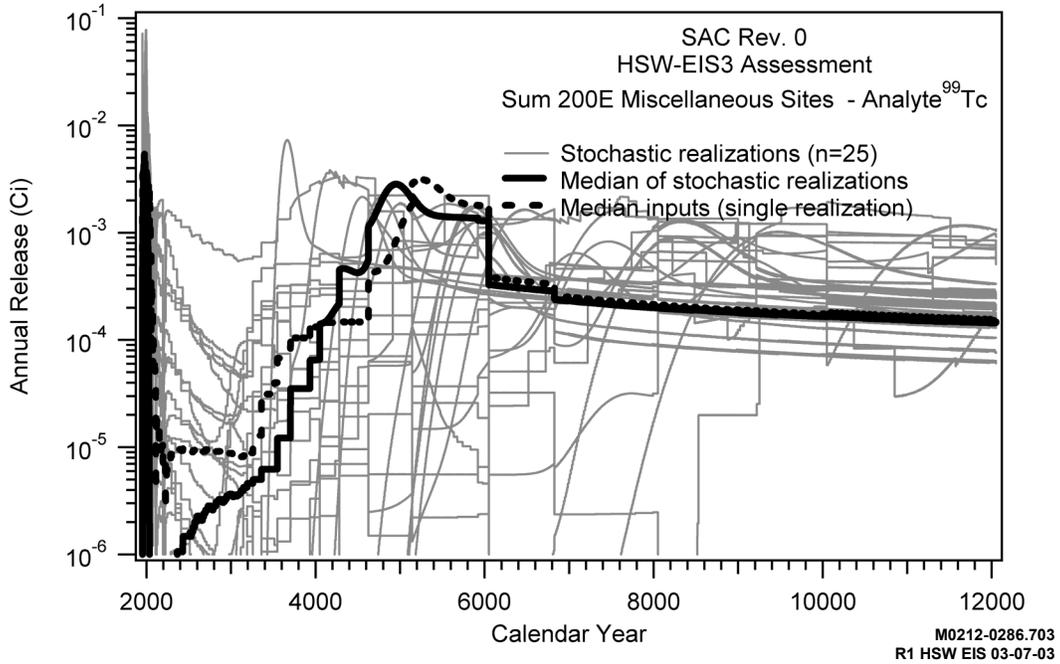


**Figure L.38.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Iodine-129 from All Liquid Discharge and Unplanned Release Sites in the 200 West Area Plus SALDS

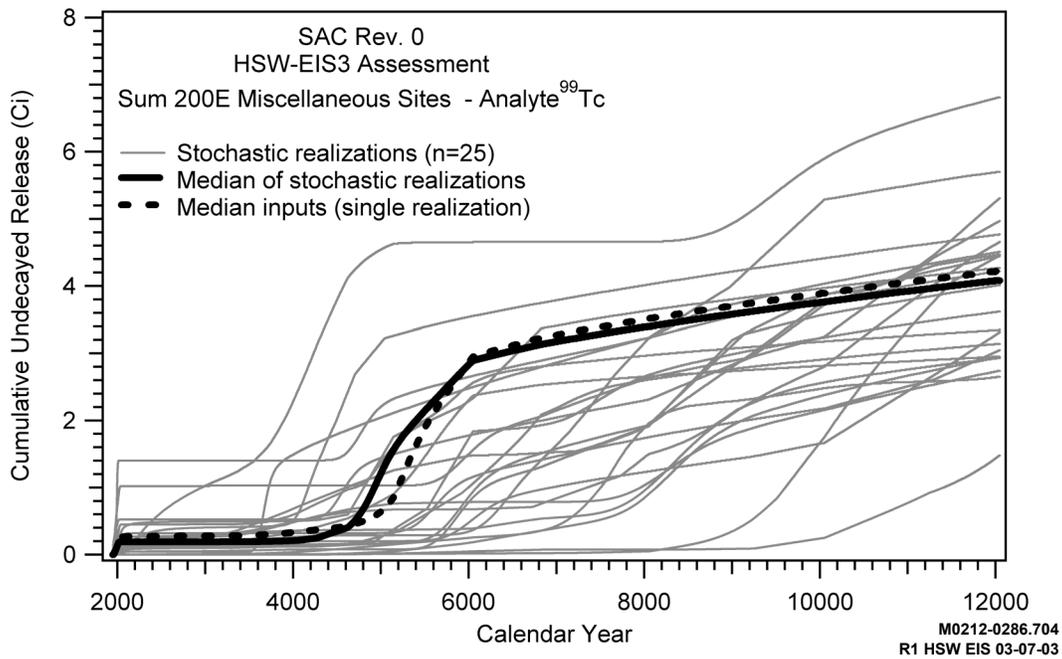
Figures L.39 through L.56 present the results for vadose zone releases to groundwater for the sum of all other sites on the Central Plateau, except for ERDF and the commercial low-level radioactive waste disposal site (that is, sites in 200 East and 200 West Areas, excluding solid waste burial ground, tank, liquid discharge, unplanned release, ERDF, and commercial low-level radioactive waste disposal sites), and for the sum of all sites outside the 200 East and 200 West Areas (that is, the 100, 300, 400, and 600 Area sites). Cumulative releases to groundwater for all other sites (for example, canyons, tunnels) on the Central Plateau range from approximately 15 to approximately 50 Ci for technetium-99 during the 10,000-year analysis period. The majority of this activity is associated with 200 West Area. Negligible releases of uranium occur from these sites. Iodine-129 releases from these sites range from 0 to approximately 0.045 Ci during the analysis period, and have median values of approximately zero for the 200 East Area and less than 0.003 Ci for the 200 West Area. Cumulative releases to groundwater from sites away from the Central Plateau (for example, river corridor sites with residual contamination) range from approximately 17 to approximately 37 Ci for technetium-99 during the 10,000-year analysis period. The release of uranium from these same sites ranges from approximately 5 to approximately 80 Ci. The release of iodine-129 from these sites ranges from approximately 0 to 0.0014 Ci, with a median value of approximately 0.0002 Ci. Note that the river corridor includes several liquid waste disposal trenches that received fuel fabrication waste streams that carried uranium to the vadose zone.

Figures L.57 through L.62 present the results for vadose zone releases to groundwater for ERDF. Cumulative releases to groundwater from ERDF range from 0 to approximately 27 Ci for technetium-99 during the 10,000-year analysis period. As in the case of solid waste, uranium in ERDF does not exhibit significant release during the 10,000-year period. Only 3 of 25 realizations exhibit any release, with no releases exhibited before 7000 years post-closure. Hence, the median case shows no uranium release to groundwater. Releases of iodine-129 to groundwater from ERDF during the 10,000 year analysis period range from approximately 0 to 0.042 Ci, with a median value of approximately 0.013 Ci.

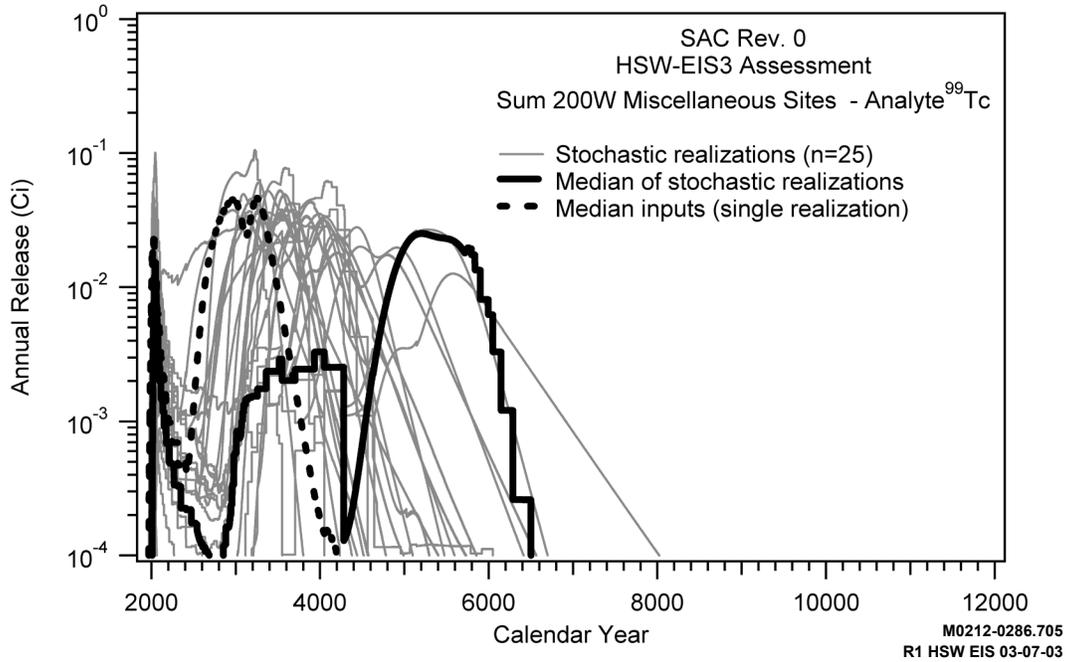
Figures L.63 through L.68 present the results for vadose zone releases to groundwater for the commercial low-level radioactive waste disposal site operated by US Ecology, Inc. Cumulative releases to groundwater from the US Ecology, Inc. site range from 0 to approximately 80 Ci for technetium-99 during the 10,000-year analysis period. The annual release curves (Figure L.63) and the cumulative plots (Figure L.64) exhibit substantial variability in the timing of release; however, the peak annual releases appear to vary between only approximately  $2 \times 10^{-2}$  and approximately  $5 \times 10^{-2}$  Ci/yr after 3000 A.D. As in the case of solid waste and ERDF, uranium in the US Ecology, Inc. site does not exhibit release to groundwater during the 10,000-year period. Releases of iodine-129 from the commercial disposal site to the groundwater range from approximately 0 to 5.3 Ci. However, few of the stochastic realizations exhibit releases to the water table, and the median value release is zero during the 10,000-year analysis period.



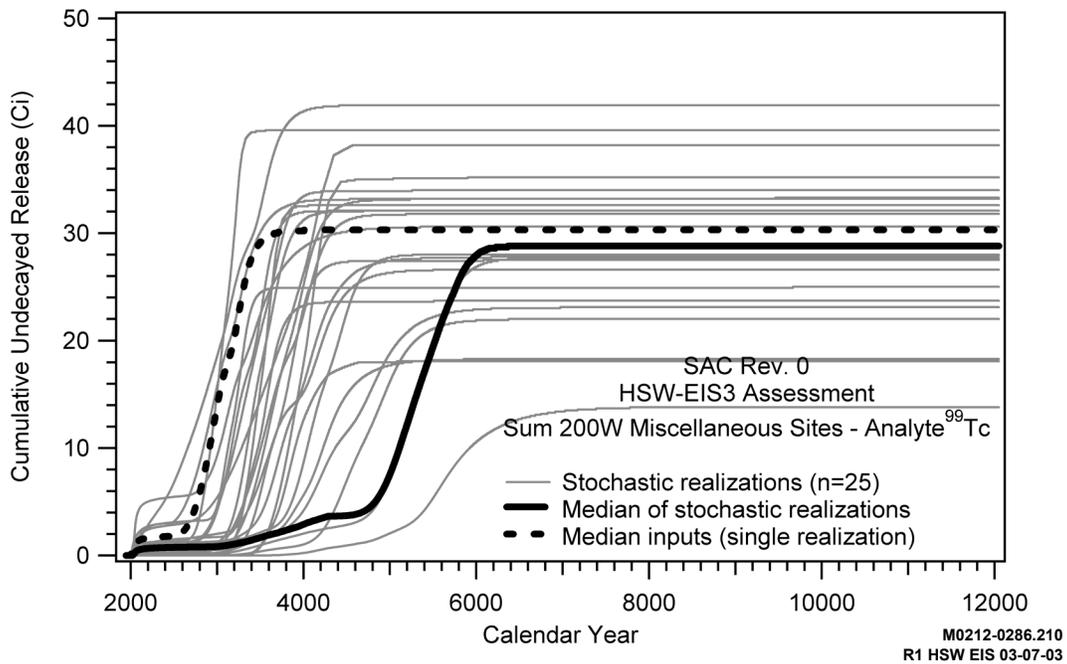
**Figure L.39.** SAC Results for Annual Vadose Zone Release of Technetium-99 from All Other Sites in the 200 East Area



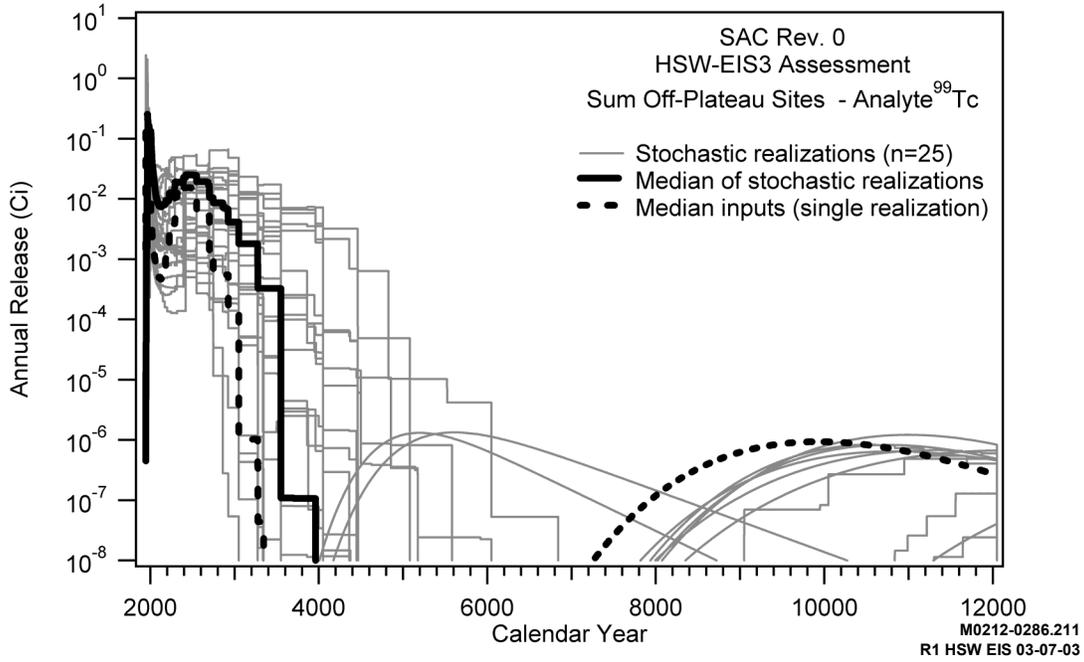
**Figure L.40.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Other Sites in the 200 East Area



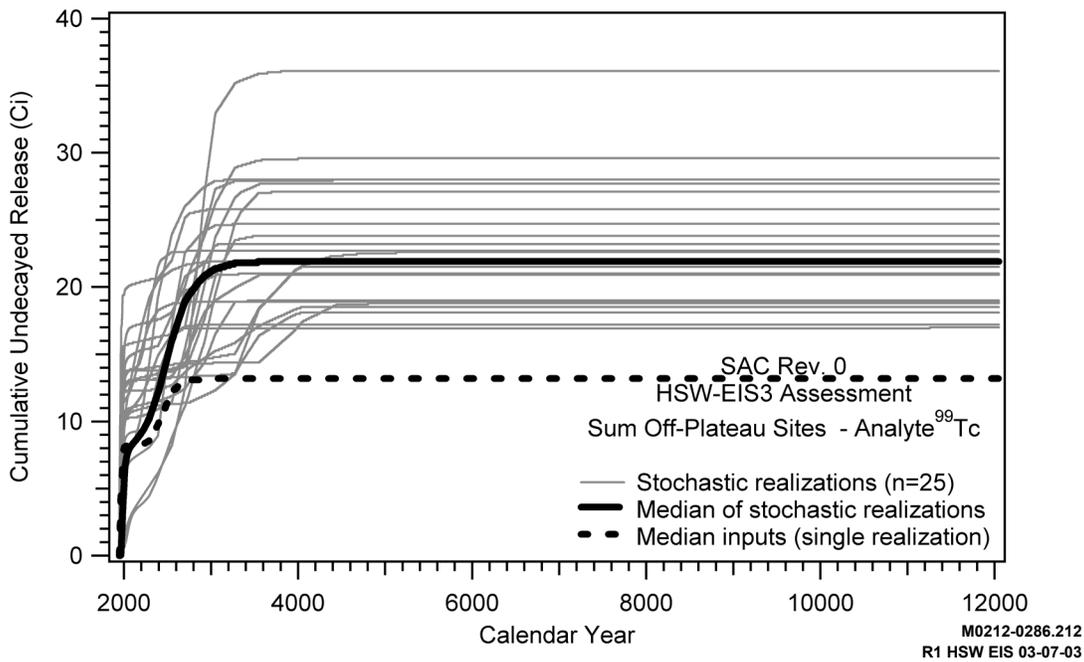
**Figure L.41.** SAC Results for Annual Vadose Zone Release of Technetium-99 from All Other Sites in the 200 West Area



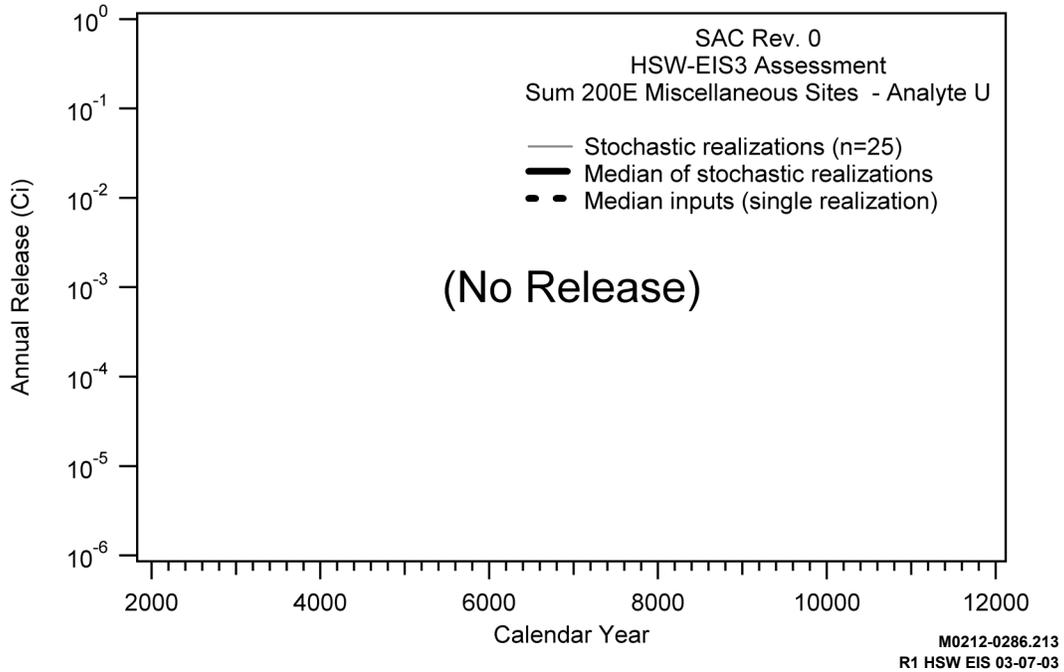
**Figure L.42.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Other Sites in the 200 West Area



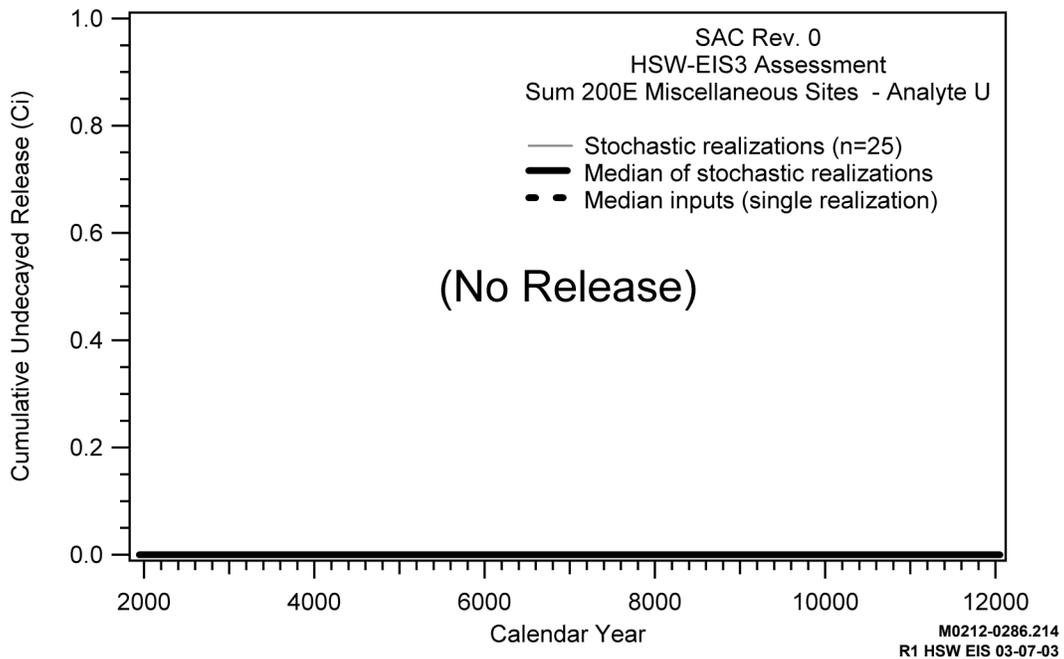
**Figure L.43.** SAC Results for Annual Vadose Zone Release of Technetium-99 from all Other Sites Outside the 200 East and 200 West Areas



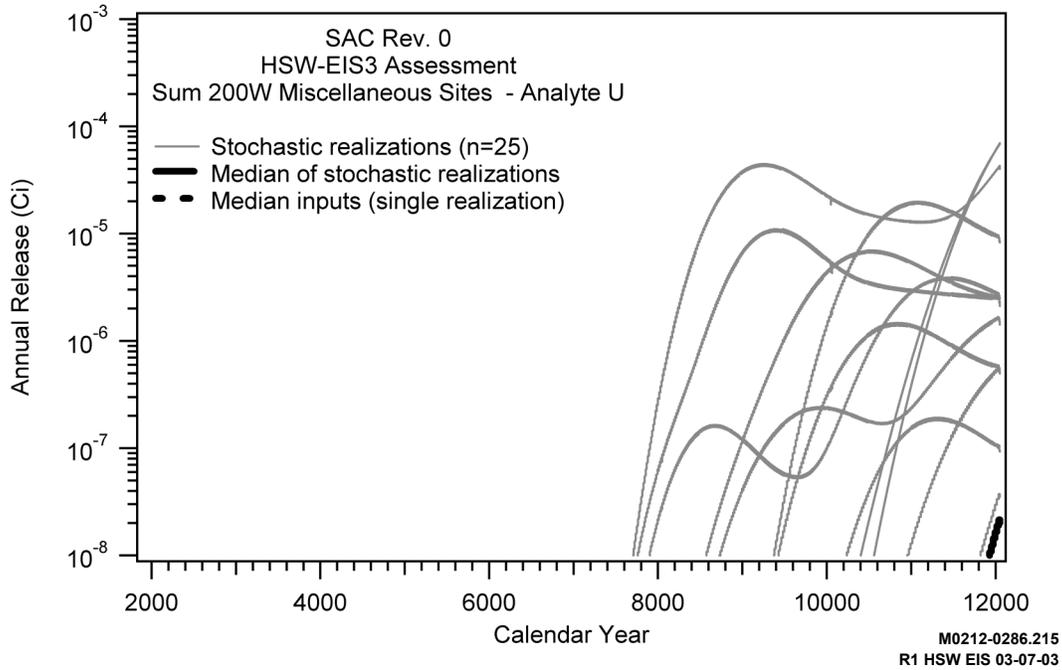
**Figure L.44.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from All Other Sites Outside the 200 East and 200 West Areas



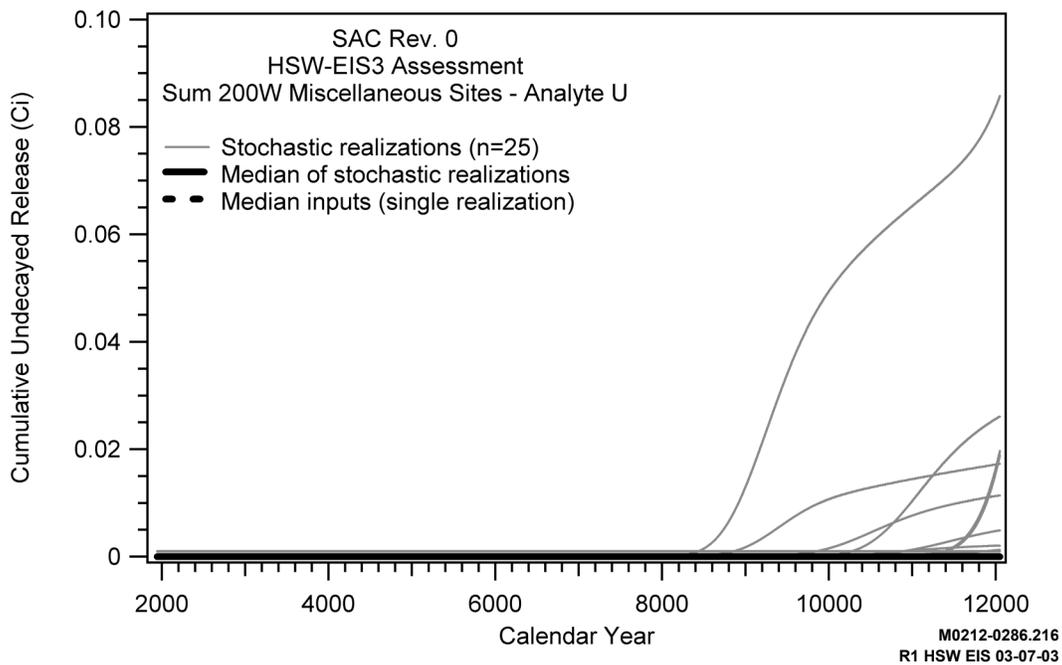
**Figure L.45.** SAC Results for Annual Vadose Zone Release of Uranium from All Other Sites in the 200 East Area



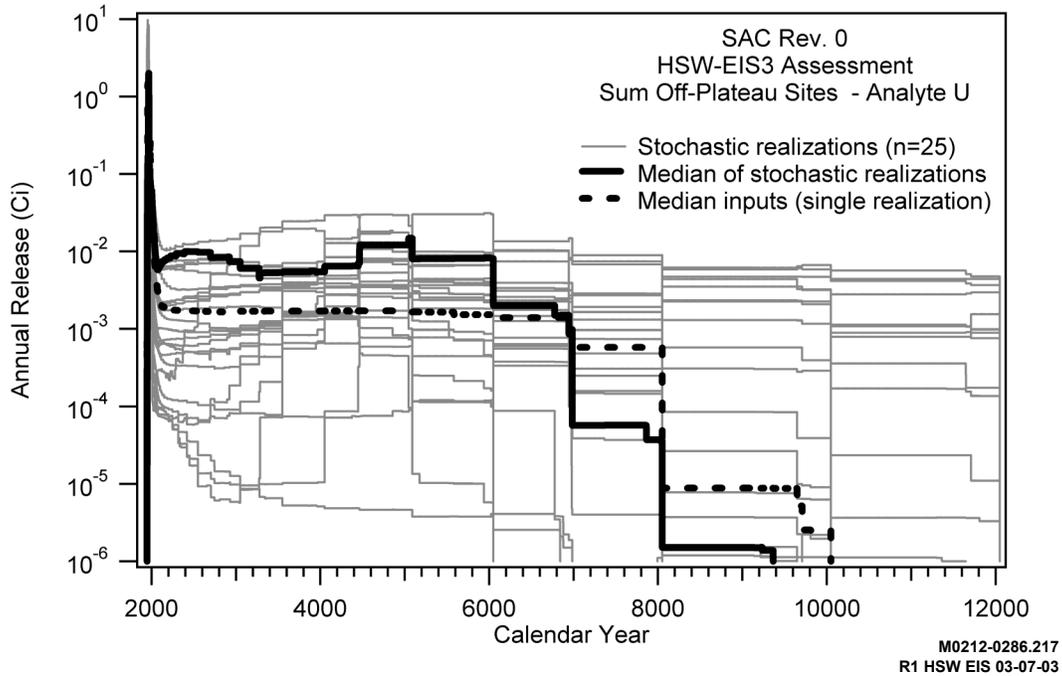
**Figure L.46.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Other Sites in the 200 East Area



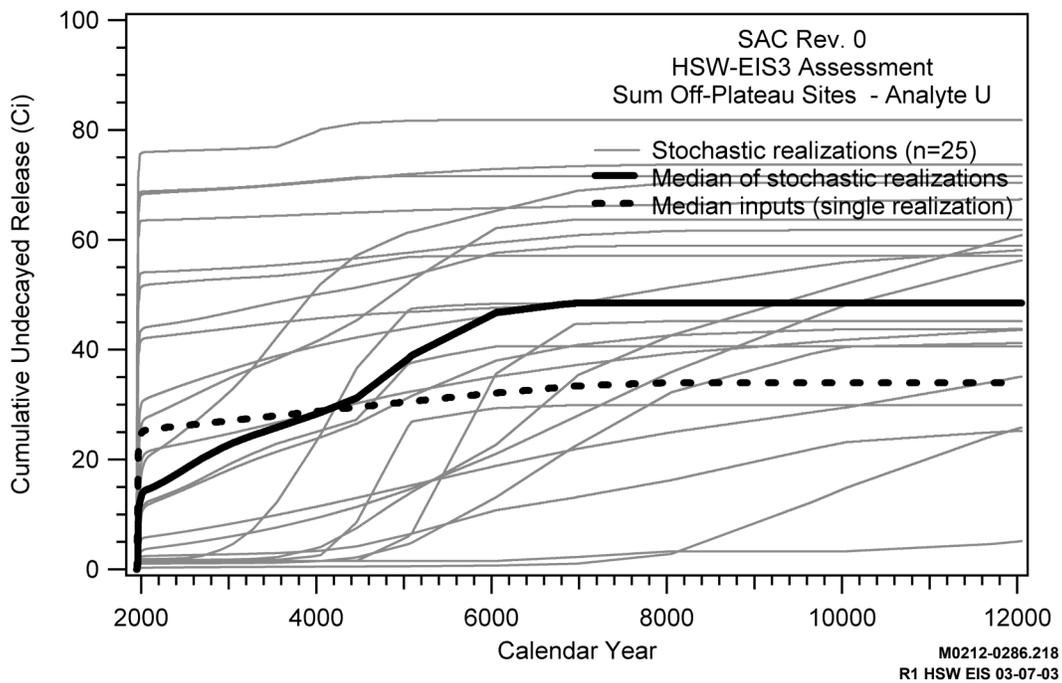
**Figure L.47.** SAC Results for Annual Vadose Zone Release of Uranium from All Other Sites in the 200 West Area



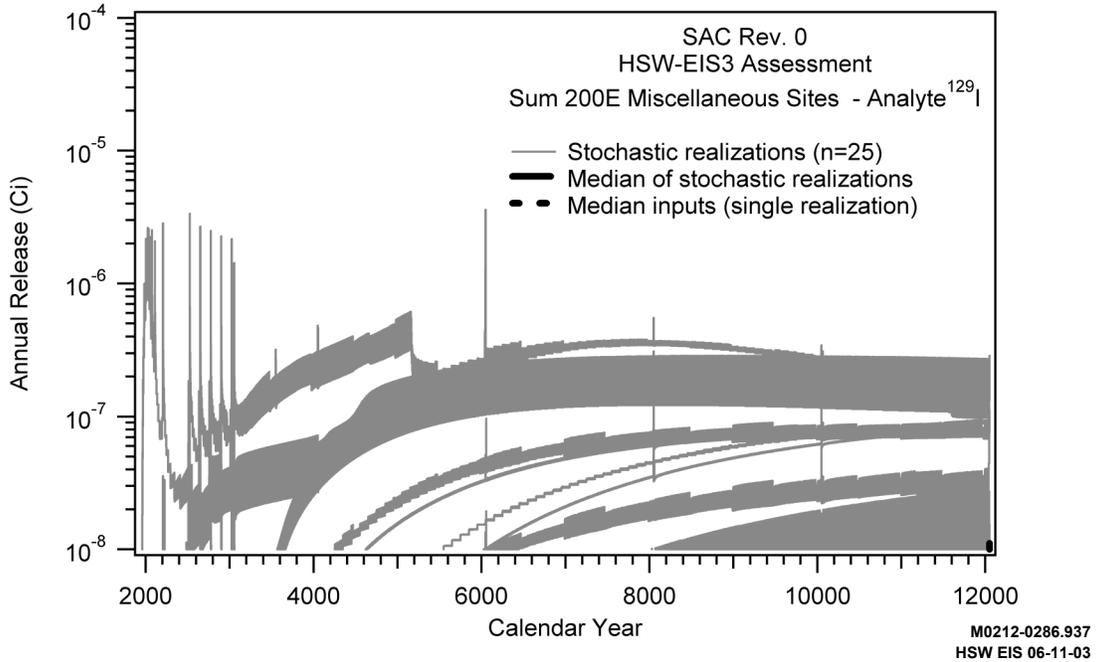
**Figure L.48.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Other Sites in the 200 West Area



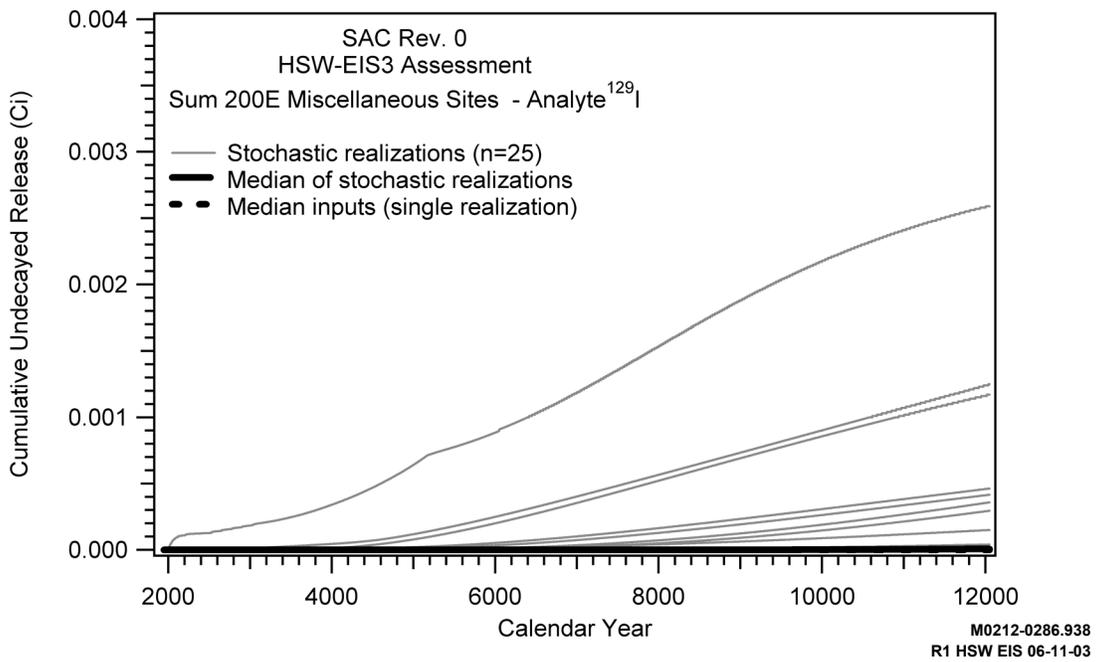
**Figure L.49.** SAC Results for Annual Vadose Zone Release of Uranium from All Other Sites Outside the 200 East and 200 West Areas



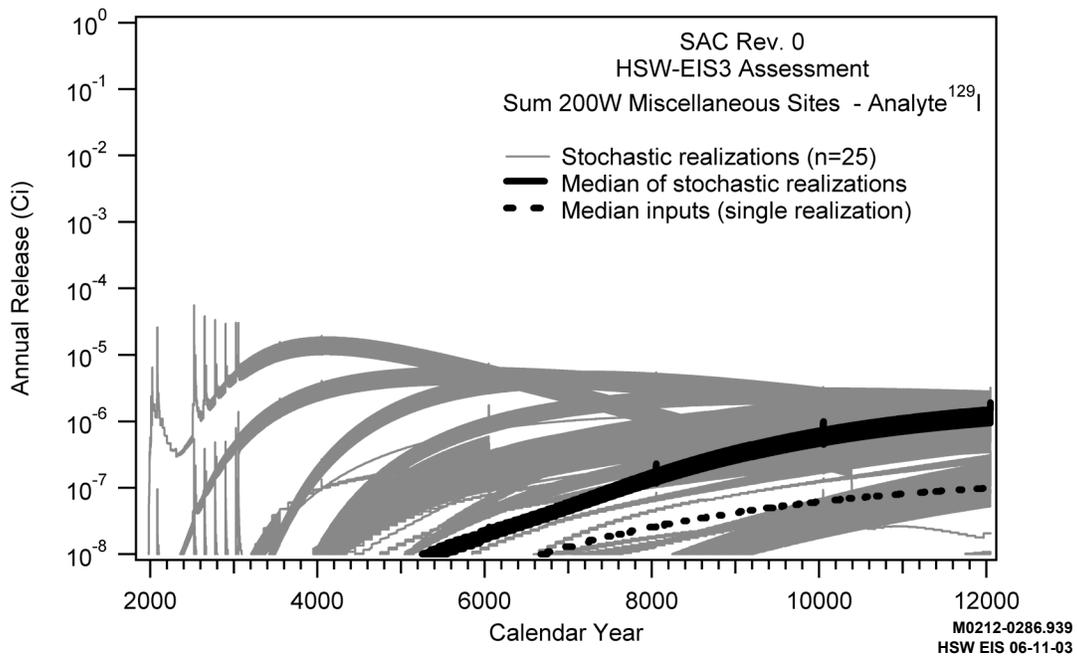
**Figure L.50.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from All Other Sites Outside the 200 East and 200 West Areas



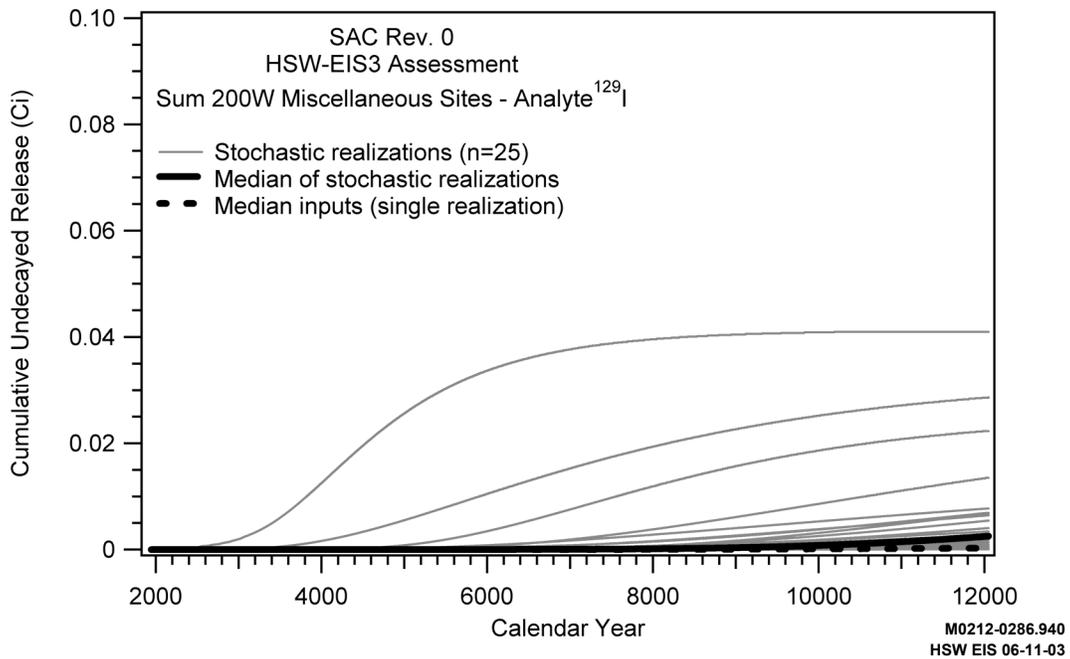
**Figure L.51.** SAC Results for Annual Vadose Zone Release of Iodine-129 from All Other Sites in the 200 East Area



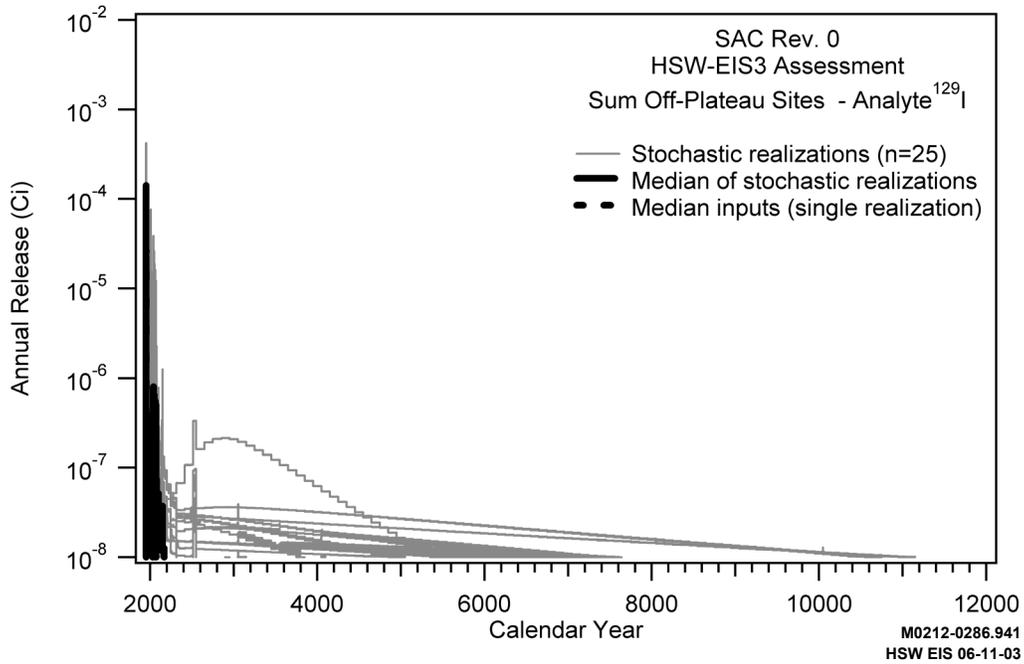
**Figure L.52.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Iodine-129 from All Other Sites in the 200 East Area



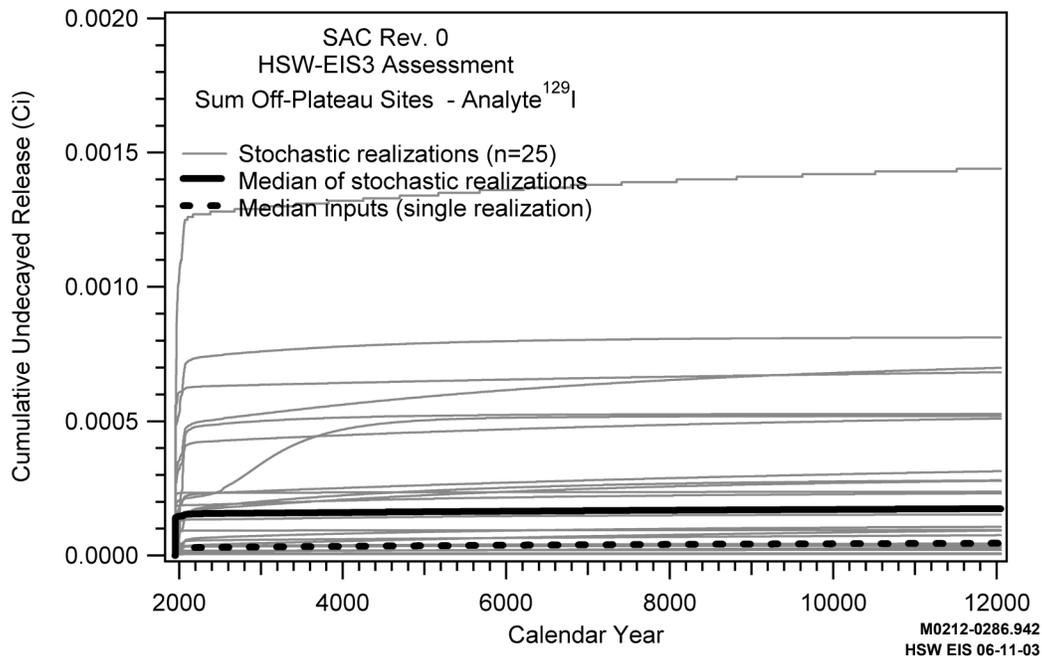
**Figure L.53.** SAC Results for Annual Vadose Zone Release of Iodine-129 from All Other Sites in the 200 West Area.



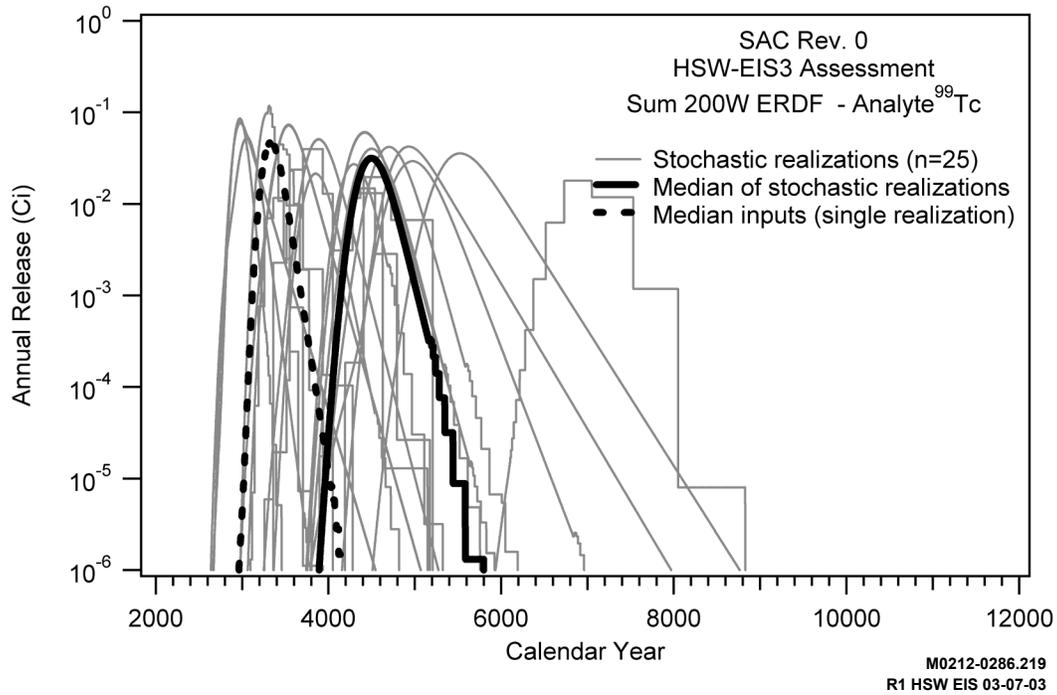
**Figure L.54.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Iodine-129 from All Other Sites in the 200 West Area.



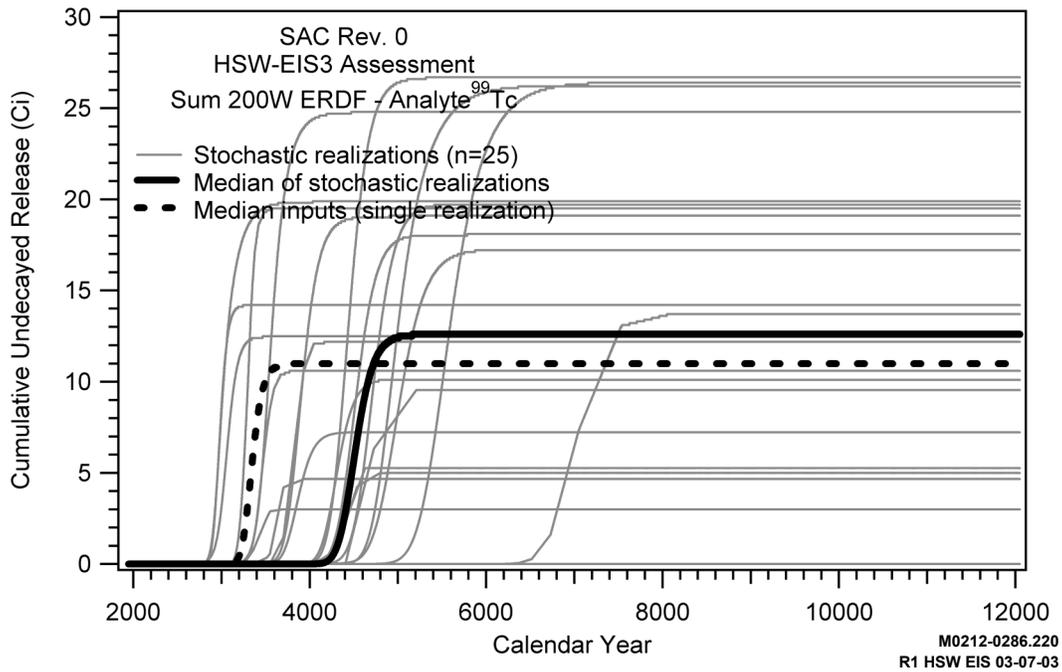
**Figure L.55.** SAC Results for Annual Vadose Zone Release of Iodine-129 from All Other Sites Outside the 200 East and 200 West Areas.



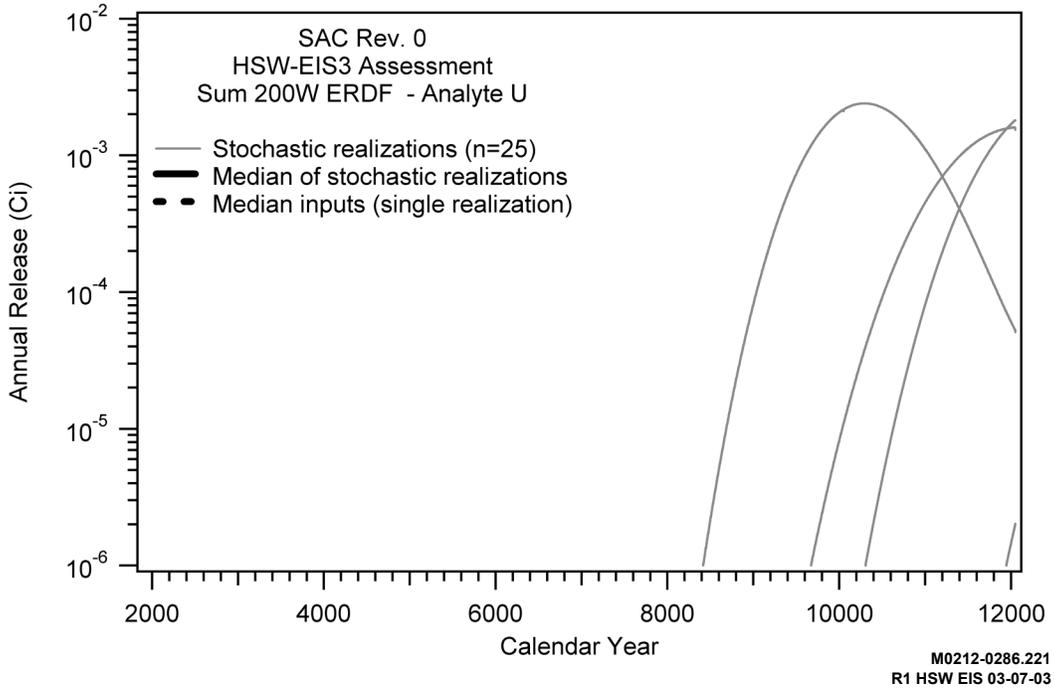
**Figure L.56.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Iodine-129 from All Other Sites Outside the 200 East and 200 West Areas.



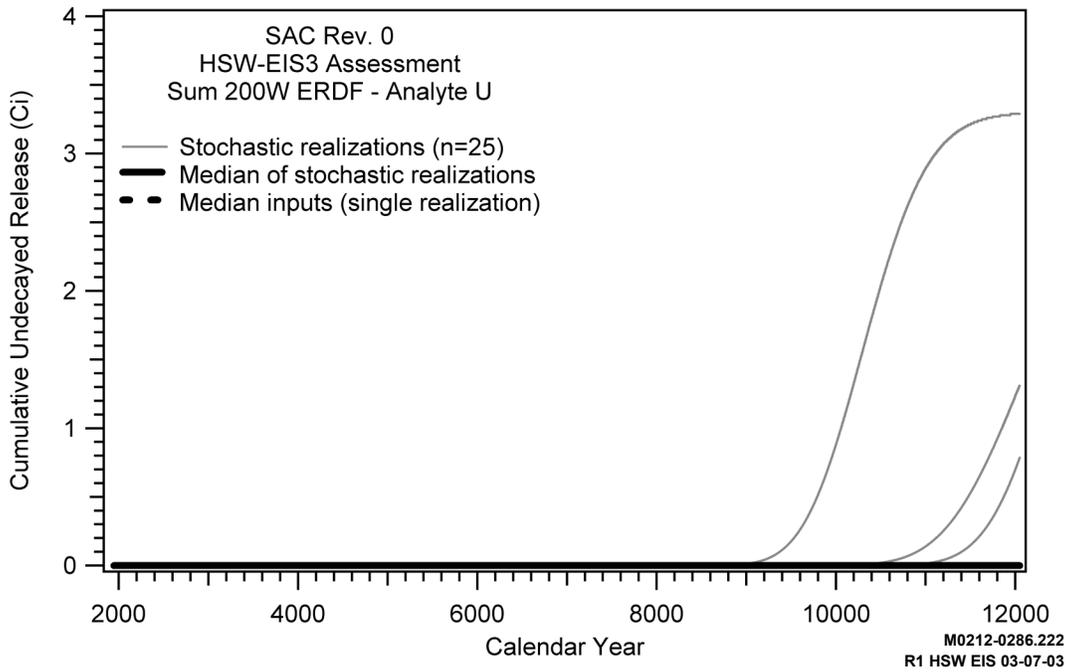
**Figure L.57.** SAC Results for Annual Vadose Zone Release of Technetium-99 from ERDF



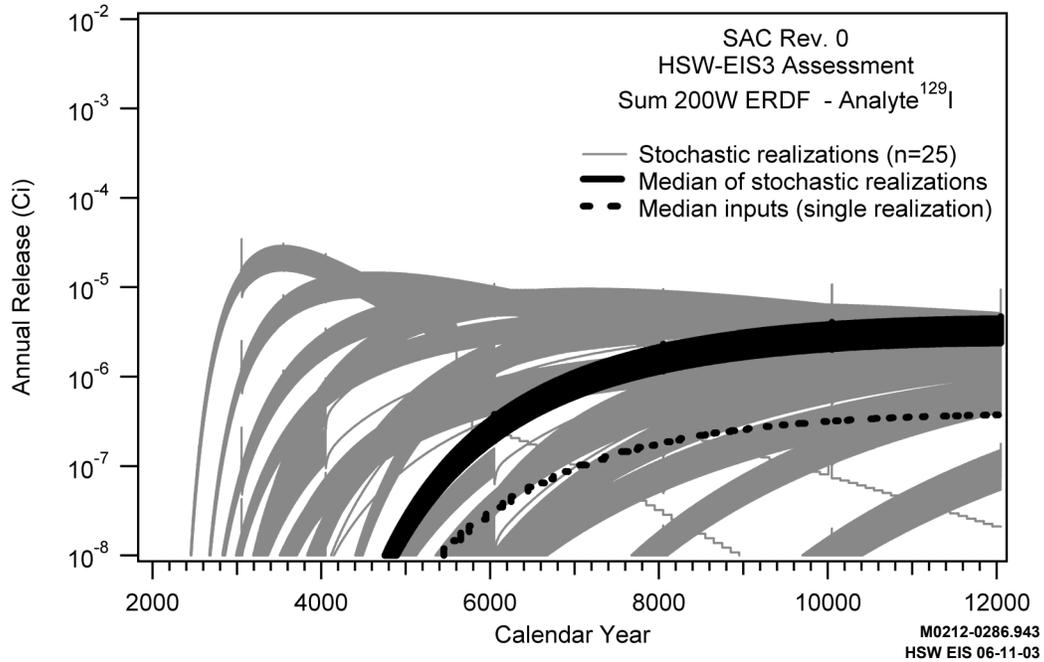
**Figure L.58.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from ERDF



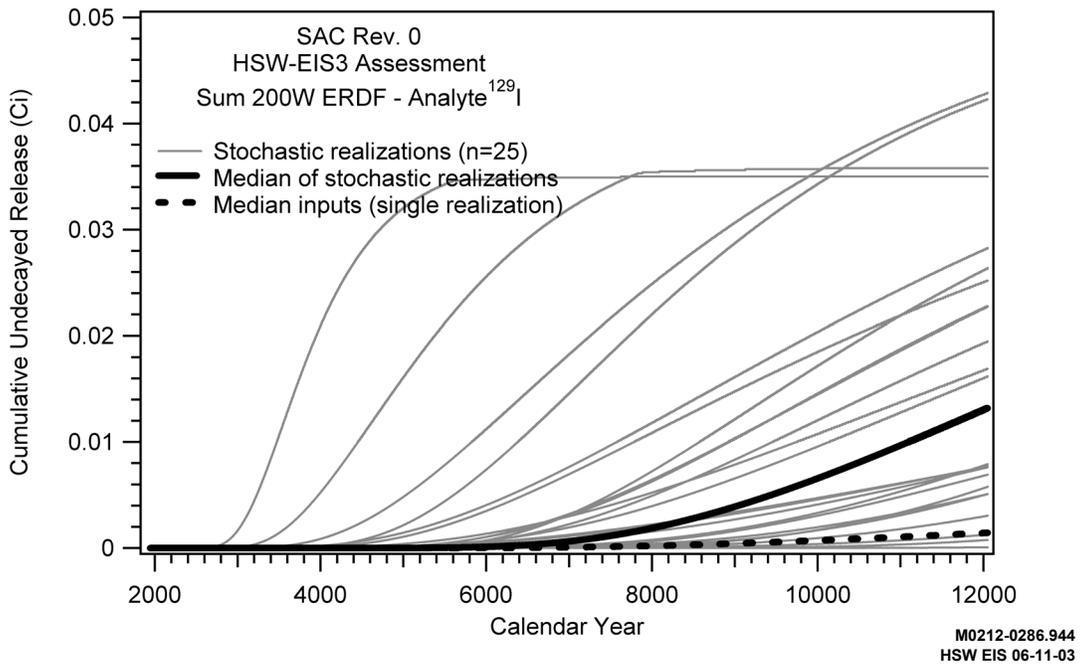
**Figure L.59.** SAC Results for Annual Vadose Zone Release of Uranium from ERDF



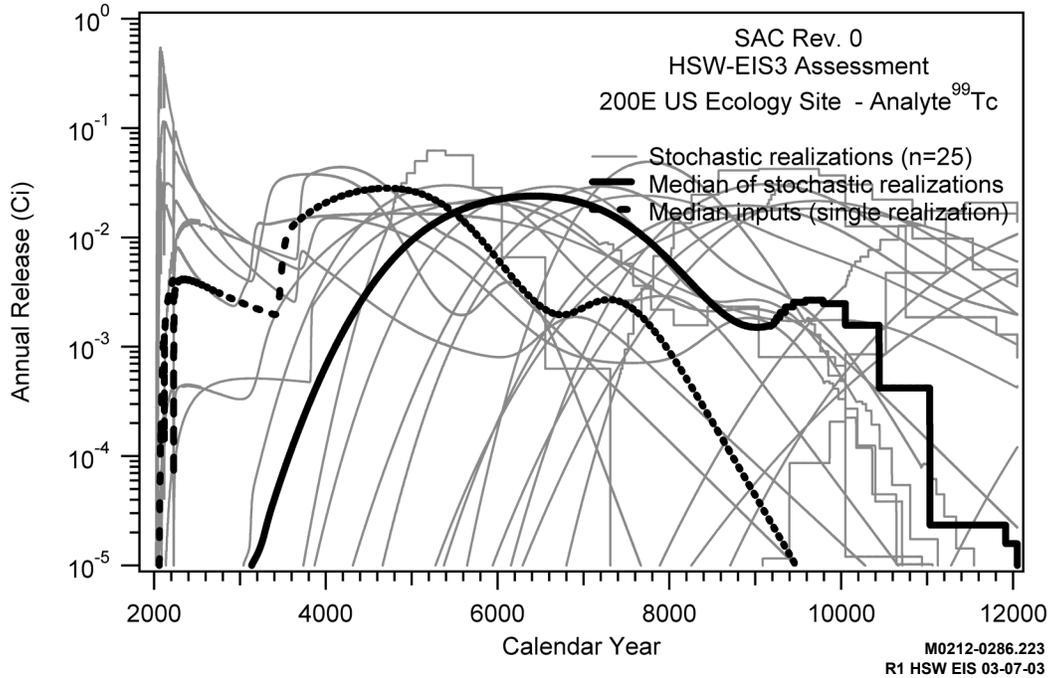
**Figure L.60.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from ERDF



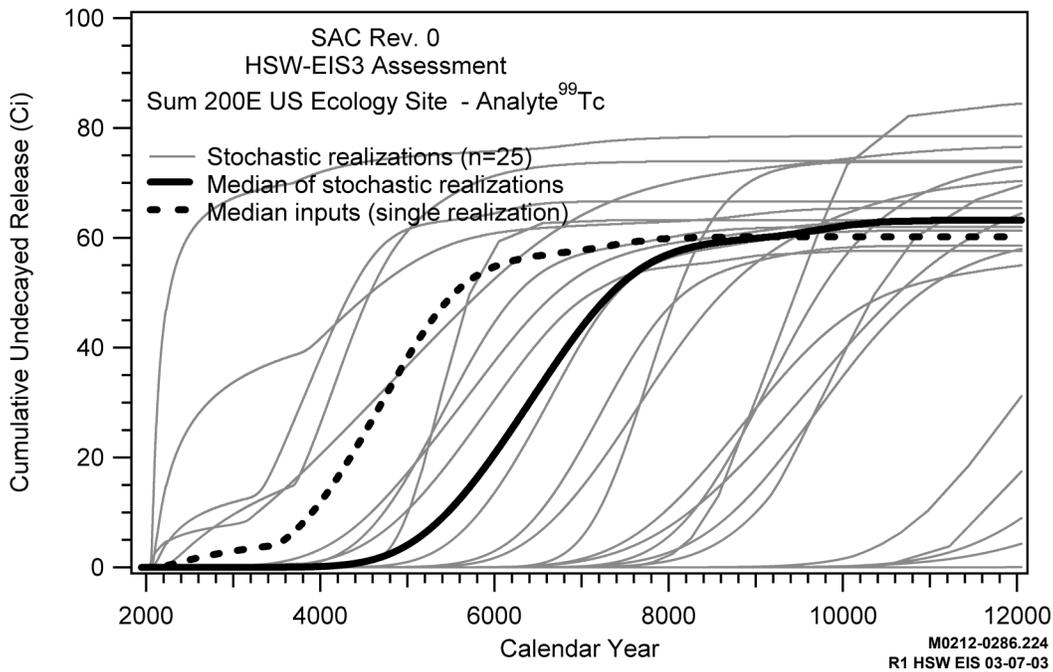
**Figure L.61.** SAC Results for Annual Vadose Zone Release of Iodine-129 from ERDF



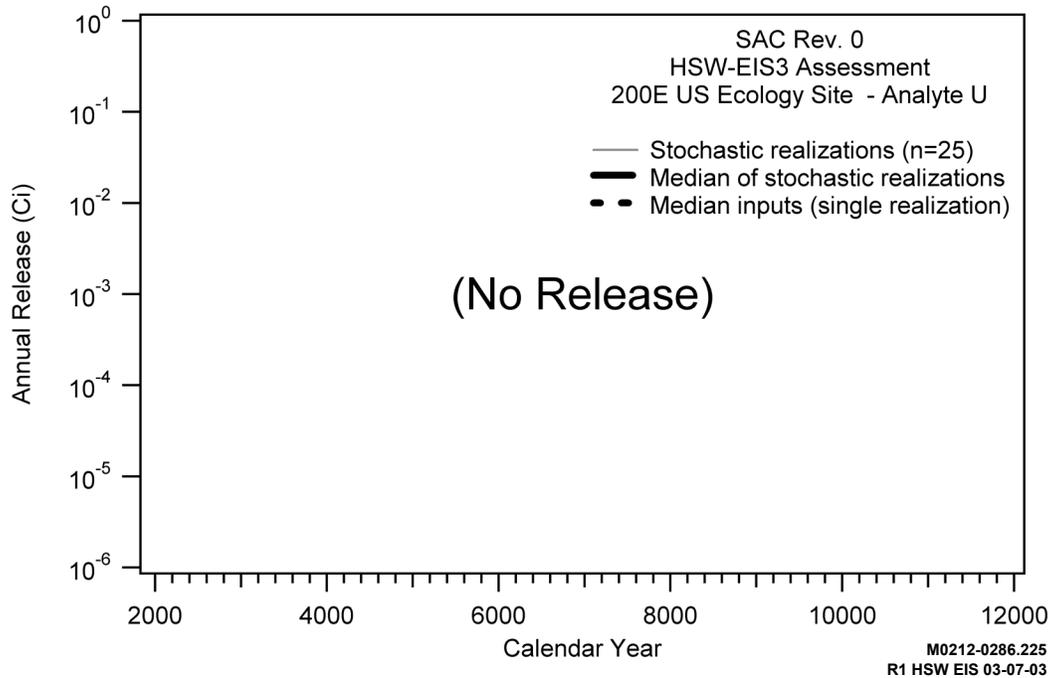
**Figure L.62.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Iodine-129 from ERDF



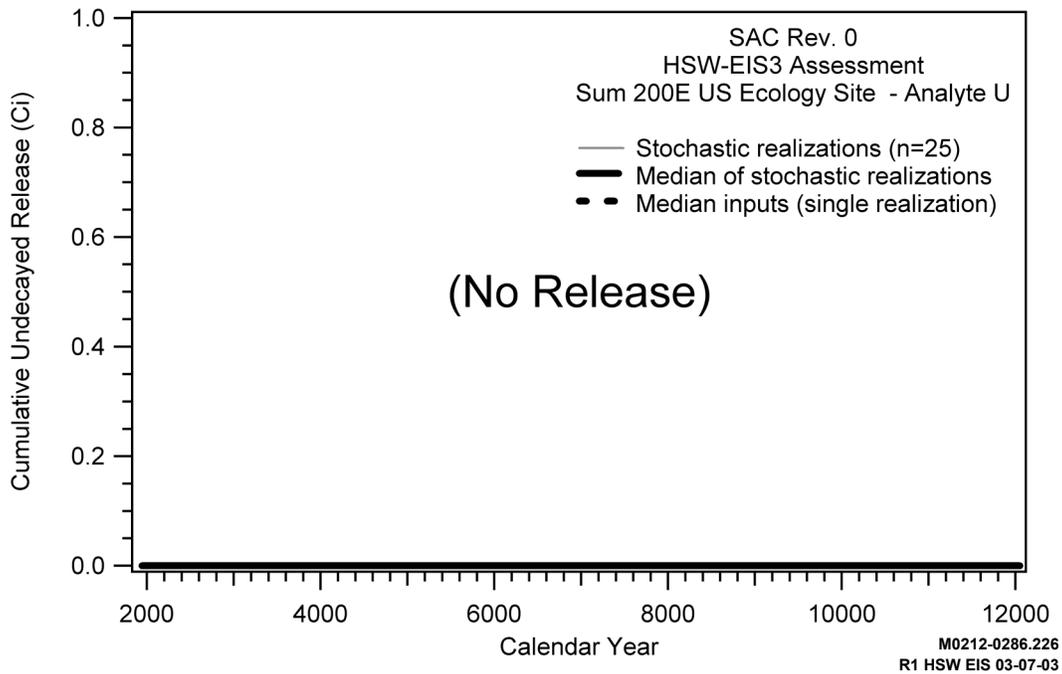
**Figure L.63.** SAC Results for Annual Vadose Zone Release of Technetium-99 from the Commercial Low-Level Radioactive Waste Disposal (US Ecology, Inc.) Site



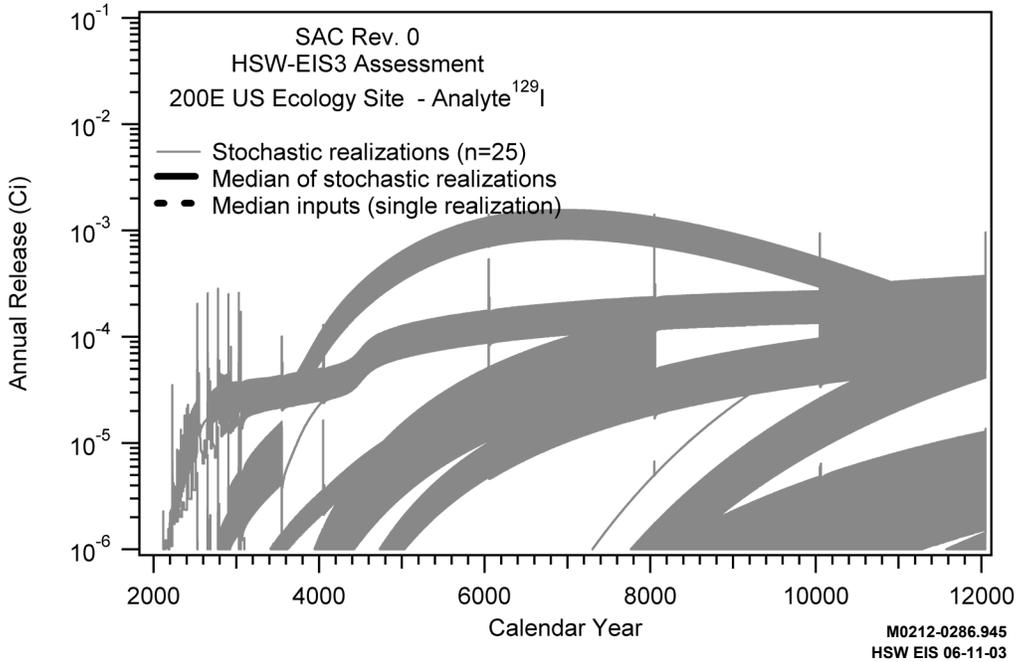
**Figure L.64.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Technetium-99 from the Commercial Low-Level Radioactive Waste Disposal (US Ecology, Inc.) Site



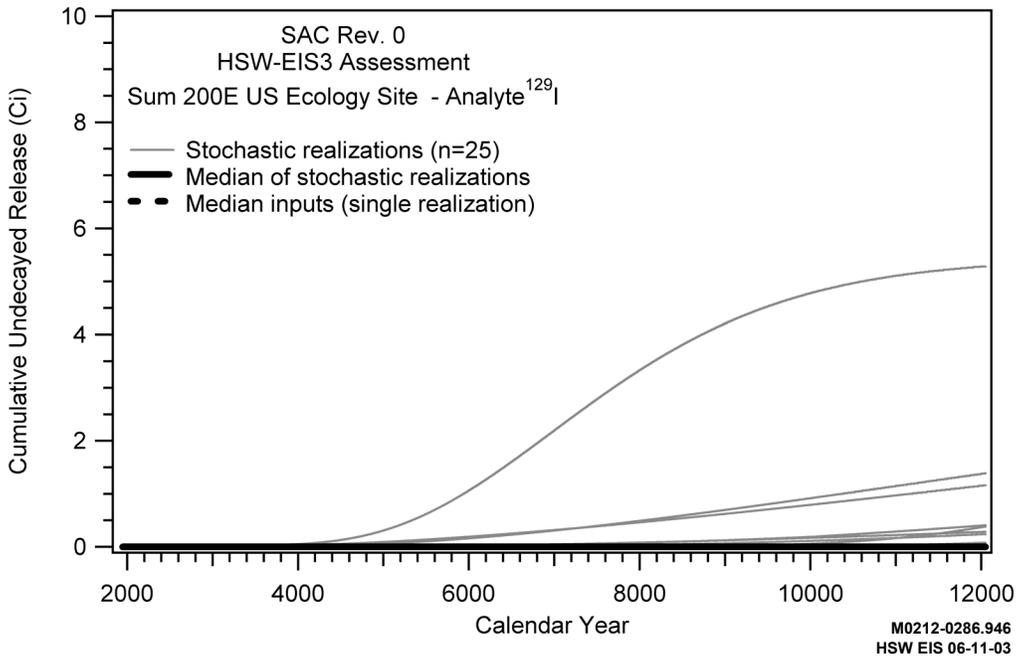
**Figure L.65.** SAC Results for Annual Vadose Zone Release of Uranium from the Commercial Low-Level Radioactive Waste Disposal (US Ecology, Inc.) Site



**Figure L.66.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Uranium from the Commercial Low-Level Radioactive Waste Disposal (US Ecology, Inc.) Site



**Figure L.67.** SAC Results for Annual Vadose Zone Release of Iodine-129 from the Commercial Low-Level Radioactive Waste Disposal (US Ecology, Inc.) Site



**Figure L.68.** SAC Results for Cumulative (undecayed) Vadose Zone Release of Iodine-129 from the Commercial Low-Level Radioactive Waste Disposal (US Ecology, Inc.)

These results indicate that technetium-99 releases from the solid waste disposal facilities to groundwater may account for approximately 323 to approximately 445 Ci in 10,000 years. This contrasts with approximately 440 to approximately 645 Ci of technetium-99 from tank sites, approximately 735 to approximately 1030 Ci from liquid releases, approximately 15 to approximately 50 Ci from other sites on the Central Plateau, approximately 17 to approximately 37 Ci from sites away from the plateau, 0 to approximately 27 Ci from ERDF, and 0 to approximately 80 Ci from the US Ecology, Inc. site. Overall, the comparison is approximately 323 to approximately 445 Ci of technetium-99 from solid waste and approximately 1530 to approximately 2310 Ci of technetium-99 released in 10,000 years from all Hanford Site sources. Thus, the contribution from Hanford solid waste would amount to about 20 percent of the cumulative technetium-99 release from all Hanford sources.

The release of uranium to groundwater from Hanford solid waste is much lower. No realizations showed any release of uranium to groundwater from Hanford solid waste in the 200 East Area, and only 5 of 25 realizations exhibit any release of uranium to groundwater from Hanford solid waste in 200 West Area. Thus, in an average, or median, sense, Hanford solid waste deposits would release no uranium to groundwater over the 10,000-year analysis period. This result compares to a median release of approximately 84 Ci and a range of release to groundwater from the 25 realizations of between approximately 10 and approximately 300 Ci of uranium for all Hanford wastes. Of the five realizations of non-zero uranium release from Hanford solid waste in the 200 West Area, the range of cumulative release was 0 to approximately 90 Ci, but the majority of realizations show zero release. As a consequence, the contribution from Hanford solid waste would amount to between 0 and 30 percent of the cumulative release from all Hanford sources. The majority of the technetium-99 and uranium release was forecast to occur from past liquid discharge sites (cribs, ponds, trenches) and unplanned releases on the plateau and from off-plateau or river corridor waste sites.

The inventory of iodine-129 and its release to groundwater from Hanford solid waste are lower than technetium-99 or uranium; however, they are just as substantial given the low production inventory and the potential health impacts of the isotope. Iodine-129 releases from the solid waste disposal facilities to the groundwater may account for approximately 0 to 2.2 Ci in 10,000 years. This amount contrasts with approximately 0.1 Ci to 0.22 Ci released from tank sites, approximately 0 to 1 Ci released from liquid discharge and unplanned release sites, approximately 0 to 0.045 Ci released from other sites on the Central Plateau, approximately 0 to 0.0014 Ci released from sites away from the plateau, approximately 0 to 0.042 Ci released from ERDF, and approximately 0 to 5.3 Ci released from the commercial low-level radioactive waste disposal site operated by US Ecology, Inc. Ci of iodine-129 from solid waste deposits, and approximately 0.1 to 8.8 Ci of iodine-129 released in 10,000 years from all Hanford sources. Using the maximum releases to the water table, the contribution from Hanford solid waste, excluding ILAW, would amount to about 25 percent of the cumulative iodine-129 release from all Hanford sources; however, the commercial disposal site dominates the estimates of maximum release. If the median result is used to estimate the role of solid waste, its role is approximately 27 percent of all releases; however, the commercial disposal site contribution is negligible, tank sites are as important as solid waste, and liquid discharge and unplanned release sites on the plateau dominate.

### L.3.2 Drinking Water Dose at Selected 200 East and 200 West Area Locations

Doses to humans calculated using the SAC software and data are summarized in this section. The exposure scenario has an adult human drinking 2 L per day of contaminated groundwater. The doses in this section are presented as total effective dose equivalents, that is, the sum of the dose equivalents to various organs and tissues of the body, each weighted by an organ-specific weighting factor. The total effective dose equivalent includes the committed effective dose equivalent from internal deposition of radionuclides (from inhalation and ingestion) and the dose equivalent from penetrating radiation from sources external to the body. The radionuclide dose conversion factors used in this report were taken from compilations established by the EPA (Eckerman et al. 1988; Eckerman and Ryman 1993). These dose conversion factors are *not* the same as those required to show compliance with the National Primary Drinking Water Regulations (40 CFR 141). However, groundwater concentrations are also shown in Section L.3.4 for comparison with the 40 CFR 141 maximum contaminant levels, or MCLs. The stochastic capability of SAC was employed for these simulations, so the following results are shown in each plot in this section:

- Individual **stochastic results** (25 realizations) are shown in black.
- The **median result** of the 25 realizations—that is, the realization that resulted in the median integrated cumulative dose in the year 12,050 A.D. (at the end of the simulation)—is shown in blue.
- The **median-inputs** simulation—a separate single-realization simulation with SAC using the median value of all stochastic input variables—is shown in red.

The variability in the stochastic results is due to variability in the inventory, release, and transport of technetium-99, iodine-129, and uranium. The human dose calculations use fixed inputs. Because active institutional control cannot be relied on after 100 years, the scenarios using groundwater begin in 2150.

The doses provided in this section are based on all waste at the Hanford Site except the ILAW, melters, and naval reactor compartments. Cumulative releases to groundwater for Hanford solid waste, excluding ILAW disposed of in the Central Plateau, range from approximately 323 to approximately 445 Ci for technetium-99 during the 10,000-year analysis period. This compares with a range of release to groundwater between approximately 1530 and 2310 Ci of technetium-99 for all Hanford wastes except ILAW. The contribution from Hanford solid waste excluding ILAW would amount to about 20 percent of the cumulative release from Hanford sources excluding ILAW. The median release of technetium-99 from Hanford solid waste excluding ILAW was approximately 390 Ci while the median release for all Hanford sources except ILAW was approximately 2000 Ci. The ILAW cumulative release of technetium-99 for the base case (Mann et al. 2001) considering the full technetium-99 inventory was approximately 86 Ci by the end of the 10,000-year post-closure period. Accordingly the contribution from Hanford solid waste including ILAW would amount to about 25 percent of the cumulative release from all Hanford sources after 10,000 years.

For uranium, the cumulative releases to groundwater for Hanford solid waste disposed of in the Central Plateau range from 0 to approximately 94 Ci. However of all realizations simulated, no

realizations showed any release to groundwater from Hanford solid waste in the 200 East Area, and only 5 of 25 realizations show any release of uranium to groundwater from Hanford solid waste in the 200 West Area. Thus in an average (or median) sense, Hanford solid waste deposits would release no uranium to groundwater over the 10,000-year period of analysis. This compares with a median release of approximately 84 Ci and a range of release to groundwater from the 25 realizations of between approximately 10 to 300 Ci of uranium for all Hanford wastes except ILAW. Of the five realizations of non-zero uranium release from Hanford solid waste in the 200 West Area, the cumulative release ranged from 0 to approximately 90 Ci. The contribution from uranium in Hanford solid waste lies between 0 and 30 percent of the cumulative release from all Hanford sources. However, the median release of uranium from Hanford solid waste was zero while the median release for all Hanford sources (except ILAW) was approximately 84 Ci. The ILAW cumulative release of uranium for the base case (Mann et al. 2001) was less than 1 Ci by the end of the 10,000-year post-closure period. Accordingly, the contribution from Hanford solid waste including ILAW would amount to less than 1.2 percent of the cumulative median release of uranium from all Hanford sources after 10,000 years.

For iodine-129, the cumulative releases to groundwater for Hanford solid waste disposal of in the Central Plateau range from approximately 0 to 2.2 Ci. The median release to groundwater is 0.1 Ci. This amount compares with a range of release to groundwater from the 25 realizations of between approximately 0.1 and 8.8 Ci of iodine-129 for all Hanford wastes (except ILAW). The median value of iodine-129 releases from all Hanford sources (except ILAW) is approximately 0.36 Ci, all of which is from DOE waste because the median release from the commercial disposal site is approximately 0 Ci. With regard to the maximum values, the contribution from iodine-129 in Hanford solid waste lies between 0 and 25 percent of the cumulative release from all Hanford sources. With regard to the median values, the contribution from solid waste is 27 percent of the total. The ILAW cumulative release of iodine-129 for the base case (Mann et al. 2001) was approximately 0.07 Ci by the end of the 10,000-year post-closure period. This is a nominal amount given the existing iodine-129 plume in groundwater and the forecast releases of other waste forms.

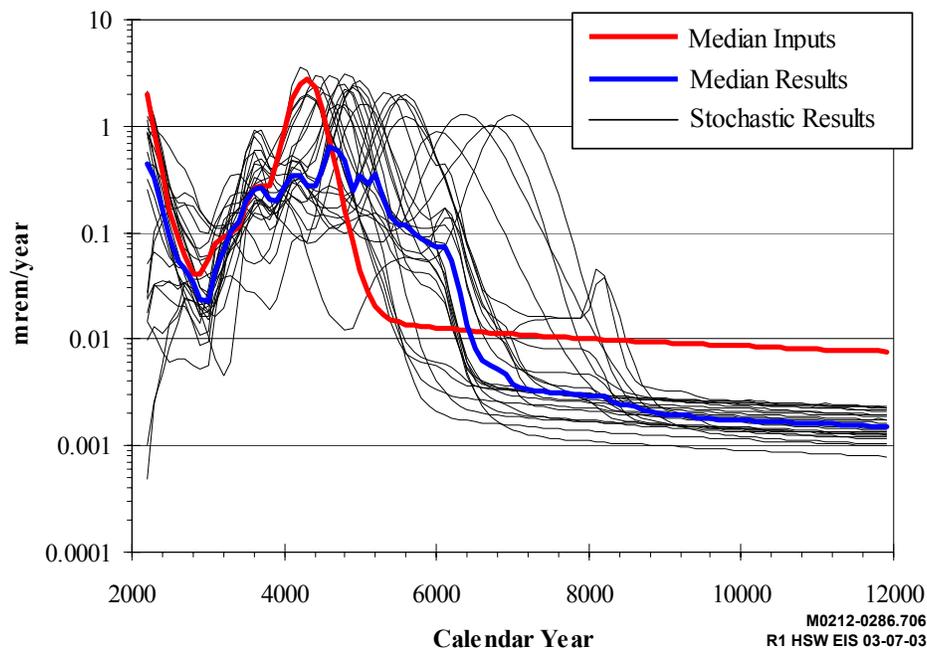
### **L.3.2.1 Drinking Water Dose at the Northeast Corner of the 200 West Area**

The drinking water dose to a human from technetium-99 using groundwater approximately 1 km (0.62 mi) outside the northeast corner of 200 West Area is provided in Figure L.69. The location was chosen to represent the highest doses from the local groundwater plume. The drinking water dose to a human from uranium and iodine-129 at the same location is provided in Figures L.70 and L.71. None of these figures includes the impact of ILAW. However, ILAW disposal occurs in the 200 East Area, and existing and future groundwater flow will conduct plumes from ILAW release away from the 200 West Area location represented in these figures. The data for technetium-99 show peaks that occur early and then again after approximately 3000 years. Figure L.69 exhibits a peak dose from technetium-99 of approximately 3.5 mrem/yr and a median of less than 1 mrem/yr with much lower consequences in the 7000 to 10,000-year time frame (that is, a range of 0.001 to 0.01 mrem/yr and a median less than 0.002 mrem/yr). Figure L.70 exhibits an early peak dose from uranium (that is, a range of less than 0.01 to 0.3 mrem/yr and a median of approximately 0.06 mrem/yr) and considerable variability in later years because of the sorption model for uranium (that is, a range of 0.0001 to 5 mrem/yr and a median of approximately 0.03 mrem/yr). Figure L.71 exhibits a peak dose from iodine-129 in the range of 0.02 to

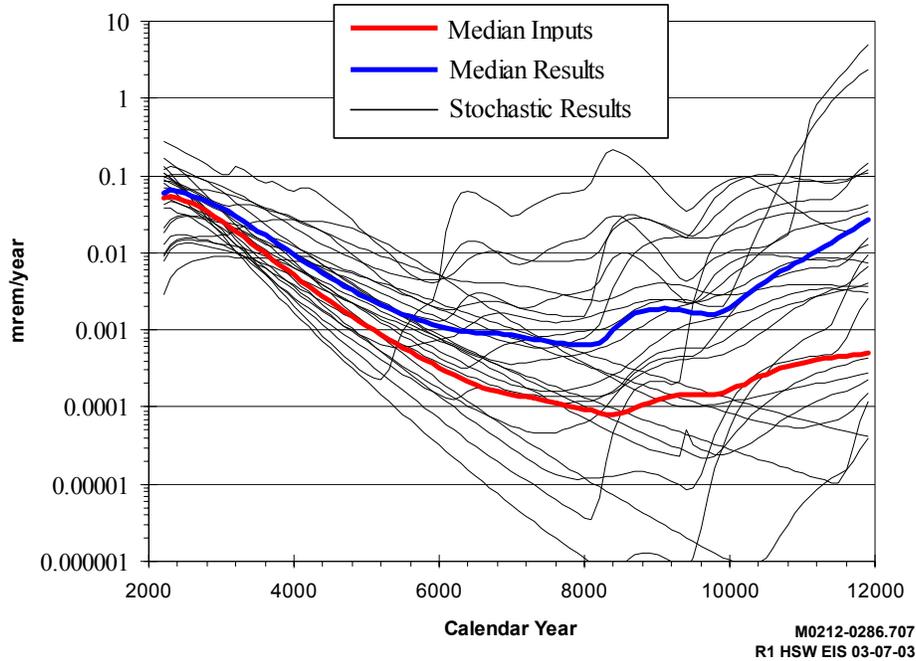
0.06 mrem/yr and a median of approximately 0.05 mrem/yr. Lower-level consequences occur in the 7000 to 12,000 A.D. time frame when a second peak or plateau in dose occurs with a long-term median value less than 0.02 mrem/yr.

### L.3.2.2 Drinking Water Dose at the Southeast Corner of the 200 East Area

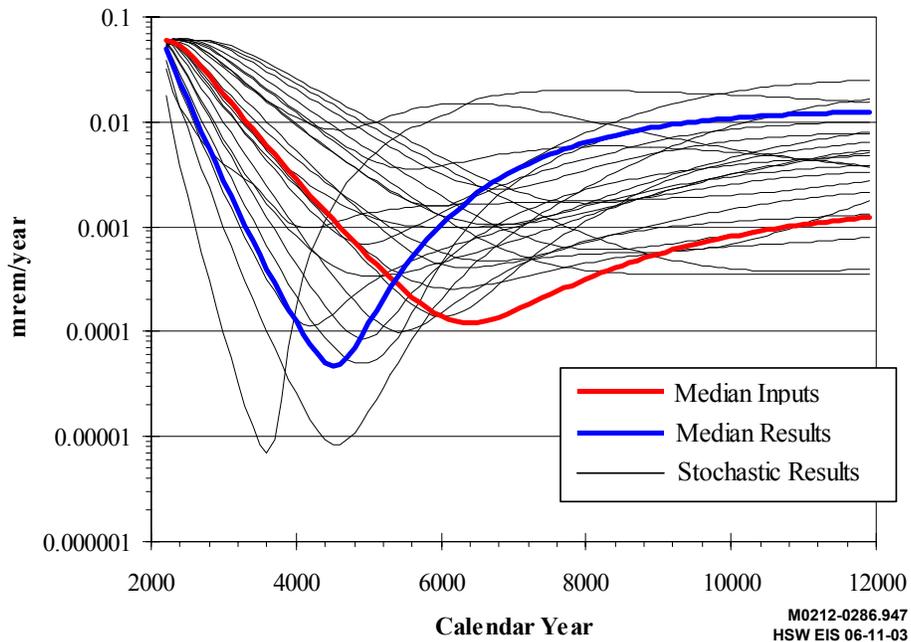
The drinking water dose to a human from technetium-99 using groundwater from approximately 1 km (0.62 mi) outside the southeast corner of 200 East Area is provided in Figure L.72. The location was chosen to represent the highest doses from the local groundwater plume. The drinking water dose to a human from uranium and iodine-129 at the same location is provided in Figures L.73 and L.74. None of these figures includes the impact of ILAW. The technetium-99 results show peaks early and again throughout the 10,000-year period. Figure L.72 exhibits a peak median dose from technetium-99 in the range of 1 to 2 mrem/yr during the 10,000-year period. Peaks of individual realizations range to 3 mrem/yr. Figure L.73 exhibits a peak median dose from uranium of less than 1 mrem/yr early with a long-term median value of less than 0.01 mrem/yr. There is considerable variability in later years because of the sorption model for uranium (that is, after 10,000 years there is a range of approximately 0.001 to 1 mrem/yr, but the median is less than 0.01 mrem/yr). Figure L.74 exhibits a peak dose from iodine-129 in the range of 0.2 to 0.25 mrem/yr and a median of approximately 0.2 mrem/yr with lower consequences after 7000 A.D. (that is, a range of 0.07 to 0.003 mrem/yr and a median of less than 0.015 mrem/yr).



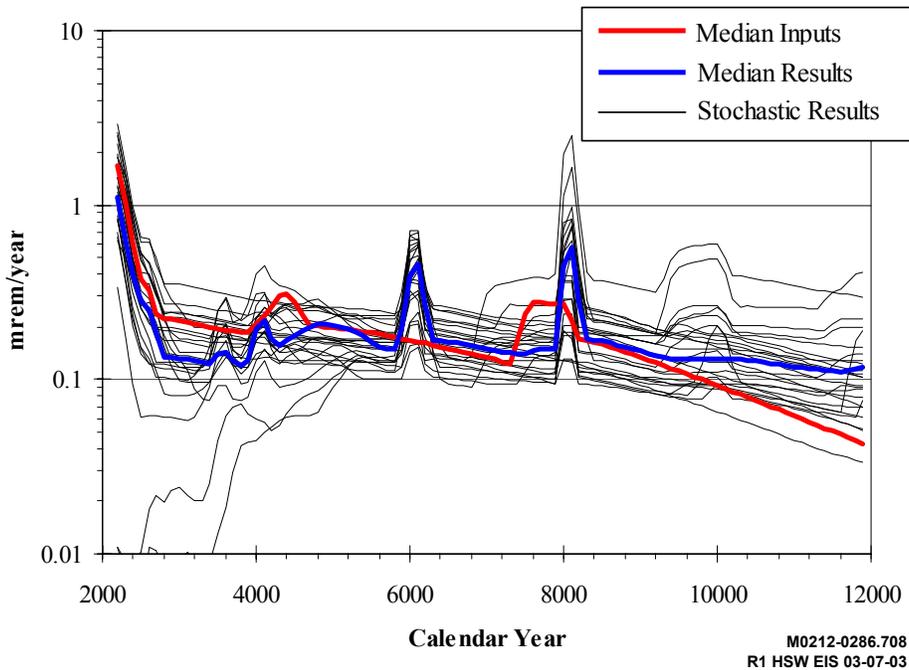
**Figure L.69.** Hypothetical Drinking Water Dose from Technetium-99 from All Hanford Sources Except ILAW, Melters, and Naval Reactors in Groundwater 1 km Downgradient Northeasterly of the 200 West Area



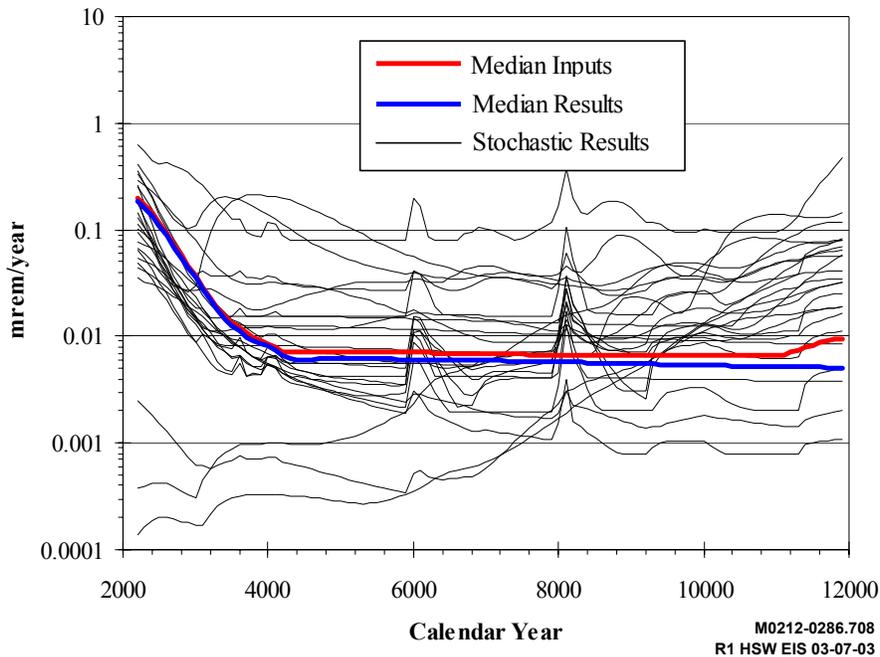
**Figure L.70.** Hypothetical Drinking Water Dose from Uranium from All Hanford Sources Except ILAW, Melters, and Naval Reactors in Groundwater 1 km Downgradient Northeasterly of the 200 West Area



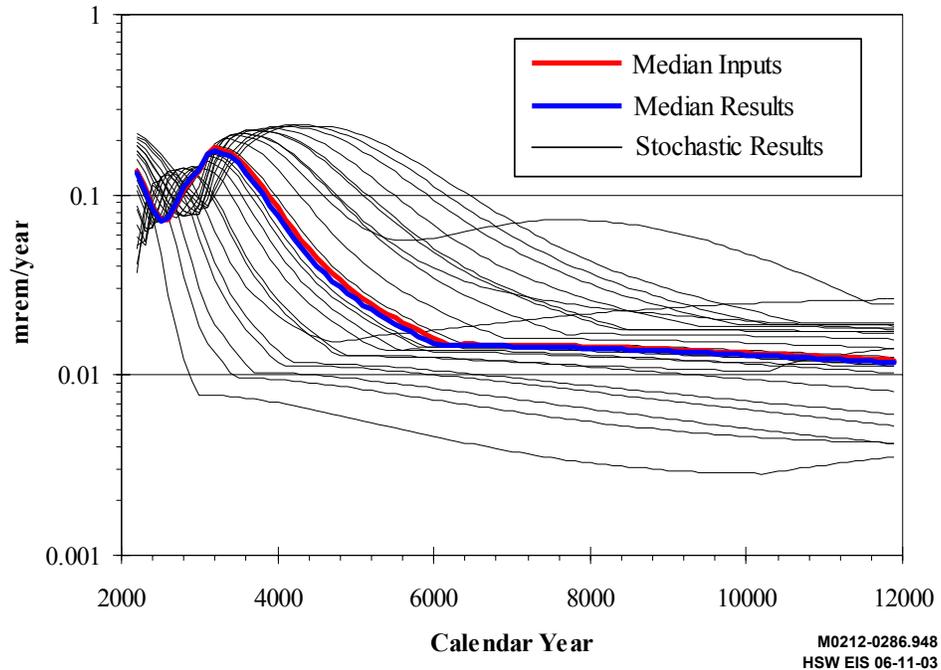
**Figure L.71.** Hypothetical Drinking Water Dose from Iodine-129 from All Hanford Sources Except ILAW, Melters, and Naval Reactors in Groundwater 1 km Downgradient Northeasterly of the 200 West Area



**Figure L.72.** Hypothetical Drinking Water Dose from Technetium-99 from All Hanford Sources Except ILAW, Melters, and Naval Reactors in Groundwater 1 km Downgradient Southeasterly of the 200 East Area



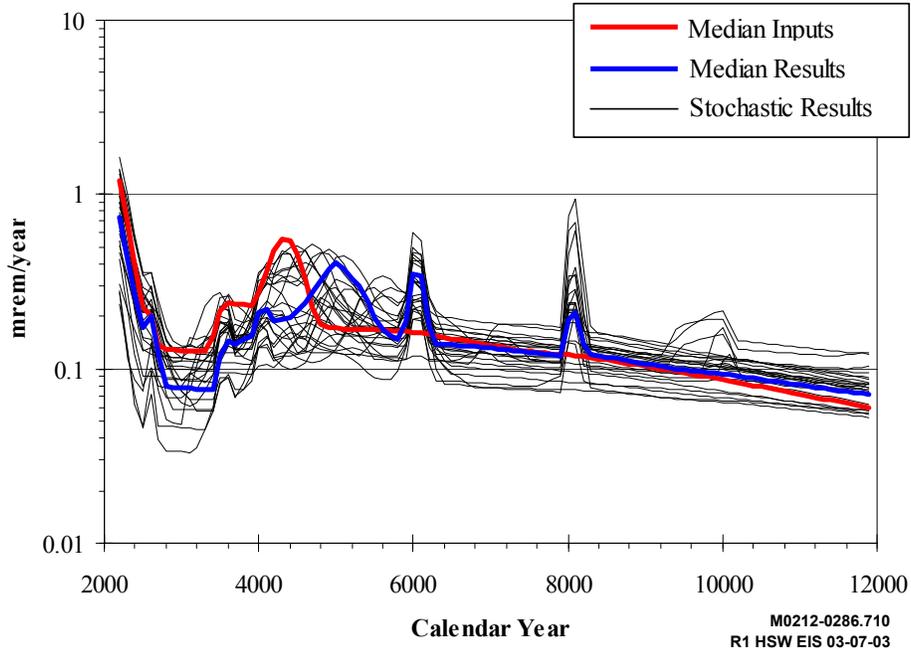
**Figure L.73.** Hypothetical Drinking Water Dose from Uranium from All Hanford Sources Except ILAW, Melters, and Naval Reactors in Groundwater 1 km Downgradient Southeasterly of the 200 East Area



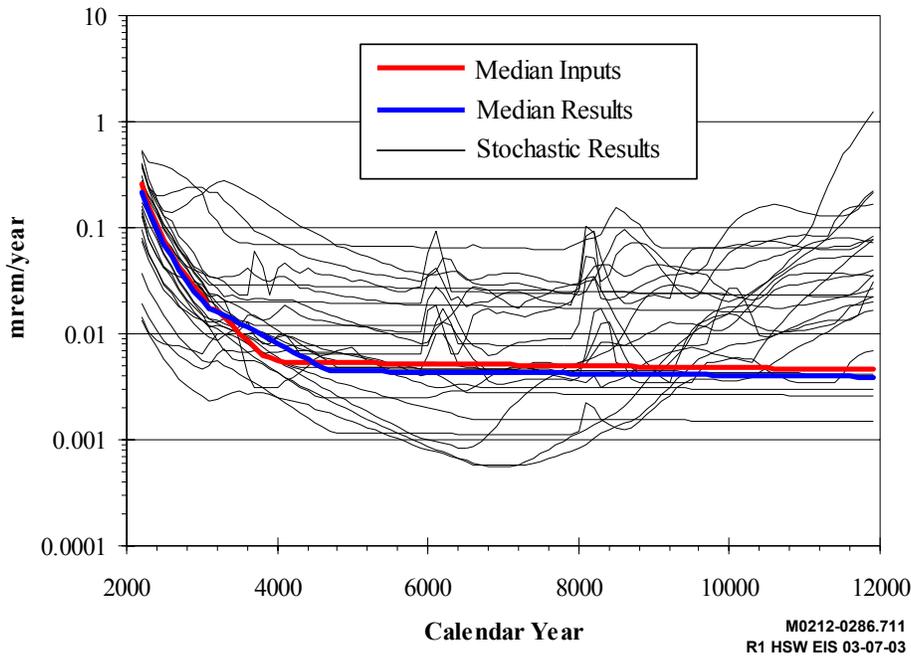
**Figure L.74.** Hypothetical Drinking Water Dose from Iodine-129 from All Hanford Sources Except ILAW, Melters, and Naval Reactors in Groundwater 1 km Downgradient Southeasterly of the 200 East Area

### L.3.2.3 Drinking Water Dose at the Northwest Corner of the 200 East Area

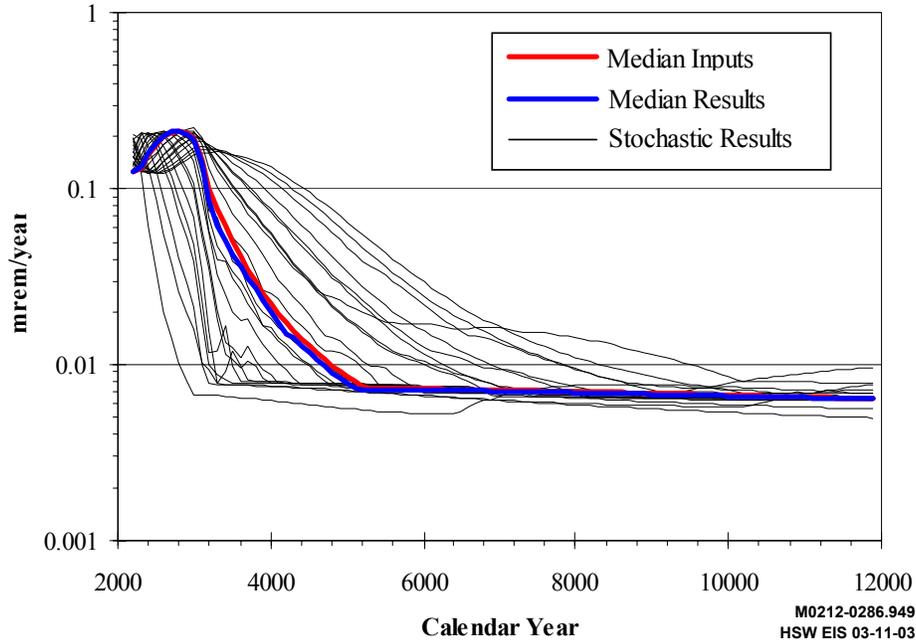
The drinking water dose to a human from technetium-99 using groundwater from approximately 1 km (0.62 mi) outside the northwest corner of 200 East Area is provided in Figure L.75. The location was chosen to represent the highest doses from the local groundwater plume. The drinking water dose to a human from uranium and iodine-129 at the same location is provided in Figures L.76 and L.77. These figures exclude the impacts of ILAW. The technetium-99 results show peaks early and again throughout the 10,000-year analysis period. Figure L.75 exhibits a peak median dose from technetium-99 in the range of 0.2 to 1 mrem/yr during the 10,000-year analysis period. Figure L.76 exhibits a peak median dose from uranium of approximately 0.3 mrem/yr with a long-term median value of less than 0.01 mrem/yr. There is considerable variability in later years because of the sorption model for uranium (that is, after 10,000 years, there is range of approximately 0.001 to 1 mrem/yr, but the median is less than 0.01 mrem/yr). Figure L.77 exhibits a peak median dose from iodine-129 of less than 0.25 mrem/yr with a long-term median value of less than 0.01 mrem/yr.



**Figure L.75.** Hypothetical Drinking Water Dose from Technetium-99 from All Hanford Sources Except ILAW, Melters, and Naval Reactors in Groundwater 1 km Downgradient Northwesternly of the 200 East Area



**Figure L.76.** Hypothetical Drinking Water Dose from Uranium from All Hanford Sources Except ILAW, Melters, and Naval Reactors in Groundwater 1 km Downgradient Northwesternly of the 200 East Area



**Figure L.77.** Hypothetical Drinking Water Dose from Iodine-129 from All Hanford Sources Except ILAW, Melters, and Naval Reactors in Groundwater 1 km Downgradient Northwesternly of the 200 East Area

### L.3.3 Dose from Columbia River Water at the City of Richland Pumping Station

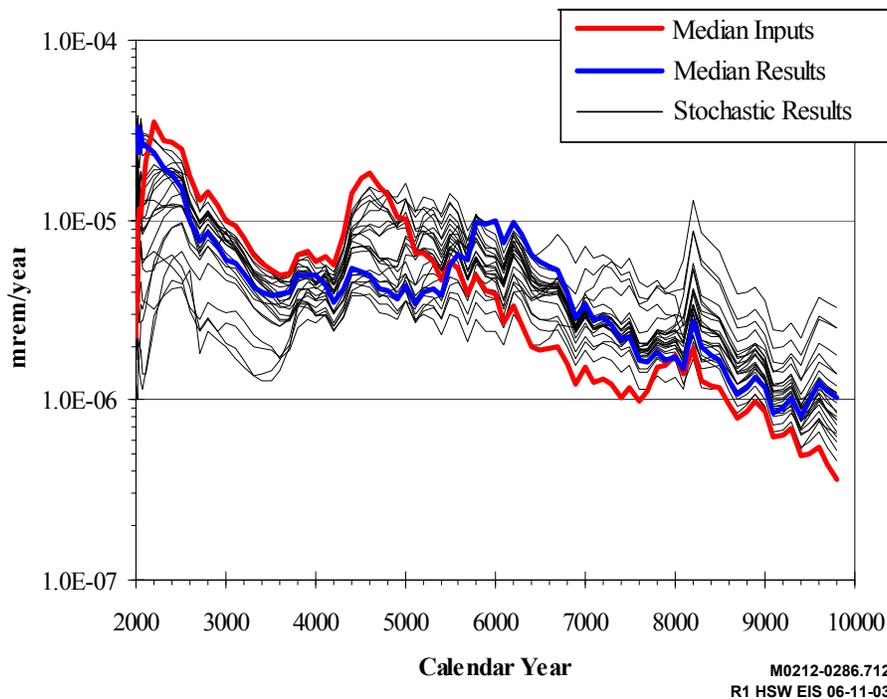
Annual dose to humans based on consumption of river water is summarized in this section. The exposure scenario has an adult human drinking 2 L/day of contaminated river water from the modeled near-shore point nearest the City of Richland pumping station. The stochastic capability of SAC was employed for these simulations, so the following results are shown in each plot in this section:

- Individual **stochastic results** (25 realizations) are shown in black.
- The **median result** of the 25 realizations—that is, the realization that resulted in the median integrated cumulative dose in the year 9900 A.D.—is shown in blue. Although the groundwater simulations continued through the year 12,050 A.D., the river simulations were terminated at the year 9900 A.D. due to software design constraints.
- The **median-inputs** simulation—a separate single-realization simulation with SAC using the median value of all stochastic input variables—is shown in red.

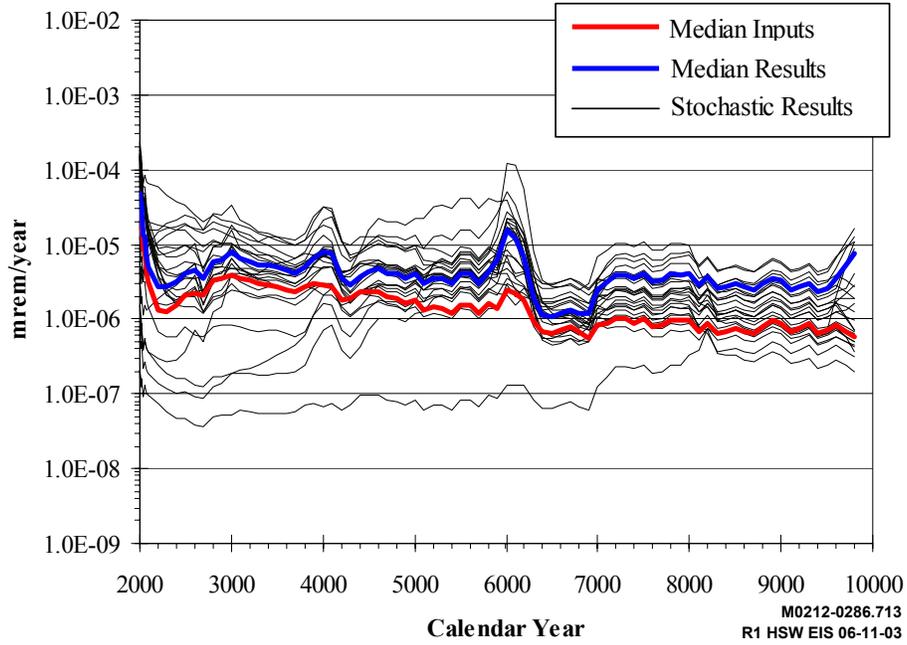
The variability in the stochastic results is due to the inventory, release, and transport of technetium-99, iodine-129, and uranium. The human dose model uses fixed inputs in the calculations. The doses provided in this section are based on all waste at the Hanford Site, except ILAW, and do not include background concentrations in the river. Thus, the doses are due entirely to Hanford contaminants, with most of the dose due to waste forms other than solid wastes.

### L.3.3.1 Drinking Water Dose at the City of Richland Pumping Station

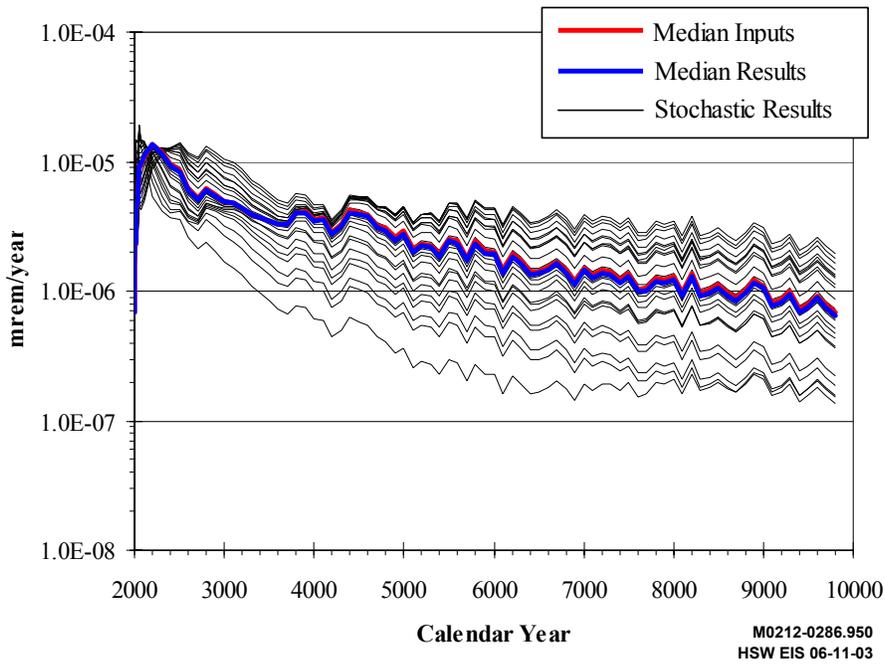
The drinking water dose to a human from technetium-99, iodine-129, and uranium using water concentrations calculated near the City of Richland pumping station in the Columbia River never gets above  $1 \times 10^{-4}$ , or 0.0001, mrem/yr. This location is downriver from all groundwater plumes of Hanford origin. The maximum estimated annual dose from technetium-99 over all realizations shown in Figure L.78 from the year 2000 through 9900 A.D. is less than  $4 \times 10^{-5}$ , or 0.00004, mrem/yr, while the peak median dose was approximately  $3.5 \times 10^{-5}$ , or 0.000035, mrem/yr. The maximum annual dose from uranium over all realizations shown in Figure L.79 from the year 2000 through 9900 A.D. is less than  $2 \times 10^{-4}$ , or 0.0002, mrem/yr, while the peak median dose was approximately  $5 \times 10^{-5}$ , or 0.00005, mrem/yr. The maximum annual dose from iodine-129 over all realizations shown in Figure L.80 from the year 2000 through 9900 A.D. is approximately  $2 \times 10^{-5}$ , or 0.00002, mrem/yr, while the peak median dose was less than  $1.5 \times 10^{-5}$ , or 0.000015, mrem/yr.



**Figure L.78.** Drinking Water Dose at the City of Richland Pumping Station from Technetium-99 Due to All Hanford Sources Except ILAW, Melters, and Naval Reactors



**Figure L.79.** Drinking Water Dose at the City of Richland Pumping Station from Uranium Due to All Hanford Sources Except ILAW, Melters, and Naval Reactors



**Figure L.80.** Drinking Water Dose at the City of Richland Pumping Station from Iodine-129 Due to All Hanford Sources Except ILAW, Melters, and Naval Reactors

### L.3.4 Annual Drinking Water Dose at Selected 200 East Area and Columbia River Locations from Hanford Sources Including ILAW

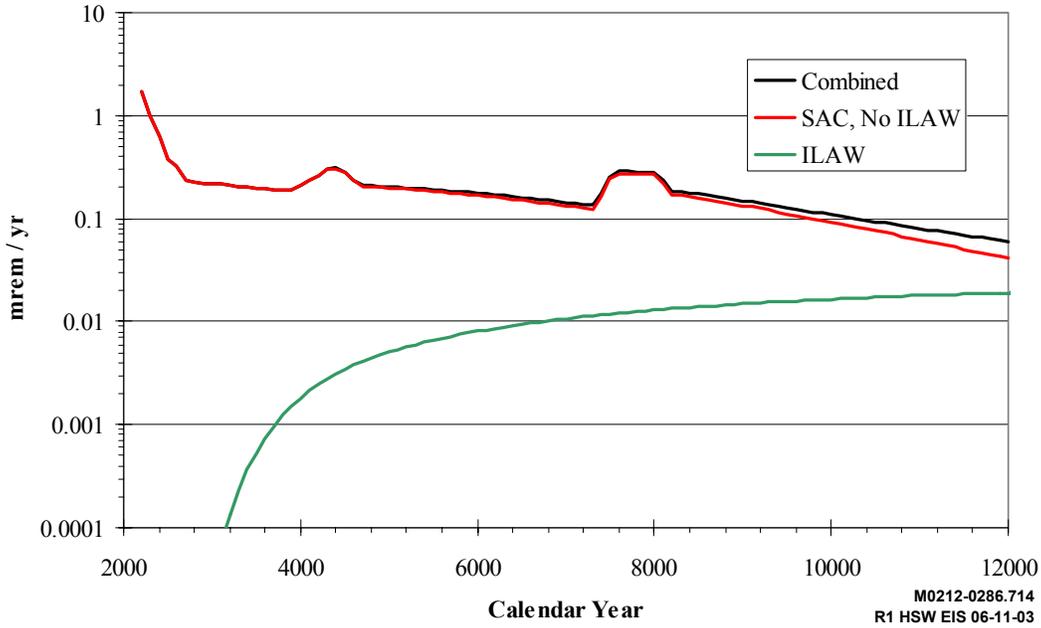
The deterministic capability of SAC was employed with results of the ILAW performance assessment (Mann et al. 2001), which were scaled to current inventory estimates, to provide an initial estimate of the cumulative impact of all Hanford sources including ILAW. These deterministic results portray the median-inputs case of the initial assessment using SAC and the base case of the ILAW performance assessment (Mann et al. 2001). Essentially, the 2-L/d dose impacts from the ILAW inventories of technetium-99, iodine-129, and uranium reported in the ILAW performance assessment (Mann et al. 2001) are superimposed on the SAC median-value simulation. A series of six plots (Figures L.81 through L.86) shows combined SAC and ILAW results at a point 1 km southeast of the 200 East Area.

The cumulative impact from technetium-99 for all Hanford sources is provided in Figure L.81. This is the annual drinking water dose from a 2-L/d drinking water scenario for technetium-99 at a point of analysis approximately 1 km (0.62 mi) southeast of the 200 East Area. The curve is a composite of the SAC initial assessment result and the base case ILAW result (Mann et al. 2001). To account for the current estimate of 25,500 Ci of technetium-99 in low-activity waste from the single- and double-shell tanks, the ILAW analysis of a 5790 Ci technetium-99 source has been scaled accordingly.

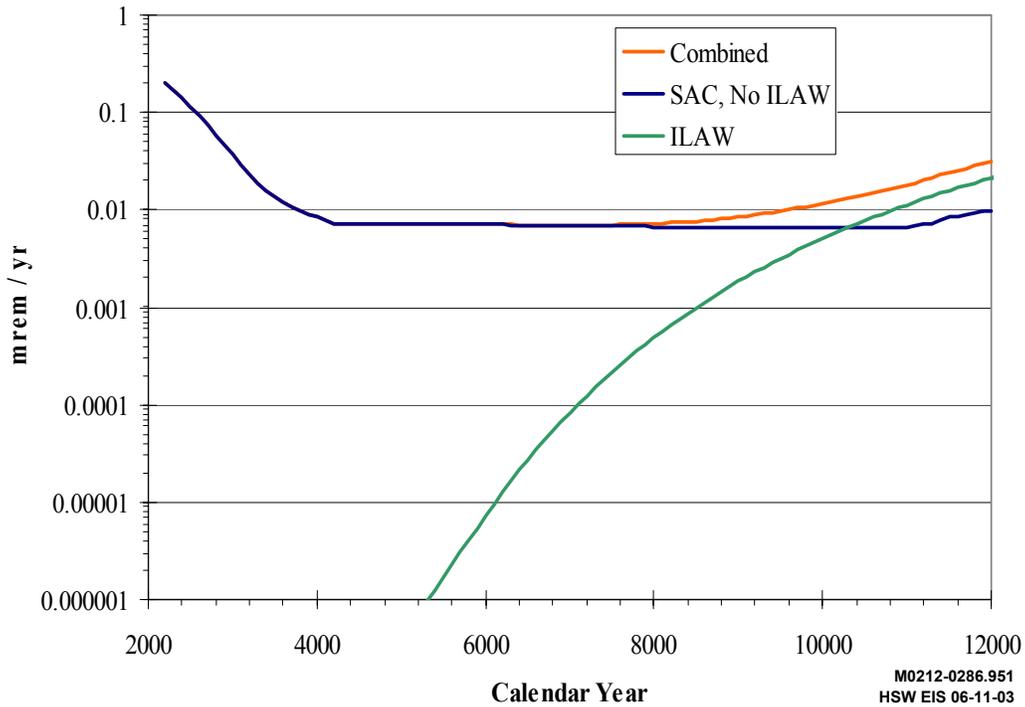
The cumulative technetium-99 result shown in Figure L.81 exhibits an initial peak in the next two centuries. The peak is approximately 2 mrem/yr and is related to releases from liquid discharge sites in the 200 East Area. Additional but lower peaks of approximately 0.3 mrem/yr, appear in approximately 4400 A.D. and 7600 A.D. Releases from solid waste disposal facilities in the 200 West Area are responsible for the earlier of these two secondary peaks. Tank waste residuals releasing from the 200 East Area, modeled as 1 percent residual tank waste volume in a salt cake waste form, are responsible for the last secondary peak.

By the end of the 10,000-year post-closure period, the cumulative dose from technetium-99 for all Hanford sources is approximately 0.06 mrem/yr, of which approximately 0.02 mrem/yr is from ILAW and 0.04 mrem/yr is from all other Hanford sources. Based on uncertainty in the groundwater conceptual model, the ILAW contribution may be four times larger. Thus, the ILAW contribution may be 0.08 mrem/yr and may be comparable to or larger than that for all other Hanford sources. For this alternate conceptual model, the cumulative 2-L/d dose would be approximately 0.12 mrem/yr at 10,000 years post closure. Note that ILAW release and associated dose impacts play a role in the last several thousand years, and do not substantially alter the secondary peaks described earlier.

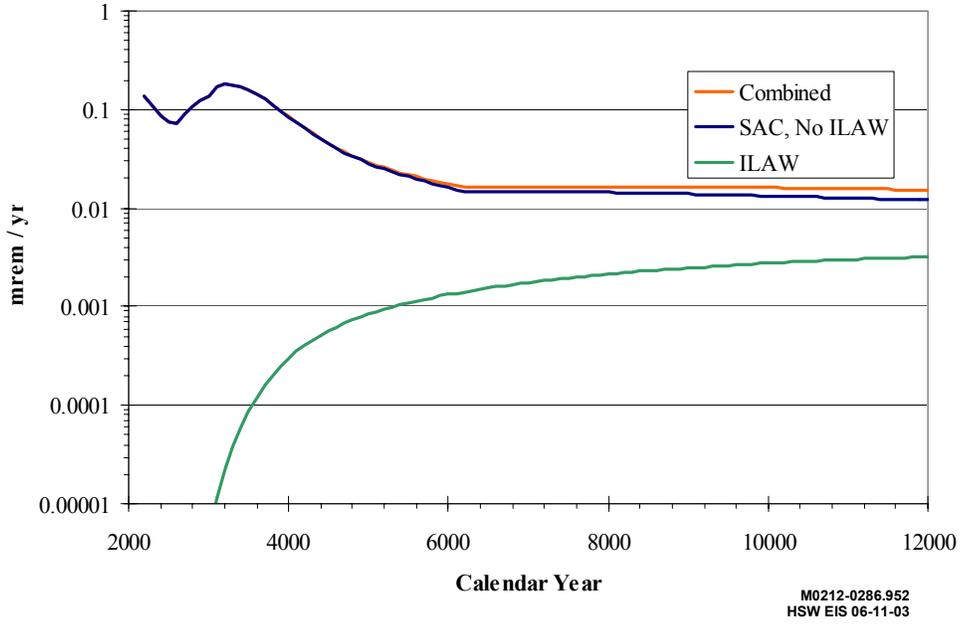
The cumulative impact from uranium for all Hanford sources at the line of analysis 1 km (0.6 mi) southeast of 200 East is provided in Figure L.82. The plot of SAC initial assessment and ILAW base-case results shows an early peak drinking-water dose of approximately 0.20 mrem/yr, and the dominance of ILAW uranium by the end of the 10,000-year analysis period. As in the case of technetium-99, uncertainty in the groundwater conceptual model could produce a fourfold increase in ILAW contributions, and the long-term uranium dose of approximately 0.02 mrem/yr could become 0.08 mrem/yr.



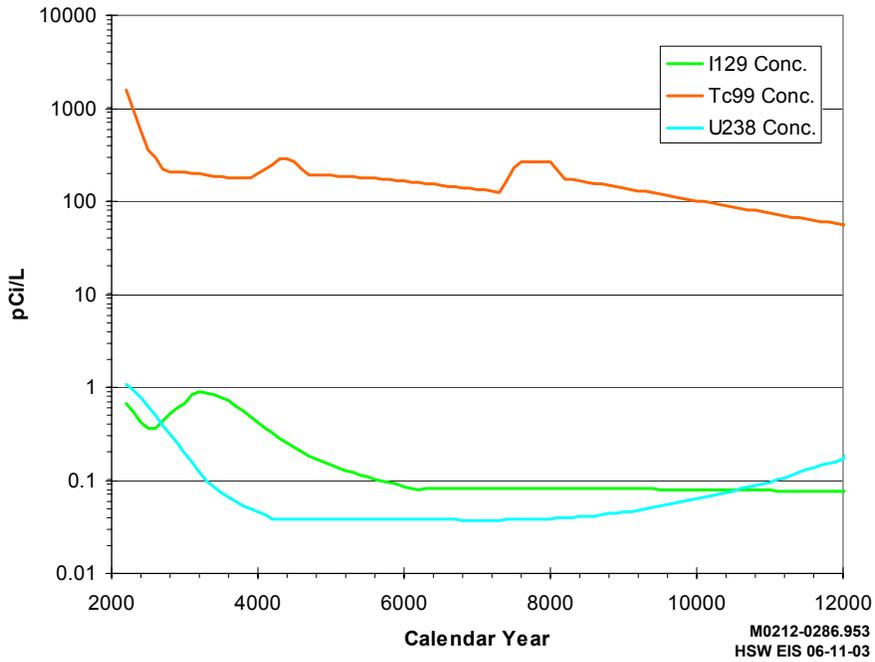
**Figure L.81.** Hypothetical Drinking Water Dose from Technetium-99 from Hanford Sources Including ILAW in Groundwater 1 km Downgradient Southeasterly of the 200 East Area



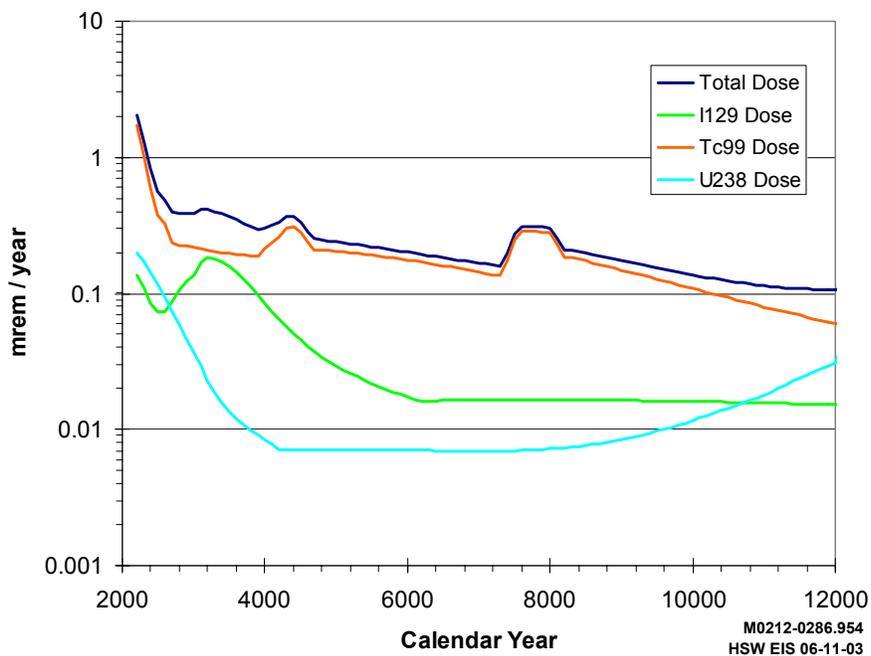
**Figure L.82.** Hypothetical Drinking Water Dose from Uranium from Hanford Sources Including ILAW in Groundwater 1 km Downgradient Southeasterly of the 200 East Area



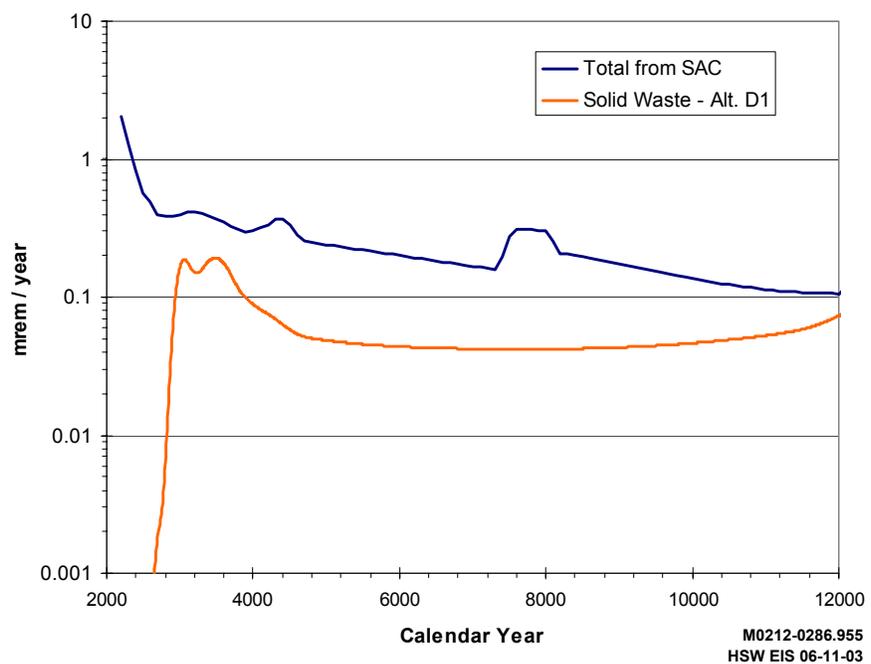
**Figure L.83.** Hypothetical Drinking Water Dose from Iodine-129 from Hanford Sources Including ILAW in Groundwater 1 km Downgradient Southeasterly of the 200 East Area



**Figure L.84.** Concentrations of Technetium-99, Iodine-129, and Uranium from All Hanford Sources in Groundwater 1 km Downgradient Southeasterly of 200 East Area.



**Figure L.85.** Hypothetical Drinking Water Dose from Technetium-99, Iodine-129, and Uranium from All Hanford Sources in Groundwater 1 km Downgradient Southeasterly of 200 East Area.



**Figure L.86.** Hypothetical Total Drinking Water Dose from All Hanford Sources and from Hanford Solid Waste Contributions in Groundwater 1 km Downgradient Southeasterly of 200 East Area.

The cumulative impact from iodine-129 for all Hanford sources at the line of analysis 1 km (0.6 mi) southeast of 200 East is provided in Figure L.83. The plot of SAC initial assessment and ILAW base-case results shows an early peak drinking-water dose of approximately 0.2 mrem/yr, and the increasing but not dominant influence of ILAW iodine-129 later (that is a peak contribution of approximately 0.003 mrem/yr). Groundwater conceptual model uncertainty could yield ILAW contributions four times larger or near 0.01 mrem/yr, hence comparable to the dose associated with other waste releases for iodine-129.

These results for technetium-99, iodine-129, and uranium are an approximation achieved by superimposing the results of two independently conducted analyses. The results indicate that the contribution from ILAW, which represents a substantial fraction of the inventory at Hanford, does not dominate the overall dose prediction made in the initial assessment for all wastes other than ILAW at a line of analysis approximately 1 km (0.6 mi) downgradient from the ILAW disposal facility. Of the three radionuclides, it appears that uranium released from ILAW may dominate uranium released from all other sites; however, the dose from technetium-99 appears to dominate the ILAW and cumulative dose curves discussed below.

Concentration profiles over time for technetium-99, iodine-129, and uranium from all Hanford sources at a line of analysis approximately 1 km (0.6 mi) downgradient southeasterly of the 200 East Area are shown in Figure L.84. Maximum concentrations for each of the radionuclides occur in the near term. Concentrations of technetium-99, iodine-129, and uranium are respectively 1600, 1.1 and, 0.90 pCi/L. The technetium-99 and iodine-129 concentrations are at or above the benchmark drinking water standards of 900 pCi/L and 1 pCi/L, respectively. The uranium concentration, approximately 3.3 µg/L, is below its benchmark drinking water standard of 30 µg/L. The cumulative impact for technetium-99, iodine-129, and uranium from all Hanford sources is provided in Figure L.85. This is the annual dose resulting from a 2 L/d drinking water scenario for each of the radionuclides. The values of maximum dose for technetium-99, iodine-129, and uranium corresponding to the maximum concentrations are 1.7, 0.18, and 0.20 mrem/yr, respectively.

The annual cumulative dose from technetium-99, iodine-129, and uranium exhibits a peak of approximately 2 mrem/yr within the next two centuries. This peak appears to be related to releases from past liquid discharge sites in the 200 East Area. Additional, but lower, peaks of approximately 0.4 mrem/yr appear in approximately years 4400 and 7600. Based on the visualization of groundwater contaminant transport in the unconfined aquifer over 10,000 years, it appears that releases of technetium-99 from Hanford solid waste disposal facilities in the 200 West Area are responsible for the peak in approximately year 4400. Tank waste residuals releasing technetium-99 in the 200 East Area from a 1-percent residual volume and a salt cake waste are responsible for the last peak. The underlying long-term dose declines to 0.1 mrem/yr by 10,000 years post closure. This dose is related to long-term releases from Hanford solid waste and other miscellaneous waste, which, when combined, account for approximately 0.07 mrem/yr, and from ILAW, which accounts for approximately 0.04 mrem/yr.

Based on uncertainty in the groundwater conceptual model, the ILAW contribution to the cumulative result may be approximately four times larger. The resulting cumulative 2 L/d drinking water dose from ILAW for technetium-99, iodine-129, and uranium would be approximately 0.2 mrem/yr at 10,000 years

post closure. Somewhat higher contributions than shown here from Hanford solid waste and other sources, (that is, 0.07 mrem/yr) may also occur because of uncertainty in the groundwater conceptual model used in the SAC; however, groundwater-model uncertainty as it relates to the Hanford solid waste contributions is addressed in Section 5.3 (of Volume I of this EIS) and Appendix G. Note that the ILAW release and associated dose impacts play a role in the last several thousand years only and do not substantially influence the peaks that occur earlier.

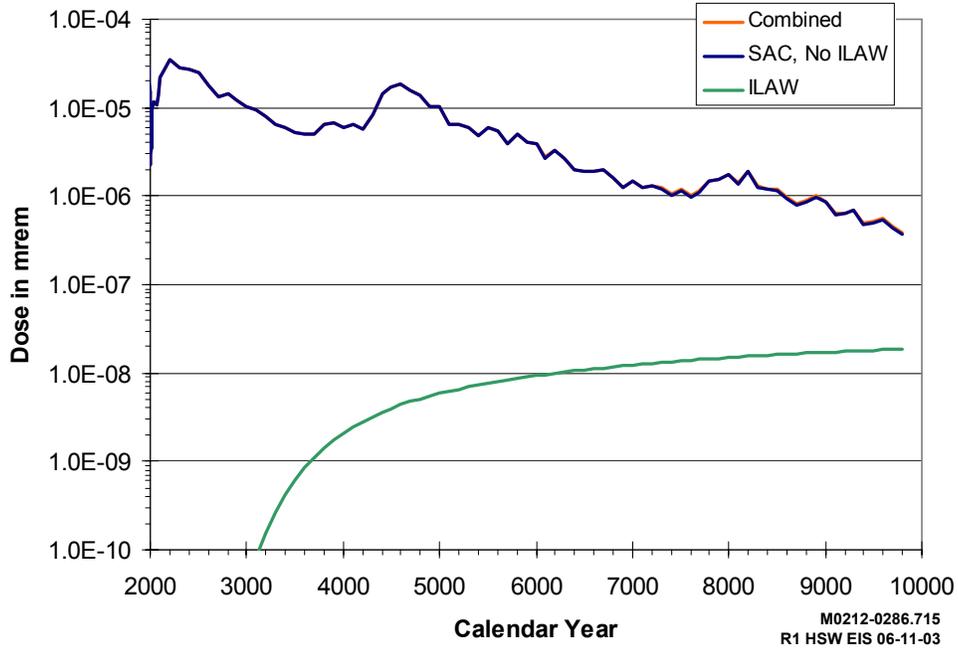
The cumulative dose from all Hanford sources and the portion attributed to solid waste at the line of analysis 1 km downgradient southeasterly of the 200 East Area are shown in Figure L.86. Differences in the two curves (that is, the slope of the curves) are attributed to somewhat different distribution coefficient values used in the simulation of solid waste disposal alternatives and in the cumulative assessment. The more rapid release and migration of uranium in the evaluation of solid waste disposal alternatives enables uranium to influence the long-term solid waste contribution between 8000 and 12,000 A.D. This uranium influence is not seen in the initial assessment simulated with SAC because of the use of somewhat higher distribution coefficients to represent median or central tendency behavior.

#### **Distribution Coefficients, $K_d$ s, of the Linear Sorption Isotherm Model**

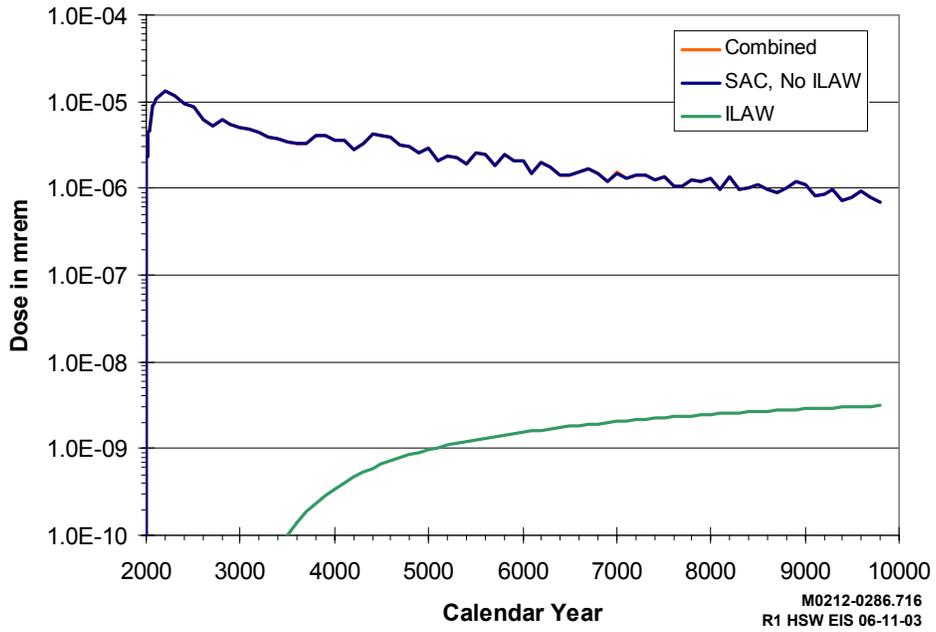
The System Assessment Capability (SAC) is designed to simulate a stochastic analysis where parameter distributions are centered around median or best estimate parameter values. For the distribution coefficient  $K_d$ , values were drawn from a recent summary of  $K_d$  data by Cantrell et al. (2002) and patterned after the  $K_d$  model in the 1998 composite analysis (Kincaid et al. 1998). The deterministic case posed and analyzed is based on median values for all stochastic data.

The HSW EIS is designed to execute a series of deterministic analyses where scenarios of waste disposal are varied but model parameters are fixed so as to produce conservative simulations, that is, fixed at lower  $K_d$  values to create more rapid and higher concentration contaminant transport. Accordingly, the conservative representation of the HSW EIS produces more rapid migration movement and higher concentrations than the median value representation of the SAC.

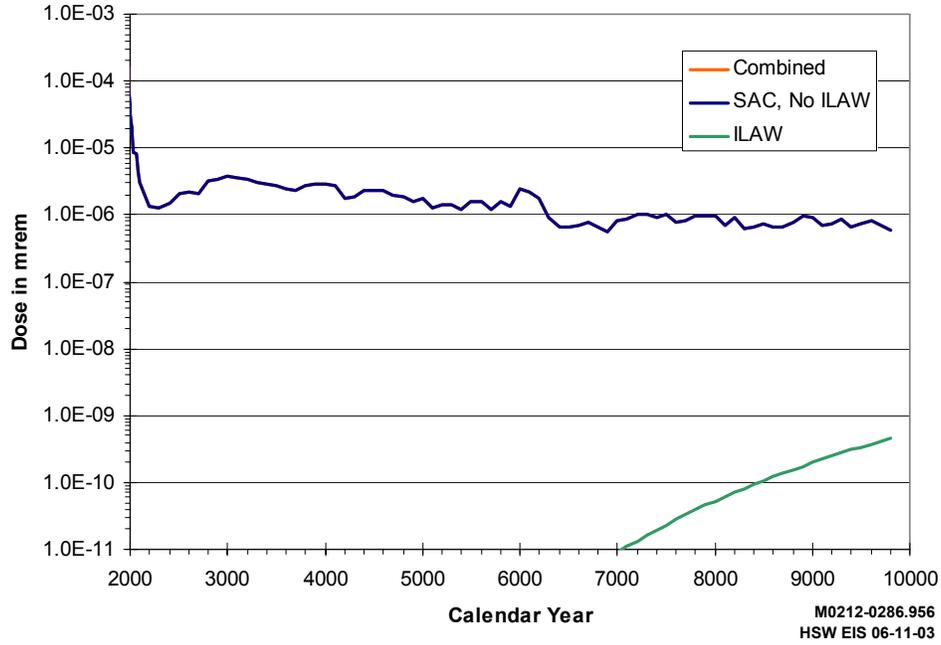
Another series of six plots (Figures L.87 through L.92) shows combined results for use of water from the Columbia River at the City of Richland pumping station located downstream of the Hanford Site. This location is downriver from all groundwater plumes of Hanford origin, and reveals the substantial dilution and dispersion that occurs because of the relatively substantial discharge of the Columbia River as compared to that of the unconfined aquifer underlying Hanford. Although groundwater simulations continued through the year 12,050 A.D. (that is, 10,000 years post closure), the river simulations were terminated at 9900 A.D. (that is 8000 years post closure) due to design constraints in the software used for the river model. Thus, river model forecasts are not available for the final 2000 years of the 10,000-year post-closure period. However, as is apparent from the simulation results achieved, trends seen in the



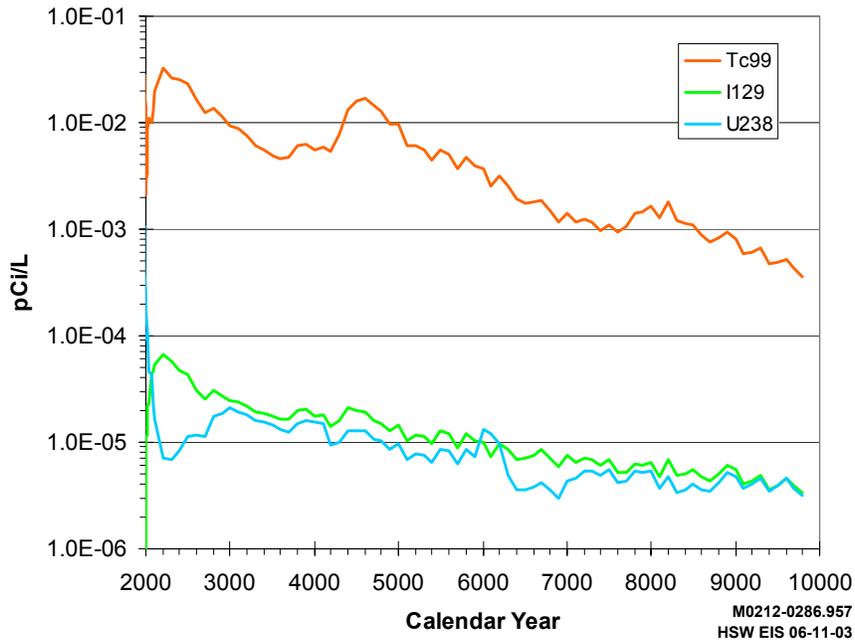
**Figure L.87.** Annual Drinking Water Dose from Technetium-99 in the Columbia River at the City of Richland Pumping Station from Hanford Sources Including ILAW



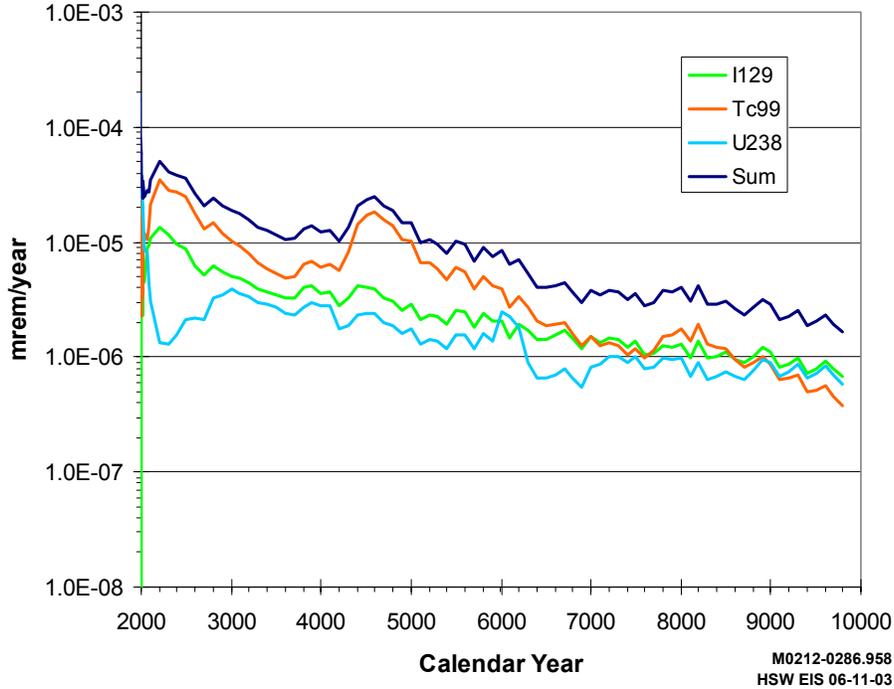
**Figure L.88.** Annual Drinking Water Dose from Iodine-129 in the Columbia River at the City of Richland Pumping Station from Hanford Sources Including ILAW



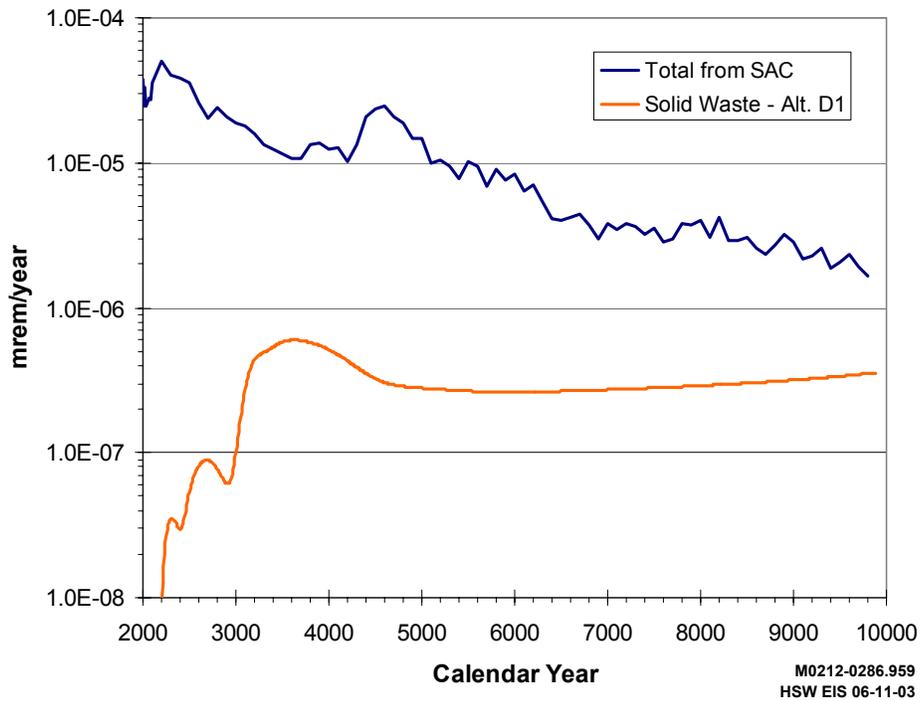
**Figure L.89.** Annual Drinking Water Dose from Uranium in the Columbia River at the City of Richland Pumping Station from Hanford Sources Including ILAW



**Figure L.90.** Concentration of Technetium-99, Iodine-129, and Uranium in the Columbia River at the City of Richland Pumping Station



**Figure L.91.** Drinking Water Dose from Technetium-99, Iodine-129, and Uranium in the Columbia River at the City of Richland Pumping Station



**Figure L.92.** Total Drinking Water Dose from All Hanford Sources and the Hanford Solid Waste Contribution in the Columbia River at the City of Richland Pumping Station.

groundwater system near the Central Plateau appear somewhat later and at much reduced concentrations in the Columbia River at the City of Richland location. Results of the dose analyses are presented as annual radiation dose.<sup>(a)</sup>

A comparison of consequences from consuming 2 L/day of river water with and without the ILAW release of technetium-99, iodine-129, and uranium are provided in Figures L.87, L.88, and L.89 for the Columbia River at the City of Richland pumping station. Results from the SAC median-input case of the initial assessment and from the ILAW performance assessment base case are shown on each figure.

Figure L.87 shows that dose originating from the ILAW containing 25,500 Ci of technetium-99 is well below the technetium-99 dose originating from all other Hanford wastes, and the cumulative dose is less than  $1.0 \times 10^{-4}$  mrem/yr. The cumulative dose from technetium-99 is less than  $1.0 \times 10^{-6}$  mrem/yr at 8000 years post closure, and this result is five orders-of-magnitude below the dose predicted at the 200 East Area location.

The comparison graphic of consequences from uranium is provided in Figure L.89. The peak value of uranium consequence occurs in the near term and is less than  $1.0 \times 10^{-4}$  mrem/yr. After 8000 years post closure and at the time of greatest ILAW uranium impact, the dose from uranium is estimated to be approximately three orders-of-magnitude below that of all other Hanford sources. Combined, the estimated dose from uranium is less than  $1.0 \times 10^{-6}$  mrem/yr after approximately 4000 years. The consequences from iodine-129 releases are shown in Figure L.88. The peak dose from iodine-129 also occurs in the near term and is less than  $2.0 \times 10^{-5}$  mrem/yr. After 8000 years post closure and at the time of greatest ILAW iodine-129 impact, the dose from ILAW iodine-129 is estimated to be more than two orders-of-magnitude below that of all other Hanford sources. Combined, the estimated dose from iodine-129 at that time is approximately  $1.0 \times 10^{-6}$  mrem/yr.

These results for technetium-99, iodine-129, and uranium are an approximation achieved by superimposing the results of two independently conducted analyses. Nevertheless, the results indicate that the contribution from ILAW does not substantially influence the overall dose prediction made in the initial assessment for all wastes other than ILAW at the City of Richland.

Figure L.90 shows the concentrations of technetium-99, iodine-129, and uranium from all Hanford sources from Columbia River water at the City of Richland pumping station for the median inputs case. A corresponding plot of the drinking water dose for technetium-99, iodine-129, and uranium is provided in Figure L.91. While having a much more variable appearance caused by river discharge variability, the peaks seen in technetium-99 plots at the 200 East Area location are also present in Figure L.91. Dose from Hanford-origin uranium and iodine-129 also exhibit a temporal variability caused by variability in Columbia River discharge. However, the peaks are subdued and delayed because these elements are

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(a) The National Council on Radiation Protection and Measurements continues to hold that a dose of 1 mrem/yr is a dose “below which efforts to reduce the radiation exposure to the individual are unwarranted (Section 17 of NCRP, 1993)” (NCRP 2002). Regardless, in this EIS doses are reported as calculated, however small they may be. Thus doses will be seen that are several to many orders of magnitude below 1 mrem/yr and, while these may be useful for comparative purposes, they should not be construed as having any physical meaning in terms of detriment to health.

sorbed, and consequently, they migrate more slowly than groundwater and non-sorbed elements such as technetium. Concentration and annual dose values are approximately five orders-of-magnitude lower at the City of Richland compared to those predicted at the 200 East Area. Figure L.91 shows that the maximum doses for the median inputs case representation of technetium-99, iodine-129, and uranium are less than or equal to  $3.5 \times 10^{-5}$ ,  $1.5 \times 10^{-5}$ , and  $5 \times 10^{-5}$  mrem/yr, respectively.

These peaks occur at different times based on the sorption of each radionuclide. The drinking water dose from Columbia River water at the City of Richland pumping station never exceeds  $1 \times 10^{-4}$  mrem/yr in the median-inputs analysis.

Figure L.92 shows the cumulative dose from all Hanford sources and the portion attributed to Hanford solid waste at the City of Richland pumping station. By the end of this analysis (8000 years post closure), the contribution from solid waste is increasing slightly while the cumulative dose from all sources is decreasing, and the overall dose from the three radionuclides is estimated to be less than  $1 \times 10^{-5}$  mrem/yr for the median-inputs case.

## L.4 References

40 CFR 141. "National Primary Drinking Water Regulations." Code of Federal Regulations. Online at: [http://www.access.gpo.gov/nara/cfr/waisidx\\_01/40cfr141\\_01.html](http://www.access.gpo.gov/nara/cfr/waisidx_01/40cfr141_01.html)

42 USC 6901 et seq. Resource Conservation and Recovery Act (RCRA) of 1976. Online at: <http://www4.law.cornell.edu>

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