

**RESPONSE TO REQUEST FOR ADDITIONAL INFORMATION ON THE
DRAFT SECTION 3116 DETERMINATION
IDAHO NUCLEAR TECHNOLOGY AND ENGINEERING CENTER
TANK FARM FACILITY**

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ACRONYMS

ALARA	as low as reasonably achievable
BTP	branch technical position
CA	composite analysis
CFR	Code of Federal Regulations
DOE	U.S. Department of Energy
DQA	data quality assessment
DQO	data quality objective
EDF	engineering design file
EPA	U.S. Environmental Protection Agency
FDF	Flood Diversion Facility
FR	Federal Register
HWMA	Hazardous Waste Management Act
ICPP	Idaho Chemical Processing Plant
INEEL	Idaho National Engineering and Environmental Laboratory
INL	Idaho National Laboratory
INTEC	Idaho Nuclear Technology and Engineering Center
LDUA	light-duty utility arm
NEA	Nuclear Energy Agency
NRC	Nuclear Regulatory Commission
ORIGEN	Oak Ridge Isotope Generation and depletion code
PA	performance assessment
PMF	probable maximum flood
PMP	probable maximum precipitation
ppmv	parts per million by volume

RCRA	Resource Conservation and Recovery Act
SDA	Subsurface Disposal Area
TFF	Tank Farm Facility
VZRP	Vadose Zone Research Park

INTRODUCTION

An Idaho draft 3116 Determination (DOE/NE-ID-11266) was provided to the Nuclear Regulatory Commission (NRC) on September 7, 2005, for review. The NRC issued a request for additional information on January 10, 2006. This document contains comprehensive responses to Request for Addition Information Comments 2, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14, and 15, as well as comprehensive responses to Clarifying Requests 2, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14, 15, 16, 17, 18, 20, and 21. The remaining responses are under development and will be provided at a later date.

NRC

Comment 1:

As recommended in SECY-03-0079 (NRC, 2003), the U.S. Department of Energy-Idaho Operations Office (DOE-ID) should investigate methods for measuring or better estimating the contaminated sand pad radionuclide inventories.

Basis:

Table 7 of the Sampling and Analysis Plan for the Post-Decontamination Characterization of the WM-184, WM-185, and WM-186 Tank Residuals (INEEL, 2003) provides anticipated sample collection points for the tanks as well as the vault sumps. The waste determination includes the following text:

“After cleaning, the water used to flush the vaults was pumped out using the steam jets in vault sumps. The sumps are generally 0.3 m (1 ft) square (some vaults have larger sumps). Review of the sampling data shows that the concentrations of radionuclides in liquid vault samples do not exceed the liquid concentration of the cleaned tanks.” (DOE-ID, 2005, Section 6.3, page 66)

DOE should provide the data and data quality assessment (DQA) used to support the last statement above.

Comparison of the sump samples for the two contaminated sand pads for tanks WM-185 and WM-187 to the sump samples for the other tanks with non-contaminated sand pads could provide useful information to confirm the estimated sand pad inventories presented in the 2003 Performance Assessment (PA) (DOE-ID, 2003b) and the Draft Section 3116 Determination Idaho Nuclear Technology and Engineering Center Tank Farm Facility (DOE-ID, 2005). Results of this sump sampling are not presented or discussed in documents such as the DQA Report for the Post-Decontamination Characterization of the Contents of Tank WM-185 at the Idaho Nuclear Technology and Engineering Center Tank Farm Facility (ICP, 2004e). This type of comparison is of particular importance given the potential significance of the sand pad inventories. For example, post-cleaning inventory estimations show the highest residual tank inventory (WM-182) to have a total of 2,394 Curies (Ci) compared to the contaminated sandpad total inventory estimate of 3,850 Ci (DOE-ID, 2005).

Path Forward:

DOE should provide the sampling data and DQAs for the vault sumps. DOE should provide the technical basis that supports the DOE-ID (2003b) conservative estimation on the contaminated sand pad inventories and describe methods used to measure and confirm these inventories.

Response:

The response to NRC Comment 1 will be provided at a later date.

NRC

Comment 2:

Several tables used to address the waste determination inventory in the Draft 3116 Determination document (DOE-ID, 2005) need further discussion regarding unexpected results.

Basis:

Table A-12 presents a comparison of the post-decontamination estimated inventory to the conservative PA inventory (DOE-ID, 2003b). Although it is noted that the total inventory is much less than that assumed in the PA (DOE-ID, 2003b) calculation, a number of radionuclides listed in Table 1 of 10 CFR 61.55 have inventories that are larger than estimated in the “conservative” inventory of the PA. These radionuclides include Curium (Cm)-242, Np-237, Pu-239, Pu-240, Pu-241, and Pu-242. On the other hand, a greater amount of mass was removed from the tanks for important radionuclides such as Sr-90, as a result of greater than expected mass transfer into the large quantity of flushing water used to clean the tanks. No credit for this removal mechanism (desorption or dissolution) was considered in estimating the inventory for the PA; therefore, the removal is underestimated for some radionuclides.

Based on these results and a review of data provided in Table 1 (page 34) and Table A-12 (page A-68) of the waste determination, it appears that the ratios of the activities of individual radionuclides to ¹³⁷Cs (analytical or Wenzel ratios) are not constant during the cleaning process or congruent between phases. This conclusion is somewhat substantiated in the text of secondary references to a limited extent (e.g., DOE-ID, ICP-2005c) but any significant discussion in the text of the waste determination appears to be lacking. Discussion regarding the potential for preferential treatment of certain radionuclides during the cleaning process, which may be based, in part, on differences in solubility and/or partitioning of the radionuclides, would increase confidence in the results of the solids analytical results. Uncertainties associated with the solids inventory based on ORIGEN2 modeling that may under-predict undissolved fuel activities in the residual heels may also help explain some of these differences.

There is significant uncertainty in the final inventory estimate that is almost entirely based on a solid sample from tank WM-183. Analytical solids results are currently missing from the Data Quality Assessment (INEEL 2004b), although they are presented in the Engineering Design File (EDF) (Portage 2005d) for Tank WM-183. The DQA report states:

“Because decontamination activities reduced the volume of solids remaining in the tank to less than 15% by volume of the total sample collected, the solids portion of the samples collected were not analyzed and compared with the action levels for regulated constituents.”

Regardless of this statement, the EDF presents the solids results; however, several key radionuclides were not analyzed, e.g., H-3, C-14, Ni-63, Np-237, Pu-240, and Pu-241, while uncertain ORIGEN2 ratios were used for all radionuclides that were not analyzed in this sample.

Path Forward:

Discussion regarding the apparent preferential treatment of certain radionuclides, e.g., Sr-90, and the apparent overestimation of removal for other radionuclides, e.g. isotopes of Pu, for the purposes of the PA (DOE-ID, 2003b) modeling is needed to provide confidence in the inventory results which are almost entirely based on one solid sample.

The Data Quality Assessment should be revised to include the solid analytical data to help ascertain if the data quality objectives (DQOs) presented in the sampling analysis plan (SAP) were met to enable use of the data in the decision-making process. Discussion regarding the inability to sample other tanks that contained greater solids mass is needed, e.g. WM-182 is estimated to contain almost twice as much solid mass and comparable maximum heel thickness, as well as discussion regarding the inability to sample several key radionuclides in the WM-183 solid sample.

Discussion regarding the limitations and uncertainties in the ORIGEN2 modeling particularly for estimating solids inventories in the tanks should be brought forward into the waste determination. Provide any information regarding potential mechanisms or sampling/modeling artifacts that may help explain why certain radionuclides appear to be preferentially removed during the tank cleaning process.

Response:

A conservative inventory was developed in 2001 for modeling purposes in the performance assessment (PA) prior to any tank cleaning activities. The current version of the Tank Farm Facility (TFF) PA (DOE-ID 2003a) was issued after NRC review (Essig 2002). That inventory was developed to provide a reasonable bound on radionuclide concentrations that may remain at closure to determine the risks associated with closure. That inventory was not meant to be the best estimate of residual activity; therefore, no attempt was made to account for preferential removal of soluble nuclides during cleaning. However, it was anticipated that soluble nuclides such as ⁹⁰Sr would be significantly reduced during cleaning. This accounts for many of the differences in the draft 3116 Determination, Table A-12 inventories (e.g., the conservative PA inventory compared to the WM-182 at closure inventory).

Fuel assembly inventories can be calculated for any particular fuel, fuel burnup, and cooling time using the Oak Ridge Isotope Generation and depletion code (ORIGEN). For typical pressurized water reactor and boiling water reactor fuels, the precision of the calculated inventories is more than adequate for use in risk analyses (NRC 2001).

Because of the precision of ORIGEN calculations, variations in the amounts of specific radionuclides in the inventory can be reliably calculated for any fuel cooling period and any fuel burnup (the small errors associated with the inventories calculated for very high burnup fuels are not significant). Because ORIGEN calculations can provide a precise inventory for any spent fuel, no research in this area is warranted. Conversely, because fuel type, burnup, and cooling period can vary widely, generic spent fuel risk assessments should probably perform risk calculations using inventories and decay heat loads for

both bounding and average pressurized water reactor and boiling water reactor spent fuels (NRC 2001).

The ORIGEN model includes a radionuclide library with approximately 1,700 entries collected into three groups: activation products, transuranics, and fission products. Nondestructive gamma-ray analyses for fission products and mass-spectrometric analyses have been compared with corresponding quantities calculated from reactor operating data using the ORIGEN model to form the basis for material balance calculations. Satisfactory agreement between ORIGEN results and measured uranium isotopic and fission product distributions was found using the ORIGEN inputs (NRC 1977).

The uncertainty of ORIGEN/Wenzel ratios in the TFF tank inventory is difficult to quantify. However, the ratios have very little impact on the inventory because they are used for less than 2% of the radioactivity in the tanks. The radionuclides affected by the use of these ratios are also not significant to the assessment of performance or waste classifications. As explained below, analytical data have been collected for all radionuclides that are significant contributors to the PA (e.g., ⁹⁹Tc and ¹²⁹I) or to the sum of the fractions calculations.

Discussion regarding the apparent preferential treatment of certain radionuclides, e.g., Sr-90, and the apparent overestimation of removal for other radionuclides, e.g. isotopes of Pu, for the purposes of the PA (DOE-ID, 2003b) modeling is needed to provide confidence in the inventory results which are almost entirely based on one solid sample.

To ensure conservatism in the PA model, individual removal efficiencies of radionuclides were not predicted for the tank cleaning process. Tank cleaning was modeled as a bulk cleaning tool. While it was known some radionuclides would be preferentially removed based on solubility (e.g., ⁹⁰Sr removal), these types of calculations were never introduced into the PA (DOE-ID 2003a).

As described below, it is clear that some soluble radionuclides are overestimated using ORIGEN/Wenzel ratios. At the time the PA was prepared, the course taken was not to reduce the inventory of soluble radionuclides because it was not clear how much preferential removal would occur. The goal was to maintain a conservative inventory.

What would appear to be an overestimation of cleaning efficiency can be attributed to other causes, since preferential removal was never assumed. The factor most likely to explain the difference in insoluble radionuclides between the PA inventory and the data for cleaned tanks can be attributed to increased analytical accuracy in the samples collected from the cleaned tanks. Sample matrix effects in concentrated samples are well-documented (EPA 1979; NRC 2004). The relatively high concentrations of ¹³⁷Cs in the samples make it difficult to quantify radionuclides in low concentration and/or with low photon energies.

Since development of the draft 3116 Determination, an additional set of solids data has become available. In 2005, after cleaning and sampling of WM-183 were completed, a few hundred gallons of tank waste that were being transferred from WM-180 to WM-187 inadvertently leaked into WM-183 through a partially open inlet valve. Final sample results were not available when the draft 3116 Determination was prepared. Results of this second sample from WM-183 are consistent with data obtained for the 2003 solids sample obtained from the same tank (PEI-EDF-1018, 2006). This is further discussed in the response to NRC Comment 17. Data available for solids associated with the TFF tanks are presented in Table RAI-2-1. These data provide support to the use of the limited analytical solids data that are available.

While the inventory in the draft 3116 Determination is based on one sample, the data from samples collected prior to cleaning were used as a foundation for the inventory. The data from all sampling events prior to cleaning, after cleaning, and the sample collected in 2005 are shown in Table RAI-2-1. The radionuclides have all been decayed to 2012. The tank solids data, as shown in Table RAI-2-1 and discussed in Section 2.4.2 of the draft 3116 Determination, are considered to be representative of solids from cleaned tanks. The information obtained from these various sampling activities provides confidence in the inventory presented in the draft 3116 Determination. Insoluble nuclides including ²⁴¹Am, ⁶⁰Co, ²³⁷Np, ²³⁸Pu, ²³⁹Pu, and ⁶³Ni are fairly consistent among all of the samples included in Table RAI-2-1. The ¹³⁷Cs concentrations in WM-183 were essentially unchanged from pre- to post-wash; however, soluble nuclides ⁹⁰Sr and ⁹⁹Tc show a significant decrease in the post-cleaning data as would be expected when using large quantities of water. The ¹³⁷Cs concentrations in Tank WM-188 samples are noticeably higher due to the tank's operational history (Loos 2004), as described in Section 2.2 of the draft 3116 Determination. The data shown in Table RAI-2-1 represent those radionuclides that contribute greatest to the PA dose and/or calculation of the sum of the fractions.

Table RAI-2-1. Tank Farm Facility tank solids data (DOE-ID 2003a; PEI-EDF-1018, 2006).

Radionuclide (Ci/kg) ^a	WM-183 Post-Cleaning Solids Data (2005)	WM-183 Post-Cleaning Solids Data	WM-182 Pre-Wash Solids Data	WM-183 Pre-Wash Solids Data	WM-188 Pre-Wash Solids Data	WM-188 Pre-Wash Solids Data	WM-188 Pre-Wash Solids Data	WM-188 Pre-Wash Solids Data
²⁴¹ Am	3.33E-04	3.34E-04	8.31E-04	2.39E-04	1.48E-04	2.10E-04	2.59E-04	ND
⁶⁰ Co	5.94E-05	5.72E-05	2.79E-05	3.79E-05	1.14E-04	ND	ND	ND
¹³⁷ Cs	6.33E-01	9.23E-01	3.30E-01	6.81E-01	9.26E-01	1.97E+00	1.59E+00	2.73E+00
¹⁵⁴ Eu	5.16E-05	3.20E-05	9.34E-05	2.77E-04	ND	ND	ND	ND
³ H	NA	NA	6.41E-06	2.07E-05	ND	ND	ND	ND
²³⁷ Np	1.01E-05	ND	1.66E-06	1.76E-06	4.68E-06	2.24E-06	1.62E-06	ND
²³⁸ Pu	9.17E-03	9.15E-03	1.77E-02	3.60E-03	6.24E-03	8.22E-03	6.44E-03	ND
²³⁹ Pu	3.17E-03	2.75E-03	1.47E-03	1.25E-03	3.32E-04	5.27E-04	4.30E-04	ND
⁹⁰ Sr	1.28E-02	1.87E-02	1.78E-01	1.41E-01	3.62E+00	5.82E+00	2.53E+00	ND
⁹⁹ Tc	1.10E-04	6.17E-04	2.63E-03	ND	5.32E-03	3.76E-03	4.41E-03	ND
²³⁴ U	NA	2.98E-06	ND	3.38E-06	ND	ND	ND	ND
¹²⁹ I	8.44E-07	6.24E-07	ND	ND	ND	ND	ND	ND

Table RAI-2-1. (continued).

Radionuclide (Ci/kg) ^a	WM-183 Post-Cleaning Solids Data (2005)	WM-183 Post-Cleaning Solids Data	WM-182 Pre-Wash Solids Data	WM-183 Pre-Wash Solids Data	WM-188 Pre-Wash Solids Data	WM-188 Pre-Wash Solids Data	WM-188 Pre-Wash Solids Data	WM-188 Pre-Wash Solids Data
¹⁴ C	2.15E-05	NA	NA	NA	NA	NA	NA	NA
⁶³ N	1.87E-04	NA	4.14E-05	1.60E-04	NA	NA	NA	NA

NA = Not analyzed.

ND = Not detected.

a. Radionuclides decayed to 2012.

The Data Quality Assessment should be revised to include the solid analytical data to help ascertain if the data quality objectives (DQOs) presented in the sampling analysis plan (SAP) were met to enable use of the data in the decision-making process. Discussion regarding the inability to sample other tanks that contained greater solids mass is needed, e.g. WM-182 is estimated to contain almost twice as much solid mass and comparable maximum heel thickness, as well as discussion regarding the inability to sample several key radionuclides in the WM-183 solid sample.

The data quality assessment (DQA) process involves performing a statistical evaluation of a set of samples collected from a tank or system. The DQAs were prepared for each cleaned tank to provide confidence that data quality objectives were met. Since only one solid sample was initially obtained, it was not possible to perform any meaningful statistical evaluation. Even with the collection of the second sample, it is not possible to perform meaningful statistical analyses using sample populations. Therefore, there is no DQA prepared for solid samples. The data related to the solid samples are provided in an engineering design file (PEI-EDF-1018, 2006).

The sampling and analysis plan (INEEL 2002) prepared for the Hazardous Waste Management Act (HWMA)/Resource Conservation and Recovery Act (RCRA) closure plan (DOE-ID 2003b) specified analysis of solids if they exceeded 15% by volume of the total sample collected. This criterion was not met for any of the tank sampling activities. Only the samples collected from WM-183 contained any visible solids. All of the WM-183 liquid samples were filtered and any solids collected were composited into a single solid sample. The solid samples collected were less than a few grams and did not meet the 15% by volume criterion. Although not required by the HWMA/RCRA closure plans, analysis of the solid samples for radionuclides determined by the PA to be risk drivers was needed to support radioactive waste classification. Since the sample sizes would only allow limited analyses, radionuclide analyses requested were focused on highly radioactive radionuclides. Analysis for gamma-emitting radionuclides, ⁹⁰Sr, ⁹⁹Tc, ¹²⁹I, and alpha spectroscopy was performed on both samples. Analysis was performed for ⁶³Ni and ¹⁴C in the second sample. The radionuclide ²³⁷Np was analyzed for in both samples but was only detected in the second sample. The radionuclides ²³⁹Pu/²⁴⁰Pu cannot be analyzed for separately; however, inventories were developed for each isotope. Analysis was not performed for ³H and ²⁴¹Pu because they are not significant contributors to dose and adequate sample volume was not available.

Complications associated with analysis of solids included the need for laboratory method development and sample cleanup to reduce interference from other nuclides and metals for analysis of ^{99}Tc and ^{129}I . Method development and analysis consumed almost 2 years.

The successes gained from the effectiveness of the tank cleaning system provided unique challenges in gathering enough residual solid material to estimate a final source term. Sampling of the tanks is a difficult process. Only a thin slice of flocculent solids is present in an uneven distribution in any of the cleaned tanks. Additionally, proximity of solids to the available risers is an important consideration. If solids are not directly below the risers, there are difficulties in collecting solid samples. Prior to tank cleaning activities, a robotic arm (light-duty utility arm [LDUA]) was deployed in Tanks WM-182, WM-183, and WM-188 to gather solids data. Although some success was achieved with the LDUA, it proved to be too costly to maintain, deploy, and operate. The first sampling method used for post-cleaning of Tank WM-182 involved using a simple sampler. This is an evacuated canister of about 1 L in volume. The sampler was lowered into the tank until the tank bottom was reached. The sampler was filled and then retrieved to the surface. This sampler was used because it was an improvement over the LDUA and it met the existing requirements to maintain radiation doses as low as reasonably achievable (ALARA). A total of approximately 27 L of liquid samples was necessary for the analytical parameters specified in the sampling and analysis plan, including those for metals and organics. While the volume of the simple sampler was greater than the LDUA to accommodate the large volume of samples required, it was thought small enough to avoid radiation control issues. The concern was that if a large volume of samples was brought to the surface, it would have significant radiation fields and violate specifications for the field work and the laboratory.

When the WM-182 samples were collected it became apparent that radiation doses were sufficient to maintain ALARA; however, a more efficient method was needed that would reduce the time to collect samples. Therefore, a small submersible pump was deployed for the remaining tank sampling activities. When this method was used in Tank WM-183, it was able to collect less than 3 g of sample as a composite of all sampling. A total of 135 L was collected and, from that, less than 3 g were removed by centrifuge. The advantage of the submersible pump is that it can be moved along the bottom of the tank during the collection of the sample, allowing more surface area of the tank bottom to be reached. Even though this sampling method was used on Tank WM-185, which contained approximately the same amount of solids as WM-183, no solids were collected. However, it must be noted that the majority of residuals in WM-185 are near the periphery of the tank (PEI-EDF-1010, 2005; PEI-EDF-1012, 2005).

Consideration was given to deploying the submersible pump in WM-182 in an attempt to gather additional solids data; however, while additional data are preferable, it is not certain that using the submersible pump to sample WM-182 would yield any solids. It was not considered practical to use the pump to resample WM-182 due to the uncertainty that solids could be obtained based on

the location of the solids relative to the location of the risers. In addition, the cost of setup and analysis would approach \$1 million.

Discussion regarding the limitations and uncertainties in the ORIGEN2 modeling particularly for estimating solids inventories in the tanks should be brought forward into the waste determination. Provide any information regarding potential mechanisms or sampling/modeling artifacts that may help explain why certain radionuclides appear to be preferentially removed during the tank cleaning process.

The ORIGEN2 model is used to predict radionuclide inventories based on nuclear fuel types (e.g., Al, Zr, and stainless steel) that have been processed at the Idaho Nuclear Technology and Engineering Center. Input parameters to the model are adjusted to allow the model to predict sodium-bearing waste (SBW) radionuclide inventories in operational tanks based on weighted averages of the different fuel types that have been processed. The Idaho National Laboratory (INL) uses radionuclide and ^{137}Cs values from the ORIGEN2 model to calculate a radionuclide-to- ^{137}Cs ratio called the ORIGEN2 ratio for each radionuclide. Analytical results from SBW samples that contain reliable data for a specific radionuclide and ^{137}Cs are used to calculate an analytical ratio. All reliable analytical data are used to generate average analytical ratios for radionuclides in SBW. These two ratios, the ORIGEN2 and analytical ratios, have been modeled and published and are informally called the Wenzel tables (Wenzel 1997). These tables have been integrated into a FORTRAN program, which is used to calculate radionuclide data (when analytical data are lacking) for SBW based on ^{137}Cs sample results. The latest compilation of the ORIGEN/Wenzel ratios is found in EDF-CRPD-001 (2005).

The PA used ORIGEN/Wenzel ratios from EDF-CPP-97080 (Wenzel 1997) and EDF-CRPD-001 (2002). The PA was being developed prior to the release of the 2002 update of the ORIGEN/Wenzel ratios. Two radionuclides that were important to the groundwater radiation dose (^{129}I and ^{99}Tc) were reviewed (Swenson 2002) in response to the NRC request for additional information in 2002 (Essig 2002). Therefore, the PA used the 1997 ratios, with the exception of ^{129}I and ^{99}Tc . There are differences between the 1997 and 2005 revisions of the ORIGEN/Wenzel ratios. The obvious difference for the 1997 and 2005 versions is the decay date. The 1997 version did not provide data decayed to 2012, while the 2005 version did provide data decayed to 2012. Early in 2005, the U.S. Department of Energy (DOE) determined that closure of the TFF would be complete by 2012, thus changing the decay date for analysis of closure. The change of the closure date affects decay and ingrowth calculations for radionuclides.

As described in “Relative Inventories of Reactor-Produced Species in INTEC Waste Types” (EDF-CRPD-001, 2005), rigorous uncertainty analyses have not been done for the ORIGEN/Wenzel ratios; however, Wenzel has generally estimated the ratios to have an accuracy of a \pm factor of 2.

In addition to the change in decay and ingrowth, some other changes were made to the assumptions for the ORIGEN/Wenzel ratios in the 1997 and 2005 revisions. Calculations for SBW did not include structural materials and activation products other than ^{55}Fe and ^{63}Ni . Structural materials of spent fuel rods were included in all ratios after 1997. The activation products of structural material added a number of radionuclides and altered the concentrations of other radionuclides.

A review of data collected from SBW was conducted in 2004 (Swenson 2004) that resulted in changes to the SBW ORIGEN/Wenzel ratios. As historical data were reviewed, some data points were corrected or excluded because they were not valid. The data review and inclusion of structural materials introduced 11 radionuclides that were previously not in the data and changed the estimated concentrations of the radionuclides.

The ORIGEN/Wenzel ratios are generally used to predict the radionuclide concentrations of SBW liquids. As described in the *Feed Composition for the Sodium-Bearing Waste Treatment Process* (Barnes et al. 2004), the ORIGEN/Wenzel ratios were used to estimate radionuclide concentrations of SBW solids. The PA used the ORIGEN/Wenzel ratios to estimate SBW solids remaining in tanks as described below. (The PA assumed a heel of SBW remained in the tanks after cleaning.) The draft 3116 Determination used the ORIGEN/Wenzel ratios to estimate the solid radionuclides remaining in the tanks. The use of the ORIGEN/Wenzel ratios to estimate residual solids after cleaning is an additional extrapolation of the data.

The use of the ORIGEN/Wenzel ratios for residual solids is examined below in context of the ultimate effect the ratios have on the inventory and what information is known about radionuclide concentrations.

The inventory for tank residual solids uses a combination of analytical data and ORIGEN/Wenzel ratios. As described in PEI-EDF-1009 (2005), the inventory for Tank WM-182 contains 2,394 Ci, of which less than 2% or 46 Ci are calculated from ORIGEN/Wenzel ratios. The other 98% of this inventory has been developed based upon analytical results. The radionuclides calculated using the ORIGEN/Wenzel ratios are not significant to performance assessment or calculation of the sum of the fractions. Of the 46 Ci, ^{241}Pu is 42% and ^{151}Sm is 37%. The radionuclide ^{151}Sm has a half-life of 90 years and decays to a stable isotope. It does not contribute to radiation dose or calculation of the sum of the fractions. The radionuclide ^{241}Pu has a half-life of 14.4 years and decays to ^{241}Am . Radionuclides ^{241}Pu and ^{241}Am are not significant contributors to the tank inventory for calculation of the sum of the fractions and ^{241}Am is a not significant radiation dose contributor (contributes only slightly in the intruder scenarios). The radionuclide ^{241}Pu does not contribute significantly to radiation in any scenario.

In summary, the effect on the PA results from radionuclides calculated using the ORIGEN/Wenzel ratios is minimal. The radionuclides, which are important to

radiation dose or calculation of the sum of the fractions, have been measured by analytical laboratory techniques.

The concentrations of radionuclides that have been detected in residual solids and SBW are shown in Table RAI-2-1. The ORIGEN/Wenzel ratios are compared to the concentration of samples collected from solid residuals in Table RAI-2-2 (WM-183 post-cleaning data [2005]). The ratios, without normalizing the ^{137}Cs concentration, predict higher concentrations than found in the laboratory data collected in 2005, with the exception of ^{129}I and ^{239}Pu . Therefore, Table RAI-2-2 shows the predicted values compared to the analytical data. The ORIGEN/Wenzel ratios for ^{137}Cs are always set to 1 so the actual concentration of ^{137}Cs can be used to normalize the remaining radionuclides.

Negative percentages indicate that the analytical data concentrations are less than those predicted by the ORIGEN/Wenzel ratios. Positive percentages indicate that the ORIGEN/Wenzel ratio concentrations are greater than the analytical values.

Table RAI-2-2. Comparison of predicted values (2005) to laboratory analysis (2005).

Radionuclide (Ci/kg) ^a	WM-183 Post-Cleaning Solids		Comparison of Values
	Data (2005)	ORIGEN Ratios (2005)	
^{241}Am	3.33E-04	2.44E-03	-86%
^{60}Co	5.94E-05	3.70E-04	-84%
^{154}Eu	5.16E-05	1.75E-03	-97%
^3H	NA	3.22E-04	NA
^{237}Np	1.01E-05	2.11E-05	-52%
^{238}Pu	9.17E-03	2.07E-02	-56%
^{239}Pu	3.17E-03	2.22E-03	+43%
^{90}Sr	1.28E-02	8.88E-01	-99%
^{99}Tc	1.10E-04	4.99E-04	-78%
^{234}U	NA	3.37E-05	NA
^{129}I	8.44E-07	5.88E-07	+44%
^{63}N	1.87E-04	1.28E-03	-85%

NA = Not analyzed.

ND = Not detected.

a. Radionuclides decayed to 2012.

Table RAI-2-3 shows the predicted values compared to the residual solids data collected from Tank WM-183 in 2003. As shown in the table ^{239}Pu , ^{99}Tc , and ^{129}I predicted values are less than the laboratory data. Negative percentages indicate that the analytical data concentrations are less than those predicted by the ORIGEN/Wenzel ratios. Positive percentages indicate that the ORIGEN/Wenzel ratio concentrations are greater than the analytical values.

The predicted values are quite comparable and within the factor of 2 for most radionuclides. The radionuclide that has been easily and repeatedly measured at TFF is ^{90}Sr , which is considerably lower than the predicted value. This low

concentration is because ⁹⁰Sr is mobile and has been removed by the tank cleaning process. This is the result that would be expected.

Other mobile radionuclides such as ⁹⁹Tc and ¹²⁹I are actually in greater concentrations than predicted. This would appear to be an anomaly. However, solid or liquid samples for ⁹⁹Tc and ¹²⁹I have rarely been analyzed at the TFF. Since data for ⁹⁹Tc and ¹²⁹I are rare, it is not surprising that the concentrations predicted in the ORIGEN/Wenzel ratios were slightly lower than those measured by laboratory analysis. The fact that the values are only slightly lower than laboratory data and within a factor of 2 (qualitative uncertainty of ORIGEN/Wenzel ratios) lends support to using the ORIGEN/Wenzel ratios.

If the ¹³⁷Cs concentrations used for the PA (1.65 Ci/kg) and the draft 3116 Determination (1.8 Ci/kg) are used to normalize the predicted concentrations, all of the predicted concentrations are greater than the laboratory analysis. (Note: The difference in the ¹³⁷Cs concentrations is the difference in decay from 2016 to 2012.)

Table RAI-2-3. Comparison of predicted values (2005) to laboratory analysis (2003).

Radionuclide (Ci/kg) ^a	WM-183 Post-Cleaning Solids Data (2003)	ORIGEN Ratios (2005)	Comparison of Values
²⁴¹ Am	3.34E-04	2.44E-03	-86%
⁶⁰ Co	5.72E-05	3.70E-04	-85%
¹⁵⁴ Eu	3.20E-05	1.75E-03	-98%
³ H	NA	3.22E-04	NA
²³⁷ Np	ND	2.11E-05	ND
²³⁸ Pu	9.15E-03	2.07E-02	-56%
²³⁹ Pu	2.75E-03	2.22E-03	+24%
⁹⁰ Sr	1.87E-02	8.88E-01	-98%
⁹⁹ Tc	6.17E-04	4.99E-04	+24%
²³⁴ U	2.98E-06	3.37E-05	-91%
¹²⁹ I	6.24E-07	5.88E-07	+6%
⁶³ N	NA	1.28E-03	NA

NA = Not analyzed.

ND = Not detected.

a. Radionuclides decayed to 2012.

Comparison of the ORIGEN/Wenzel ratios from 1997 and 2005 also provides insight into why it appears there is larger uncertainty than is observed. The ratios established are not the same for the reasons described above. In Table A-12 of the draft 3116 Determination, the total inventory for some radionuclides in Tank WM-182 inventory increases despite an approximate factor of 10 decreases in the mass of residual solids and a large reduction in liquid concentrations. The reason for this change is primarily due to changes in the ORIGEN/Wenzel ratios from 1997 to 2005.

Table RAI-2-4 shows the ORIGEN/Wenzel ratios from 1997 and 2005 without using ¹³⁷Cs to modify the results. In addition, the table shows percentages of 1997 values to 2005 values. Negative percentages indicate that the 2005 ORIGEN/Wenzel ratios are greater than the 1997 ORIGEN/Wenzel ratios. Positive percentages indicate that the 2005 ORIGEN/Wenzel ratios are less than the 1997 ORIGEN/Wenzel ratios. This table shows the radionuclides detected by laboratory analysis but can be used to compare the 1997 to 2005 ratios. Radioactive decay to a common date is not included in these values. Some of the 1997 values are lower, while some are higher than predicted in 2005.

Table RAI-2-4. Comparison of ORIGEN/Wenzel ratios from 1997 and 2005.

Radionuclide	2005 Ratios	1997 Ratios	Comparison of Values
²⁴¹ Am	2.44E-03	1.40E-03	-43%
⁶⁰ Co	3.70E-04	1.80E-04	-51%
¹⁵⁴ Eu	1.75E-03	7.10E-04	-59%
³ H	3.22E-04	1.40E-04	-56%
²³⁷ Np	2.11E-05	7.80E-05	+270%
²³⁸ Pu	2.07E-02	9.10E-03	-56%
²³⁹ Pu	2.22E-03	1.50E-03	-33%
⁹⁰ Sr	8.88E-01	1.00E+00	+13%
⁹⁹ Tc	4.99E-04	2.60E-04	-48%
²³⁴ U	3.37E-05	2.20E-05	-35%
¹²⁹ I	5.88E-07	1.40E-06	+138%
⁶³ N	1.28E-03	7.60E-04	-41%

Conclusions

The inventory for tank residual solids uses a combination of analytical data and ORIGEN/Wenzel ratios. As described in PEI-EDF-1009 (2005), the inventory for Tank WM-182 contains 2,394 Ci, of which less than 2% or 46 Ci are calculated from ORIGEN/Wenzel ratios. The other 98% of this inventory has been developed based upon analytical results. The radionuclides calculated using the ORIGEN/Wenzel ratios are not significant to performance assessment or calculation of the sum of the fractions.

In summary, the effect on the PA results from radionuclides calculated using the ORIGEN/Wenzel ratios is minimal. The radionuclides, which are important to radiation dose or calculation of the sum of the fractions, have been measured by analytical laboratory techniques.

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NRC

Comment 3:

Additional justification for the initial inventory for the sand pad and vault sumps appears warranted in light of the fact that no sampling data has been provided.

Basis:

ORIGEN2 modeling to estimate the inventory for WM-181 which stored SBW was used to estimate inventories for the sand pads underneath tanks WM-185 and WM-187 that stored aluminum and zirconium clad fuel reprocessing waste. Sampling data from the February 14, 1962, pre-event sampling, and samples taken from the vault during siphoning are provided in Latchum et al. (1962) but are very limited. The results of the WM-181 tank inventory estimates were calibrated to WM-185 and WM-187 analytical results just prior to the event; however, the uncertainty associated with the inventory calculation is not clear. The performance assessment (DOE-ID, 2003b) documents calculation of initial sand pad inventories based on a diffusion model. This model was thought to be appropriate to estimate the initial inventory in the sand pad because the sand pad was saturated just prior to the event and no hydraulic gradient existed between the vault sump and sand pad. However, event documentation (Latchum et al., 1962) hypothesizes that drainage of water from the sand pad into the sump following vault sump pumping may have helped initiate the event. Additionally, according to Latchum et al., the maximum level in the annular region of the tank following the event was approximately 3.5 feet, which is well above the height of the sump (1 foot). Under these circumstances it would appear that the hydraulic head in the annulus would lead to radial, advective flow of contaminated water into the sand pad. Additionally, even if the sand pad was saturated prior to the event and diffusion was the most important transport mechanism for mass transfer of contaminants into the sand pad, the appropriateness of a one-dimensional (1-D) diffusion model is questionable. If first-cycle extraction fluids filled the vault above the sand pad, diffusive mass transport from both above the sand pad and through the sand pad/curb interface should have been considered. An illustration showing the initial and boundary conditions for the diffusion model presented in equation 2-22 on page 2-69 of the PA is needed (DOE-ID, 2003b). There are significant inconsistencies or unexplained base case modeling assumptions that require further clarification. As it is argued that the vaults cannot be sampled to determine the amount of residual inventory in the sand pads because they also contain diverted contamination from the valve box sumps, how was diffusion of contamination between the vault sump and the sand pad considered in the modeling of the inventory in the sand pad? Is contamination in the vault sump thought to be higher or lower than contamination in the sand pad? How is the inventory in the sumps considered in the performance assessment?

Path Forward:

Provide all of the analytical data available to substantiate the initial starting concentrations for radionuclides present in the first-cycle extraction fluid that entered the vault during the event. Provide information regarding the uncertainty of the estimated starting concentrations based on ORIGEN2 modeling. Intermediate outputs from the sand pad inventory calculations are needed, such as data or figures showing the initial inventory in the sand pad after the event, and the change in inventory over time as a result of flushing events.

Additional justification for use of a one-dimensional diffusion model to estimate the base case sandpad inventory also appears warranted in light of the information reported in Latchum et al.(1962) surrounding the contamination event. DOE-ID should provide a cross-sectional diagram of the vault, tank, sandpad, curb, curb drain holes, and sump to allow review of the applicability of the diffusion model presented in the PA, along with an illustration showing the initial and boundary conditions for the 1-D diffusion model. Full explanation regarding the estimated water-level in the vault and saturation of the sand pad prior to, during, and immediately following the event should be provided. Finally, justification regarding the lack of consideration of vault sump contamination from valve box and piping encasement drainage is necessary.

Response:

The response to NRC Comment 3 will be provided at a later date.

NRC

Comment 4: Sensitivity of model results to choice of sand pad K_d values was not discussed.

Basis: For sorption coefficients in the sand pad, which affect both inventory and transport calculations, DOE used default sand soil values from Sheppard and Thibault (1990). No justification for this choice, nor discussion of uncertainties, was provided. Conservatism is not simply defined for these choices; for example, a lower K_d is conservative for transport but not for inventory. In the absence of site-specific data, use of default literature values needs to be justified. Sorption characteristics of the relatively clean sand likely used for the pad may differ substantially from the sand solids included in the Sheppard and Thibault compilation, which may contain up to 30 percent non-sand particles. Thus, the sand pad (if it is indeed made from clean, well-sorted sand) may have lower K_d values than sand soils.

Path Forward: Technical bases should be provided that will build confidence in the calculation of the sand pad inventory used in the performance assessment (PA), as this inventory is estimated and does not use sampling data to build confidence that a bounding inventory has been used. Provide a discussion justifying the choice of sand pad K_d values, taking into consideration the competing conservatisms inherent in the two model uses of the parameters.

Response: The response to NRC Comment 4 will be provided at a later date.

NRC

Comment 5:

Additional information regarding Criterion 2 evaluation is needed.

Additional information regarding analysis of tank cleaning effectiveness, estimation of residual solid volume, and remaining cleaning activities for tanks, vault sumps, piping encasements, valve boxes, and process waste lines is needed.

Basis:

Criterion 2 of the National Defense Authorization Act (NDAA), *"has had highly radioactive radionuclides removed to the maximum extent practical,"* is an essential part of the NDAA. The criteria used to demonstrate removal of highly radioactive radionuclides to the maximum extent practical, taking into consideration the costs and benefits to public health and safety and the environment, plays a key role in the waste incidental to reprocessing determination process. Section 5.2 of the draft Section 3116 determination, page 50-51, defines the maximum extent practical approach as *"not removal to the extent 'practical' or theoretically 'possible'."* Instead, it will be based on *"exercising expert judgement"*. Expert judgement is often necessary and has an important role in waste determination processes. However, the evaluation of Criterion 2 should also include appropriate analyses that consider available waste removal technologies; technology selection, implementation, and removal effectiveness; and consideration of the costs and benefits of attempting additional removal of highly radioactive radionuclides.

An exponential decrease in concentration in the waste stream has been used to demonstrate that Criterion 2 of the NDAA has been met; however, achievement of an asymptote in the effluent concentration doesn't necessarily show that continued cleaning activities would be ineffectual. The waste determination states the following:

"When radiation levels decrease to the lowest value and remains [sic] constant, cleaning is stopped and the tanks are inspected. If visual inspection via a remote-controlled camera confirms that the tank has been cleaned to the extent practical [sic], then samples are collected and analyzed to verify performance objectives are met." (DOE-ID 2005, page 25)

There may be areas of the tanks where additional solids removal can be accomplished but the waste stream detector is only indicating diluted aqueous activities in areas of the tank that have already been cleaned. As stated in the waste determination, use of video inspection during cleaning, as well as observance of a plateau in the activity in the rinse water, needs to be used in conjunction to determine when cleaning to the maximum extent practical with the current technology has been achieved. However, the statement above is rather ambiguous, and the actual process used to determine that cleaning has progressed to the "maximum extent practical" is not clear. It is not clear how verification that performance objectives have been met via sampling is accomplished (e.g., are action levels set and if so, what are these action levels based on), or if sampling and cleaning is an iterative process. If interim sampling data exists, this data should be provided to help estimate removal efficiencies. Nonetheless,

achievement of performance objectives, in itself, may not obviate the need for further cleaning.

Review of the kriged surfaces in the EDFs for each tank (Portage, 2005a; Portage, 2005b; Portage, 2005c; Portage, 2005d; Portage, 2005e; Portage, 2005f; Portage, 2005g), reveals various patterns of residual contamination in cleaned tanks. The surface for WM-182 was mistakenly included in the EDF for WM-181 (Portage, 2005b). Tanks that did not contain cooling coils appear to have less residual contamination. Some tanks appear to have a ring around the outside of the tank (e.g. WM-185), while most tanks have certain areas where cleaning appears to be less effective. Please comment on the difficulty or inability to slurry remaining solids materials to jet and airlift pumps in tanks such as WM-182, which contains significantly higher residual radioactivity than other cleaned tanks.

The modeling assumptions related to the estimation of total solids volume remaining in the tanks requires further clarification. The uncertainty associated with the estimated solid volume needs to be quantified, e.g. how accurate is visual examination of reflective surfaces and reference points to solids levels in determining the measurements on this scale? More detailed information regarding the interpolation approach, e.g. grid spacing and density of reference points, is needed. Tanks with no cooling coils have a much larger density of interpolation points in some cases. Every EDF states that cooling coil brackets were used as reference points, although not all tanks have cooling coils. Please clarify these statements.

In Section 5.2.1, page 54, and Table 6 (DOE-ID, 2005), DOE uses the total production of radionuclides generated at INTEC during its operations to determine the percentages of radionuclides removed from the TFF. While it is factual that DOE did remove much of the key radioactivity from the waste stream derived from the processing of spent nuclear fuel, use of these percentages may exaggerate the effectiveness of treatment and cleaning activities. Comparing radionuclide activities present in the tanks prior to cleaning versus the activity remaining after cleaning would be useful.

Remaining cleaning activities for piping encasements, valve boxes, and vault sumps is not clear. Information regarding decontamination fluids to be used and the sequential cleaning and sampling steps left in the closure process is needed.

Provide the basis for the assumption that remaining tanks, particularly tank WM-187 that was used to collect SBW waste from other tanks undergoing cleaning operations, will be cleaned as effectively as the first seven tanks. The estimated costs associated with final clean-up activities should be provided.

DOE-ID should explicitly state whether any new technologies to clean the sand pads or tanks have been identified since the last waste determination. Provide any additional information regarding the costs of deployment of new technologies.

Path Forward:

Establish criteria that will be consistently implemented throughout the cleaning process for tanks, vaults, associated piping, and ancillary equipment. Although the final inventory status for each component after cleaning can be different, the approach should be consistent in terms of the cost/benefit analysis and the criteria employed in determining that radionuclides have, indeed, been removed to the maximum extent practical. Provide more specific details regarding the iterative process used to evaluate effectiveness of the current cleaning technology, including use of visual tools, sampling, estimation of radionuclide removal efficiencies, and any action levels used in the decision-making process used to terminate cleaning operations. Additionally, provide an evaluation of the costs and benefits of attempting additional removal of highly radioactive radionuclides.

Provide discussion regarding cleaning difficulties due to geometry or other physical limitations that make it more or less difficult to clean individual tanks. Discuss any changes in operational parameters during the cleaning process to help facilitate heel removal.

Provide the surface plots for WM-181.

Provide clarifying information regarding the data input and interpolation parameters for kriging; and the uncertainty associated with the solid surfaces used to estimate the residual solid volume remaining in the tanks.

Provide tables of percent removal based on inventory of heel prior to cleaning and after cleaning.

Discuss any remaining cleaning and sampling activities for piping encasements, valve boxes, and vault sumps. Provide additional justification regarding the expected removal efficiencies for remaining tanks to be cleaned and estimated costs for final cleaning.

Discuss the evaluation of any new technologies to remove waste from the sand pads or tanks. Please discuss whether any technologies were considered to stabilize the sand pads without removing the sand, e.g., in-situ grout injection?

Response:

Establish criteria that will be consistently implemented throughout the cleaning process for tanks, vaults, associated piping, and ancillary equipment. Although the final inventory status for each component after cleaning can be different, the approach should be consistent in terms of the cost/benefit analysis and the criteria employed in determining that radionuclides have, indeed, been removed to the maximum extent practical. Provide more specific details regarding the iterative process used to evaluate effectiveness of the current cleaning technology, including use of visual tools, sampling, estimation of radionuclide removal efficiencies, and any action levels used in the decision-making process used to terminate cleaning operations. Additionally, provide an evaluation of the costs and benefits of attempting additional removal of highly radioactive radionuclides.

Establishment of Cleaning Criteria

The following discussion describes the development of cleaning criteria for the TFF components, which was performed prior to the initiation of cleaning activities. A summary of analysis for selecting tank cleaning technologies is presented in Appendix B of the draft 3116 Determination. The next section of the response describes specific details regarding the iterative process used to evaluate the effectiveness of the current cleaning technology.

300,000-gal Tanks

A consistent goal was established to determine what an acceptable level of cleaning was for each TFF component. This goal was to remove radionuclides to the maximum extent practical, while (1) meeting “action levels” for specific hazardous constituents of concern under the RCRA^a and (2) demonstrating that any remaining residual waste would meet performance objectives consistent with those found in 10 Code of Federal Regulations (CFR) 61.55.

The RCRA actions levels were established through State of Idaho-issued partial closure plans, and included concentrations of organic and inorganic hazardous waste constituents to be met to achieve a RCRA performance-based “clean closure” of the TFF. The HWMA/RCRA closure plans for each tank or set of tanks to be closed included action levels for approximately 50 different constituents of concern.

A PA (DOE-ID 2003) was developed prior to any tank cleaning experience as part of DOE’s tank closure plans, which assumed, for conservative modeling purposes only, limited tank cleaning success. Cleaning of the 300,000-gal tanks was assumed to remove tank waste to within 1 in. of the tank bottom. This 1-in. layer of waste residual was modeled in the PA as containing 25% solids and 75% interstitial liquids (EDF-TST-001, 2000), assuming approximately 24,000 Ci of activity remain in each tank. At that level of tank residuals, the PA demonstrated that performance objectives found in 10 CFR 61 were met.

In an attempt to maximize the effectiveness of the water washing system in the tank cleaning process, a mockup demonstration was used prior to deployment. The mockup included a full-height, half-circumference tank, which included cooling coils, simulated solids, washball/spray nozzles, and a steam jet, that were used to test the effectiveness of the cleaning equipment (INEEL 2001a). Lessons learned from the mockup included the need for additional directional spray nozzles to be deployed through available tank risers near the sides of the tanks to provide a mechanism to “sweep” residual solids away from the outer areas of the tank floor and toward the steam jet at the center of the tank, allowing for “spot cleaning” for areas of the tank floor as necessary.

To achieve the goal of removal of radionuclides to the maximum extent practical from the tanks, and based in part upon lessons learned from the mockup

a. This information should not be construed to suggest that RCRA-hazardous wastes are subject to Section 3116 of the Ronald W. Reagan Defense Authorization Act for Fiscal Year 2005 (Public Law 108-375, 2004).

demonstration, procedures were developed to operate the washball system, together with two (or three, depending upon the number of available riser locations) spray nozzles simultaneously with operation of the steam jet. This process would then provide a comprehensive washing of all internal tank surfaces, the ability to “sweep” tank solids towards the steam jet, and allow remaining liquid waste and slurried tank solids to be removed from the tank. The planned process of tank cleaning was to maintain a depth of flush water a few inches above the depth of solids during cleaning. This depth of flush water was thought to be optimum in terms of keeping enough depth to maintain solid particles in a slurry for removal. Too much flush water would prevent effective movement of solids by the washball/spray nozzles. As discussed in the draft 3116 Determination, radiation detectors on each tank’s outlet piping would provide an indication of the rate of waste removal compared to the volume of flush water used.

Cameras installed on the spray cleaning equipment allowed real-time visual examination of the tank internals. Tank cleaning was to continue until (1) radiation levels decreased to the lowest value and remained constant, which would be indicative that further flushing of the tank internals provided no further waste removal (see Figure RAI-5-1), and (2) comprehensive remote visual examinations of the tank internals after each day’s cleaning showed that the spray washing was no longer removing further waste residuals.

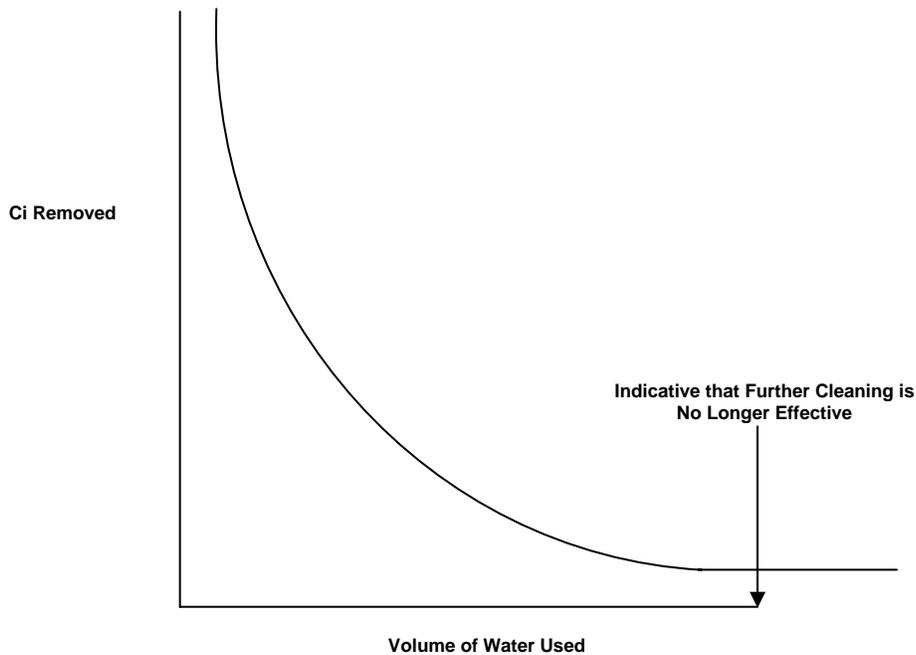


Figure RAI-5-1. Graph indicating the point at which tank cleaning is to be stopped.

At that point, tank cleaning would be stopped and the sampling and analysis program would commence. (Minimizing the generation of new waste volumes is one of the goals in any DOE activity; therefore, it was judged not to be prudent to

continue to clean tanks and other TFF components once the cleaning becomes ineffective.) Once analysis of the samples was complete, the results would be used to ensure that actions levels in the HWMA/RCRA closure plans were met. In addition, a computer-aided mapping process (kriging) would be used to estimate the volume of remaining waste residuals.

If radiation levels decreased to the lowest value and remained constant, and the remote visual inspections of the tank internals showed that further flushing of the tank internals would not be effective, tank cleaning activities would be stopped and radionuclides would be considered removed to the maximum extent practical.

30,000-gal Tanks

As described in the draft 3116 Determination, the 30,000-gal tanks are significantly different in terms of design and operational history than the larger 300,000-gal tanks. In 1983, these tanks were taken out of service, and subsequently, inspected, flushed, and sampled. Those activities showed that these four tanks did not have any remaining buildup of solid residuals (EDF-2614, 2002). As a result, the cleaning program for these tanks differed markedly from the larger 300,000-gal tanks. Since the previous sampling activities indicated that the tanks contained insignificant levels of radioactivity, any development and deployment of additional cleaning systems was judged to be not cost-effective. Therefore, instead of deploying a spray cleaning system, flush water was to be introduced into each of the tanks through the existing piping and then jetted out. A sampling and analysis program was to be implemented to validate previous sample results and verify that all RCRA action levels were met and determine the concentrations of radionuclides. Following sampling activities, remote visual inspections would be performed and, if no visible solids were noted, no further tank cleaning activities would be performed, and radionuclides would be considered removed to the maximum extent practical.

Tank Vaults

In general, tank vaults were known by operational history to contain only insignificant contamination, because the tank vaults only provide secondary containment for the stored waste. The WM-185 and WM-187 tank vaults were contaminated from back-siphon events; however, as discussed in the response to NRC Comment 1, these tank vaults, exclusive of the sandpads, were also thought to contain only minor contamination. Each tank vault has only two 12-in. riser locations, and this lack of access to the vaults prohibited deployment of any spray wash system similar to the tank washball/spray nozzles into the vault area. Rather, the vault floors were to be cleaned by introduction of sufficient volumes of water to completely cover the vault floors. The vault sumps would be jetted back into an operating tank or the process equipment waste system. Such vault cleaning techniques, combined with the existing low levels of vault contamination, were judged to be an effective method of removing radionuclide to the maximum extent practical. A sampling and analysis program was to be implemented to verify that all RCRA action levels were met and determine the

concentrations of radionuclides. If all RCRA actions levels were met, no further cleaning activities would be performed, and the radionuclides would be considered removed to the maximum extent practical.

Transfer Piping

Due to the acidic nature of the waste in the TFF, the stainless steel construction materials, the gravity-drain design of the piping system, and the operational practice to flush piping after each waste transfer, the transfer piping was thought to contain only very small amounts of residual waste materials. Tank closure planning included a process to flush each length of transfer piping with a minimum of three volumes of water to ensure any remaining waste material would be flushed from the piping. A sampling and analysis program (INEEL 2001b) was to be implemented to remove several sections of transfer piping from Tank WM-182, verify there was no buildup of solids materials, and that all RCRA action levels were met and determine the concentrations of radionuclides. If all RCRA actions levels were met and the visual inspections of the removed piping section showed no visible solids remaining, no further flushes would be performed, and the radionuclides would be considered removed to the maximum extent practical.

Other TFF Components

Other TFF components such as valve boxes, which provide access to transfer piping valve assemblies, were planned to be cleaned by spraying the interior surfaces of each valve box with several hundred gallons of water, which would flush any accumulated waste materials from the valve boxes and to the associated tank vaults. (Any waste transferred to the tank vaults would then be removed from the vault sumps as described above). Visual inspection of the accessible areas of the valve boxes would be performed to ensure no buildup of solids materials remained. In addition, a sampling and analysis program was to be implemented to verify that all RCRA action levels were met and determine the concentrations of radionuclides. If all RCRA actions levels were met, and the visual inspections showed no visible solids remaining the valve boxes, no further cleaning activities would be performed, and the radionuclides would be considered removed to the maximum extent practical.

A specific sequence of cleaning was established for each set of tanks to ensure waste materials removed from cleaned areas of the TFF were transferred to existing operations tanks for future treatment.

Evaluating Cleaning Effectiveness

300,000-gal Tanks

As planned, and described above, a cleaning approach was used for the tanks that would clean the tank interior and remove solids. High-pressure water from the washball/spray nozzles was used to push the solids on the tank floor and wash tank walls. Based on post-cleaning sample results and visual examination of the

interior of the tanks, the tank-cleaning washball was effective in cleaning all surfaces of the tank interior and the cooling coils along the tank walls. The directional nozzles were used to sweep the solids on the tank bottom toward the steam jet for removal from the tank. The liquid heel level was a factor that determined the efficiency of the wash. Too little liquid would not provide a medium to move solids and, if the liquid heel became too deep, the spray would not be able to penetrate the heel and move the solids. Each day, the tank was visually inspected to map the areas that contained the largest volumes to solids. Prior to each day's cleaning, project personnel met to review the remote video inspection of the tank internals and develop an approach to appropriately target the cleaning effort. The directional nozzles were concentrated on those areas. The discharge piping was monitored with a radiation monitor to provide a real-time verification of cleaning efficiencies. When the radiation levels dropped, the washing moved to another area containing solids. As washing began, an increase in radiation levels verified that the directional nozzles were moving solids to the jet. This process of visual inspection before and during cleaning in conjunction with the radiation detectors provided an efficient process for cleaning the tanks. When no areas of concentrated solids were visible, the project continued to spray the entire tank bottom and verified that no change in the radiation monitor occurred during the wash. When both the visual inspection and radiation monitor verified the cleaning was no longer removing solids, tank cleaning activities were discontinued (see Figure RAI-5-2 for WM-182 and Appendix RAI-5-A for all cleaned tanks). In general, tank washing occurred over a period of several weeks and used between 40,000 and 130,000 gal of flush water per tank. As described above, a sampling and analysis program was implemented. The results of these sampling activities were used to compare against the actions levels in the respective HWMA/RCRA closure plans. A computer-aided mapping process (kriging) was used to estimate the volume of remaining waste residuals and confirm that residual activity was below that projected in the PA.

Once radiation levels decreased to the lowest value and remained constant, the remote visual inspections of the tank internals showed that further flushing of the tank internals would not be effective and all RCRA action levels for the tanks were met, tank cleaning activities were stopped and the radionuclides were considered removed to the maximum extent practical.

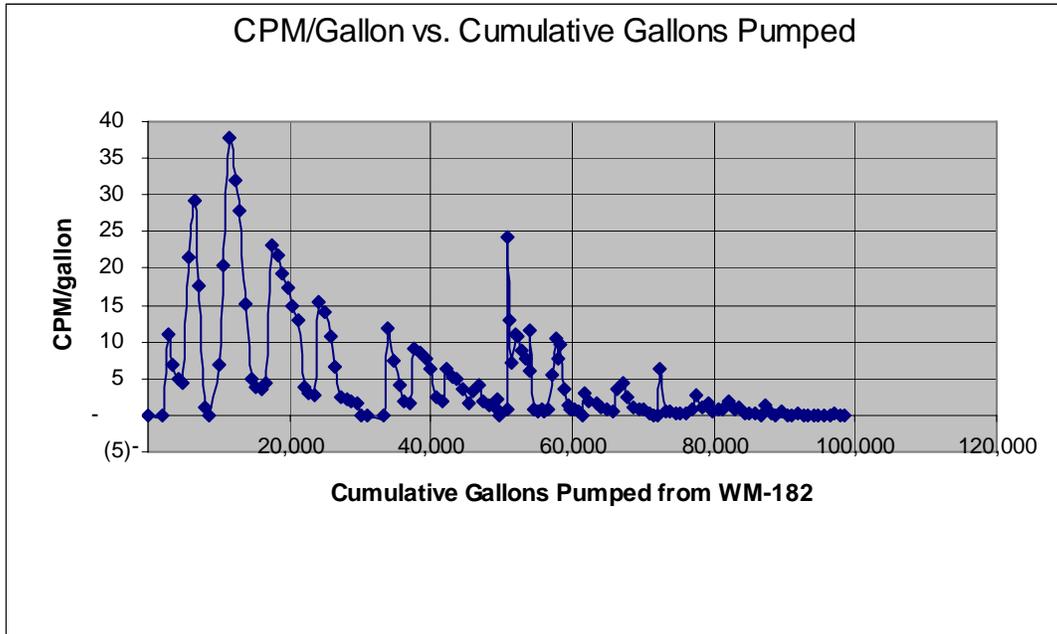


Figure RAI-5-2. Radioactivity concentration [counts per minute per gallon] curve.

30,000-gal Tanks

As discussed above, these tanks had been previously flushed and sampled. Flush water was introduced into each of the tanks through the existing piping and then jetted out. A number of considerations were taken into account to determine when additional cleaning would no longer be reasonably useful or reasonably effective. A sampling and analysis program was implemented to validate previous sample results and verify that all RCRA action levels were met. The sampling and analysis program was also used to determine the concentrations of radionuclides and to confirm that concentrations were well below that projected in the PA. Moreover, remote visual inspections were also performed and the inspections showed no visible solids inside the tanks.

Tank Vaults

As discussed above, the vault floors were washed by introduction of sufficient volumes of water to completely cover the vault floors and then jetted out. A number of considerations were taken into account to determine when additional cleaning would no longer be reasonably useful or reasonably effective. A sampling and analysis program was implemented to verify that all RCRA action levels were met. The sampling and analysis program was also used to determine the concentrations of radionuclides and to confirm that concentrations were well below that projected in the PA.

Transfer Piping

As described above, the transfer piping was thought to contain only very small amounts of residual waste materials. Several sections of transfer piping were removed from Tank WM-182 and it was verified that there was no buildup of solids materials. A sampling and analysis program was implemented to verify that all RCRA action levels were met (INEEL 2001b). These piping sections were judged to be representative of all transfer piping. To ensure that transfer piping did not contain significant radioactivity or RCRA constituents a flush was completed with a minimum of three volumes of water. Triple rinsing was specified in the HWMA/RCRA closure plans. The volume of each pipe transfer route to be flushed was calculated and then the piping associated with the transfer route was rinsed with at least three times the calculated volume. Flow meters were used to measure flush volumes. This process reduces radiation levels for all transfer piping and the radionuclides are considered to be removed to the maximum extent practical.

Other TFF Components

The valve boxes were flushed with a pressure water wand. Each valve box was washed top to bottom three times with a minimum of 200 gal of water. Representative valve boxes that contain sumps were sampled to verify that all RCRA action levels were met and determine the concentrations of radionuclides. If all RCRA action levels were met no further cleaning activities were performed, and the radionuclides were considered removed to the maximum extent practical.

Evaluation of Costs/Benefits of Additional Cleaning

As described above, decisions regarding when to terminate further cleaning of tanks and other TFF components were based upon monitoring of radiation detectors in tank outlet piping, visual inspections, and results from implementation of sampling and analysis plans. Monetary costs of additional cleaning were not a factor in decisions to stop the cleaning. Minimizing the generation of new waste volumes is one of the goals in any DOE activity; therefore, it was judged not to be prudent to continue to clean tanks and other TFF components once the cleaning becomes ineffective. The largest components of cleaning costs are associated with removal of existing equipment, installation of cleaning equipment, and sampling and analysis activities, not by flushing operations. Flush water, along with removed waste materials, from the cleaning activities was jetted to operational waste systems and then to facility evaporator systems to boil off the flush water. The incremental costs of evaporating the flush water are not significant. Therefore, additional spray cleaning of tank components would not have been prohibitive in terms of cost, but rather, were judged to be ineffective in removal of additional waste.

An evaluation of the costs and benefits for additional cleaning with a new technology is provided in Section 5.2.2 of the draft 3116 Determination.

Provide discussion regarding cleaning difficulties due to geometry or other physical limitations that make it more or less difficult to clean individual tanks. Discuss any changes in operational parameters during the cleaning process to help facilitate heel removal.

There are several factors that affect the ability to clean individual tanks with the same efficiency. The most significant factors are the physical differences in the tanks and the riser locations, which determine spray nozzle, steam jet, and washball locations, and the tank bottom configuration.

The configuration of the tank bottoms varied with each tank to be cleaned. Eight of the 11 300,000-gal tanks contain cooling coils. The design of the cooling coils is similar from tank to tank, but there are differences in the exact location of cooling coil routing and the number of cooling coil supports. These differences in cooling coil systems present slightly different challenges in terms of the ability to move tank solids around the tank floor as was observed in WM-182. Three of the 300,000-gal tanks do not contain cooling coils, which presented fewer challenges in terms of the ability to move tank solids towards the steam jet locations.

The remote video inspection of all the tanks' as-is configurations shows areas of the tank bottoms that are slightly lower than other areas of the tank bottoms, creating a wave or rippling effect, which results in the tank having low spots in differing locations on the tank floor. Each tank's stainless steel floor has different low spots. For example, remote visual inspection shows the floor outer edge of Tank WM-185 was several inches lower than the center of the tank. Therefore, as the tank was washed, any solids not jetted out would settle around the outer edge of the tank floor. When the tank heel was emptied to its lowest level, bare spots were observed in the center of the tank floor.

There were no areas of the tanks where the washwater did not move solids. As the cleaning was terminated in each tank, several inches of flush water was left in the tanks and any solids that could not be jetted out of the tank settled onto the tank floor. These solids settled out differently in the various tanks depending upon where the water from the spray nozzles and washball last contacted the tank floor and where the low spots and interferences exist on the tank bottom. The areas of the tank bottoms that have more solids are usually the low spots in the tank and the cleanest areas usually are the last spot the nozzle sprayed or a high spot on the floor. Additional washing would likely redistribute the solids to different areas of the tank floor. Therefore, differences in the kriging maps for each tank demonstrate slight differences in cleaning effectiveness between tanks as well as differences in the exact operation of the spray wash components.

Lastly, differences in the final configuration of remaining tank solids in the various tanks is due to incorporation of lessons learned, which resulted in some changes in the operational parameters during cleaning. As the cleaning activities progressed to different tanks, lessons learned meetings were held to discuss changes that could be used to increase cleaning efficiency and productivity. The major lesson that was learned from the first tank cleaning efforts was to remove as much heel with the new steam jet before cleaning started. Cleaning of the first

two tanks required slow removal of tank heels and solids to allow for evaluation of radiation levels in the transfer piping and valve boxes. After cleaning of these first two tanks, it was determined that the radiation levels would not impact operational activities, which allowed for removal of heels as rapidly as possible. This new approach did not result in a cleaner tank but required less water to complete the cleaning.

Provide the surface plots for WM-181.

A new engineering design file (EDF) for WM-181, which includes correct surface plots, was provided to the NRC during a February 1, 2006, meeting between DOE and NRC. In addition, the new EDF is attached to this response.

Provide clarifying information regarding the data input and interpolation parameters for kriging; and the uncertainty associated with the solid surfaces used to estimate the residual solid volume remaining in the tanks.

As described in the EDFs for each cleaned tank, a kriging method was used to estimate tank solids volumes for each tank. In general, two approaches were used to provide reference points for kriging estimates. The first approach, for tanks with cooling coils, utilized the cooling coil support structures, which provided numerous reference points for estimating depth of residual solids on the tank bottom. The tanks with cooling coils used the 3/8-in. cooling coils support base plate to estimate solids depths. In the tanks without cooling coils, 1-in.-diameter stainless steel pipe was cut into 1-in.-long sections and placed in the tanks. The pipe sections moved with the solids during the cleaning process and settled with the solids in each tank. The areas that did not contain pipe sections were generally clean and the tank bottom welds were used to estimate solids depths. The solids depths were then used to calculate solids volume.

Figure RAI-5-3 provides the cooling coil support spacing for the tanks containing cooling coils. The cooling coil support base plates were used as control points in the kriging analyses. Based on video inspection of the tanks, additional control points were established between base plates where no solids were noted and input into the kriging analyses. Based on video inspection, areas in the video that indicated presence of solids materials were outlined with control points between the cooling coils. The number of control points was based on the video observation and the degree to which solids materials were dispersed between cooling coils. If the solids materials were visually consistent in depth between several cooling coil base plate locations, then fewer control points were needed in the kriging analyses.

A second approach was used for tanks (WM-181, WM-184, and WM-186) that do not contain cooling coils. Cleaning of these tanks resulted in very light layers of solids on the tanks bottoms. The solids layer in these tanks was very sporadic and thin, with large areas of solid bare tank bottom visible, allowing the welds on the tank bottom to be visible. The kriging analyses for these tanks involved using the welds on the tank bottom and 1-in.-diameter pipe sections that were placed in the tanks. These tanks contained less solids materials on average than the tanks

with cooling coils, and were easier to visually inspect the welds between the stainless steel plates making up the bottom of the tank.

During this kriging analysis of the tank bottoms to estimate residual solids, every effort was made to provide a reasonably accurate, yet conservative estimate of the depth of remaining solids. The video and kriging analysis of the tank residual solids materials has inherent uncertainties. These uncertainties are due to the difficulty in estimating the fraction of the solids depth near the cooling coil support base welds and estimating the interface of solids materials between welds, especially in areas where the solids are thinning to a zero thickness. However, the uncertainty in the depth of residual solids materials was controlled by using conservative estimates of the depth of solids materials. For example, solids materials near the cooling coil base plates were assigned a depth of 3/8 in. when the solids materials were near the surface of the base plate, and the base plate was still visible. In cases where the base plate was slightly visible, the depth of solids materials were assigned a depth of 0.5 in. The control points for areas that had a reflective bottom surface were feathered at a fairly steep rate toward the areas containing solids materials. This slope of material is assumed to be much more gradual in a water deposition environment. In all cases, only areas that could be clearly defined as containing minimal solids residuals were assigned values less than the maximum from the nearest control point. The uncertainty in the solids estimate is also mitigated by the methodology that was used to estimate mass of the solids. The supernate, or interstitial liquids in solids, was not introduced in the calculation of solid mass. Analysis of the pre-wash samples showed that 25% of the heel solids were actual solids and the remaining 75% were interstitial liquids (EDF-TST-001, 2000). Therefore, by assuming the entire depth of solids from the kriging analysis is all solids and not discounting that much of the volume is interstitial liquid, a conservative estimate is developed.

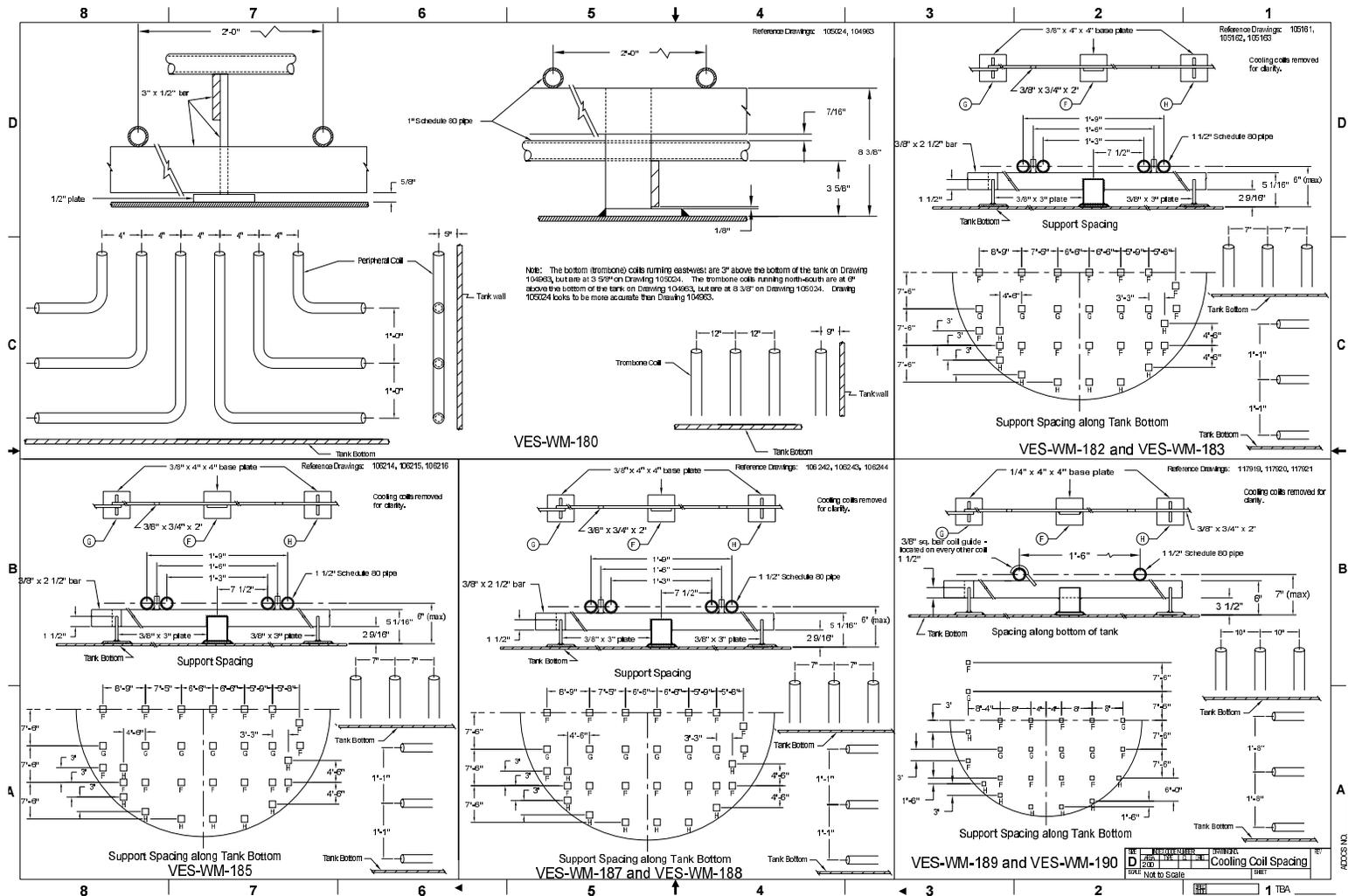


Figure RAI-5-3. Cooling coil support plate locations.

Provide tables of percent removal based on inventory of heel prior to cleaning and after cleaning.

Prior to the initiation of the tank cleaning program, and while the tanks were still in an operational status, a robotic arm system was deployed into two of the tanks (WM-182 and WM-183) to inspect the tank internals and draw samples of the tank solids for characterization purposes. However, estimates of the volume of solids on the tank floors were not developed. Pre-wash tank sampling and inspection did not occur with any of the other tanks that have been cleaned to date. As a result, a “beginning” inventory of tank waste at the start of tank cleaning was not developed for the tanks that have been cleaned. Without this “beginning inventory,” a percent removal comparison of the before and after cleaning inventories is not possible.

Discuss any remaining cleaning and sampling activities for piping encasements, valve boxes, and vault sumps. Provide additional justification regarding the expected removal efficiencies for remaining tanks to be cleaned and estimated costs for final cleaning.

The pipe encasements, valve boxes, and vault sumps that are associated with a tank are cleaned during the same time as the tanks. Therefore, the pipes, pipe encasements, valve boxes, and vault sumps that support Tanks WM-187, WM-188, WM-189, and WM-190 will be cleaned when the tanks have been emptied and the tanks are being closed. The vaults that support these tanks will be sampled after flushing is completed. Tank WM-190 has been used as a spare tank and is currently empty. Tanks WM-187, WM-188, and WM-189 will be cleaned and sampled after the waste has been removed and treated.

As discussed in the draft 3116 Determination, it is anticipated that the cleaning efficiencies in these four remaining tanks to be cleaned will be very similar to that experienced to date. Cleaning in the remaining 300,000-gal tanks is assumed to be as effective as in those that have been cleaned already for the reasons explained in the draft 3116 Determination. The same process of cleaning these tanks will be used. The physical properties of the solid residuals are well known (EDF-TST-001, 2000). There is no significant difference between the solids in the remaining tanks to be cleaned (INEEL 2004), and solids were effectively removed from the tanks cleaned to date. More volume of water or a longer cleaning time may be necessary because of the amount of solids now stored in remaining tanks (with the exception of Tank WM-190). Due to the large volumes of solids in WM-187 (since this tank has been used to accept flush solutions from previously cleaned tanks), additional liquids may be required, but the efficiency to remove solids should be similar to the previous tanks. The solids in Tank WM-187 are a collection of solids that have previously been removed using the same process. Based on previously incurred costs for tank cleaning, it is estimated that cleaning of the remaining four tanks will average approximately \$1 million per tank. The additional flush water, which may be required for cleaning of Tank WM-187, will not significantly increase the average cost for cleaning.

Discuss the evaluation of any new technologies to remove waste from the sand pads or tanks. Please discuss whether any technologies were considered to stabilize the sand pads without removing the sand, e.g., in-situ grout injection?

The project continues to review technology used in the DOE complex and new technology available through participation in technical exchanges and weekly EM conference calls. No new technology that has not previously been considered has been identified. In late March 2006, DOE will be conducting a complexwide tank cleaning workshop in which INL intends to participate.

Current plans to grout the tank vaults have been shown by the PA to adequately immobilize the estimated radioactive inventory in the two contaminated sandpads without direct stabilization of the sandpads. The limited access to the tank vaults and the location of the sandpads prohibit direct stabilization (e.g., in situ grout injection) of the sandpads.

References:

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EDF-TST-001, 2000, "Solids Characterization," September 20, 2000.

INEEL, 2001a, *INTEC Tank Farm Facility Closure Mockup Test Report Project File No. 015722*, INEEL/EXT-2001-00299, March 2001.

INEEL, 2001b, *Sampling and Analysis Plan for the Post-Decontamination Characterization of the Process Waste Lines from INTEC Tank Farm Facility Tanks WM-182 and WM-183*, INEEL/EXT-01-01543, November 2001.

INEEL, 2004, *Feed Composition for the Sodium-Bearing Waste Treatment Process*, INEEL/EXT-2000-01378, Rev. 4, June 2004.

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APPENDIX RAI-5-A

RADIOACTIVITY CONCENTRATION [COUNTS PER MINUTE PER GALLON] CURVES FOR CLEANED TANKS

Note: The figures included in this appendix have varying scales for the x axis and the y axis.

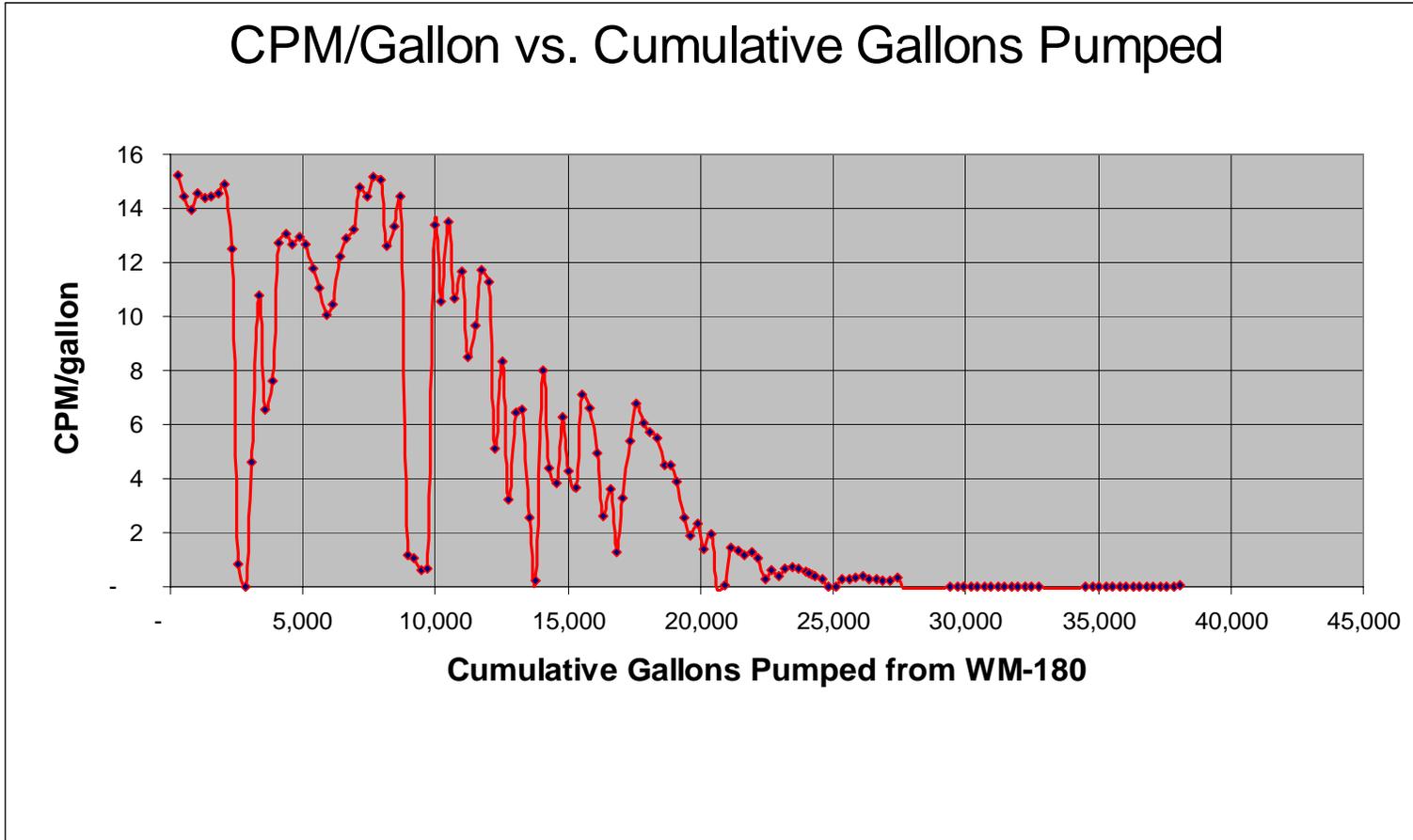


Figure RAI-5-A-1. Cumulative gallons pumped from WM-180.

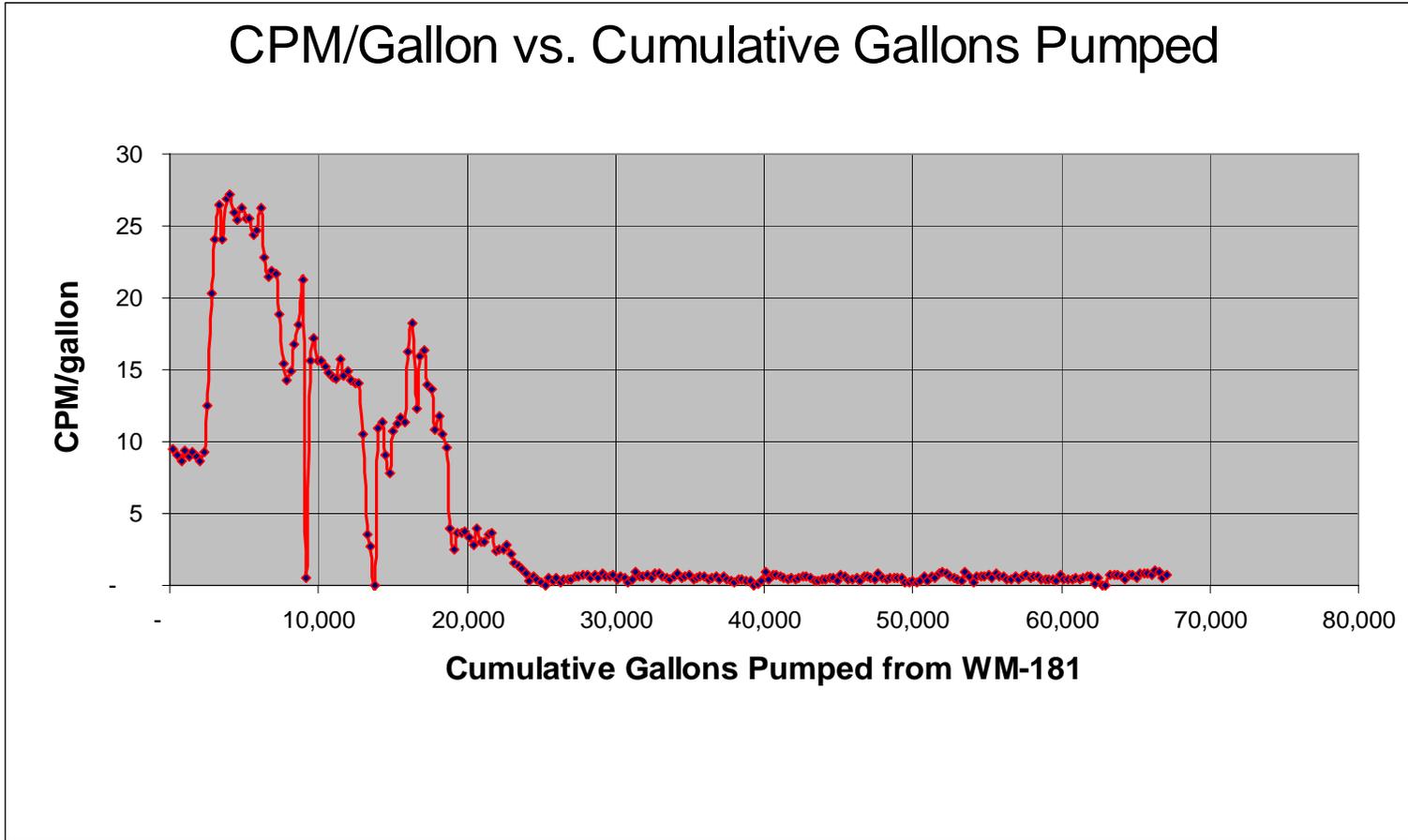


Figure RAI-5-A-2. Cumulative gallons pumped from WM-181.

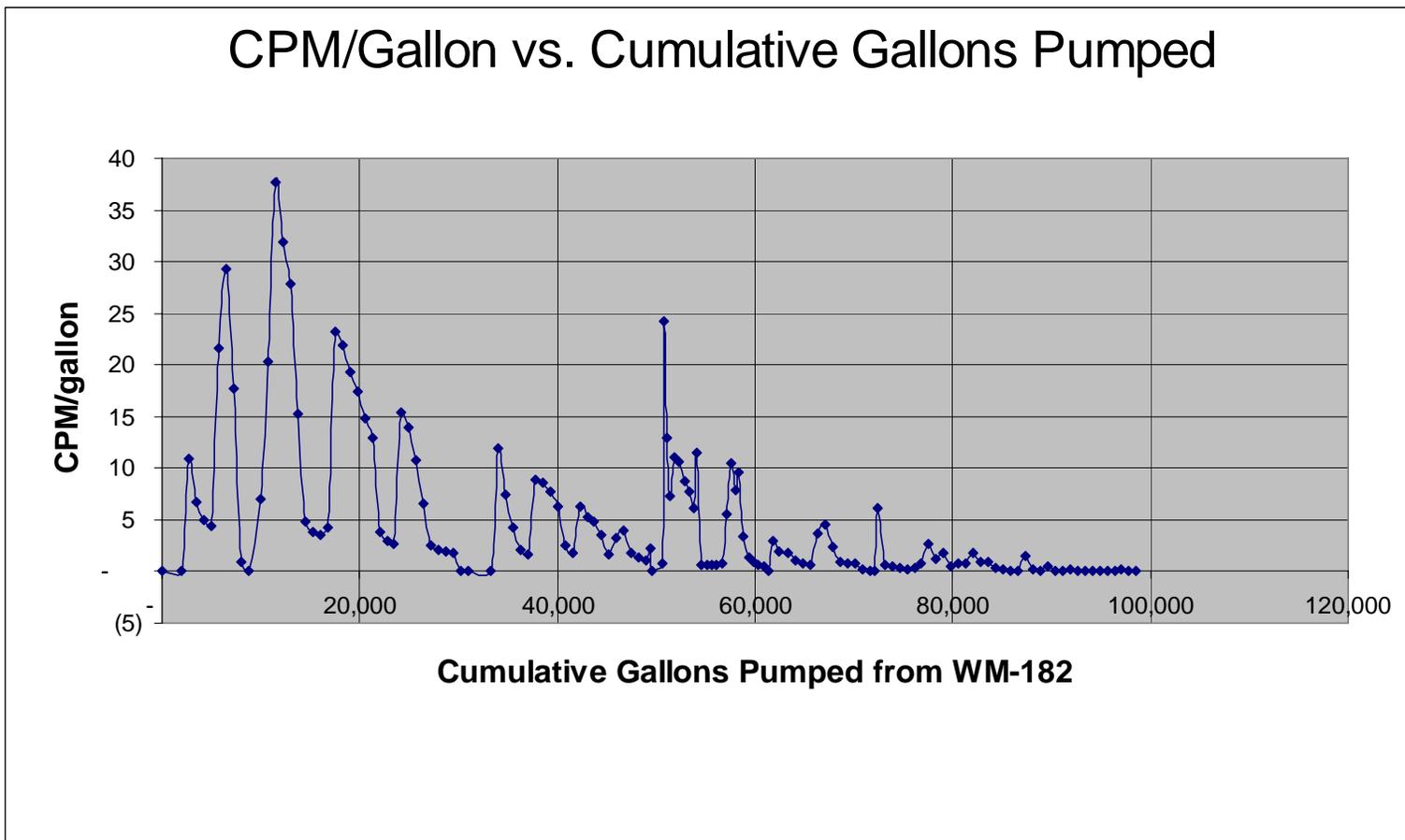


Figure RAI-5-A-3. Cumulative gallons pumped from WM-182.

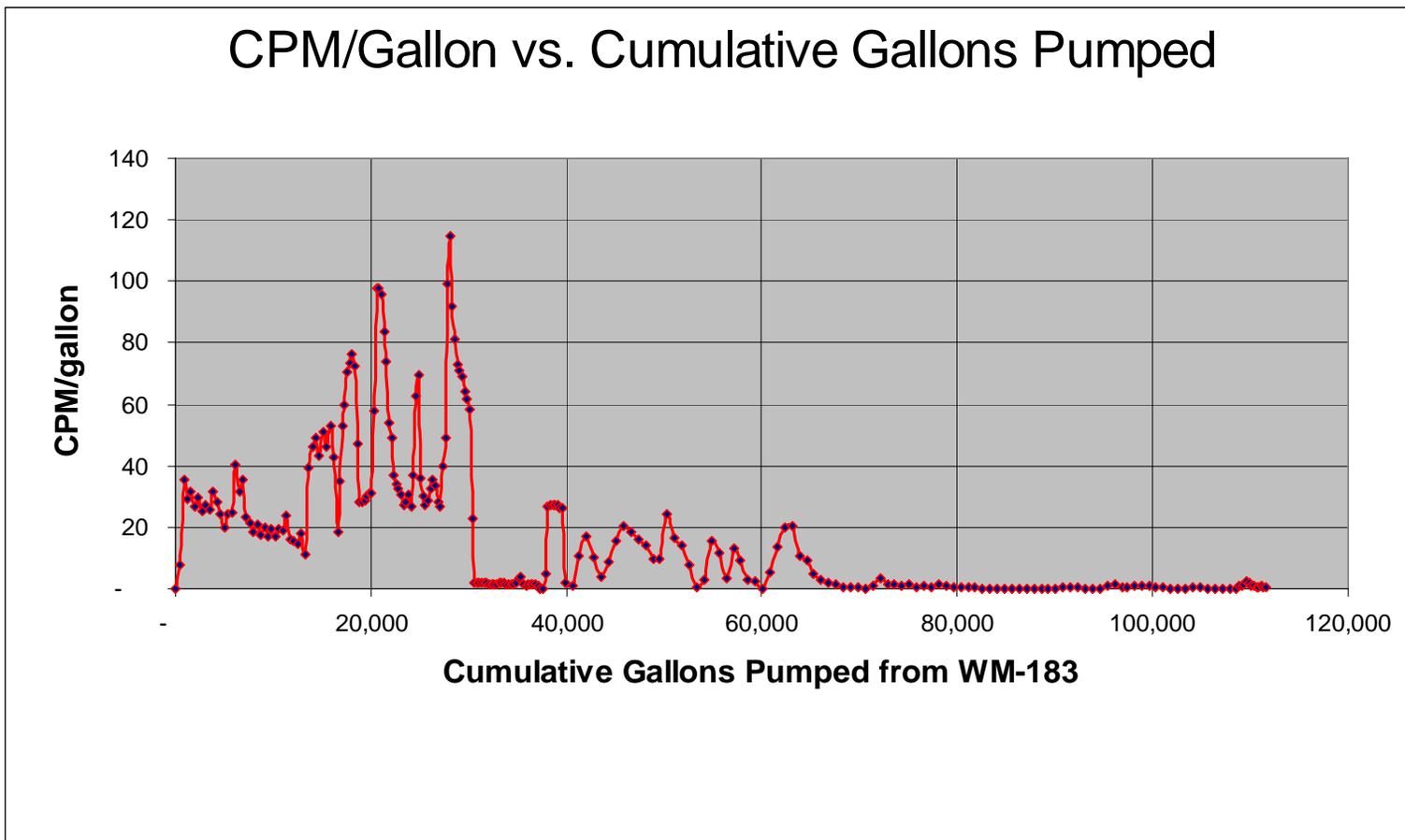


Figure RAI-5-A-4. Cumulative gallons pumped from WM-183.

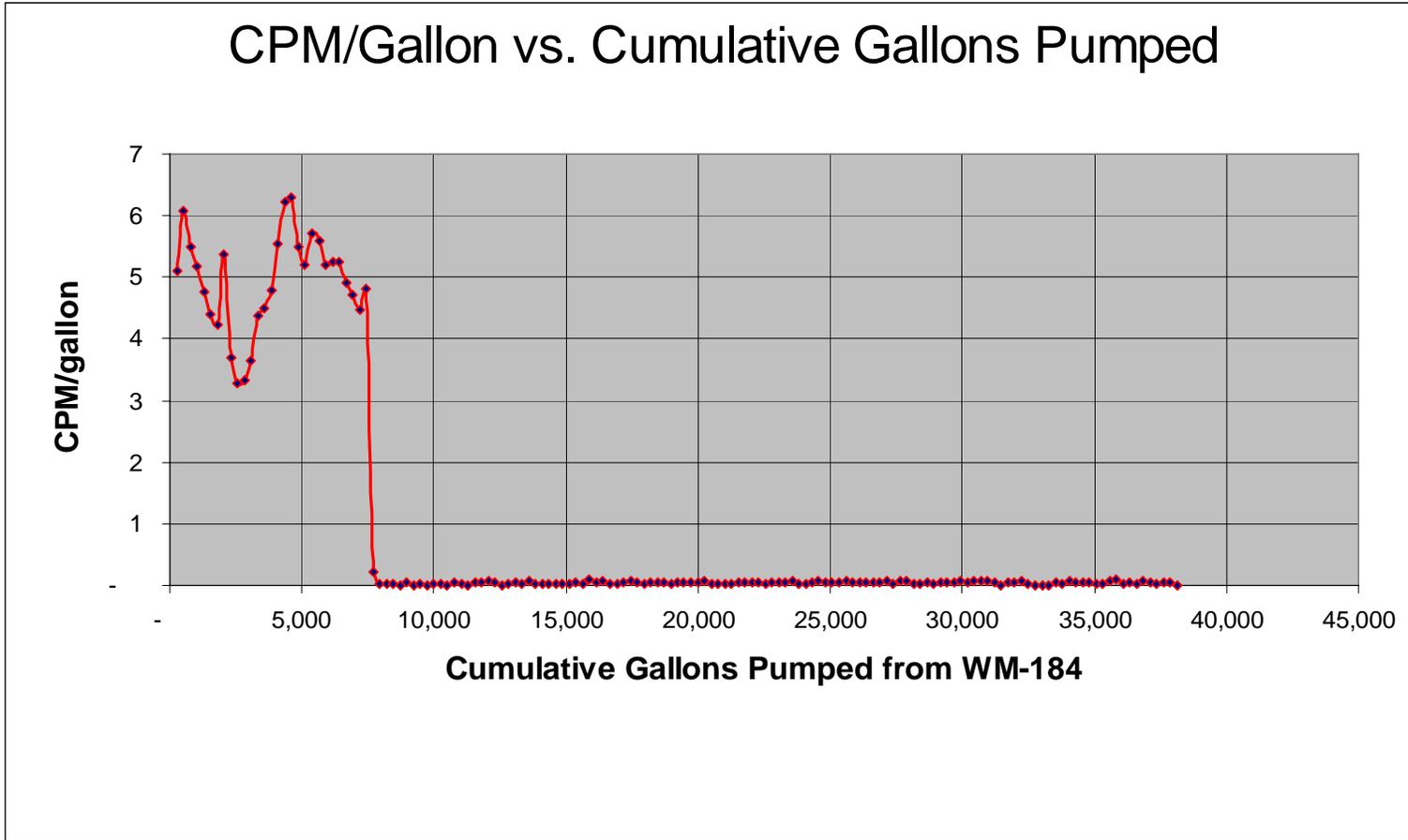


Figure RAI-5-A-5. Cumulative gallons pumped from WM-184.

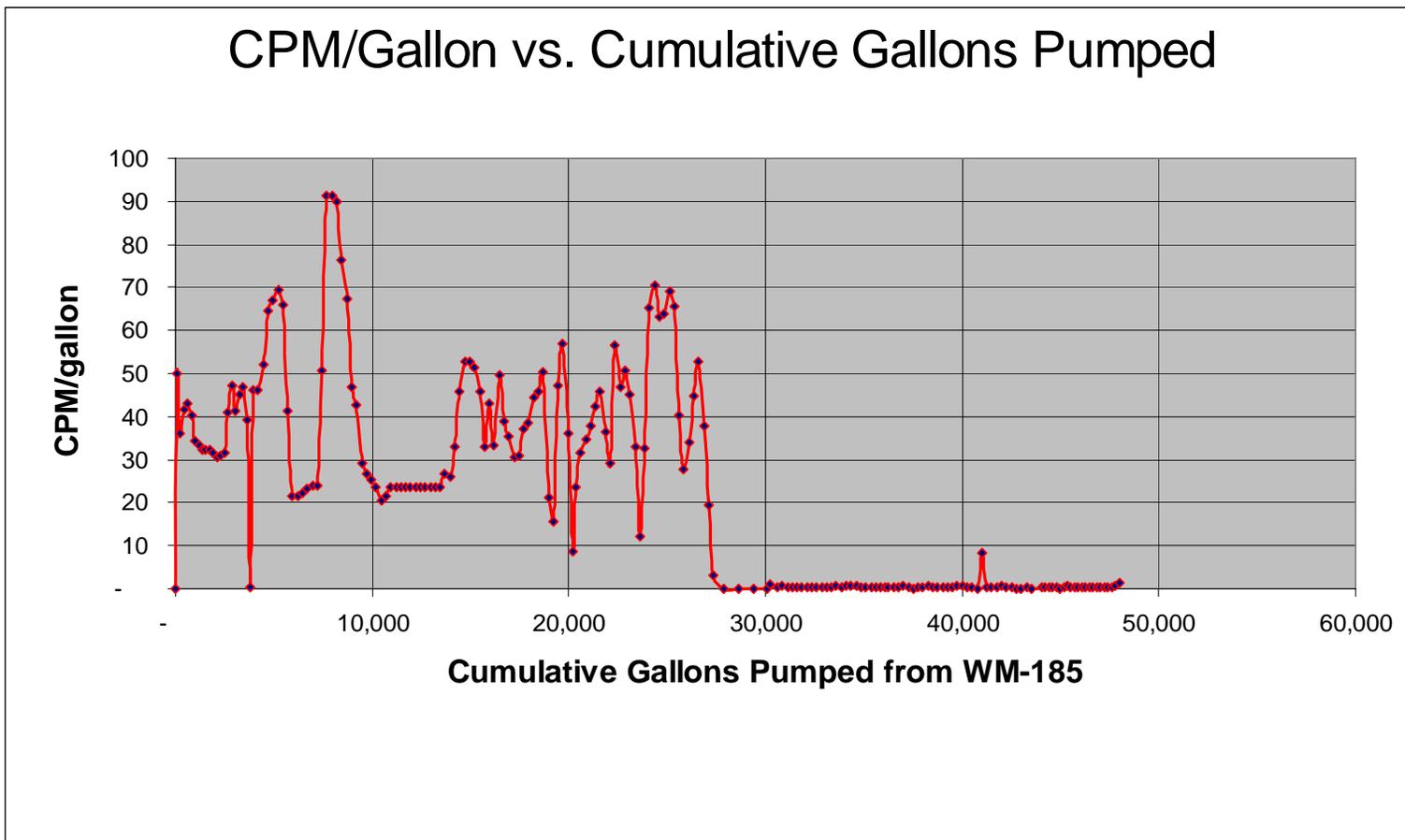


Figure RAI-5-A-6. Cumulative gallons pumped from WM-185.

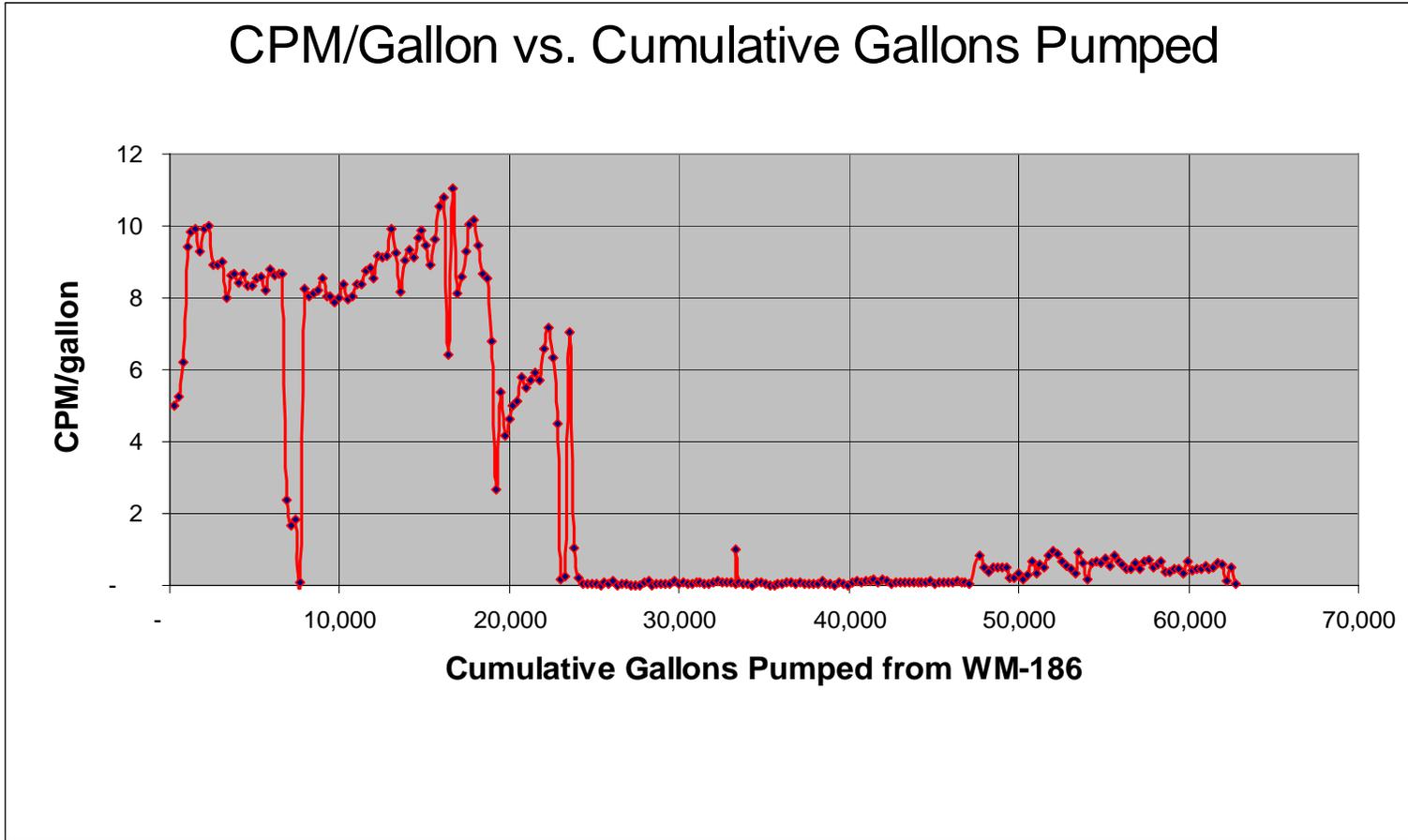


Figure RAI-5-A-7. Cumulative gallons pumped from WM-186.

NRC

Comment 6:

The conceptualized model showing the evolution of the vault [DOE-ID, 2003b, Figure E-1(a) and associated text depicts the outer concrete vault as uncracked in the initial phase, but shows the presence of "shrinkage cracks" on the vault and grout after the grout pour [DOE-ID, 2003b, Figure E-1(b)]. Information would be helpful regarding how the conceptual model accounts for the formation of voids and cracks due to the thermal regime resulting from the heat of hydration of the grout pour and due to the physical coupling between concrete and grout.

Basis:

The evolution of concrete and grout properties depends partly on the extent of physical coupling between two materials of different compositions and partly on the thermal regime initiated by heat of hydration of the grout pour. The situation is further complicated by the fact that the concrete is subject to restraint, by reinforcement, whereas the grout is apparently unrestrained. The evolution of concrete and grout properties occurring at this stage may have an important influence on the final grout and concrete properties. Laboratory simulations do not simulate the coupling between materials at full scale and properties measured on these simulants may not scale up.

Path Forward:

Provide a technical basis for the anticipated thermal and mechanical history of the various cementitious materials. Provide any available data on the isothermal shrinkage of the two main material types (concrete and grout) and assess the additional impact of the thermal excursion arising in the course of emplacing the grout and its subsequent hardening. Explain how the dimensional changes would translate to the incidence of voids and cracking and the five stages illustrated in DOE-ID (2003b, Figure E-1).

Response:

The degradation analysis in Appendix E of the PA (DOE-ID 2003) provides a range of potential degradation times for each system component. The concrete/grout degradation analysis provided in Appendix E of the PA supports much longer degradation times than were assumed in the PA analyses. The vault and outer grout (between the stainless steel tank and vault) are assumed to degrade completely to rubble after 100 years. The tank and inner grout are assumed to degrade completely to rubble after 500 years. Therefore, the concrete degradation analysis presented in Appendix E of the PA provides confirmation of the very conservative nature of the PA degradation assumptions (i.e., complete degradation times of 100 and 500 years).

Provide a technical basis for the anticipated thermal and mechanical history of the various cementitious materials. Provide any available data on the isothermal shrinkage of the two main material types (concrete and grout) and assess the additional impact of the thermal excursion arising in the course of emplacing the grout and its subsequent hardening.

The tank vaults were constructed approximately 50 years ago. Since the model assumed the existing concrete vaults will be rubble after only 100 years, no detailed evaluation has been performed of the anticipated thermal and mechanical history of the vault concrete or for data on the isothermal shrinkage of the vault concrete. Plans for grouting of the TFF components have included analysis of

grout behavior (EDF-6715, 2006). The grout is scheduled to be poured in a series of nine separate layers, each with a thickness of approximately 1 m (see Figure RAI-6-1). Since the volume of grout contacting the vault during each pour will be relatively small, there will not be any significant thermal or mechanical stresses to the vault walls. Time between pours will allow for cooling. Procedures will be in place during grout pouring to protect the integrity of the tank and vault during grout pours by limiting temperature rise and volume change.

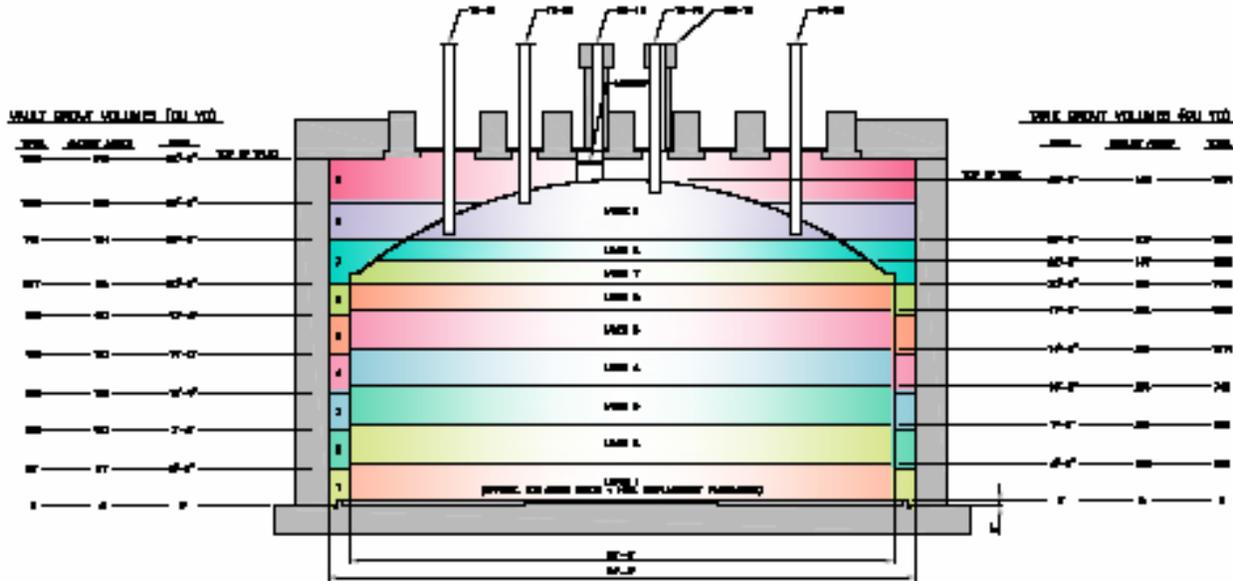


Figure RAI-6-1. Grout pour lift sequences and dimensions.

The specifications for grout shrinkage (EDF-6715, 2006) are provided in Table RAI-6-1.

Table RAI-6-1. Grout shrinkage specifications (EDF-6715, 2006).

Parameter	Performance Values	Selection of Value
Shrinkage	<1%	Maximum
	<0.5%	Expected
	<0.1%	Desirable

A full-scale field test of a grout pour has been conducted (INEEL 1999). Figures RAI-6-2 through RAI-6-6 show photographs that were taken of the grout after a test pour and give a visual indication of anticipated cracking. Cracks occur on approximately 1-m spacing. Each pour will seal some of the cracks developed in the previous pour, but cracks will exist nonetheless.

The presence of cracks does not change the degradation calculations because of the conservative basis of the calculations. Mass transport is assumed to occur through degraded rubble, not intact concrete or grout. The fact that cracks will be

present means that the intact portion of the system is not as resistant as one would like; however, the degradation calculations are based upon the properties of the altered outer zone, not the intact inner zone.

The performance of the grout has been summarized in EDF-6715 (2006).



Figure RAI-6-2. Grout Pour Photograph #1.



Figure RAI-6-3. Grout Pour Photograph #2.



Figure RAI-6-4. Grout Pour Photograph #3.



Figure RAI-6-5. Grout Pour Photograph #4.



Figure RAI-6-6. Grout Pour Photograph #5.

Explain how the dimensional changes would translate to the incidence of voids and cracking and the five stages illustrated in DOE-ID (2003b, Figure E-1).

Figure E-1 (attached) is a artistic conceptual drawing showing the evolution of system degradation in a general sense. The stages of degradation in terms of the numbering in the figure from (a) to (e) are as follows:

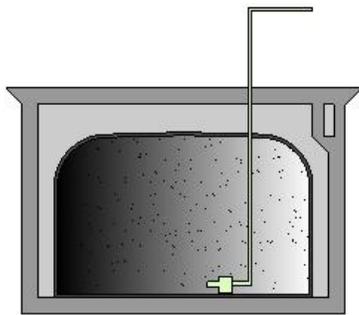
- a) Initial condition after grout pours are finished: Cracks are present in the grout between the tank and the vault, and inside the tank. Grout pour does not crack the vault. The vault system is buried beneath the land surface, such that structural support is provided by the soils surrounding the vault. Therefore, cracking of the vault during the grout pour is not expected.
- b) Steel reinforcement in the outer vault corrodes. Volume expansion from corrosion causes spalling around corroded reinforcement.
- c) Degradation front passes through the vault and reaches the outside of the stainless steel tank.
- d) The stainless steel tank may take more than 20,000 years to fully corrode, but is assumed to have initial localized penetrations.
- e) Entire system converts to rubble.

References:

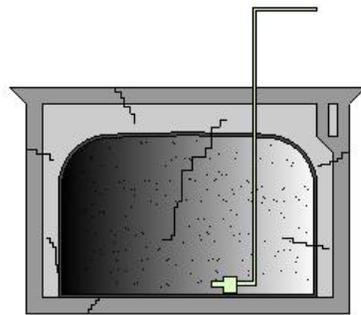
DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

EDF-6715, 2006, "Grout/CLSM Testing and Selection for the INTEC Tank Farm Closure," Project No. 15722, March 7, 2006.

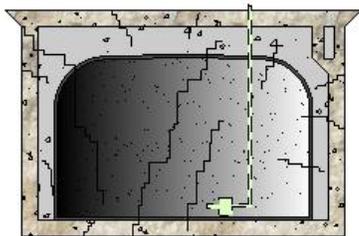
INEEL, 1999, *Idaho Nuclear Technology and Engineering Center Tank Farm Facility (TFF) Closure TFF WM-182 Grout Mock-Up*, INEEL/EXT-99-01067, Rev. 0, October 1999.



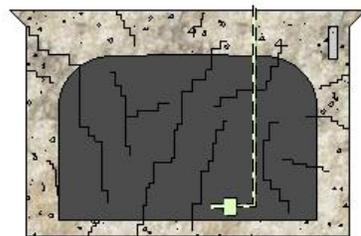
**(a) Conceptualized system—
tank filled with grout and grout
between tank and vault.**



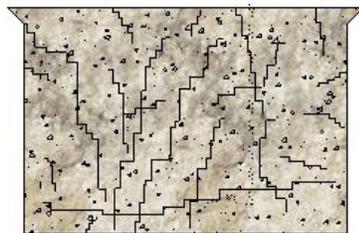
**(b) Following grout pour—
shrinkage cracks exist in the outer
vault and grout and there are
penetrations in the tank wall.**



**(c) The steel reinforcement in the
outer concrete vault corrodes and
the concrete loses its structural
properties.**



**(d) The grout outside the tank
fails from the chemical attack.**



**(e) The tank corrodes and all
concrete and grout turn to rubble.**

Figure E-1. Conceptualization of the representative TFF tank, grout, vault, and piping system shown in (a) and the degradation sequence (b) through (e) at the TFF.^b

b. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

NRC

Comment 7:

In the conceptual model, degradation is assumed to occur from the outside and propagate inwards, and a homogeneous degradation front is modeled (DOE-ID, 2003b). This conceptual model may not be appropriate if the transport of aggressive agents is through cracks or if degradation occurs from internal factors (e.g., by alkali-aggregate reaction).

Basis:

As indicated in Comment 6, there is a potential for cracking that apparently has not been evaluated. If cracking occurs, and especially if cracking is confined to a relatively few but wide cracks, each crack may serve as a rapid conduit for migration of water deep into the grout mass. Cracks may eventually dominate the rate at which degradation occurs, especially in grout which is unrestrained.

Path Forward:

How have alternative conceptual models of degradation that consider the effects of the emplacement of grout and the resulting thermal regime, as well as the intrinsic dimensional stability (i.e., under isothermal conditions) of the grout as a function of time and hydration of its components been considered? If these alternative conceptual models have not been considered, explain how the analysis presented in Appendix E is expected to bound the potential results of these alternative conceptual models?

Response:

As explained below, alternative conceptual models of degradation that consider the effects of the placement of grout and the resulting thermal regime, as well as the intrinsic dimensional stability (i.e., under isothermal conditions) of the grout as a function of time and hydration of its components, have not been considered. The degradation analysis in Appendix E of the PA (DOE-ID 2003) provides a range of potential degradation times for each system component. The concrete/grout degradation analysis provided in Appendix E of the PA supports much longer degradation times than were assumed in the PA analyses. The vault and outer grout (between the stainless steel tank and vault) are assumed to degrade completely to rubble after 100 years. The tank and inner grout are assumed to degrade completely to rubble after 500 years. Therefore, the concrete degradation analysis presented in Appendix E of the PA provides confirmation of the very conservative nature of the PA degradation assumptions (i.e., complete degradation times of 100 and 500 years).

Cracks represent preferential pathways for corrosive agents including water, oxygen, chloride, sulfate, and magnesium. The cracks illustrated in Figures RAI-7-1 through RAI-7-5 will cause the system to degrade more rapidly than it would without cracks and actual degradation will occur more rapidly along some of the cracks. The presence of these cracks does not change the degradation calculations presented in Appendix E of the PA because of the conservative basis of the calculations. Mass transport in the degradation calculations is assumed to occur through degraded rubble, not intact concrete or grout. The presence of initial cracks may reduce lifetime relative to an uncracked system; however, the calculations are based upon the properties of the altered outer zone (vault and vault grout), not the intact inner zone (tank and tank grout). Thus, the degradation calculations are conservative. Cracking will affect actual performance, but will

not affect the lower bound degradation estimates provided in Appendix E of the PA.

The rate of actual degradation of concrete appears to be related to the properties of the cementitious materials near the corrosion front. The difficulty is that actual transport is through the degraded concrete/grout and through and into an active weathering zone at the boundary with the intact concrete/grout. Sufficient knowledge and models do not exist to accurately predict actual degradations rates. In order to keep the models conservative, it was assumed that transport always occurs through weathered concrete/grout.



Figure RAI-7-1. Grout Pour Photograph #1.



Figure RAI-7-2. Grout Pour Photograph #2.



Figure RAI-7-3. Grout Pour Photograph #3.



Figure RAI-7-4. Grout Pour Photograph #4.



Figure RAI-7-5. Grout Pour Photograph #5.

References:

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

NRC

Comment 8:

DOE has not fully justified its choice of the surface rinsing release option in DUST-MS.

Basis:

The surface rinsing model uses an equilibrium sorption relationship to calculate pore-water radionuclide concentrations in the grout and sand pad sources. Among alternative release models is one that assumes that radionuclide water concentrations are controlled by solubility. It is not always clear which type of model is most appropriate or conservative. Typically both models are considered, and the model that provides the most conservative result is used. On page 2-57 of DOE-ID (2003b), DOE makes clear that the solubility limit model was not used, but also refers to comparisons between tank grout pore-water concentrations based on solubility and those based on sorption coefficients. The details of this comparison are not provided, but are necessary for evaluating the choice of release model. Discussions of release model choice in Section 7.7 of DOE-ID (2003b) do not include a comparison with the solubility model.

Path Forward:

Provide a discussion and tabulation of the pore-water concentration comparison (DOE-ID, 2003b, Section 2.1.6.3, p. 2-57) and justification for the model selected.

Response:

As the PA (DOE-ID 2003) was developed, consideration was given to both a surface rinse model and a solubility model with the goal of using the most appropriate model for the TFF. Release modeling, in the PA, for the near-field environment involved the use of the DUST-MS surface rinse model based on sorption coefficients. It would be reasonable to limit the DUST-MS surface rinse model concentrations based on solubility limits as applicable. However, to maintain model conservatism, solubility limits were not used to limit the releases in the PA, since additional constraint on radionuclide releases was not necessary to maintain doses below the performance objectives.

The example comparison of the solubility limits and the sorption coefficient-based pore-water concentrations in a grouted tank, which was inadvertently omitted in the PA, is provided in Table RAI-8-1. The solubility limit ranges for concrete were obtained from Kaplan and Serne (2000) and Allard et al. (1985). The PA pore-water concentrations, based on the sorption coefficient, in the grouted waste form at 500 years post-closure (i.e., beginning of the tank releases in the PA) are also provided for comparison to the solubility limit ranges (see Table RAI-8-1). Radionuclides such as ^{129}I and ^{99}Tc that are dose drivers in the PA as well as ^{137}Cs and ^3H are not assumed to be solubility limited and are not compared in Table RAI-8-1. However, some actinides are known to have low solubility limits in cement and are presented in Table RAI-8-1. Comparison of the values in Table RAI-8-1 indicates that using the Allard et al. (1985) solubility limits would result in lower release rates than the surface rinse model for ^{239}Pu , ^{240}Pu , ^{234}U , ^{235}U , ^{236}U , and ^{238}U . The surface rinse model chosen for use in the PA provides more conservative release rates for these radionuclides. When comparing solubility limits and surface rinse model concentrations, it is not reasonable to always use the highest value. The

geochemical behavior of radionuclides in the TFF tank grout were described in the PA using the distribution coefficient (K_d value) in the DUST-MS surface rinse model. When radionuclide concentrations exceed the solubility constraint, K_{sp} , precipitation can be expected and subsequent radionuclide aqueous concentrations and behavior is controlled by solubility. At concentrations below this limit, the radionuclide concentration is controlled by the K_d construct. The PA did not consider limiting the radionuclide releases from the grout by the solubility constraint, thus providing conservative release rates.

Table RAI-8-1. Radionuclide solubility in grout compared to the PA concentrations in the tank pore water.

Nuclide	Half-Life (yr)	Solubility Limit in Concrete		Grout Pore-Water Concentrations ^c ($\mu\text{Ci/L}$)
		$\mu\text{Ci/L}^a$	$\mu\text{Ci/L}^b$	
²⁴¹ Am	4.32E+02	2.49E+02	3.06E+03	1.11E-03
²⁴³ Am	7.38E+03	1.46E+01	1.79E+02	4.42E-06
²³⁸ Pu	8.78E+01	2.04E+03	1.30E-01	6.87E-04
²³⁹ Pu	2.41E+04	7.43E+00	4.74E-04	2.79E-03
²⁴⁰ Pu	6.54E+03	2.74E+01	1.75E-03	2.20E-03
²⁴¹ Pu	1.44E+01	1.24E+04	7.93E-01	6.36E-08
²³⁴ U	2.45E+05	1.46E+00	3.64E-05	4.85E-04
²³⁵ U	7.04E+08	5.09E-04	1.27E-08	2.40E-06
²³⁶ U	3.42E+06	1.05E-01	2.61E-06	1.45E-05
²³⁸ U	4.47E+09	8.01E-05	2.00E-09	1.81E-06
²³⁰ Th	7.70E+04	2.33E+00	1.16E-03	1.06E-06

a. Kaplan, D. I., and R. J. Serne, 2000, *Geochemical Data Package for the Hanford Immobilized Low-Activity Tank Waste Performance Assessment (ILAW PA)*, PNNL-13037, Rev. 1. February 2000.

b. Allard, B., G. Persson, and B. Torstenfelt, 1985, *Technical Reports 85-18 Actinide Solubilities and Speciation in a Repository Environment; 85-19 Organic Complexing Agents in Low- and Medium-Level Radioactive Waste; 85-20 Radionuclide Sorption on Carbonate-Clayish Rock; and 85-21 Radionuclide Sorption on Concrete*, NAGRA, Radiochemistry Consultant Group for Kematka Consultants Co., Stockholm, Sweden.

c. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

References:

Allard, B., G. Persson, and B. Torstenfelt, 1985, *Technical Reports 85-18 Actinide Solubilities and Speciation in a Repository Environment; 85-19 Organic Complexing Agents in Low- and Medium-Level Radioactive Waste; 85-20 Radionuclide Sorption on Carbonate-Clayish Rock; and 85-21 Radionuclide Sorption on Concrete*, NAGRA, Radiochemistry Consultant Group for Kematka Consultants Co., Stockholm, Sweden.

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

Kaplan, D. I., and R. J. Serne, 2000, "Geochemical Data Package for the Hanford Immobilized Low-Activity Tank Waste Performance Assessment (ILAW PA)," PNNL-13037, Rev. 1. February 2000.

NRC

Comment 9:

Flooding at the TFF could increase the downward mobility of subsurface contamination, potentially influencing flow regimes, transport pathways, transport times, and contaminant concentrations. Additional technical basis is needed to support the flooding analysis.

Basis:

Figure 2-18 shows the areal coverage of a Probable Maximum Flood at INL (DOE-ID, 2003b). It is difficult to review Figure 2-18 because its resolution is low. The figure would be more useful if shown at a higher resolution. The location of Idaho Nuclear Technology and Engineering Center is not superimposed on the inundation map shown in Figure 2-18, yet its location apparently near a margin of the flooded area is supposed to support the flooding analysis.

It is not clear in the performance assessment (DOE-ID, 2003b) what assumptions are made regarding dam maintenance, effectiveness of the INL spreading areas and flood diversion facility, and landscape evolution over the period of performance. Such assumptions would be relevant to severity and implications of flooding at the TFF.

Section 2.1.5.3.6 pertains to Potential Dam Failures. DOE-ID (2003b) in addressing the study performed by Van Hafften, Koslow, and Naretto (1984) for the flooding analysis of the New Production Reactor site, includes the following text: "The flood hydrograph for the base-case discharge shows a peak flow that lasts only 7.5 hours (Figure 2-19). Since the INTEC facility is located at the boundaries of the flood, the area would be inundated for a lesser period of time" (p. 2-38). The information conveyed in the second sentence called out above does not necessarily follow from the information conveyed in the first sentence: The duration of peak discharge does not necessarily imply how long water may pond at the surface of the TFF or elsewhere (see also Section 7.4.2, page 7-26).

The maximum depth to which ponded flood water is expected to infiltrate into the unsaturated zone below the TFF requires a technical basis (DOE-ID, 2003b, p. 2-39). The concept of a "wetting front," which is common to conceptual models of flow through sedimentary porous media, may not be appropriate for the fractured basalt portions of the unsaturated zone below the TFF (e.g., Faybishenko, et al., 2000).

For the Large Scale Infiltration Test conducted at INL, Newman and Dunnivant (1995) were able to model only 10 of 26 breakthrough curves using a one-dimensional transport model, which suggests use of an inadequate method (Wood and Faybishenko, 2001).

Path Forward:

DOE-ID should provide a higher resolution figure of the relevant portions of Figure 2-18. DOE-ID should also show the location of INTEC superimposed on the higher-resolution inundation map.

DOE-ID should provide information regarding their assumptions for landscape evolution, which are relevant to the severity and implications of flooding at the

TFF. This information may include assumptions by the Army Corps of Engineers regarding the useful lifetime of Mackay Dam, and whether Mackay Dam is assumed to be dredged repeatedly so that it does not fill with sediment. This information may include an analysis of flooding scenarios and effect on dose if Mackay Dam was no longer maintained starting at $t = 500$ years (not a one-time dam failure, but rather the dam is assumed out-of-commission over the long term). DOE-ID should provide its assumptions and technical bases regarding whether the spreading areas at INL will not fill with sediment such that capacity is reduced. DOE-ID should provide an estimate regarding the longevity of the effectiveness of the flood diversion facility in the absence of maintenance at the end of the period of institutional control.

DOE-ID should provide the text on page 2-35 of the DOE-ID (2003b) that is currently hidden behind a figure.

DOE-ID should provide information that demonstrates that the impacts of the probable maximum flooding condition on the INTEC facility have been bounded. DOE-ID should provide the analysis by Dunnivant et al., (1998) on which they, in part, base their expectation of minimal impact. DOE-ID should address the potential for higher levels of contamination to persist for a longer period of time in the flooding scenario. DOE-ID should provide an analysis that clearly demonstrates the expected amount of time water might pond at the surface of the TFF after a dam failure or probable maximum flood.

Response:

DOE-ID should provide a higher resolution figure of the relevant portions of Figure 2-18. DOE-ID should also show the location of INTEC superimposed on the higher-resolution inundation map.

A higher resolution figure of the relevant portions of Figure 2-18 from the PA (DOE-ID 2003) is included as Figure RAI-9-1, with the location of the Idaho Nuclear Technology and Engineering Center (INTEC) (formerly CPP) superimposed. Figure RAI-9-2 is a topographic map of the TFF. The ground elevation around the tanks varies between 4,914 and 4,916 ft (amsl). The estimated flood elevation is 4,917 ft, providing between 1 and 3 ft of flood water inundation over the tanks.

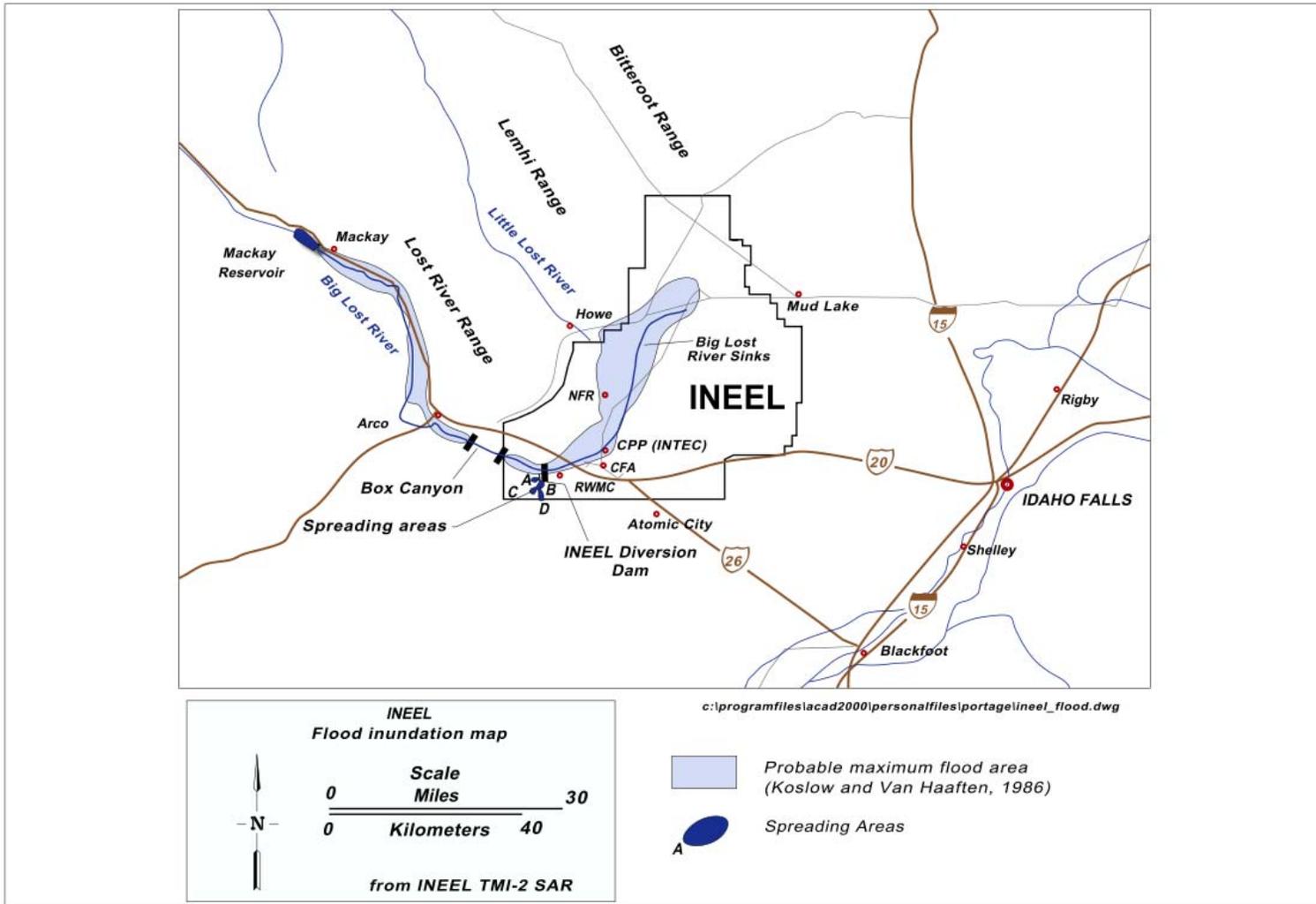


Figure RAI-9-1. Probable maximum flood Idaho National Engineering and Environmental Laboratory (INEEL) (now INL) inundation map.

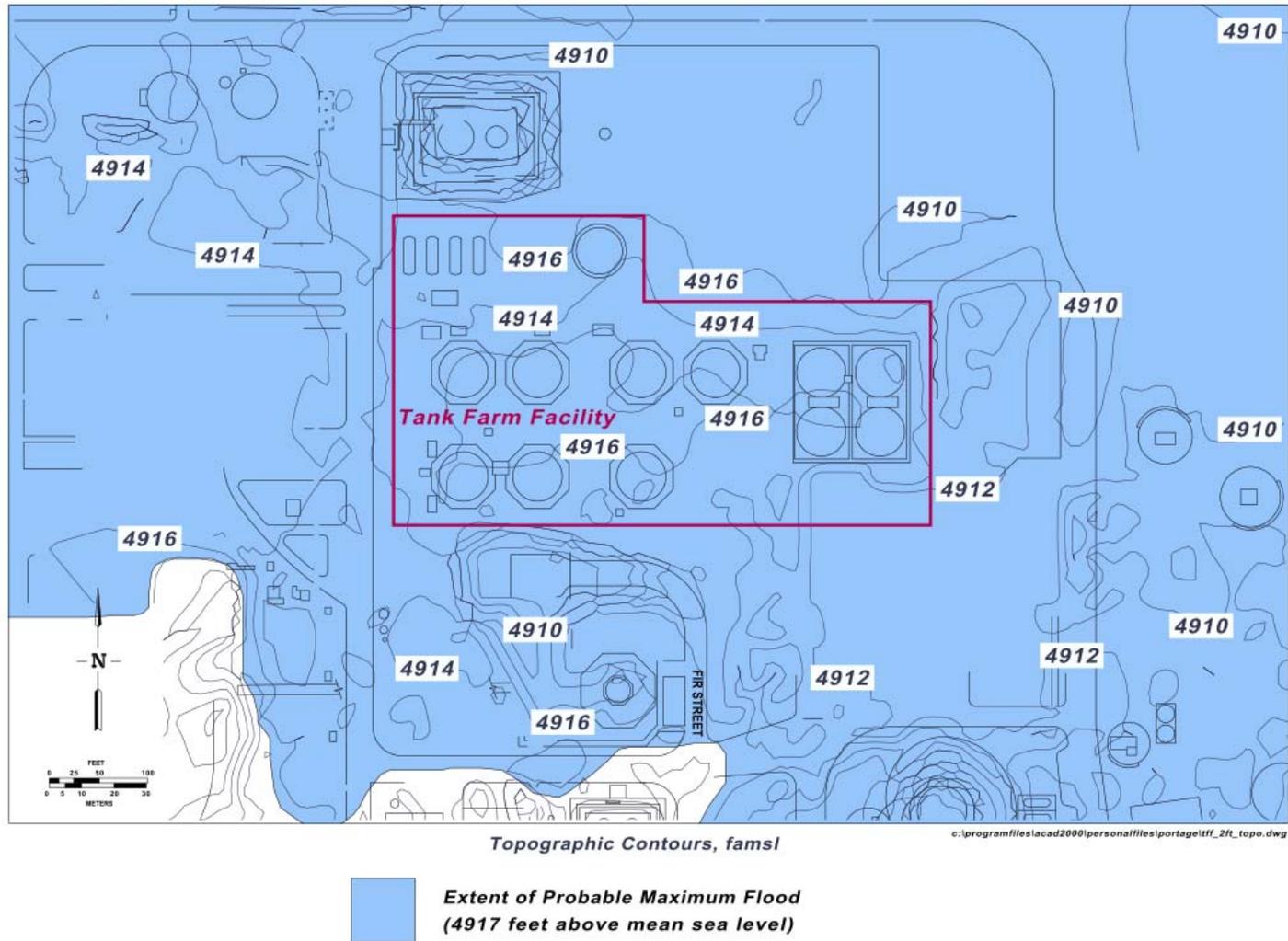


Figure RAI-9-2. Topographic map of the Tank Farm Facility (flood depth is 4,917 ft in elevation).

DOE-ID should provide information regarding their assumptions for landscape evolution, which are relevant to the severity and implications of flooding at the TFF. This information may include assumptions by the Army Corps of Engineers regarding the useful lifetime of Mackay Dam, and whether Mackay Dam is assumed to be dredged repeatedly so that it does not fill with sediment. This information may include an analysis of flooding scenarios and effect on dose if Mackay Dam was no longer maintained starting at $t = 500$ years (not a one-time dam failure, but rather the dam is assumed out-of-commission over the long term). DOE-ID should provide its assumptions and technical bases regarding whether the spreading areas at INL will not fill with sediment such that capacity is reduced.

NRC requested that INL provide information regarding the PA assumptions for landscape evolution for the flooding analysis. The flooding analyses presented in the PA are based on the existing Mackay Dam failure and the existing Flood Diversion Facility (FDF) spreading areas in place and maintained. The reservoir behind Mackay Dam has not been dredged to remove sediment and dredging is not anticipated in the future (Beller 2006). The useful life of the reservoir is not known. If it were assumed that the landscape had evolved over time due to sediment infilling of Mackay Dam and the FDF, then the flood analysis would be solely due to runoff from a probable maximum precipitation (PMP) event. It is important to remember that the probable maximum flood (PMF) event used in the PA assumed that the Mackay Dam failed during a PMP event. Mackay Dam has a storage capacity of $5.5 \times 10^7 \text{ m}^3$. These flood waters are assumed to be released from the dam due to overtopping and subsequent failure of the dam. The FDF spreading areas have a capacity of $2.2 \times 10^7 \text{ m}^3$. It is reasonable to assume that if the FDF spreading areas are filled with sediments over time so that the ability to divert flood waters is nonexistent, then it also must be assumed that Mackay Dam has been filled with sediments and the storage capacity has been reduced to a natural stream channel. In this case, the flood analysis is reduced to a simple PMP event. In other words, the storage capacity loss from Mackay Dam would not be included in the flood analysis. Therefore, the difference between the dam storage capacity and the FDF spreading area capacity of $3.3 \times 10^7 \text{ m}^3$ would not be included in the PMP analysis. Therefore, the PMF presented in the PA is considered bounding for potential flood changes due to landscape evolution over time.

DOE-ID should provide an estimate regarding the longevity of the effectiveness of the flood diversion facility in the absence of maintenance at the end of the period of institutional control.

The longevity and effectiveness of the FDF would be minimal after the loss of institutional control. However, as explained above, the flooding analysis is bounding for the case where Mackay Dam and the FDF are no longer in operation. In addition, due to the length of time for the degradation of the tanks and vaults (i.e., 100–500 years), it is unlikely that Mackay Dam would be in operation and the FDF would be out of operation. It is more reasonable to assume that after the institutional control period Mackay Dam and the FDF would no longer be viable structures.

DOE-ID should provide the text on page 2-35 of the DOE-ID (2003b) that is currently hidden behind a figure.

The hidden text on page 2-35 from the TFF PA (DOE-ID 2003) is attached.

DOE-ID should provide information that demonstrates that the impacts of the probable maximum flooding condition on the INTEC facility have been bounded.

As stated previously, the probable maximum flooding condition presented in the PA is bounding for additional flooding considerations such as landscape evolution. The probable maximum flooding scenario also bounds the future sediment infilling and the loss of Mackay Dam and the FDF spreading areas.

DOE-ID should provide the analysis by Dunnivant et al., (1998) on which they, in part, base their expectation of minimal impact. DOE-ID should address the potential for higher levels of contamination to persist for a longer period of time in the flooding scenario.

The Dunnivant et al. (1998) reference is being provided in the response to NRC Clarifying Request 21. The TFF contains 13 m of alluvial sediments resting on top of a basalt layer. This alluvium is assumed to rapidly transmit the flood water to the underlying basalt. Dunnivant et al. (1998) performed a large-scale infiltration test for basalt with ponded water, which resulted in an average wetting front advance of 5 m/day. Dunnivant et al. (1998) note that the wetting front was non-uniform and fracture flow may have resulted in the wetting front advancing rapidly in some areas and slower in others. However, the wetting front advance was an average rate that included the contributions from all potential flow mechanisms in the basalts. As noted by Dunnivant et al. (1998), the advance of the wetting front was interrupted by the location of the interbed sediments, which caused perched water zones. The interbed sediments in the PA two-dimensional model also result in the interruption of the wetting front for the PA flooding analysis. The subsequent perched water zones created by the flood waters also resulted in a dilution of the contaminants migrating from tanks during the flood analysis. This resulted in a dilution of the increased contaminant releases from the tanks in the flood analysis. Large-scale increases in recharge, due to a major flooding event, will result in a dilution effect as illustrated in the PA (see Figure 7-15; attached).

Peak concentrations at the water table are lower for a flooding scenario compared with steady-state simulations. Larger floods provide more water for recharge and subsequent dilution of contaminants. The PA (see Figure 7-16; attached) shows the impact on perched water zones due to the PMF flooding event. The increase in the number, size, and location of perched water zones will increase the dilutions of the contaminants.

DOE-ID should provide an analysis that clearly demonstrates the expected amount of time water might pond at the surface of the TFF after a dam failure or probable maximum flood.

The NRC notes that PA Section 2.1.5.3.6 states “The flood hydrograph for the base-case discharge shows a peak flow that lasts only 7.5 hours (Figure 2-19).” The NRC also notes that the PA states “Since the INTEC facility is located at the boundaries of the flood, the area would be inundated for a lesser period of time (p. 2-38).” The NRC states “The information conveyed in the second sentence called out above does not necessarily follow from the information conveyed in the first sentence: The duration of peak discharge does not necessarily imply how long water may pond at the surface of the TFF or elsewhere (see also Section 7.4.2, page 7-26).” The INL agrees with NRC that the statement is in error. The peak flow rate duration is not an indication of how long water may pond at the TFF. The peak flow rate would be expected to be less at the margins of the flood channel.

The NRC’s concern that an increase in the ponded water residence time will result in higher infiltration rates and subsurface transport is addressed in Section 7 of the PA. Large-scale increases in recharge, due to a major flooding event, will result in a dilution effect as illustrated in the PA (see Figure 7-15; attached). Peak concentrations at the water table are lower for a flooding scenario compared with steady-state simulations. Concentrations for the flooding scenario after the peak has been reached are slightly higher than the steady-state simulation due to the large lateral spreading of the contaminant in the increased perched water zones. However, these small concentration increases in the later years are still bounded by the PA analysis. Larger floods provide more water for recharge and subsequent dilution of contaminants. The PA (see Figure 7-16; attached) shows the impact on perched water zones due to the PMF flooding event. The increase in the number, size, and location of perched water zones will increase the dilutions of the contaminants.

References:

Beller, John, Portage, Inc., Personal Communication with Bob Duke, Big Lost River Irrigation District Watermaster, March 1, 2006.

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

Dunnivant, F. M., et al., 1998, “Water and Radioactive Tracer Flow in a Heterogeneous Field-Scale System,” *Ground Water*, Vol. 36, No. 6, pp. 949-958.

The following text was hidden behind a figure on page 2-35 in the electronic copy of the PA (DOE-ID 2003) and is provided here as requested by NRC.

PMF-induced overtopping failure of the Mackay Dam caused by an extreme precipitation event, the general storm PMP. Figure 2-17 represents the PMF hydrograph, and Figure 2-18 is the inundation map for the PMF-induced failure of the Mackay Dam. Table 2-7 lists information on the peak water surface elevation, peak flow, water velocity, and time of arrival at several downstream locations for this dam failure scenario. The following discussion was taken from the *TMI-2 Safety Analysis Report* (INEEL 1999b), which is applicable to the assessment of the INTEC TFF maximum flood.

2.1.5.3.2 Probable Maximum Precipitation—The general storm PMP for the drainage basin above Mackay Dam resulted from a 48-hr general storm in June, preceded three days before by an antecedent storm with a magnitude of 40% of the 48-hr storm. This scenario provides for no flow losses to the ground in order to be conservative and represent situations in which the ground may be frozen or fully saturated. The peak flow for the PMF is 82,100 ft³/s, occurring 154 hr after the beginning of the storm. The PMF estimate falls within the 50,000–200,000 ft³/s Myers envelope curve used by the U.S. Army Corps of Engineers (INEEL 1999b). The PMF peak flow is almost 20 times higher than the highest flow of 4,420 ft³/s recorded at Howell

Ranch, a USGS station located approximately 17 mi (27 km) northwest of the dam. The PMF is based on the maximum potential for critical hydro-meteorological conditions to occur, not on probabilities or historical flood frequencies.

2.1.5.3.3 Precipitation Losses—The Big Lost River leaves the mountains at Arco. Below this point, the topography and drainage characteristics change along the river. The area is a low, flat plain with basalt bedrock. The drainage from most of the area in Pioneer Basin is integrated with the Big Lost River. Locally, some depressions in the basalt receive intermittent runoff. There is seldom enough precipitation in this area to exceed the infiltration capacity of the soil to create intermittent streams to the Big Lost River.

2.1.5.3.4 Runoff Model—The combined Big Lost River Basin and Pioneer Basin range in elevation from 4,784 ft (1,458 m) to over 12,600 ft (3,840 m) amsl. Thus, this area has over 7,546 ft (2,300. m) of relief, resulting in large differences in temperature and climate at any given time. The low land in the Pioneer Basin is subjected to periods of warm wind, rain, and snowmelt during the winter months. These conditions cause runoff and minor flooding in the lower basins during regional storms and

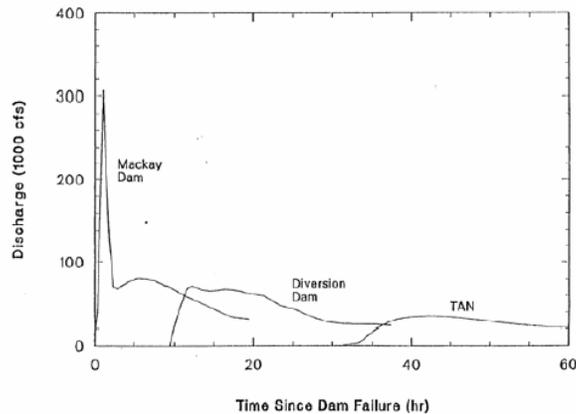


Figure 2-17. Probable maximum flood hydrograph (INEEL 1999b).

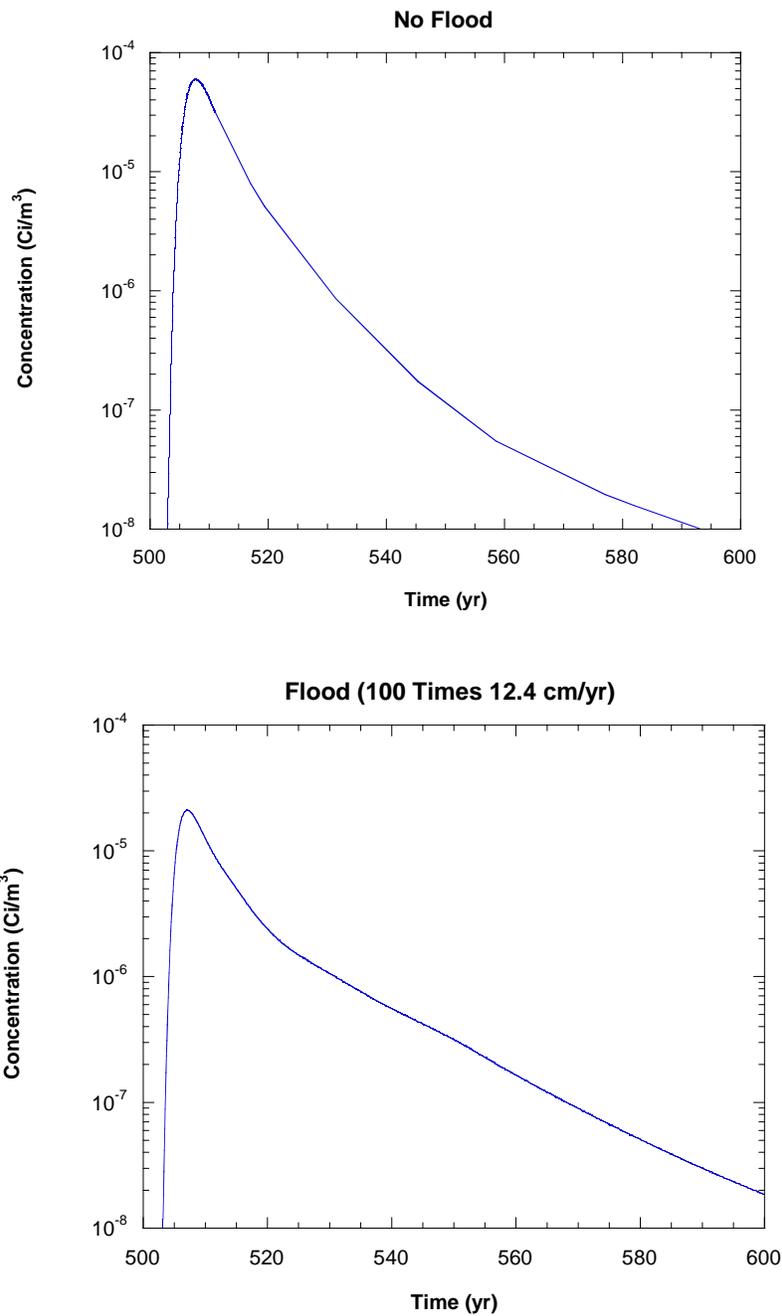


Figure 7-15. Comparison of radionuclide concentrations at the water table for non-flooding and flooding conditions.^c

c. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

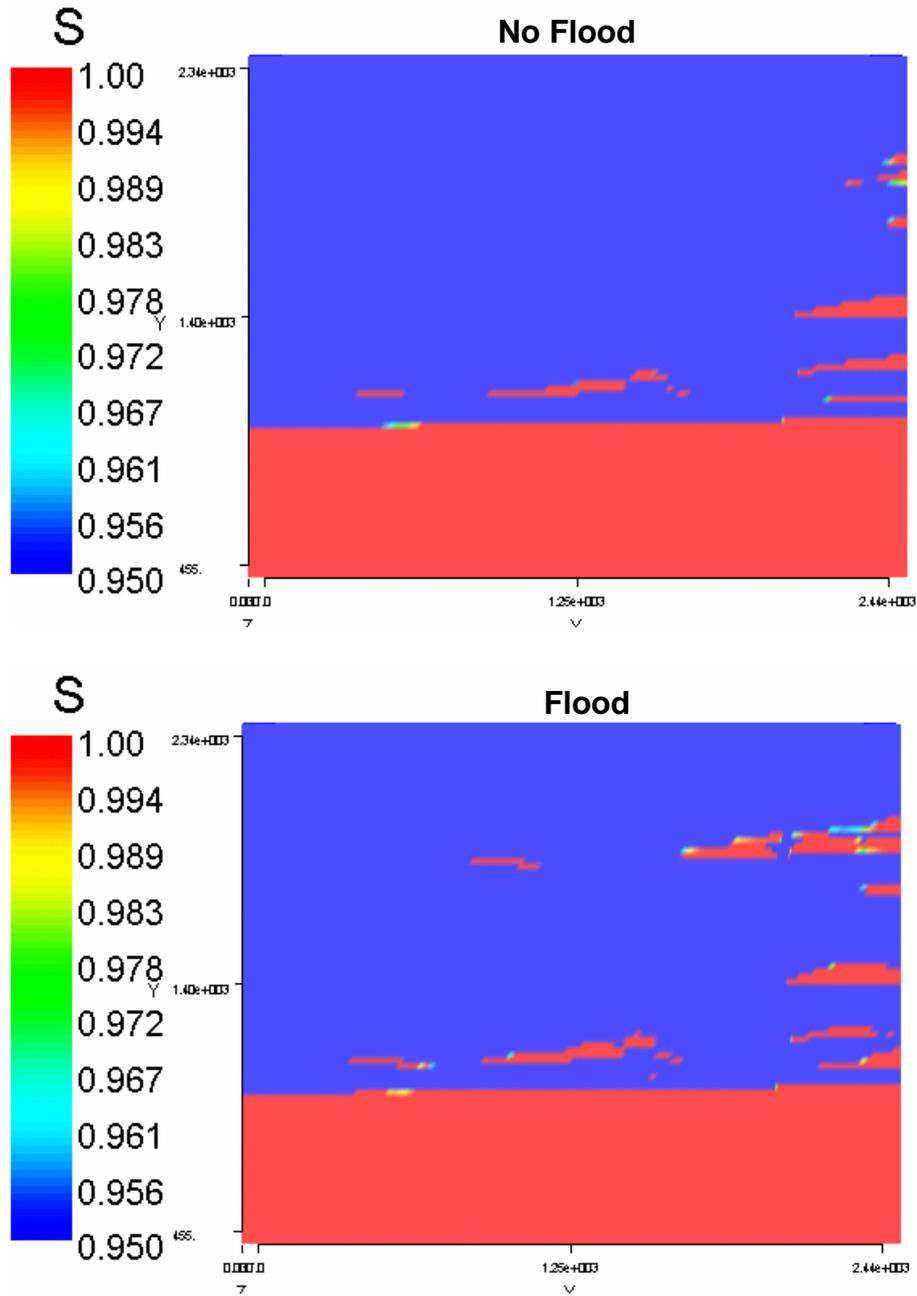


Figure 7-16. Extent of perched water due to non-flooding conditions and flooding conditions (saturated conditions are denoted as red).^d

d. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

NRC

Comment 10:

Based on review of new information, it is not clear that the current two-dimensional unsaturated zone groundwater model (DOE-ID, 2003b, p. 3-39) is adequate for estimating the all pathways dose to a member of the public, and additional information is needed to determine if results from the two-dimensional model are conservative.

Basis:

The three-dimensional geometry of the *intra*-basalt fracture system and inter-basalt rubble zones is important (Faybishenko et al., 2001), but the geometry is not accounted for in the DOE ID two-dimensional groundwater model. At the scale of the TFF groundwater model, fracture patterns likely are not isotropic in map view. Basalt flows typically have a lobate distribution 20-60-m wide and up to 1,000-m long (Sorenson, et al., 1996). Detailed vertical stratigraphy in two-dimensions cannot account for tortuous flow paths expected in three dimensions due to the presence of basalt flow fingers or lenticulae, where upper margins of flows are rubbly and fractured (due to fast cooling) whereas flow interiors and bottoms are massive and less fractured (due to slow cooling). Rubble zones associated with individual flow tops may also conduct lateral groundwater flow, and the decreasing fracture spacing with depth into a single basalt flow leads to fewer flow paths and potential funneling—a process observed to continue with increasing depth as water leaves shallow basalt flows and moves into deeper basalt flows (Faybishenko, et al., 2000). Physical fast transport pathways could occur in the study area because of heterogeneity in active lateral basalt rubble pathways and lateral sedimentary interbed pathways, and heterogeneity in active subvertical fracture pathways. Multiple peaks in breakthrough curves during field tests at INL suggest water follows a number of different transport pathways (e.g., Jones, et al., 2004 and LSIT results). While it may be correct in part to suggest that lack of transverse dispersion in a two-dimensional model could contribute to achieving a conservative dose estimate, adopting this methodology neglects that a two-dimensional model may be prone to artificially “trapping” water and radionuclide contaminants because the connectivity of actively conducting portions of the model domain is too low in the absence of the third dimension (Doughty, 2000). To the extent that radionuclide-bearing water may be artificially trapped in the model domain, this methodology is nonconservative. Capturing a large level of detail in two dimensions may be of less value than capturing an appropriate level of detail in three dimensions.

Wood and Faybishenko (2001) say that at INL “without an overall understanding of the geometry and physics of flow at the macro-scale, we can describe what is seen, but cannot use this information for making further predictions, and therefore cannot make meaningful assessments of contaminant transport.”

Previous recommendations of Lawrence Berkeley National Laboratory and INL staff (Faybishenko, et al., 2001; Doughty 2000) for achieving realism in vadose zone flow and transport models of INL include simulating the structured, nonrandom geometry of basalt fracture patterns using deterministic fracture models in addition to simulating flow in the system using stochastic hydrologic parameter models.

Path Forward:

DOE-ID should provide any available information on the geometry of the fracture system in the unsaturated zone downgradient from the TFF. DOE-ID should provide any available information on the anisotropy of basalt flows in the TFF vicinity, and to what degree the long lateral axis of basalt flows are oriented obliquely to hydraulic gradients at the scale of the TFF groundwater model. Information and data from other sites at INL (e.g., Radioactive Waste Management Complex, Large Scale Infiltration Test, and Vadose Zone Research Park) may provide a supporting basis for the selected modeling approach. DOE-ID should provide information that demonstrates that they are either realistic or conservative in their dose estimates for the groundwater pathway.

Response:

Based on review of new information, it is not clear that the current two-dimensional unsaturated zone groundwater model (DOE-ID, 2003b, p. 3-39) is adequate for estimating the all pathways dose to a member of the public, and additional information is needed to determine if results from the two-dimensional model are conservative.

The selection of a two-dimensional model to simulate water and contaminant transport at the INTEC facility was based on the complexity of the site geology and the type and availability of hydraulic data. Previous modeling programs at the INL were reviewed, including three-dimensional models. A three-dimensional model was considered; however, it was determined that a two-dimensional model would provide the accuracy necessary for defining the numerous basalt and unconsolidated deposits that control water movement in the Snake River Plain. The use of a three-dimensional model representing a large area such as INTEC and the surrounding area requires that the analyst considers both the controlling geologic features and the model computational requirements. Therefore, in three-dimensional modeling, simplifying assumptions are generally made, such as grouping interbed sedimentary units and basalt units. This grouping is generally done to reduce the discretization (i.e., number of computational nodes) required for representing the geologic units in a three-dimensional model. This results in the loss of both the accurate delineation of interbeds and the important discontinuities within the interbeds that allow a reasonable approximation of downward movement of contaminant plumes.

Recent publications of studies conducted at the INL support the conceptual model used as the basis for predictive modeling in the PA (DOE-ID 2003). Mattson et al. (2004) interpretations on vadose zone water movement based on large-scale field test and long-term monitoring “suggest that sedimentary interbeds with the basalt formations have a larger effect on the creation of perched water and its lateral extent.” Nimmo et al. (2004) provide further support to the PA conceptual model that “sedimentary interbeds are likely to impede vertical flow and to cause preferential flow...their layered structure may be conducive to funneled flow.” It is clear from recent publications that the conceptual model that forms the basis of the PA model agrees with the earlier publications describing the controlling hydrostratigraphic features for unsaturated flow and the principles described in the PA for constructing the two-dimensional predictive model.

Additional support for the PA model is provided in Bartholomay et al. (1997), which shows the location of the percolation ponds (see Figure 3; attached) and selected monitoring wells. In addition, Bartholomay et al. (1997) present chloride data for several monitoring wells surrounding the percolation ponds (see Table 7; attached). Wells 40 and 47, located approximately 1,000 m north of the percolation ponds but near the disposal well, show elevated chloride in late 1983, presumably due to the disposal well, but water quality data from 1984 to 1995, after the disposal well was no longer in service, showed chloride concentrations that are near background water quality. These data indicate that wastewater from the disposal well impacted Wells 40 and 47 in 1984, but after disposal operations switched to the percolation ponds, infiltrating water from the ponds no longer impacted these wells.

Using the tritium plume predicted by the two-dimensional model calibration simulation from the PA (DOE-ID 2003) (see Figure 3-20; attached), and assuming that the distribution of chloride discharged to the percolation ponds is similar to the tritium distribution since both analytes are conservative, the predicted plume is approximately 350 m north, or upgradient, of the percolation ponds. Once again, Wells 40 and 47 are approximately 1,000 m north of the percolation ponds and, based on water quality data, are beyond the influence of chloride infiltration from the ponds—in agreement with the PA model calibration predictions. Well 59, located upgradient of the percolation ponds and within the 350-m upgradient distance predicted by the model, shows elevated chloride concentrations—once again matching the PA model predictions. As expected, all of the downgradient wells, 37, 57, 111, and 113, show elevated concentration of chloride—consistent with the PA model predictions.

Three-dimensional modeling studies at the INL Site were conducted by Unger et al. (2004) using 23 by 21 by 21 nodes of dimension 1.0 by 1.0 by 1.0 m in the *x*, *y*, and *z* directions. At this local scale it is possible to imbed fractures into the porous media, but at the regional scale of the INTEC site, 2,500 by 200 m, this approach is impractical. In addition, there is little information on the location, density, and aperture of fractures, especially at depth, to reliably simulate the fracture network at the INTEC facility. Instead, the PA used the conservative assumption of high vertical hydraulic conductivities, ranging up to 100 m/d, to rapidly move water through fractured units in the subsurface. Magnuson (2004) compared unsaturated flow curves at a field test site and found the Cory-based curves, similar to those used in the PA, resulted in a conservative approach to simulating flow in the unsaturated zone. Furthermore, Magnuson used a saturated hydraulic conductivity of approximately 0.1 m/d. The PA two-dimensional model used a minimum vertical hydraulic conductivity for the fractured basalt of 10 m/d, which is two orders of magnitude higher to allow water to move rapidly through the basalts.

The current understanding of the vadose zone indicates that the stratigraphy underlying the site and recharge from the Big Lost River have the greatest influence on the rate and direction of contaminant transport. Some of the best information available for the site includes a series of U.S. geologic reports by Anderson (1991). Detailed geologic cross-sections are available for the INL Site

showing the structure control of interbeds. Using a two-dimensional model allowed sufficient detail to accurately digitize the stratigraphy of the site. This approach is supported by recent work by Mattson et al. (2004), which states “long-term monitoring and field tests results suggest that sedimentary interbeds within the basalt formations have a large effect on the creation of perched water and its lateral extent.” Mattson et al. further state, “Gaps in the sedimentary interbeds will allow some of the recharge water to bypass the interbeds and rapidly continue toward the aquifer.” This scenario is exactly how the PA two-dimensional model behaves during flow and transport simulations.

The conservative nature of the PA two-dimensional model is illustrated in the following description. The PA takes a 1-m-thick vertical slice of the center of a contaminant plume emanating from the tanks. This 1-m slice represents the highest concentrations of contaminants at the maximum radius of two tanks. Only longitudinal and transverse dispersivity in the x - z plane are considered to lower the concentrations of contaminants as the plume spreads during transport. Lateral dispersivity in the y -direction is ignored as well as diffusion from the higher concentrations in the 1-m slice, both conservative assumptions. The PA two-dimensional model incorporates detailed stratigraphic information that controls vadose zone water movement and places the highest contaminant concentration to the groundwater in a “funnel” manner as shown in Figure 4-2 (attached).

The NRC’s concern that “Physical fast transport pathways could occur in the study area because of heterogeneity in active lateral basalt rubble pathways and lateral sedimentary interbed pathways, and heterogeneity in active subvertical fracture pathways” would actually result in the lateral spread of contaminants and reduce the funneling effect observed in the two-dimensional model.

Water-producing zones in basalt aquifers commonly occur along basalt contacts where rubble zones or permeable sedimentary units occur. Thick basalt beds commonly exhibit significantly lower permeabilities. These lateral or horizontal preferential flow zones would spread water and contaminants over a large area, dispersing the plume and resulting in lower concentration levels entering the groundwater. The PA approach does not take advantage of these flow characteristics but rather allows water to move rapidly through the basalts with a dominant vertical anisotropy of 10 to 100 times greater vertical permeabilities compared to the corresponding horizontal permeabilities. Water trapped in dead-end fractures or by capillary barriers at the porous/fracture boundary is not included in the analysis, resulting in a very conservative modeling approach.

The two-dimensional slice of the problem domain is aligned in a north-south direction. The Big Lost River, the major source of recharge in the area, is located on the northern boundary of the problem domain. Recharge from the Big Lost River results in a dominant southern flow component for the unsaturated zone in approximately the same direction as the model domain. In addition, regional groundwater flow in the Snake River Plain Aquifer flows north to south, also approximately aligned with the model domain.

The three-dimensional geometry of the intra-basalt fracture system and inter-basalt rubble zones is important (Faybishenko et al., 2001), but the geometry is not accounted for in the DOE-ID two-dimensional groundwater model. At the scale of the TFF groundwater model, fracture patterns likely are not isotropic in map view.

In map view, there can be anisotropy in the vertical and horizontal planes. Vertical fractures are the dominant hydraulic feature of the basalt units. The PA model incorporates this feature by assigning a dominant vertical anisotropy that is one to two orders of magnitude higher than the horizontal components. It is important to reiterate that preferential flow in the horizontal direction, as implied by the NRC comments on the importance of rubble zones, would spread the contaminant plume horizontally, resulting in trapped water along the rubble zones and in dead-end fractures and pores, a feature that is not incorporated into the PA model and, as such, is a conservative approach. If a dominant fracture pattern of northwest/southeast results in the preferential movement of water in the horizontal direction that does not coincide with the alignment of the two-dimensional model domain, then the controlling hydraulic parameters in the unsaturated zones need to be considered. First, if it is assumed that vertical fractures are the dominant hydraulic feature for the basalts, then gravity drainage is the dominant force moving water through the basalts. The vertical anisotropy and the moisture characteristic curves that release water with minor pressure changes that are inherent in the PA model in order to replicate gravity drainage as the dominant hydraulic force moving water through the basalts. It is difficult to conceive of a viable mechanism that could move water horizontally in fractures that are vertical but aligned in a dominant horizontal direction. For this conceptual model to be valid, then it must be assumed that capillarity will move water along the fractures trending northwest with a force greater than gravitational forces, an assumption without technical merit. Since there is no porous structure in the fractures, gravity drainage is dominant and horizontal water movement in fractures with no porous structure is nonexistent.

Basalt flows typically have a lobate distribution 20-60-m wide and up to 1,000-m long (Sorenson, et al., 1996). Detailed vertical stratigraphy in two-dimensions cannot account for tortuous flow paths expected in three dimensions due to the presence of basalt flow fingers or lenticulae, where upper margins of flows are rubbly and fractured (due to fast cooling) whereas flow interiors and bottoms are massive and less fractured (due to slow cooling). Rubble zones associated with individual flow tops may also conduct lateral groundwater flow, and the decreasing fracture spacing with depth into a single basalt flow leads to fewer flow paths and potential funneling—a process observed to continue with increasing depth as water leaves shallow basalt flows and moves into deeper basalt flows (Faybishenko, et al., 2000). Physical fast transport pathways could occur in the study area because of heterogeneity in active lateral basalt rubble pathways and lateral sedimentary interbed pathways, and heterogeneity in active subvertical fracture pathways. Multiple peaks in breakthrough curves during field tests at INL suggest water follows a number of different transport pathways (e.g., Jones, et al., 2004 and LSIT results). While it may be correct in part to suggest that lack of transverse dispersion in a two-dimensional model could

contribute to achieving a conservative dose estimate, adopting this methodology neglects that a two-dimensional model may be prone to artificially “trapping” water and radionuclide contaminants because the connectivity of actively conducting portions of the model domain is too low in the absence of the third dimension (Doughty, 2000). To the extent that radionuclide-bearing water may be artificially trapped in the model domain, this methodology is nonconservative. Capturing a large level of detail in two dimensions may be of less value than capturing an appropriate level of detail in three dimensions.

Wood and Faybishenko (2001) say that at INL “without an overall understanding of the geometry and physics of flow at the macro-scale, we can describe what is seen, but cannot use this information for making further predictions, and therefore cannot make meaningful assessments of contaminant transport.”

Anderson (1991) provides location maps and the geologic cross-sections (see Figures 4 and 7; attached), which illustrate a north-south cross-section and a roughly east-west cross-section of the area in and around the INTEC facility. Comparing the area between Wells CPP-1 and PW-1, on the north-south cross-section (Figure 7; attached), with the area between Wells 43 and 49 on the east-west cross-section (Figure 4; attached), shows that there is no preferential dip direction. The statement that basalts have a lobate distribution of 20–60 m in width and up to 1,000 m in length is clearly not the case for majority of the basalt flows as illustrated in both cross-sections, which show massive basalt structures extending over several thousand meters in every direction. By comparing the two cross-sections, there is no definitive structure that would yield preferential flow in either direction.

There is no evidence that “trapping” of water is occurring in the two-dimensional model. Extensive model calibration matched the known extent of upper perched water zones that are associated with low permeable sedimentary interbeds. As shown in the PA (see Figure 3-19; attached), perched water zones are associated with recharge from the Big Lost River and the percolation ponds. The break in the two upper perched zones occurs near the discontinuity of the upper sedimentary interbeds in agreement with the two-dimensional model. There is no evidence from the distribution of the perched water zones illustrated in Figure 3-19 (attached) that indicates preferential flow is occurring either east or west of the two-dimensional model domain since the continuity of the perched zones trend in an east-west direction. If there was a significant preferential pathway located either east or west of the model domain, then the extent of the upper perch zones would terminate in that area, which is not the case based on water-level data.

Previous recommendations of Lawrence Berkeley National Laboratory and INL staff (Faybishenko, et al., 2001; Doughty 2000) for achieving realism in vadose zone flow and transport models of INL include simulating the structured, nonrandom geometry of basalt fracture patterns using deterministic fracture models in addition to simulating flow in the system using stochastic hydrologic parameter models.

The preceding responses to this NRC comment have clarified the basis of the deterministic model used in the PA, which utilized the most reliable site information. Stochastic models are gaining favor at selected sites, such as Yucca Mountain, where extensive field studies have provided a reasonable level of detail of the fracture geometry sufficient to reduce the uncertainties inherent in the stochastic approach. At the INL Site, there are only a limited amount of direct technical data on the location, density, and aperture of fractures in the numerous basalt units underlying the site. Only limited coring of the bedrock has occurred at the site and trying to characterize vertical fractures with vertical borings is inherently limited. Using a stochastic approach to estimate the distribution of fractures in the numerous basalt flows that exhibit individual characteristics based on thickness, depositional characteristics, and weathering patterns with limited field data presents a level of uncertainty that would be difficult if not impossible to quantify. The deterministic model used in the PA utilized reliable geology and hydraulic data that reduce uncertainty. The few boreholes drilled at the site, to accurately define the fracture distribution, cannot be relied upon to quantify the distribution of fractures over thousands of square meters and the numerous distinctive basalt units. It would be difficult to quantify or bound the uncertainty of a stochastic model at the INL Site due to the lack of site data on fracture characteristics.

DOE-ID should provide any available information on the geometry of the fracture system in the unsaturated zone downgradient from the TFF. DOE-ID should provide any available information on the anisotropy of basalt flows in the TFF vicinity, and to what degree the long lateral axis of basalt flows are oriented obliquely to hydraulic gradients at the scale of the TFF groundwater model. Information and data from other sites at INL (e.g., Radioactive Waste Management Complex, Large Scale Infiltration Test, and Vadose Zone Research Park) may provide a supporting basis for the selected modeling approach. DOE-ID should provide information that demonstrates that they are either realistic or conservative in their dose estimates for the groundwater pathway.

There is limited information available for the direct measurement of field-scale anisotropy for the site. Even recent work at the Box Canyon site by Unger et al. (2004) using pneumatic tests concedes that the method only estimated the mean permeability of the various test zones and does little to determine spatial continuity in the surrounding field. The dominating forces controlling water movement in the vadose zone are recharged from the Big Lost River combined with the low-permeable discontinuous sedimentary interbeds. The Big Lost River provides a hydraulic boundary that forces water to the south, parallel to the two-dimensional slice used in the model. In comparison, basalt fracture anisotropy provides only a minor influence that is not well quantified.

Consequently, vertical permeabilities for the basalt were set at high values to rapidly conduct water through the basalts for a conservative PA modeling approach. The model results presented in the PA show a funneled plume that rapidly moves through the unsaturated zone and impacts the water table at concentrations that are conservative.

The calibrated and verified PA model is supported by the most reliable site data including lithologic samples, water-level data for perched wells, and contaminant discharges at the site. David W. Esh, Anna H. Bradford, Kristina L. Banovac, and B. Jennifer Davis of NRC, Washington, D.C., published a paper entitled, "Risks and Uncertainties Associated with High-Level Waste Tank Closure" (2005). In this paper, the NRC staff developed a generic PA model, applicable to high-level waste tank closure, which NRC utilized to complete its independent review. The model was developed using the generic simulation software, GoldSim, because of its probabilistic capabilities and its adaptability to different problems. The findings of this report are:

The PA model has proven to be an effective tool in risk-informing reviews of the HLW tank closure impacts.

The PA model provides an accurate representation of hydrologic processes controlling water movement and contaminant transport in the Snake River Plain underlying the INTEC facility and is verified by reliable site data and independent probabilistic modeling studies.

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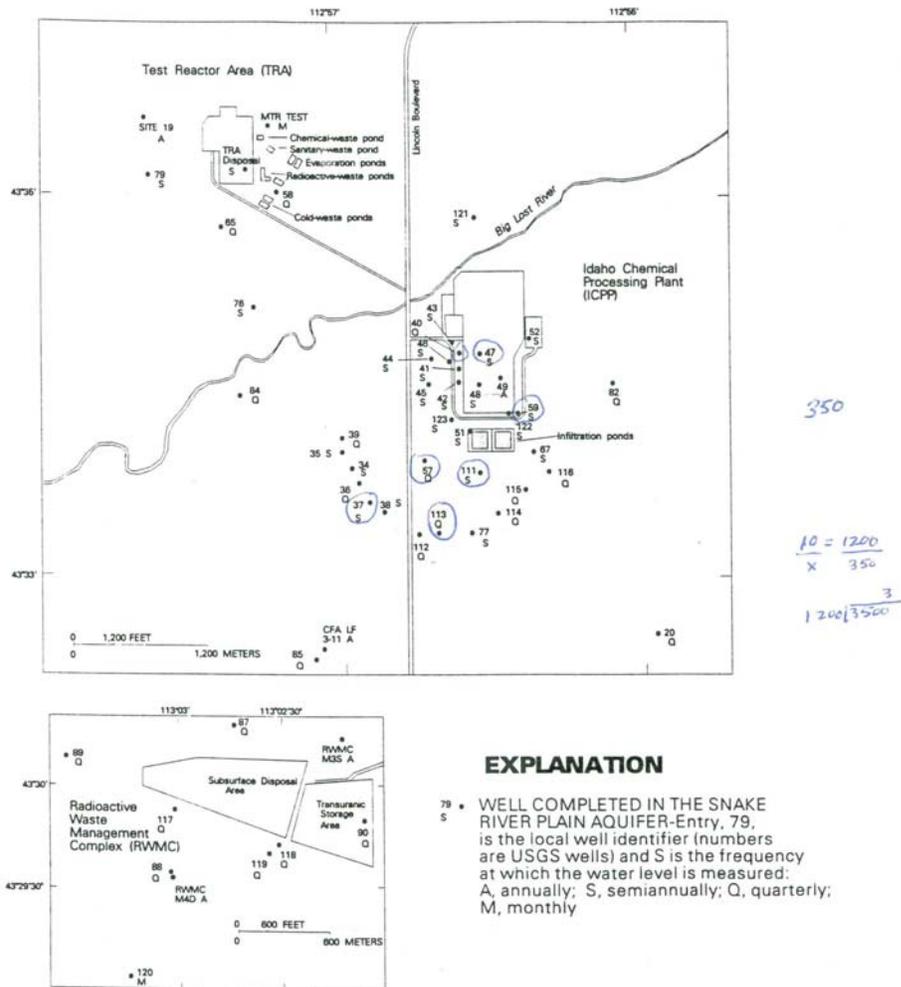


Figure 3. Location of wells and frequency of water-level measurements (as of December 1995) in the Snake River Plain aquifer, Test Reactor Area, Idaho Chemical Processing Plant [now INTEC], and Radioactive Waste Management Complex. [Note: The percolation ponds are designated as infiltration ponds on the figure.]^e

e. Bartholomay, R. C., B. J. Tucker, D. J. Ackerman, and M. J. Liszewski, 1997, *Hydrologic Conditions and Distribution of Selected Radiochemical and Chemical Constituents in Water, Snake River Plain Aquifer, Idaho National Engineering Laboratory, Idaho, 1992 through 1995*, DOE/ID-22137, U.S. Geological Survey Water Resources Investigations 97-4086, April 1997.

Table 7. Chloride concentrations in water from selected wells at the Idaho National Engineering Laboratory [now INL], 1981-95 [concentrations in mg/L].^f

Well CFA-1		Well 37		Well 40		Well 47	
Date	Concentration	Date	Concentration	Date	Concentration	Date	Concentration
10/13/81	82	10/07/81	66	10/12/81	150	10/81	NS
10/11/82	86	10/07/82	85	10/07/82	150	10/07/82	32
10/06/83	78	10/10/83	59	10/10/83	150	10/17/83	160
10/12/84	53	10/10/84	53	10/16/84	44	10/23/84	24
10/25/85	51	10/28/85	37	10/29/85	23	10/29/85	21
10/31/86	78	10/27/86	50	10/30/86	23	10/29/86	23
10/15/87	83	10/05/87	70	10/13/87	24	10/26/87	23
10/28/88	86	10/07/88	86	11/09/88	23	9/30/88	23
10/26/89	93	9/29/89	92	10/18/89	23	10/19/89	24
10/15/90	86	10/17/90	78	10/16/90	27	10/31/90	31
10/10/91	100	10/07/91	75	10/29/91	31	10/24/91	30
10/08/92	100	10/02/92	120	11/17/92	29	10/21/92	36
10/93	NS	10/21/93	140	10/08/93	30	10/26/93	31
10/94	NS	10/07/94	140	10/18/94	30	10/19/94	38
10/11/95	100	10/11/95	100	10/19/95	37	10/16/95	35

Well 57		Well 59		Well 111		Well 113	
Date	Concentration	Date	Concentration	Date	Concentration	Date	Concentration
10/81	NS	10/06/81	37	--	--	--	--
10/06/82	140	10/06/82	47	--	--	--	--
10/13/83	130	10/13/83	60	--	--	--	--
10/10/84	68	10/10/84	28	--	--	--	--
10/29/85	42	10/30/85	40	11/05/85	50	11/01/85	67
11/14/86	64	11/14/86	35	10/27/86	100	10/27/86	110
10/09/87	67	10/06/87	23	9/25/87	120	10/02/87	150
10/05/88	74	10/21/88	24	10/04/88	120	9/27/88	160
12/22/89	67	10/23/89	22	10/04/89	120	10/06/89	160
10/29/90	85	10/12/90	44	9/27/90	140	10/03/90	180
10/24/91	120	10/21/91	190	10/25/91	110	10/08/91	170
10/06/92	160	10/23/92	67	10/09/92	130	10/22/92	200
10/12/93	180	10/25/93	44	10/21/93	140	10/23/93	220
11/03/94	170	11/01/94	47	10/13/94	130	10/17/94	210
10/11/95	150	10/23/95	150	10/26/95	100	10/17/95	190

f. Bartholomay, R. C., B. J. Tucker, D. J. Ackerman, and M. J. Liszewski, 1997, *Hydrologic Conditions and Distribution of Selected Radiochemical and Chemical Constituents in Water, Snake River Plain Aquifer, Idaho National Engineering Laboratory, Idaho, 1992 through 1995*, DOE/ID-22137, U.S. Geological Survey Water Resources Investigations 97-4086, April 1997.

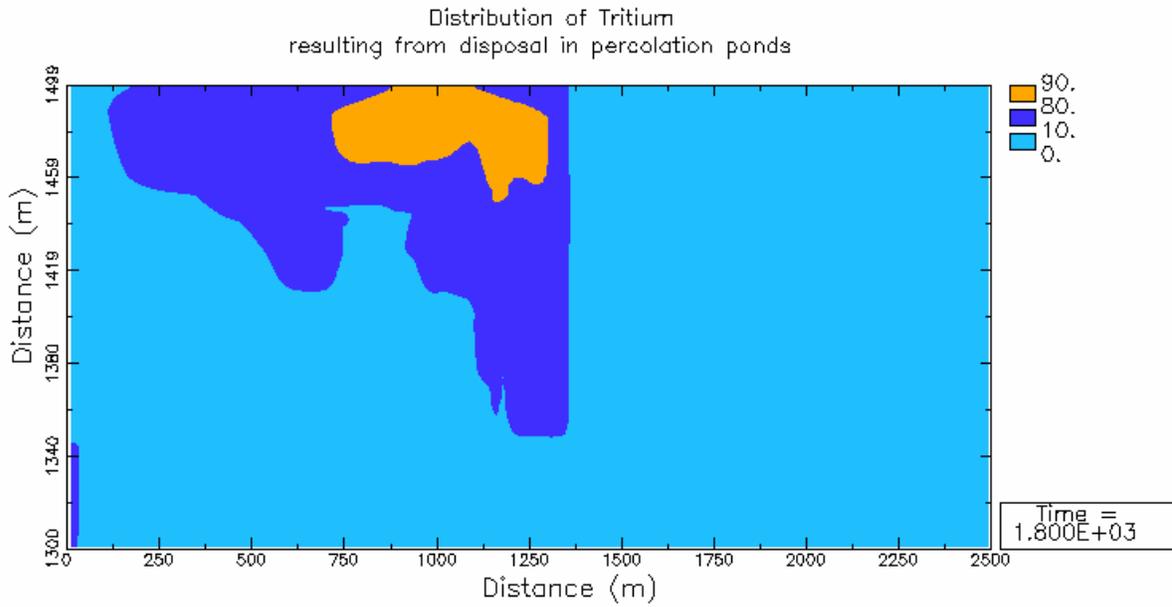


Figure 3-20. Distribution of tritium resulting from discharge into percolation ponds (units in pCi/mL).^g

g. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

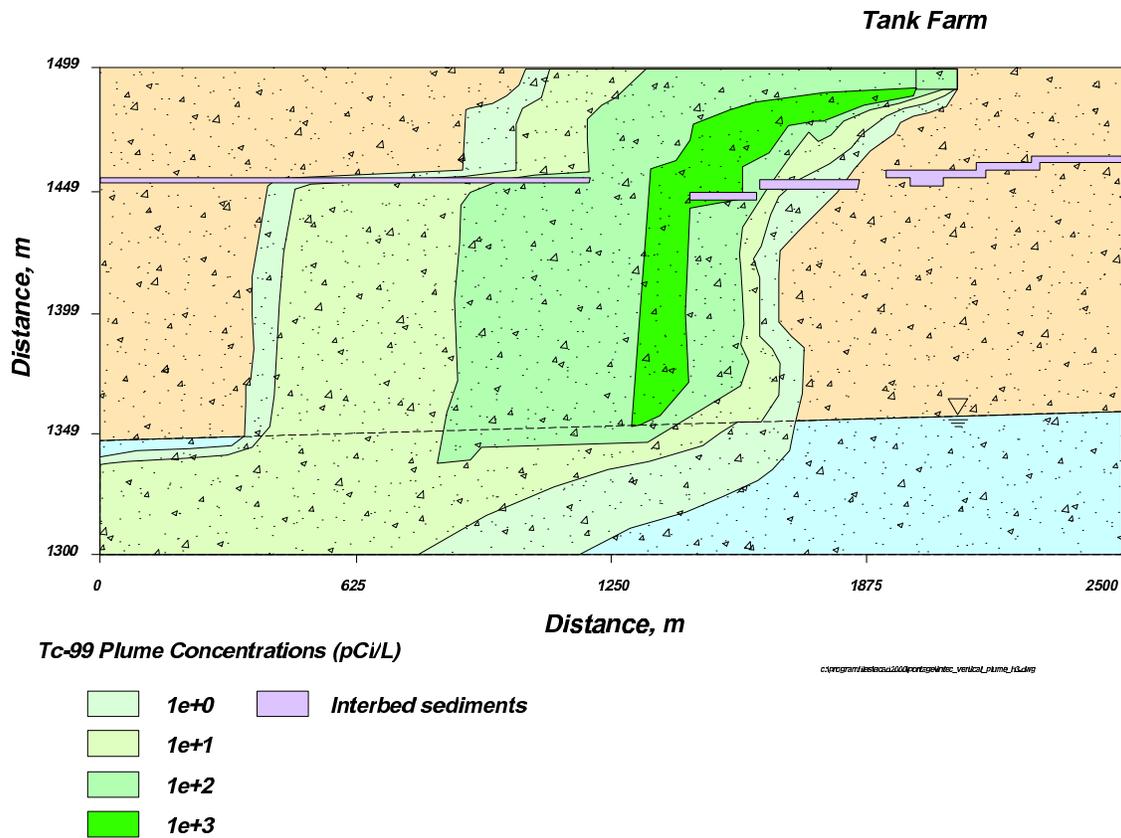
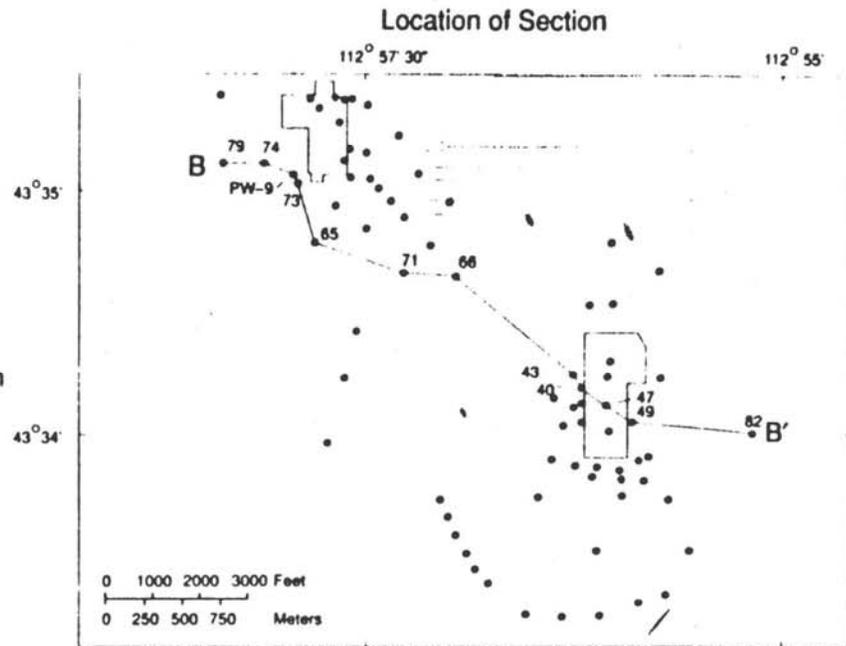
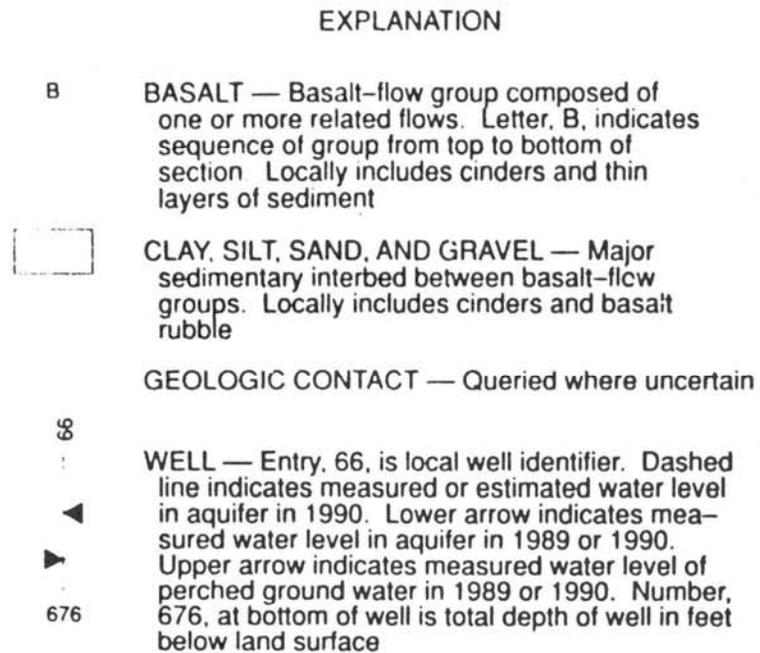


Figure 4-2. Groundwater modeling domain showing ⁹⁹Tc concentrations and location of maximum concentrations (all concentrations based on a unit source inventory).^h

h. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).



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Figure 4. Geologic section B – B' at the Idaho Chemical Processing Plant [now INTEC] and Test Reactors Area.ⁱ

i. Anderson, S. R., 1991, *Stratigraphy of the Unsaturated Zone and Uppermost Part of the Snake River Plain Aquifer at the Idaho Chemical Processing Plant and Test Reactor Area, Idaho National Engineering Laboratory, Idaho*, DOE/ID-22095, U.S. Geological Survey Water-Resources Investigations Report 91-4010, 1991.

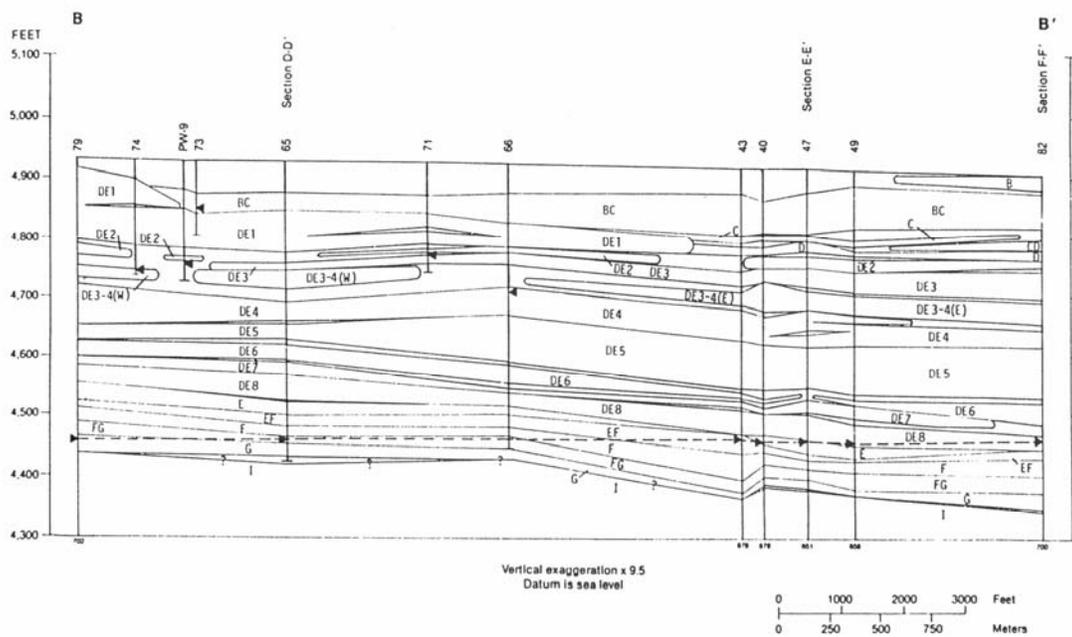


Figure 4. Geologic section B – B’ at the Idaho Chemical Processing Plant [now INTEC] and Test Reactors Area.--Continued¹

j. Anderson, S. R., 1991, *Stratigraphy of the Unsaturated Zone and Uppermost Part of the Snake River Plain Aquifer at the Idaho Chemical Processing Plant and Test Reactor Area, Idaho National Engineering Laboratory, Idaho*, DOE/ID-22095, U.S. Geological Survey Water-Resources Investigations Report 91-4010, 1991.

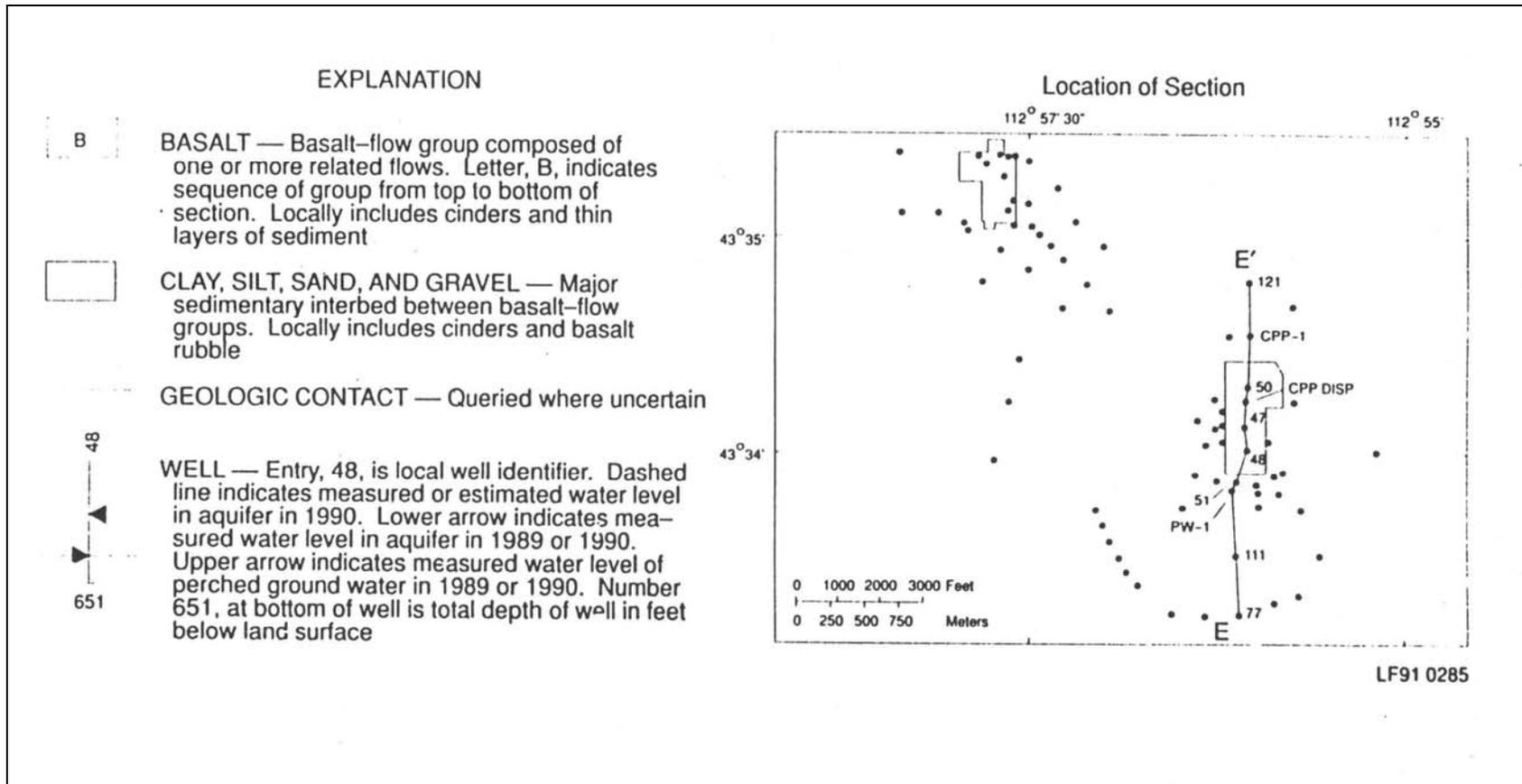


Figure 7. Geologic section E – E' at the Idaho Chemical Processing Plant [now INTEC] and Test Reactors Area.^k

k. Anderson, S. R., 1991, *Stratigraphy of the Unsaturated Zone and Uppermost Part of the Snake River Plain Aquifer at the Idaho Chemical Processing Plant and Test Reactor Area, Idaho National Engineering Laboratory, Idaho*, DOE/ID-22095, U.S. Geological Survey Water-Resources Investigations Report 91-4010, 1991.

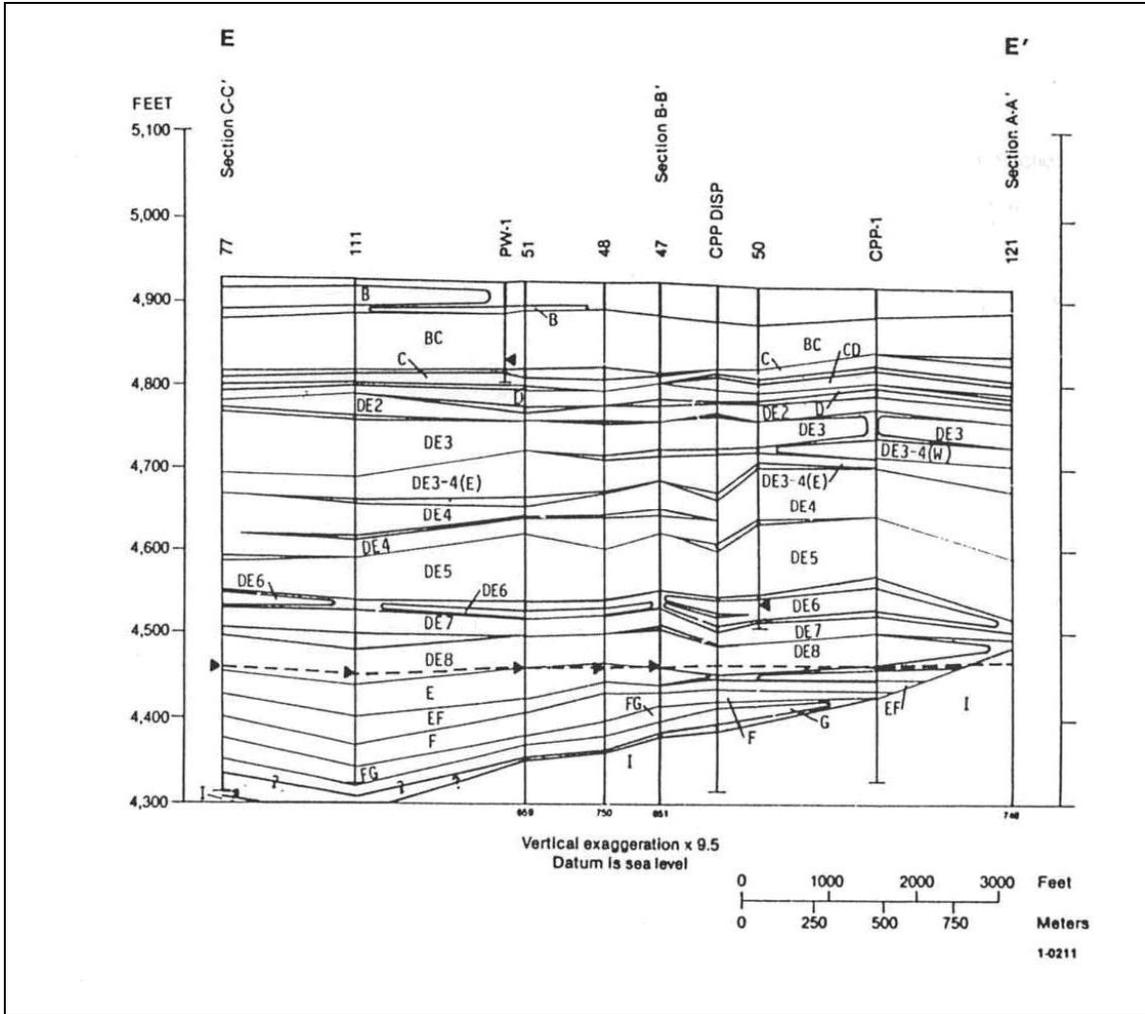


Figure 7. Geologic section E – E’ at the Idaho Chemical Processing Plant [now INTEC] and Test Reactors Area.--Continued¹

¹ Anderson, S. R., 1991, *Stratigraphy of the Unsaturated Zone and Uppermost Part of the Snake River Plain Aquifer at the Idaho Chemical Processing Plant and Test Reactor Area, Idaho National Engineering Laboratory, Idaho*, DOE/ID-22095, U.S. Geological Survey Water-Resources Investigations Report 91-4010, 1991.

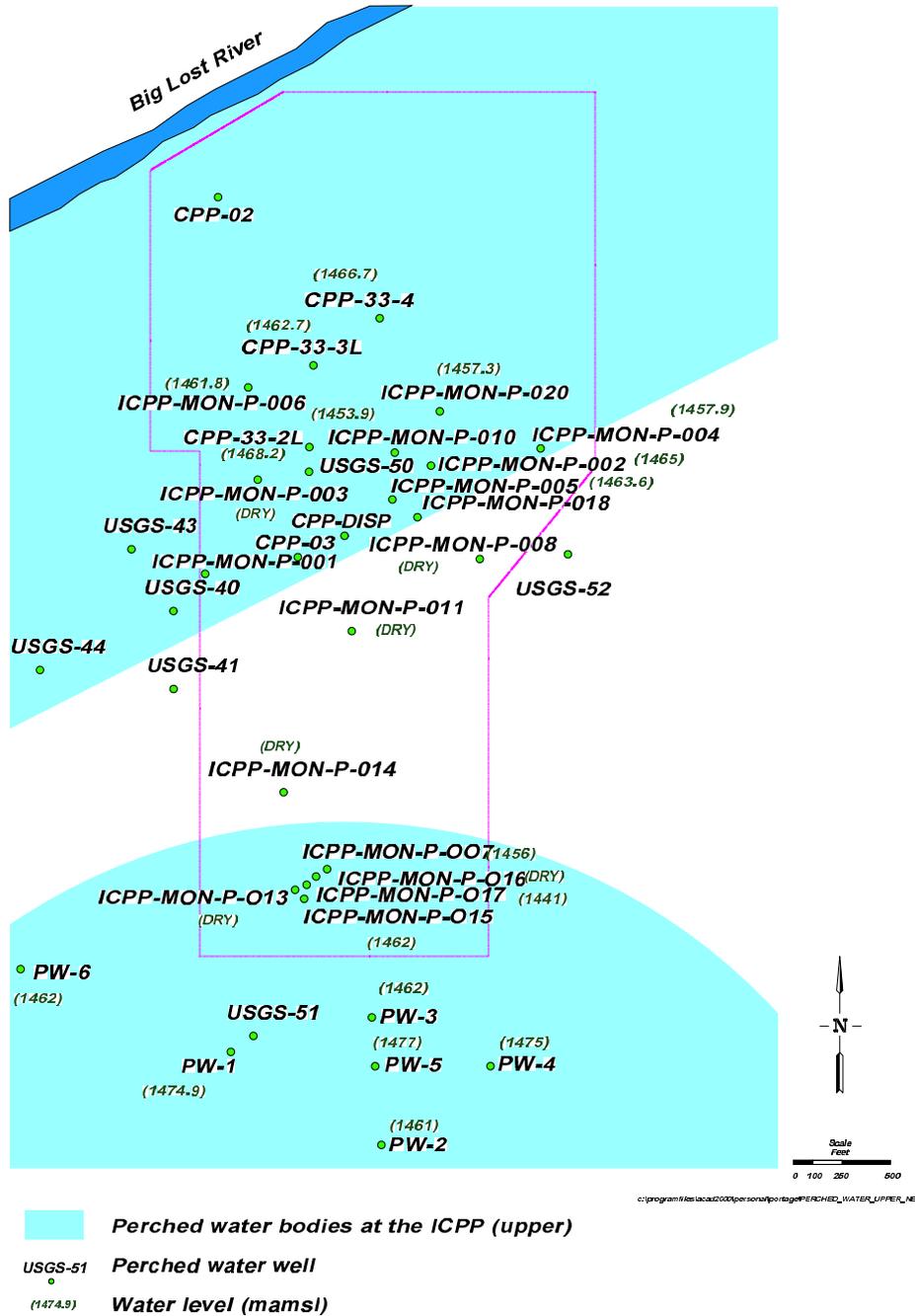


Figure 3-19. The extent of upper perched water at the INTEC facility based on perched water well data in Table 3-7.^m

m. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

NRC

Comment 11:

Unsaturated zone hydrologic properties of massive-to-fractured basalts, basaltic rubble zones, and sedimentary interbeds are highly uncertain, as are the initial conditions for radionuclide release, and the nature of the flow history prior to initial release. These sources of uncertainty are not reflected in the deterministic model results. In the current model, significant emphasis is placed on the sedimentary interbeds but, during the Large Scale Infiltration Test, intra-basalt flow, column-bounding vertical fractures were shown to contribute significantly to the vertical permeability of the vadose zone at INL (Faybishenko, et al., 2000).

Basis:

The unsaturated zone parameter values, constitutive relationships, and the treatment method for vadose zone air in the flow and transport model will affect unsaturated zone flow and radionuclide transport model estimates. Field-scale evidence from tests at INL or just outside the INL boundary at the Box Canyon analog site suggests that flow patterns and zones of elevated saturation change over relatively short time-scales, and that they are sensitive not just to lithology, but also to initial conditions and the nature of previous episodes of wetting and drying (Faybishenko, et al., 2000).

Parameter ranges and constitutive relationships used in the flow model should be clear to the reviewer, and parameter uncertainty should be evaluated to determine a defensible unsaturated zone parameter set. Information, data, and model results from other sites at INL (e.g., Radioactive Waste Management Complex, Large Scale Infiltration Test, Vadose Zone Research Park) may provide a supporting basis for parameter values used in the tank farm groundwater model.

Path Forward:

DOE should provide the lithologic analysis of Anderson, et al. (1999). DOE-ID should clearly identify how they treat the gaseous phase in their PORFLOW model (i.e., is air treated as stagnant, or is the groundwater model for the TFF evaluation truly a multiphase flow simulation?). DOE-ID should evaluate the significance of individual components of a recommended multi-geological-component conceptual model (Faybishenko et al., 2000) for unsaturated zone flow at INL, including (1) fracture-to-matrix diffusion, (2) vesicular basalt-to-massive basalt diffusion, (3) preferential flow through conductive fractures and the effect of flow funneling, (4) vesicular basalt-to-nonconductive fracture diffusion, (5) conductive fracture-to-vesicular basalt advection and diffusion, (6) lateral flow and advective transport in the central fracture zone, (7) lateral flow and advective transport in the rubble zone, and (8) vertical flow and advective transport into underlying basalt flows (Faybishenko et al., 2000). Justification should be provided for not including significant components in modeling analyses for estimating groundwater dose to the public.

Response:

It is agreed that column-bounding vertical fractures contribute to the vertical permeability of the vadose zone. As noted in NRC Comment 10, conservative values were assigned to the vertical hydraulic conductivities in the basalts. The more important point is that column-bounding vertical flow is only one of several hydraulic parameters that contribute “significantly” to vadose zone transport.

Furthermore, the Box Canyon site varies in scale, geology, boundary conditions, and the type and availability of data.

Unger et al. (2004) and Faybishenko et al. (2000) conducted their investigations on an exposed basalt unit to a depth of 16 m. By contrast, the TFF area is covered with 13 m of alluvium underlain by numerous basalts and sedimentary interbeds that encompasses a modeling domain 2,500 by 200 m. In addition, the Big Lost River provides a critical boundary condition influencing vadose zone flow over a large portion of the problem domain. The PA model (DOE-ID 2003) is based on reliable field data, including stratigraphic, lithologic, and hydraulic data, to accurately digitize stratigraphic units, to identify low-permeable sediments, and to assign realistic flow and transport parameters. The result is a model that is calibrated and verified to field data and provides a realistic quantification of water movement in the subsurface due to boundary conditions (Big Lost River), the location of perched water zones (low permeable sedimentary interbeds), and the rapid water movement through the basalt units to reduce uncertainty and provide a conservative and bounding analysis.

The uncertainty of intra-basalt flow, column-bounding vertical fractures, and the overall vertical permeability of the basalts is bounded by the incorporation of high vertical hydraulic conductivity values that exhibit a dominant vertical anisotropy that ranges from one to two orders of magnitude higher than horizontal permeability values. As noted in the response to NRC Comment 10, Magnuson (2004) used a saturated hydraulic conductivity of approximately 0.1 m/d. The PA two-dimensional model used a minimum vertical hydraulic conductivity for the fractured basalt of 10 m/d, which is two orders of magnitude higher, allowing water to move rapidly through the basalts. Lower horizontal hydraulic conductivity values resist the preferential movement of water and contaminants along rubble zones, which would spread the contamination in the unsaturated zone and reduce maximum contaminant concentrations in the plume entering the groundwater—a conservative assumption.

The model is conducted under steady-state conditions that calculate initial moisture contents based on boundary conditions and moisture characteristics of the soils and basalts. It is agreed that initial moisture conditions have an impact on flow conditions near the surface, but the unsaturated zone at the site is over 150 m thick. At depth, conditions tend to remain relatively stable in response to surface precipitation events. Another important factor is the difference between the Box Canyon site and the INTEC facility. At the Box Canyon site, the basalt is at the surface where the observations of Faybishenko et al. (2000) are important. At the INTEC facility, there are over 13 m of alluvium covering the underlying basalts. Many of the surface features relative to infiltration at the Box Canyon site are distinctly different for the INTEC facility.

Recent publications of studies conducted at the INL Site support the conceptual model used as the basis for predictive modeling in the PA. The Mattson et al. (2004) interpretation on vadose zone water movement based on large-scale field tests and long-term monitoring “suggest that sedimentary interbeds with the basalt formations have a larger effect on the creation of perched water and its

lateral extent.” Nimmo et al. (2004) provide further support to the PA conceptual model that “sedimentary interbeds are likely to impede vertical flow and to cause preferential flow...their layered structure may be conducive to funneled flow.” It is clear from recent publications that the conceptual model that forms the basis of the PA model agrees with the earlier publications describing the controlling hydrostratigraphic features for unsaturated flow and the principles described in the PA for constructing the two-dimensional predictive model.

DOE should provide the lithologic analysis of Anderson, et al. (1999).

The requested reference (Anderson et al. 1999) was provided in the response to NRC Clarifying Request 21.

DOE-ID should clearly identify how they treat the gaseous phase in their PORFLOW model (i.e., is air treated as stagnant, or is the groundwater model for the TFF evaluation truly a multiphase flow simulation?).

No parameters were entered into the PA two-dimensional model for the air component in the unsaturated groundwater flow zone. The only implication for the transport of water is that the effect of trapped air will reduce the infiltration rates for wetting fronts. As previously noted, the TFF area contains 13 m of alluvium covering the basalt, contrary to the Box Canyon site where the basalt is exposed at the surface. This trapped air effect will reduce the penetration of infiltrating water, allowing higher losses by evaporation, and reducing the annual recharge from precipitation. Since the PA model does not incorporate air density, viscosity, and compressibility, it is inherently conservative regarding infiltration from precipitation. To further increase the conservatism of the model, instead of assigning one of the few actual field infiltration measurements conducted at the site as reported by Cecil et al. (1992) using measured tritium and ³⁶Cl profiles in the soil, which yielded a range of 0.4 to 1.1 cm/yr, INL used as a base-case infiltration rate of 4.1 cm/yr.

DOE-ID should evaluate the significance of individual components of a recommended multi-geological-component conceptual model (Faybishenko et al., 2000) for unsaturated zone flow at INL, including (1) fracture-to-matrix diffusion, (2) vesicular basalt-to-massive basalt diffusion, (3) preferential flow through conductive fractures and the effect of flow funneling, (4) vesicular basalt-to-nonconductive fracture diffusion, (5) conductive fracture-to-vesicular basalt advection and diffusion, (6) lateral flow and advective transport in the central fracture zone, (7) lateral flow and advective transport in the rubble zone, and (8) vertical flow and advective transport into underlying basalt flows (Faybishenko et al., 2000). Justification should be provided for not including significant components in modeling analyses for estimating groundwater dose to the public.

The following are the individual responses to the specific questions presented by the NRC.

(1) Since the fractures were simulated as a porous media, diffusion coefficients were based on porous media. Diffusion from a fracture without a porous structure would be higher than the values assigned in the model; consequently, diffusion would be slightly higher, thereby reducing the core concentrations in the plume—a conservative assumption.

(2) Vesicular basalt is typically not an interconnected porosity and would not provide a migration pathway. However, if diffusion did occur under this scenario, then contaminant concentrations would decrease in the main plume as contaminants diffuse into dead-end pores and dead-end fractures in the massive basalts. This component was not included in the conceptual model, making it more conservative.

(3) A high vertical hydraulic conductivity value was assigned to the basalts, allowing the rapid movement of water through the basalts. Funneling of the plume by the discontinuity of the interbeds is illustrated in Figure 4-2 of the PA (see response to NRC Comment 10) and is supported by recent work by Nimmo et al. (2004), which states “the interbeds...layered structure may be conducive to funneled or unstable flow.”

(4) Diffusion from vesicular basalt to nonconductive fractures would reduce plume concentrations and is not incorporated into the PA model—a conservative assumption.

(5) Diffusion and advection does occur laterally in the basalts in the PA model. These processes are specified in the PA model as vertical and horizontal anisotropy.

(6), (7), and (8) The PA model, which incorporates vertical hydraulic conductivities that range upward to three orders of magnitude higher than those reported by Magnuson (2004) and dominant vertical anisotropies that range as much as two orders of magnitude higher, allows the plume to move rapidly through the basalt layers with minimal lateral spreading. The net result is a confined plume at high concentrations that rapidly reaches the regional groundwater system. This approach reduces uncertainty regarding the complexity of the basalt structure and results in a conservative prediction of the groundwater dose to the public.

References:

Cecil, L. D., et al., 1992, “Water Infiltration Rates in the Unsaturated Zone at the Idaho National Engineering Laboratory Estimated by Chlorine-36 and Tritium Profiles, and Neutron Logging,” *Proceedings of the 7th International Symposium on Water-Rock Interactions, Park City, Utah, July 13–18, WRI-7*, Y. K. Kharaka and A. S. Meest (eds.).

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

Faybishenko, B., C. Doughty, M. Steiger, J. C. S. Long, T. R. Wood, J. S. Jacobsen, J. Lore, and P. T. Zawislanski, 2000, "Conceptual Model of the Geometry and Physics of Water Flow in a Fractured Basalt Vadose Zone," *Water Resources Research*. Vol. 36, No. 12. pp. 3,499–3,520, 2000.

Magnuson, S., 2004, "Regulatory Modeling for the INEEL's Subsurface Disposal Area and Conceptual Model Uncertainty Treatment," *Vadose Zone Journal*, 3:59-74.

Mattson, E. D., S. O. Magnuson, and S. L. Ansley, 2004, "Interpreting INEEL Vadose Zone Water Movement on the Basis of Large-Scale Field Tests and Long-Term Vadose Zone Monitoring Results," *Vadose Zone Journal*, 3:35-46.

Nimmo, J. R., J. P. Rousseau, K. S. Perkins, K. G. Stollenwerk, P. D. Glynn, R. C. Bartholomay, and L. L. Knobel, 2004, "Hydraulic and Geochemical Framework of the INEEL Vadose Zone," *Vadose Zone Journal*, 3:6-34.

Unger, A. J., B. Faybishenko, G. S. Bodvarsson, and A. M. Simmons, 2004, "Simulating Infiltration Tests in Fractured Basalt at the Box Canyon Site, Idaho," *Soil Science of America, Vadose Zone Journal*, 3:75-89.

NRC

Comment 12:

Overall, information pertaining to geologic controls on hydrology for the TFF vicinity are lacking from DOE-ID (2003b) (see Comments 10 and 11). Information provided does not address whether there is a known southerly dip of the sedimentary interbeds, consistent with the selected model domain. The regional potentiometric map is not shown, and while there is likely a dominant direction of basalt flow fingers or lenticulae, their orientation is not addressed in the performance assessment.

Basis:

An important geologic control on unsaturated zone flow is the dominant dip direction of sedimentary interbeds. While flow in the aquifer may be to the south-southwest, it is unclear whether perched water flow along sedimentary interbeds would also preferentially flow to the south.

The regional potentiometric map is a source of groundwater model boundary conditions, but a map illustrating this information is not provided. Without this information the reviewer must rely on statements that saturated groundwater flow is to the south-southwest for a model that predicts flow only in a due south direction. The dominant orientation of basalt fingers or lenticulae should also be noted on the map.

Path Forward:

DOE-ID should provide reports cited on page 2-47. These documents include Walker (1960), Ackerman (1991), and Anderson et al. (1999). DOE-ID should provide information that addresses the open question of whether the sedimentary interbeds dip in a known direction, and if so, are they known to have a dominant southerly dip consistent with the modeled domain? If the sedimentary interbeds are known to have a dominant dip in a direction that is not consistent with the modeled domain, DOE-ID should address the implications of modeling water flow in a direction that is oblique to the dominant dip direction.

DOE-ID should provide a map of the regional potentiometric surface, and an indication of the dominant direction of basalt fingers or lenticulae.

Response:

DOE-ID should provide reports cited on page 2-47. These documents include Walker (1960), Ackerman (1991), and Anderson, et al. (1999).

The requested reports cited on page 2-47 were provided in the response to NRC Clarifying Request 21. These documents include Walker (1960), Ackerman (1991), and Anderson et al. (1999).

DOE-ID should provide information that addresses the open question of whether the sedimentary interbeds dip in a known direction, and if so, are they known to have a dominant southerly dip consistent with the modeled domain? If the sedimentary interbeds are known to have a dominant dip in a direction that is not consistent with the modeled domain, DOE-ID should address the implications of modeling water flow in a direction that is oblique to the dominant dip direction.

Anderson (1991) presents a north-south cross-section and a roughly east-west cross-section of the area in and around the INTEC facility (see Figures 4 and 7;

attached). Comparing the area between Wells CPP-1 and PW-1, on the north-south cross-section (see Figure 7 [continued]; attached), with the area between Wells 43 and 49 on the east-west cross-section (see Figure 4 [continued]; attached), shows that the sedimentary interbeds do not dip in any known direction. The interbed sediments at INTEC are less continuous across the INTEC area than at the southern portions of the INL Site, which include the Subsurface Disposal Area. Construction of a dip surface map of the sedimentary interbed layers would not be appropriate, since methods such as kriging would indicate a surface dip where actual breaks are occurring between well data points. It is important to note that the controlling mechanism for the location of the perched water bodies at INTEC is directly related to the breaks in the interbed sediments. More important in controlling unsaturated flow directions than minor dips in the interbeds is recharge from the Big Lost River located north of the facility. By volume, the largest of the recharge sources are the Big Lost River and the former service wastewater disposal ponds located south of the INTEC facility. The INTEC service wastewater is currently discharged in new percolation ponds located approximately 3 km (2 miles) southwest of the old ponds, thereby eliminating the largest anthropogenic recharge in the area (Mattson et al. 2004). The Big Lost River recharge is responsible for the perched water zones and results in a hydraulic force that moves water in a southern direction, which is consistent with the alignment of the two-dimensional model slice (see the response to NRC Comment 10 for a detailed discussion).

DOE-ID should provide a map of the regional potentiometric surface, and an indication of the dominant direction of basalt fingers or lenticulae.

The regional potentiometric map for the Snake River Plain Aquifer is presented in Robertson et al. (1974) (see Fig. 74; attached). In the area surrounding the INTEC facility, the hydraulic gradient is in a southwest direction. Groundwater flow in the regional aquifer is properly represented in the PA model (DOE-ID 2003). Regional groundwater flow is not critical to groundwater dose calculations in the PA since the model analysis selects the area where the highest contaminant concentrations enter the groundwater. In other words, the aquifer only acts as a dilution factor at the entry point of contaminants from the vadose zone.

Existing groundwater plumes in the regional Snake River Plain Aquifer, such as those presented by Beasley et al. (1998) (see Figure 5; attached), show a dominant south-southwest flow direction, which is also consistent with the alignment of the two-dimensional model slice. The regional northwest-trending basalt fracture is probably responsible for the more southern flow component compared with the southwest dipping hydraulic gradient in the aquifer. In map view, the importance of anisotropy is clear in the aquifer.

The dominant direction of basalt flow fingers or lenticulae has been discussed by Nimmo et al. (2004) and Smith (1994). Nimmo et al. (2004) note that basalt flows in the Snake River Plain were deposited during individual, brief eruptive events (Kuntz et al. 1980) from numerous fissures and small shield volcanoes on and near the Arco-Big Southern Butte volcanic rift zone (Kuntz et al. 1992)

Some flows are as long as several kilometers (Kuntz et al. 1994). Smith (2004) notes studies by Hackett et al. (2004) that indicate the mean lava flow dimensions are 12.4 km in length and 96.5 km² in area. Basalt flows and other volcanic deposits combine into basalt flow groups, each of which is a complex assemblage of overlapping flows and deposits related to a single eruption. A basalt flow group comprises several, perhaps hundreds, of distinct basalt flows that occurred in a time interval brief enough that little or no sediment accumulated between them.

Smith (2004) notes that most of the volcanic activity occurs in the so-called Axial Volcanic Zone and in those portions of the volcanic rift zones closest to the axis of the Eastern Snake River Plain. Resulting lava flows tend to be elongated in a direction perpendicular to the length of the Eastern Snake River Plain and to the general direction of groundwater flow.

It is acknowledged that there are basalt-flow orientations on the INL Site that regionally trend northwest and southeast from the rift zones. The basalt units may show a general trend in orientation; however, the unsaturated flow through the basalts would be vertical and dominated by gravity. Flow at INTEC is controlled by infiltration from the Big Lost River and associated perched water zones located above sedimentary interbed units. As the perched water flows off of these interbed sediments, it is rapidly transmitted vertically through the vadose zone basalts. Since the perched water zones are controlled by interbed sediments, which are not controlled by the rift zone orientation, basalt orientation is not significant in the vadose zone analysis.

References:

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Beasley, T. M., P. R. Dixon, and L. J. Mann, 1998, "⁹⁹Tc, ²³⁶U, and ²³⁷Np in the Snake River Plain Aquifer at the Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho," *Environmental Science and Technology*, Vol. 32, pp. 3,875–3,881, 1998.

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Kuntz, M. A., G. B. Covington, and L. J. Schorr, 1992, "An Overview of Basaltic Volcanism of the Eastern Snake River Plain, Idaho" p. 227-267, *Geological Society of America Memoir 179*, Geological Society of America, Boulder, Colorado.

Kuntz, M. A., G. B. Dalrymple, D. E. Champion, and D. J. Doherty, 1980, *Petrography, Age, and Paleomagnetism of Volcanic Rocks at the Radioactive Waste Management Complex, Idaho National Engineering and Environmental Laboratory, Idaho, with an Evaluation of Potential Volcanic Hazards*, U.S. Geological Services Open-File Report 80-388. 63 p.

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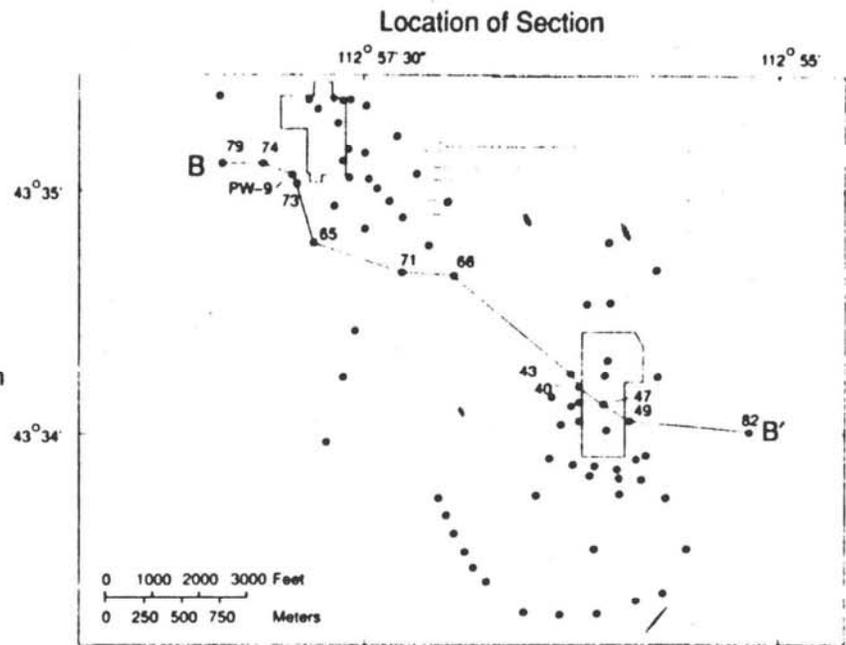
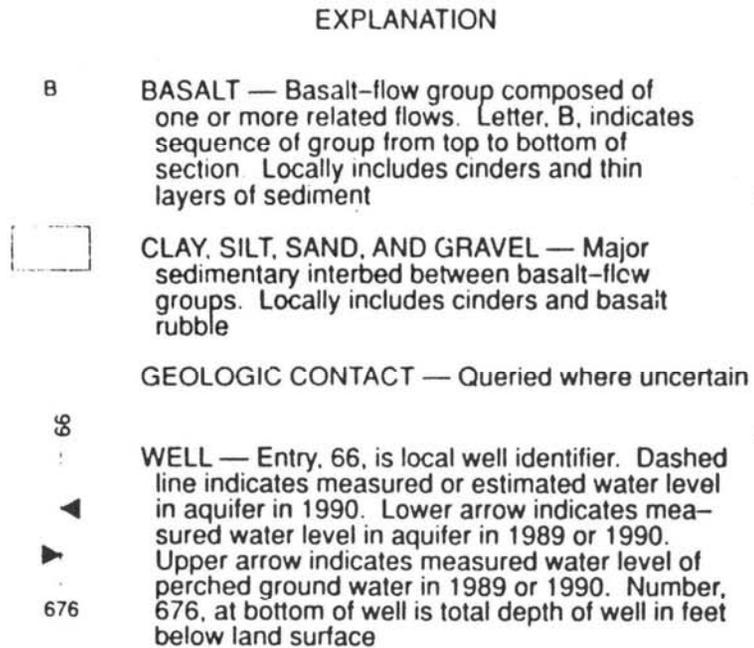
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Figure 4. Geologic section B – B' at the Idaho Chemical Processing Plant [now INTEC] and Test Reactors Area.ⁿ

n. Anderson, S. R., 1991, *Stratigraphy of the Unsaturated Zone and Uppermost Part of the Snake River Plain Aquifer at the Idaho Chemical Processing Plant and Test Reactor Area, Idaho National Engineering Laboratory, Idaho*, DOE/ID-22095, U.S. Geological Survey Water-Resources Investigations Report 91-4010, 1991.

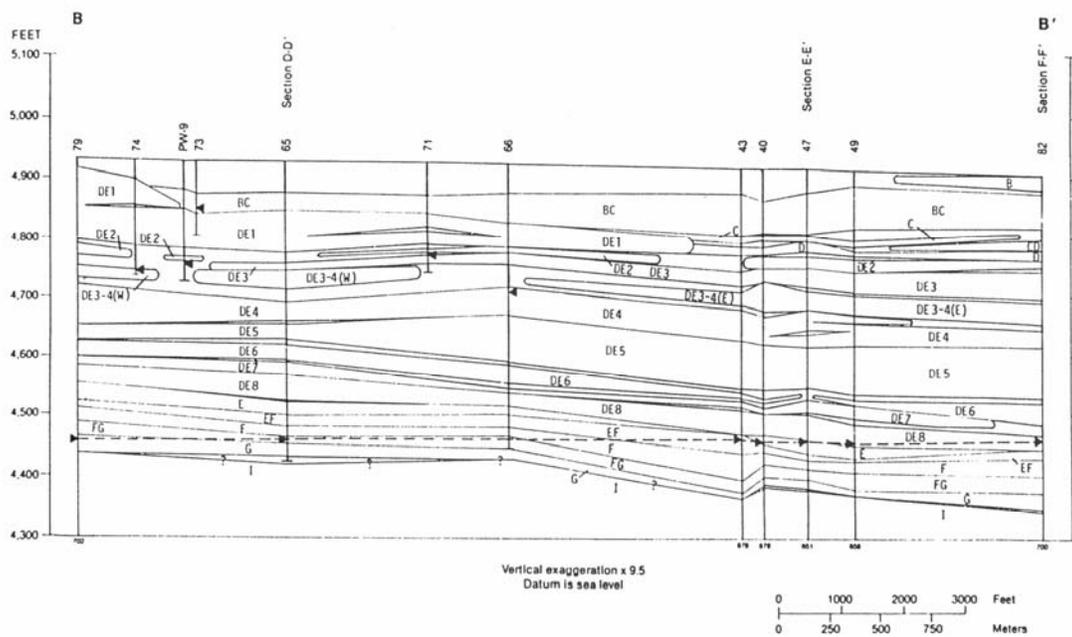


Figure 4. Geologic section B – B’ at the Idaho Chemical Processing Plant [now INTEC] and Test Reactors Area.--Continued^o

^o Anderson, S. R., 1991, *Stratigraphy of the Unsaturated Zone and Uppermost Part of the Snake River Plain Aquifer at the Idaho Chemical Processing Plant and Test Reactor Area, Idaho National Engineering Laboratory, Idaho*, DOE/ID-22095, U.S. Geological Survey Water-Resources Investigations Report 91-4010, 1991.

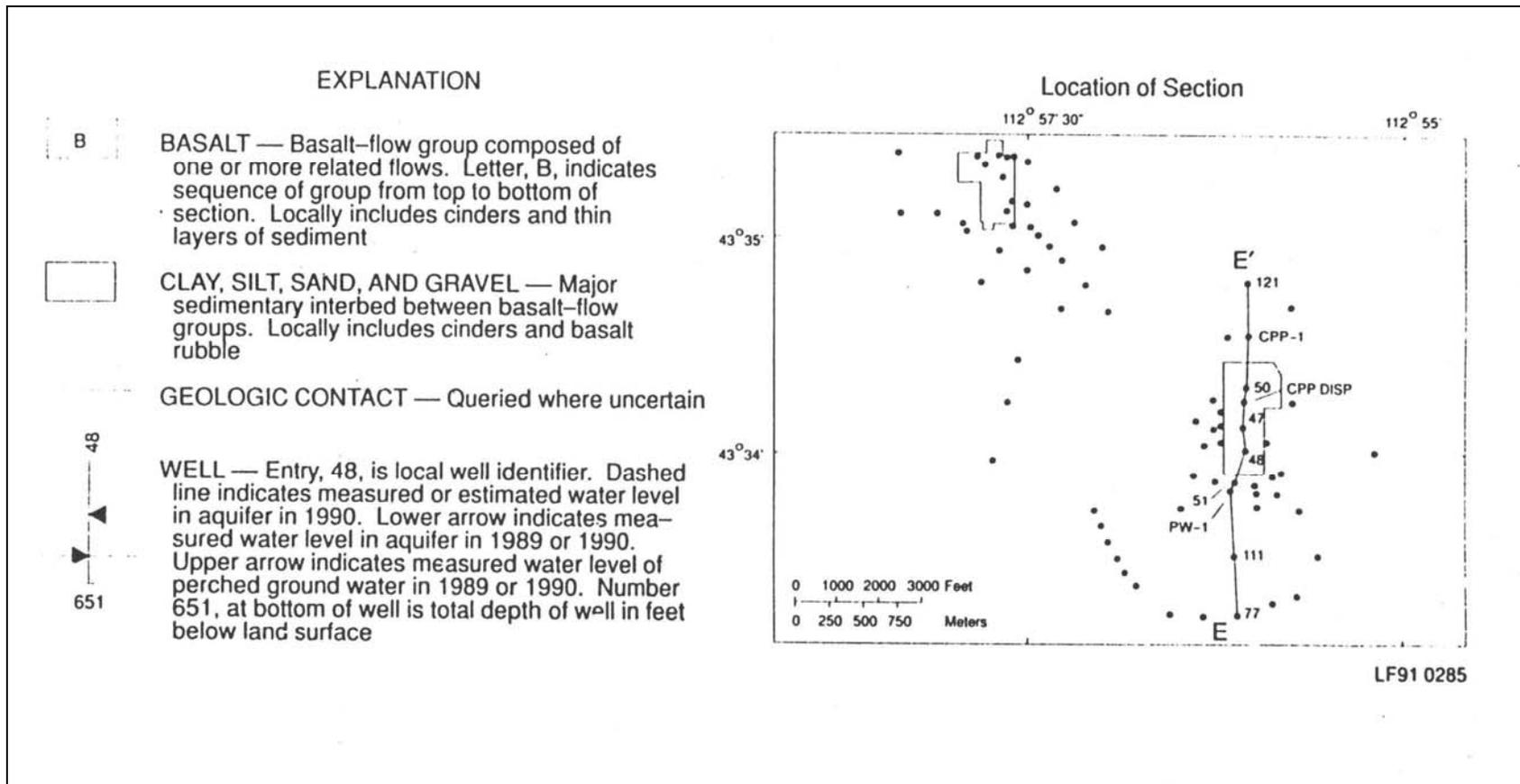


Figure 7. Geologic section E – E’ at the Idaho Chemical Processing Plant [now INTEC] and Test Reactors Area.^p

p. Anderson, S. R., 1991, *Stratigraphy of the Unsaturated Zone and Uppermost Part of the Snake River Plain Aquifer at the Idaho Chemical Processing Plant and Test Reactor Area, Idaho National Engineering Laboratory, Idaho*, DOE/ID-22095, U.S. Geological Survey Water-Resources Investigations Report 91-4010, 1991.

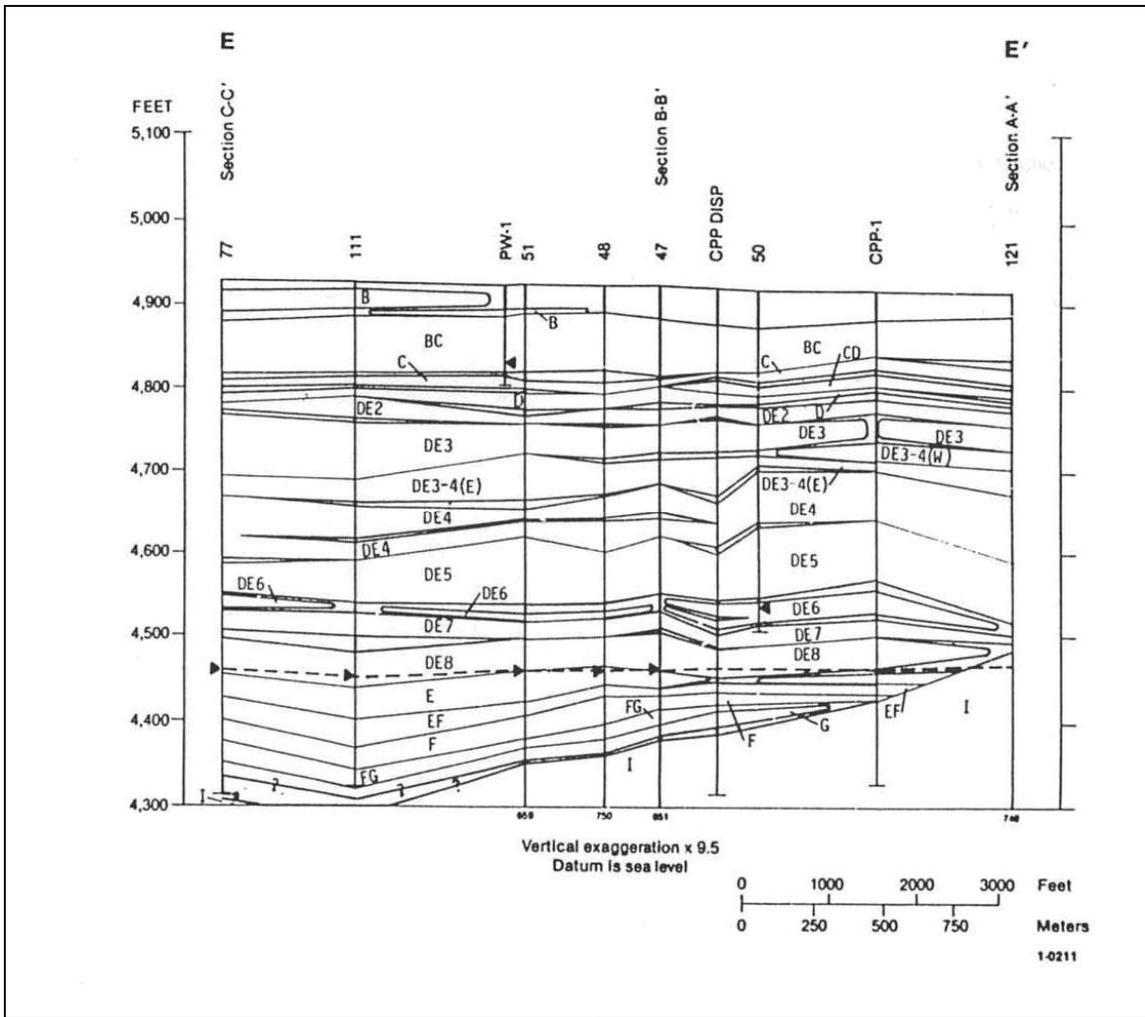


Figure 7. Geologic section E – E' at the Idaho Chemical Processing Plant [now INTEC] and Test Reactors Area.--Continued^q

q. Anderson, S. R., 1991, *Stratigraphy of the Unsaturated Zone and Uppermost Part of the Snake River Plain Aquifer at the Idaho Chemical Processing Plant and Test Reactor Area, Idaho National Engineering Laboratory, Idaho*, DOE/ID-22095, U.S. Geological Survey Water-Resources Investigations Report 91-4010, 1991.

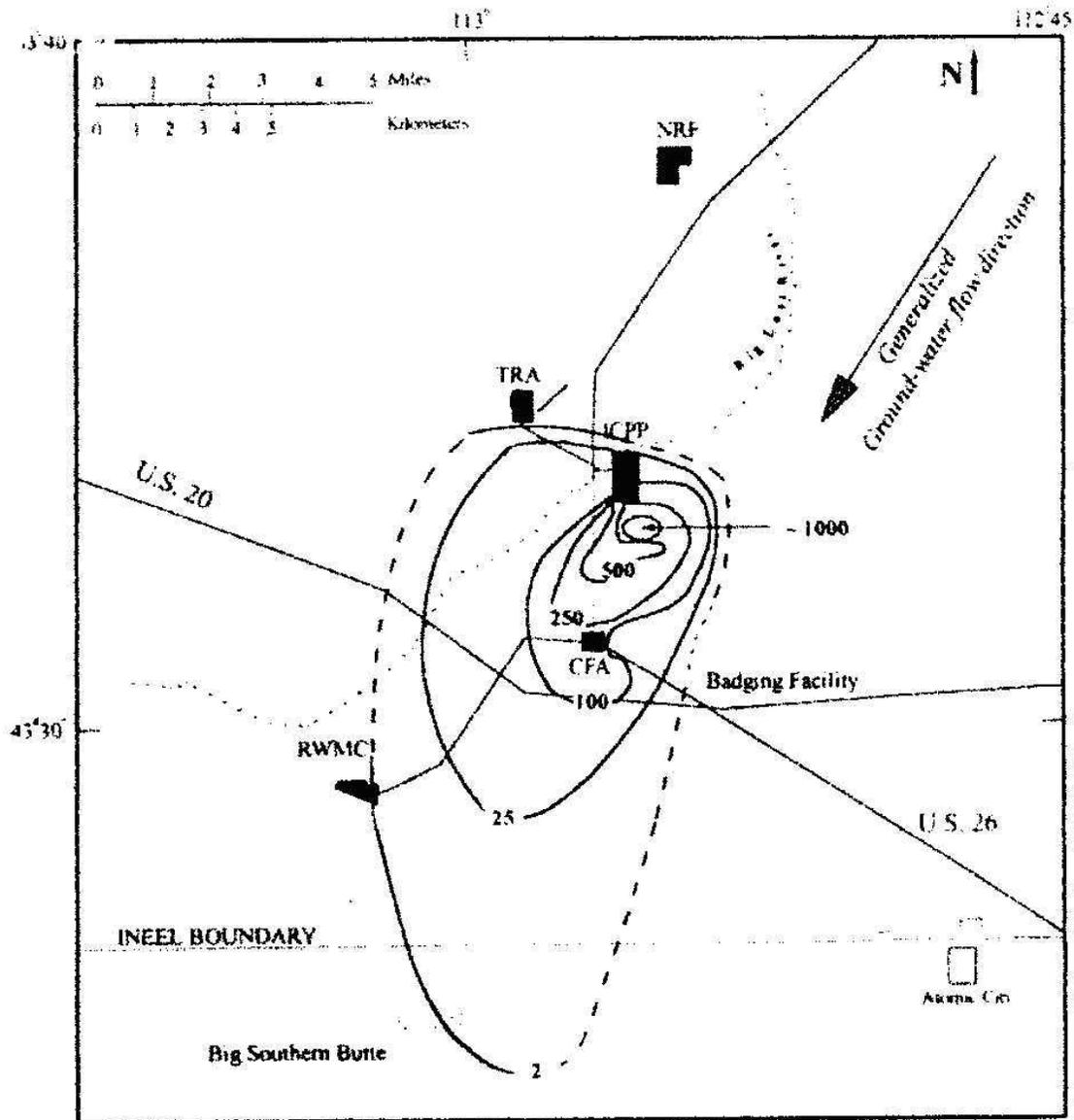


Figure 5. Isopleths of ^{99}Tc (mBq L^{-1}) in the Snake River Plain aquifer. The total plume area is 136 km^2 (53 mi^2) and contains 554 GBq ($\sim 15 \text{ Ci}$) of ^{99}Tc . The dashed line for the 2 mBq L^{-1} contour is intended to indicate uncertainty in its exact position because of the number of values used in its construction. The inventory and plume area are for the period 1991–1992.[†]

† r. Beasley, T. M., P. R. Dixon, and L. J. Mann, 1998, " ^{99}Tc , ^{236}U , and ^{237}Np in the Snake River Plain Aquifer at the Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho," *Environmental Science and Technology*, Vol. 32, pp. 3,875–3,881, 1998.

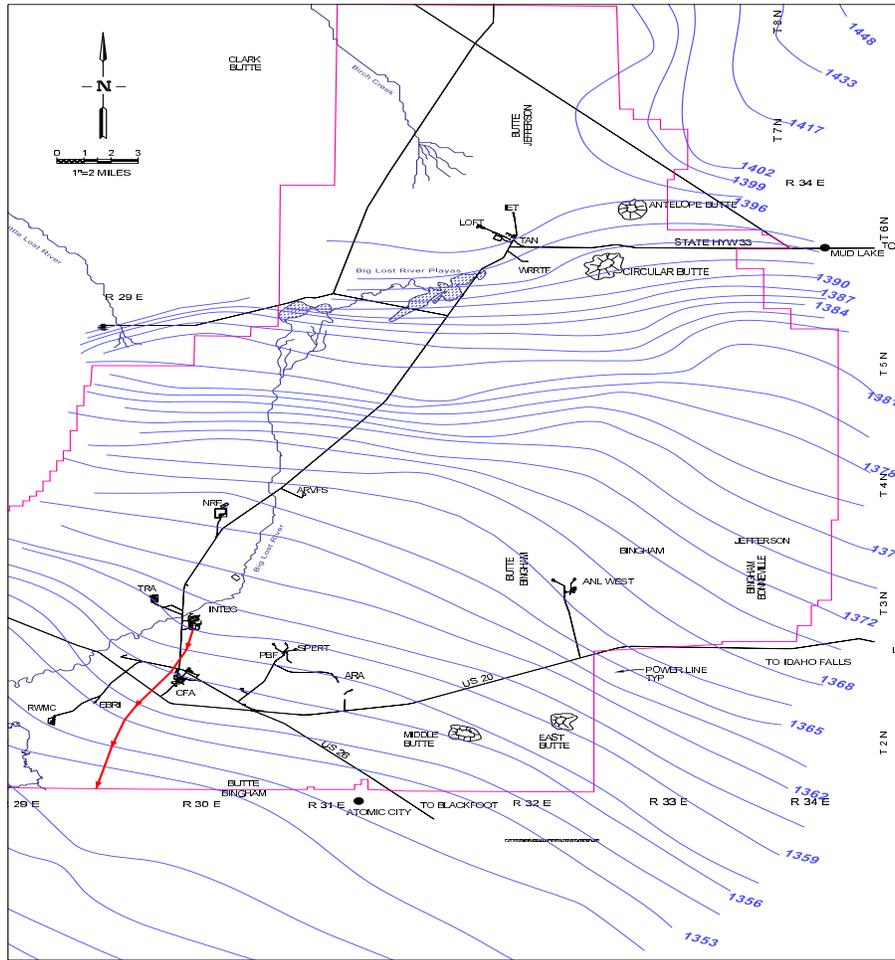


Figure. 74 Map showing potentiometric surface (water table elevation) in the ICPP-CFA vicinity.^s

s. Robertson, J. B., R. Schoen, and J. T. Barraclough, 1974, *The Influence of Liquid Waste Disposal on the Geochemistry of Water at the National Reactor Testing Station, 1952-1970*, U.S. Geological Survey Open-File Report, IDO-22053, TID-4500, February 1974.

NRC

Comment 13:

Existing sampling data for radionuclides currently contaminating the subsurface of the TFF are not provided. This data may provide useful information regarding the accuracy or degree of conservatism of the flow and transport model.

Basis:

Existing soil contamination (from piping leakage) has resulted in the transport of Tc from the tank farm source area to the saturated zone underneath the TFF in less than fifty years. It is not clear that this information is consistent with unsaturated zone flow and transport modeling predictions.

The initial radionuclide concentrations are set to zero at the model boundaries and in the model domain, even though there is known contamination of the subsurface at the TFF (DOE-ID, 2003b, p. 3-36). This treatment is said to be based on existing sampling data that shows existing contamination of the site is minimal, especially for future estimates.

Path Forward:

DOE-ID should provide existing sampling data for radionuclides currently contaminating the subsurface at the TFF and information regarding saturated groundwater contamination within the model domain. DOE-ID should discuss to what extent monitoring data in the vicinity of the TFF corroborates the modeling predictions presented in the 2003 PA. Contaminant releases to the aquifer from the injection well, WAG-3 soils, and TRA warm waste ponds were evaluated in the WAG-3 Remedial Investigation/Feasibility Study (RI/FS) (Rodriquez et al., 1997). DOE-ID should provide this reference.

Response:

The initial radionuclide concentrations are set to zero at the model boundaries and in the model domain, even though there is known contamination of the subsurface at the TFF (DOE-ID, 2003b, p. 3-36). This treatment is said to be based on existing sampling data that shows existing contamination of the site is minimal, especially for future estimates.

Setting initial radionuclide concentrations to zero at the model boundaries and at the model domain is appropriate for the purpose of the TFF PA (DOE-ID 2003a). The purpose of the PA is to evaluate the potential doses from contaminants released from the closure of the tanks, not for the evaluation of historical releases. Historical releases of tritium from the percolation ponds were evaluated in the PA for calibration purposes only. The evaluation of the all-pathways dose from potential combined doses from tank releases after closure, historical releases at INTEC, and other INL contaminant plumes is described in the TFF composite analysis (CA) (DOE-ID 2003b). The all-pathways dose calculated in the CA demonstrates compliance with the established performance objectives and is detailed in Table 6-2 of the CA (DOE-ID 2003b). Therefore, existing contaminants in the groundwater system are not input into the PA model and initial contaminant concentrations are set to zero in the groundwater environment. This is required since the purpose of the PA is to evaluate the future doses from the closed tanks. Evaluating existing contaminants along with the tank releases would provide a combined evaluation of radionuclide concentrations at the receptor point without knowledge of the contribution from

the tanks. For example, if the existing ⁹⁹Tc sources in the soils were input into the PA model along with the source in the tank, then the resulting ⁹⁹Tc concentration at the receptor location would be a combined value and the contribution from the tanks would not be known. This would be counter to the PA evaluation of the doses from the tank releases for purposes of supporting a waste determination for stabilized residuals in tanks and other TFF components.

Existing soil contamination (from piping leakage) has resulted in the transport of Tc from the tank farm source area to the saturated zone underneath the TFF in less than fifty years. It is not clear that this information is consistent with unsaturated zone flow and transport modeling predictions.

The ⁹⁹Tc contamination issue is discussed in the *Evaluation of Tc-99 in Groundwater at INTEC: Summary of Phase 1 Results* (ICP 2004). The historical releases from the TFF piping to the adjacent soils are suspected as the source of contamination detected in an aquifer well that is upgradient from the TFF. The conclusions regarding the appearance of ⁹⁹Tc in a monitoring well upgradient from the TFF have not yet been finalized. A possible reason is from the upgradient water production wells (i.e., pumping and well capture). That is, pumping from upgradient water production wells may be drawing contaminants toward these upgradient monitoring wells. The pumping of the production wells is not considered in the PA, since releases from the vaults and tanks are not expected until 100 and 500 years, respectively, and the use of these water production wells after institutional control is not expected. Therefore, the use of the ⁹⁹Tc release events is not considered suitable for use in the calibration of the PA model. The NRC notes that the “Existing soil contamination (from piping leakage) has resulted in the transport of Tc from the tank farm source area to the unsaturated zone underneath the TFF in less than fifty years.” This is in agreement with the modeled travel time to the aquifer for ⁹⁹Tc after release from the tanks of approximately 31 years.

It should be noted that there is an error in the Table 4-1 of the PA (DOE-ID 2003a). The peak release time for ⁹⁹Tc should be 14,559 years instead of the stated time of 12,206 years. Travel times calculated from the dates provided in this table would result in an incorrect travel time. A corrected Table 4-1 is attached.

DOE-ID should provide existing sampling data for radionuclides currently contaminating the subsurface at the TFF and information regarding saturated groundwater contamination within the model domain.

Several contaminants exist in the vadose zone perched water bodies and in the aquifer at the INTEC facility. However, the origin of the contaminants in the vadose zone is difficult to determine. Rodriguez et al. (1997) provide the most comprehensive evaluation of these existing contaminants to date at INTEC. As stated in the above responses, existing contaminants in the groundwater system are not input into the PA model and initial contaminant concentrations are set to zero in the groundwater environment. This is required since the purpose of the PA is to evaluate the future doses from the closed tanks. The evaluation of the

potential combined doses from tank releases after closure, historical releases at INTEC, and other INL contaminant plumes are evaluated in the TFF composite analysis (DOE-ID 2003b).

DOE-ID should discuss to what extent monitoring data in the vicinity of the TFF corroborates the modeling predictions presented in the 2003 PA.

Monitoring data for the percolation ponds were used in the PA to assess the accuracy of the modeling predictions. The PA presents an analysis of the transport of tritium from the percolation ponds. The waste percolation ponds were used until 2002, when the ponds were relocated 10 km southwest of INTEC. Prior to construction of the waste percolation ponds, waste was directly injected into the aquifer via the CPP injection well.

Bartholomay et al. (1997) show the location of the percolation ponds (see Figure 3; attached) and selected perched water monitoring wells. In addition, Bartholomay et al. (1997) present chloride data for several perched water monitoring wells surrounding the percolation ponds (see Table 7; attached). Perched water data from Wells 40 and 47, located approximately 1,000 m north of the percolation ponds but near the injection well, showed elevated chloride in late 1983, presumably due to the injection well, but water quality data from 1984 to 1995, after the injection well was no longer in service, showed chloride concentrations that are near background water quality. These data indicate that wastewater from the injection well impacted Wells 40 and 47 in 1984, but after disposal operations switched to the percolation ponds, infiltrating water from the ponds no longer impacted these wells.

Using the tritium plume predicted by the two-dimensional model calibration simulation from the PA (see Figure 3-20; attached), and assuming that the distribution of chloride discharged to the percolation ponds is similar to the tritium distribution since both analytes are conservative, the predicted plume is approximately 350 m north, or upgradient, of the percolation ponds. Wells 40 and 47 are approximately 1,000 m north of the percolation ponds and, based on perched water quality data (Bartholomay et al. 1997), are beyond the influence of chloride infiltration from the ponds. The PA model also predicts that Wells 40 and 47 are beyond the influence of the percolation ponds. Well 59, located north of the percolation ponds and within the 350-m upgradient distance predicted by the model, shows elevated chloride concentrations, once again matching the PA model predictions. As expected, all of the downgradient wells, 37, 57, 111, and 113, show elevated concentration of chloride, consistent with the PA model predictions.

Contaminant releases to the aquifer from the injection well, WAG-3 soils, and TRA warm waste ponds were evaluated in the WAG-3 Remedial Investigation/Feasibility Study (RI/FS) (Rodriquez et al., 1997). DOE-ID should provide this reference.

The Rodriquez et al. (1997) reference is being provided in the response to NRC Clarifying Request 21.

References:

Bartholomay, R. C., B. J. Tucker, D. J. Ackerman, and M. J. Liszewski, 1997, *Hydrologic Conditions and Distribution of Selected Radiochemical and Chemical Constituents in Water, Snake River Plain Aquifer, Idaho National Engineering Laboratory, Idaho, 1992 through 1995*, DOE/ID-22137, U.S. Geological Survey Water Resources Investigations 97-4086, April 1997.

DOE-ID, 2003a, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

DOE-ID, 2003b, *Composite Analysis for Tank Farm Facility Closure*, DOE/ID-10974, April 2003.

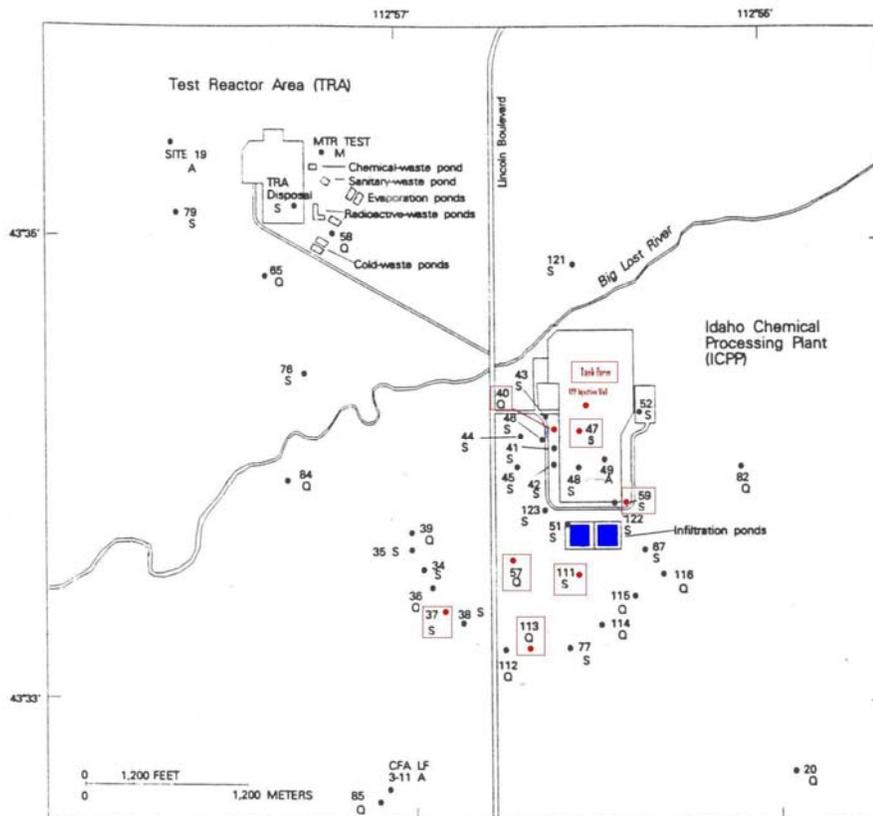
ICP, 2004, *Evaluation of Tc-99 in Groundwater at INTEC: Summary of Phase 1 Results*, ICP/EXT-04-00244, Rev. 0, September 2004.

Rodriguez, R. R., A. L. Shafer, J. McCarthy, P. Martian, D. E. Burns, D. E. Raunig, N. A. Burch, and R. L. VanHorn, 1997, *Comprehensive RI/FS for the Idaho Chemical Processing Plant OU 3-13 at the INEEL – Part A, RI/BRA Report (Final)*, DOE/ID-10534, Binders 1–3, November 1997.

Table 4-1. Peak releases rates, groundwater concentration, and doses for the groundwater pathway analyses.^t

Nuclide	Peak Release Rate		Peak Groundwater Concentration		Peak Drinking Water Dose		Peak All-Pathway Dose	
	Time (yr)	Ci/yr	Time (yr)	pCi/L	Time (yr)	mrem/yr	Time (yr)	mrem/yr
⁹⁰ Sr/ ⁹⁰ Y	232	0.005	551	0.01	551	0.001	551	0.006
⁹⁹ Tc	14,559	5.0E104	1.459E+04	116	1.459E+04	0.12	1.459E+04	0.87
¹²⁹ I	867	1.3E105	890	3.8	890	0.77	890	1.35
¹⁴ C	1.1998E+05	1.6E114	1.216E+05	4.1E109	1.216E+05	6.3E112	1.216E+05	2.9E111

t. DOE-ID, 2003a, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).



EXPLANATION

- 79 S • WELL COMPLETED IN THE SNAKE RIVER PLAIN AQUIFER-Entry, 79, is the local well identifier (numbers are USGS wells) and S is the frequency at which the water level is measured: A, annually; S, semiannually; Q, quarterly; M, monthly

Figure 3. Location of wells and frequency of water-level measurements (as of December 1995) in the Snake River Plain aquifer, Test Reactor Area, Idaho Chemical Processing Plant [now INTEC], and Radioactive Waste Management Complex. [Note: The percolation ponds are designated as infiltration ponds on the figure.]^u

u. Bartholomay, R. C., B. J. Tucker, D. J. Ackerman, and M. J. Liszewski, 1997, *Hydrologic Conditions and Distribution of Selected Radiochemical and Chemical Constituents in Water, Snake River Plain Aquifer, Idaho National Engineering Laboratory, Idaho, 1992 through 1995*, DOE/ID-22137, U.S. Geological Survey Water Resources Investigations 97-4086, April 1997.

Table 7. Chloride concentrations in water from selected wells at the Idaho National Engineering Laboratory [now INL], 1981-95 [concentrations in mg/L].^v

Well CFA-1		Well 37		Well 40		Well 47	
Date	Concentration	Date	Concentration	Date	Concentration	Date	Concentration
10/13/81	82	10/07/81	66	10/12/81	150	10/81	NS
10/11/82	86	10/07/82	85	10/07/82	150	10/07/82	32
10/06/83	78	10/10/83	59	10/10/83	150	10/17/83	160
10/12/84	53	10/10/84	53	10/16/84	44	10/23/84	24
10/25/85	51	10/28/85	37	10/29/85	23	10/29/85	21
10/31/86	78	10/27/86	50	10/30/86	23	10/29/86	23
10/15/87	83	10/05/87	70	10/13/87	24	10/26/87	23
10/28/88	86	10/07/88	86	11/09/88	23	9/30/88	23
10/26/89	93	9/29/89	92	10/18/89	23	10/19/89	24
10/15/90	86	10/17/90	78	10/16/90	27	10/31/90	31
10/10/91	100	10/07/91	75	10/29/91	31	10/24/91	30
10/08/92	100	10/02/92	120	11/17/92	29	10/21/92	36
10/93	NS	10/21/93	140	10/08/93	30	10/26/93	31
10/94	NS	10/07/94	140	10/18/94	30	10/19/94	38
10/11/95	100	10/11/95	100	10/19/95	37	10/16/95	35

Well 57		Well 59		Well 111		Well 113	
Date	Concentration	Date	Concentration	Date	Concentration	Date	Concentration
10/81	NS	10/06/81	37	--	--	--	--
10/06/82	140	10/06/82	47	--	--	--	--
10/13/83	130	10/13/83	60	--	--	--	--
10/10/84	68	10/10/84	28	--	--	--	--
10/29/85	42	10/30/85	40	11/05/85	50	11/01/85	67
11/14/86	64	11/14/86	35	10/27/86	100	10/27/86	110
10/09/87	67	10/06/87	23	9/25/87	120	10/02/87	150
10/05/88	74	10/21/88	24	10/04/88	120	9/27/88	160
12/22/89	67	10/23/89	22	10/04/89	120	10/06/89	160
10/29/90	85	10/12/90	44	9/27/90	140	10/03/90	180
10/24/91	120	10/21/91	190	10/25/91	110	10/08/91	170
10/06/92	160	10/23/92	67	10/09/92	130	10/22/92	200
10/12/93	180	10/25/93	44	10/21/93	140	10/23/93	220
11/03/94	170	11/01/94	47	10/13/94	130	10/17/94	210
10/11/95	150	10/23/95	150	10/26/95	100	10/17/95	190

v. Bartholomay, R. C., B. J. Tucker, D. J. Ackerman, and M. J. Liszewski, 1997, *Hydrologic Conditions and Distribution of Selected Radiochemical and Chemical Constituents in Water, Snake River Plain Aquifer, Idaho National Engineering Laboratory, Idaho, 1992 through 1995*, DOE/ID-22137, U.S. Geological Survey Water Resources Investigations 97-4086, April 1997.

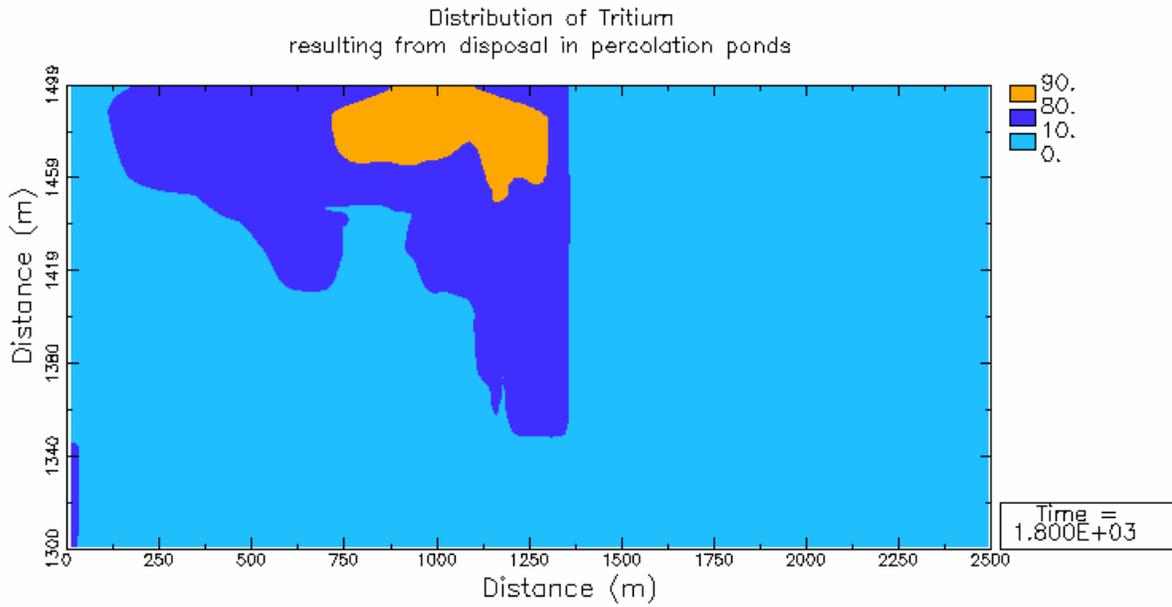


Figure 3-20. Distribution of tritium resulting from discharge into percolation ponds (units in pCi/mL).^w

w. DOE-ID, 2003a, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

NRC

Comment 14:

It is not clear that the unsaturated zone flow and transport model accounts for potentially limited solute-rock interaction during flow along basalt fractures.

The potential for colloid-facilitated release and transport was not addressed.

Basis:

As mentioned in Portage (2005j, p. 12), a previous modeling effort at INL employed zero sorption coefficients for all contaminants in the unsaturated zone because of the relatively rapid velocity of water in vertical fractures in the basalt (Rodriguez, et al., 1997). In subsequent discussions, DOE does not justify its use of non-zero sorption coefficients in the unsaturated zone.

Colloids may enhance release and transport of otherwise relatively immobile radioelements such as plutonium. While it is not clear that colloids could be important for system performance, no justification was provided for neglecting colloidal effects.

Path Forward:

DOE should explicitly justify the use, in its compliance calculations, of non-zero sorption coefficients in unsaturated basalt in light of the conclusions of Rodriguez, et al. (1997). If retardation in unsaturated basalts can be shown to have a negligible effect on performance, further justification is not needed.

Provide a discussion supporting neglect of colloidal effects on radionuclide release and transport.

Response:

DOE should explicitly justify the use, in its compliance calculations, of non-zero sorption coefficients in unsaturated basalt in light of the conclusions of Rodriguez, et al. (1997). If retardation in unsaturated basalts can be shown to have a negligible effect on performance, further justification is not needed.

The assumption by Rodriguez et al. (1997) for using a non-zero sorption coefficient in the unsaturated basalts was presented in the PA (DOE-ID 2003) as background information on previous modeling studies at the INL. The assumption by Rodriguez et al. (1997) was not based on studies of vadose zone basalt sorption characteristics; instead, it was a highly conservative assumption. For the expected case, the PA model assumed that the vadose zone basalts exhibited some degree of radionuclide sorption. Since site-specific studies (Del Debbio and Thomas 1989; Newman 1996) exist that quantify non-zero basalt sorption coefficients for INL, it is not reasonable to assume a zero sorption value in the INL vadose zone basalts. The NRC is referred to Section 7 of the PA, in which the sensitivity analysis used zero sorption coefficients for ⁹⁹Tc and ¹²⁹I in the basalts for the worst-case analysis (see Table 7-3; attached). The sensitivity analysis showed that reduced sorption coefficients (i.e., for grout, interbeds, and basalts) in the worst-case scenario did not result in drinking water doses or all-pathways doses that exceeded the performance objectives (see Tables 7-10 and 7-11; attached). Although the sensitivity analysis presented in Section 7 of the PA lumped the sorption coefficient changes together (i.e., sorption coefficients lowered simultaneously for grout, interbed sediments, and basalts), the change in predicted doses for a reduction in the vadose zone basalts would

provide a smaller change in total dose presented in the PA sensitivity/uncertainty analysis.

Additional justification for use of non-zero sorption coefficients is demonstrated by the fact that iron oxides are present in the basalt fractures (see Figure RAI-14-1). As noted in EPA (1999), iron oxide minerals are abundant in the subsurface environment and mineral coatings consisting of Fe(III)-oxides can be a significant source of reactivity for contaminant adsorption in a variety of soil systems (e.g., Smith and Jenne [1991] as summarized by Tompson et al. [1996]). The effect of chemical heterogeneity arising from spatially variable Fe(III)-oxide abundance has been considered in modeling the transport of cobalt (Brusseau and Zachara 1993) and reactive transport of uranyl-citrate (Tompson et al. 1996) and co-ethylene diaminetetraacetic acid (Szecsody et al. 1998a, 1998b) complexes. Modeling studies of reactive solute transport in a saturated system have shown that chemical heterogeneities result in non-ideal transport (i.e., contaminant sorption) behavior (Bosma et al. 1993; Sugita et al. 1995).

Sorption coefficient studies are based on crushed basalt, which provides fresh, unmineralized surfaces that do not represent the original fractured basalt or consider sorption effects of iron oxides. Figure RAI-14-1 provides a photo of an INL basalt core showing the extensive iron oxides present. Discussions on the selection of basalt sorption coefficients are explained in detail in PEI-EDF-1023 (2005).

Provide a discussion supporting neglect of colloidal effects on radionuclide release and transport.

A review of recent articles on the role of colloidal transport of contaminants (DeNovio et al. 2004; Rousseau et al. 2004) reveals that there is still a great deal of uncertainty regarding the role of colloidal effects in transport processes. On a qualitative basis, there are several factors that reduce the potential importance of colloidal transport. First, the grout that contains the radionuclides is not a source of colloidal particles, which are commonly comprised of clay, iron oxides, or microorganisms. Second, the high ionic strength of the pore water within the grout will reduce colloidal concentrations (Batcheller and Redden 2004). Highly immobile radionuclides such as plutonium need abundant colloids present in the source area to mobilize to the surrounding environment. As discussed in DeNovio et al. (2004), air-water interfaces present within the unsaturated porous media can serve as collectors of colloidal particles. Colloids that are transported to the air-water interface are retained by either capillary or electrostatic forces. Since there are over 150 m of unsaturated zone, there are numerous air-water interfaces underlying the site. Finally, the fine-grain sedimentary interbeds can remove colloids by pore straining where colloids are removed within small water-filled pores spaces that are smaller than the colloid diameter, or by the adsorbed water film where surface tension retains colloids against the mineral surfaces.



Figure RAI-14-1. Basalt core photo showing iron oxide staining.

Magnuson (2004) evaluated facilitated transport by colloids at the Subsurface Disposal Area (SDA) at the INL Site. The SDA is a low-level waste disposal site located on the INL Site, approximately 6 miles southwest of the INTEC facility. Magnuson (2004) notes:

Facilitated transport via sorption onto colloids or organic complexation has long been suspected as a possible mechanism to explain sporadic, very low concentrations of plutonium detected in the aquifer near SDA. These aquifer concentrations of Pu at the SDA are generally just at or slightly above the routine analysis method detection limit of about 0.02 pCi/L. Some of these low level detections have been confirmed with the same analytic method through reanalysis of a duplicate sample, indicating that Pu may indeed be present (Holdren et al., 2002). However, analysis with more sensitive analytical methods has not confirmed Pu in the aquifer (Roback et al., 2000). Column studies with SDA interbed sediments to evaluate if facilitated transport is a viable mechanism have been completed. The studies have shown that a very small fraction of Pu may become mobile (Fjeld et al., 2000).

Magnuson (2004) tested the effect of conceptual uncertainty associated with facilitated transport. Magnuson (2004) conducted a series of simulations to evaluate if small fractions of Pu released from the buried waste would reasonably explain the aquifer measurements. Magnuson (2004) notes:

Comparisons of the simulated aquifer concentrations to the low-level concentrations detected in the aquifer show that none of the simulated mobile fractions results are plausible. The simulated predict that ²³⁹Pu should be widespread and easily detected in the aquifer at the current date, which is not the case. This representation of facilitated transport is simplistic and did not include processes that could filter colloids being transported across the interbeds. Inclusion of attenuating processes would yield slower breakthrough and lower simulated concentrations in the aquifer. However, model calibration would still be unachievable because monitoring for Pu contaminants yield typically yield nondetect results. Nonetheless, the public interest in Pu transport remains high, even though modeling and monitoring results show transport is not likely.

There is still a great deal of uncertainty on preferential flow of colloids in unsaturated flow systems. In addition, there is currently no strong evidence that colloids provide a significant transport mechanism for highly sorbed contaminants in the unsaturated zone at the INL Site.

References:

Batcheller, T.A., and G.D. Redden, 2004, *Colloidal Plutonium at the OU 7-13/14 Subsurface Disposal Area: Estimate of Inventory and Transport Properties*, ICP/EXT-04-00253, 2004.

Bosma, W. J. P., and S. E. A. T. Van der Zee, 1993, "Transport of Reacting Solute in a One-Dimensional Chemically Heterogeneous Porous Medium," *Water Resources Research*, 29:117-131.

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Table 7-3. Overview of the parameter values for the sensitivity/uncertainty analysis.^x

		Nuclide Best Scenario	Realistic Scenario	Conservative Scenario	Worst-Case Scenario
Solid Radionuclide Inventory		50% reduction from worst case	25% reduction from worst case	10% reduction and ¹²⁹ I and ⁹⁹ Tc inventories reduced	Depicts sodium-bearing waste (undiluted tank-heel residual)
Liquid Radionuclide Inventory		95% reduction from worst case	80% reduction from worst case	50% reduction and ¹²⁹ I and ⁹⁹ Tc inventories reduced	Depicts sodium-bearing waste (undiluted tank-heel residual)
Infiltration		1.1 cm/yr	1.1 cm/yr	4.1 cm/yr	12.4 cm/yr
Grout Sorption Coefficients (m ³ /kg)	Sr	0.006	0.006	0.003	0.001
	Tc	5	5	2.5	1
	I	0.03	0.03	0.008	0.002
	C	10	10	5.0	1.0
Unsaturated Zone Longitudinal Dispersivities (m)		0.52 (sediment) 3.36 (basalt)	0.52 (sediment) 3.36 (basalt)	0.29 (sediment) 1.85 (basalt)	0.052 (sediment) 0.34 (basalt)
Unsaturated Zone Transverse Dispersivities (m)		0.26 (sediment) 1.7 (basalt)	0.26 (sediment) 1.7 (basalt)	0.14 (sediment) 0.94 (basalt)	0.026 (sediment) 0.17 (basalt)
Interbed Sediment Sorption Coefficients (mL/g)	Sr	24	24	18	12
	Tc	0.1	0.1	0.01	0
	I	5	5	0.1	0.01
	C	20	20	10	2
Basalt Sorption Coefficients (mL/g)	Sr	13	13	6	1
	Tc	0.24	0.24	0.01	0
	I	1	1	0.1	0
	C	7.1	7.1	5.0	1.7

x. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

Table 7-10. Drinking water doses for sensitivity/uncertainty groundwater analyses. (Note: Conservative-case used in dose analysis is highlighted in yellow; green shows the worst-case scenario with the conservative inventory.)^y

Parameterization			Drinking Water Dose (yr post-closure) (mrem/yr)					
Grout K _d	Transport K _d	Infiltration	Inventory	¹²⁹ I	⁹⁹ Tc	⁹⁰ Sr/ ⁹⁰ Y	¹⁴ C	Total (yr post-closure)
Worst-Case	Worst-Case	12.4	Worst-Case	23.1 (538)	1.07 (2370)	17.3 (294)	0.008 (1.4E-04)	23.1 (538)
			Conservative	8.7	0.99	17.3	0.004	17.3 (294)
			Realistic	6.7	0.09	17.3	0.002	17.3
			Best	4.4	0.05	17.3	0.0004	17.3
Worst-Case	Worst-Case	4.1	Worst-Case	9.09 (607)	0.38 (5060)	3.02 (342)	1E-4 (3.78E+04)	9.09 (607)
			Conservative	3.42	0.35	3.02	5E-05	3.42
			Realistic	2.64	0.03	3.02	2E-05	3.02 (342)
			Best	1.75	0.02	3.02	5E-06	3.02
Worst-Case	Worst-Case	1.1	Worst-Case	2.66 (844)	0.09 (1.75E+04)	0.04 (461)	9E-10 (1.0E+05)	2.66(844)
			Conservative	1.0	0.09	0.04	4E-10	1.0
			Realistic	0.77	0.008	0.04	2E-10	0.77
			Best	0.51	0.005	0.04	5E-11	0.51
Conservative	Conservative	12.4	Worst-Case	5.72 (635)	0.47 (4270)	0.02 (453)	4E-06 (5.53E+04)	5.72 (635)
			Conservative	2.15	0.43	0.02	2E-06	2.15
			Realistic	1.66	0.04	0.02	9E-07	1.66
			Best	1.10	0.02	0.02	2E-07	1.10
Conservative	Conservative	4.1	Worst-Case	2.05 (890)	0.13 (1.46E+04)	0.001 (551)	1E-11 (1.22E+05)	2.05 (890)
			Conservative	0.77	0.12	0.001	6E-12	0.77
			Realistic	0.60	0.011	0.001	2E-12	0.60
			Best	0.39	0.007	0.001	6E-13	0.39
Conservative	Conservative	1.1	Worst-Case	0.49 (1890)	0.04 (4.13E+04)	3.4E-07 (891)	0	0.49 (1890)
			Conservative	0.18	0.03	3.4E-07	0	0.18
			Realistic	0.14	0.003	3.4E-07	0	0.14
			Best	0.09	0.002	3.4E-07	0	0.09
Realistic/Best	Realistic/Best	12.4	Worst-Case	1.49 (1060)	0.23 (8100)	4.77E-05 (856)	5E-09 (9.14E+04)	1.49 (1060)
			Conservative	0.56	0.21	4.77E-05	2E-09	0.56
			Realistic	0.43	0.02	4.77E-05	9E-10	0.43
			Best	0.29	0.01	4.77E-05	2E-10	0.29
Realistic/Best	Realistic/Best	4.1	Worst-Case	0.50 (1960)	0.07 (2.33E+04)	3.53E-07 (988)	2E-17 (1.83E+05)	0.50 (1960)
			Conservative	0.19	0.07	3.53E-07	9E-18	0.19
			Realistic	0.14	0.006	3.53E-07	4E-18	0.14
			Best	0.09	0.004	3.53E-07	9E-19	0.09
Realistic/Best	Realistic/Best	1.1	Worst-Case	0.14 (5670)	0.02 (8.05E+04)	1.14E-12 (1310)	0	0.14 (5670)
			Conservative	0.05	0.01	1.14E-12	0	0.05
			Realistic	0.04	0.001	1.14E-12	0	0.04
			Best	0.03	0.0008	1.14E-12	0	0.03

y. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

Table 7-11. All-pathway doses for sensitivity/uncertainty groundwater analyses. (Note: Conservative-case used in dose analysis is highlighted in yellow; green shows the worst-case scenario with the conservative inventory.)^z

Parameterization				All-Pathways Dose (yr post-closure) (mrem/yr)				
Grout K _d	Transport K _d	Infiltration	Inventory	¹²⁹ I	⁹⁹ Tc	⁹⁰ Sr/ ⁹⁰ Y	¹⁴ C	Total (yr post-closure)
Worst-Case	Worst-Case	12.4	Worst-Case	40.4 (538)	7.52 (2370)	85.8 (294)	0.04 (1.41E+04)	85.8 (294)
			Conservative	15.2	6.98	85.8	0.02	85.8
			Realistic	11.7	0.630	85.8	0.008	85.8
			Best	7.76	0.38	85.8	0.002	85.8
Worst-Case	Worst-Case	4.1	Worst-Case	15.9 (607)	2.65 (5060)	15.0 (342)	4.95E-04 (3.78E+04)	15.9 (607)
			Conservative	5.97	2.46	15.0	2.48E-04	15.0 (342)
			Realistic	4.61	0.22	15.0	9.94E-05	15.0
			Best	3.05	0.13	15.0	2.48E-05	15.0
Worst-Case	Worst-Case	1.1	Worst-Case	4.65 (884)	0.685 (1.75E+04)	0.18 (461)	4.31E-09 (1.0E+05)	4.65 (884)
			Conservative	1.75	0.64	0.18	2.16E-09	1.75
			Realistic	1.35	0.06	0.18	8.66E-10	1.35
			Best	0.89	0.03	0.18	2.16E-10	0.89
Conservative	Conservative	12.4	Worst-Case	9.98 (635)	3.29 (4270)	0.12 (453)	2.01E-05 (5.53E+04)	9.98 (635)
			Conservative	3.75	3.05	0.12	1.01E-05	3.75
			Realistic	2.89	0.28	0.12	4.03E-06	2.89
			Best	1.92	0.16	0.12	1.01E-06	1.92
Conservative	Conservative	4.1	Worst-Case	3.59 (890)	0.94 (1.46E+04)	0.006 (551)	5.75E-11 (1.22E+05)	3.59 (890)
			Conservative	1.35	0.87	0.006	2.88E-11	1.35
			Realistic	1.04	0.08	0.006	1.15E-11	1.04
			Best	0.69	0.05	0.006	2.88E-12	0.69
Conservative	Conservative	1.1	Worst-Case	0.86 (1890)	0.25 (4.13E+04)	1.68E-06 (891)	0	0.86 (1890)
			Conservative	0.32	0.23	1.68E-06	0	0.32
			Realistic	0.25	0.21	1.68E-06	0	0.25
			Best	0.17	0.01	1.68E-06	0	0.17
Realistic/Best	Realistic/Best	12.4	Worst-Case	2.61 (1060)	1.62 (8100)	2.36E-04 (856)	2.23E-8 (9.14E+04)	2.61 (1060)
			Conservative	0.98	1.50	2.36E-04	1.11E-08	1.5
			Realistic	0.76	0.14	2.36E-04	4.47E-09	0.76
			Best	0.50	0.08	2.36E-04	1.11E-09	0.50
Realistic/Best	Realistic/Best	4.1	Worst-Case	0.87 (1960)	0.5 (2.33E+04)	1.75E-06 (988)	9.02E-17 (1.83E+05)	0.87(1960)
			Conservative	0.33	0.46	1.75E-06	4.51E-17	0.46
			Realistic	0.25	0.04	1.75E-06	1.81E-17	0.25
			Best	0.17	0.02	1.75E-06	4.52E-18	0.17
Realistic/Best	Realistic/Best	1.1	Worst-Case	0.24 (5670)	0.11 (8.05E+04)	5.65E-12 (1310)	0	0.24 (5670)
			Conservative	0.088	0.1	5.65E-12	0	0.10
			Realistic	0.068	0.009	5.65E-12	0	0.07
			Best	0.045	0.005	5.65E-12	0	0.04

z. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

NRC

Comment 15:

It is not clear to what degree existing and more recent sorption data relevant to INL were used in developing and evaluating values used for the performance assessment.

Basis:

Site-specific sorption data used in developing and justifying performance assessment parameters for carbon, iodine, strontium, and technetium are limited to only two references (Del Debbio and Thomas, 1989; Rodriguez, et al., 1997). The K_d evaluation (Portage, 2005j) mentions a field-based study that provided K_d information (Beasley, et al., 1998), but does not quote values nor use this study in developing performance assessment parameters. In addition, it is notable that the most recent sorption reference in Portage (2005j) is from 2000. It is possible that more recent information may be available; for example, Fjeld, et al. (2001) present data on strontium sorption in basalt and sediment column experiments.

Path Forward:

Confirm that no available sorption data relevant to INL have been neglected in developing and supporting performance assessment parameters.

Also, please provide the following references:

Del Debbio, J. A. and T. R. Thomas. "Transport Properties of Radionuclides and Hazardous Chemical Species in Soils at the Idaho Chemical Processing Plant." WINCO-1068. Idaho Falls, Idaho: Westinghouse Idaho Nuclear Company, Inc. 1989.

Rodriguez, R.R., A.L. Shafer, J. McCarthy, P. Martian, D.E. Burns, D.E. Raunig, N.A. Burch, and R.L. VanHorn. "Comprehensive RI/FS for the Idaho Chemical Processing Plant OU 3-13 at the INEEL—Part A, RI/BRA Report (Final)." DOE/ID-10534, Binders 1-3. Idaho Falls, Idaho: DOE. 1997.

Response:

Confirm that no available sorption data relevant to INL have been neglected in developing and supporting performance assessment parameters.

The INL has recently conducted a search of literature for site-specific data and identified five additional studies with site-specific sorption data (Colello et al. 1998; Fjeld et al. 2001; Hawkins and Short 1965; Leecaster and Hull 2004; Liszewki et al. 1997). These studies provide sorption coefficients for americium, cobalt, cesium, neptunium, plutonium, strontium, and uranium. None of these studies contained additional sorption values for carbon, iodine, or technetium. The additional sorption values identified for strontium are shown in Table RAI-15-1.

Table RAI-15-1. Summary of sorption coefficient measurements at the INL.

Element	K _d Values (mL/g)			Reference
	Alluvium	Interbed Sediment	Basalt	
Sr	35–52	110–186	1.1–2.7	Del Debbio and Thomas (1989)
	23–26			Schmalz (1972)
	8.3–16.6			Hawkins and Short (1965)
	61–134			Liszewski et al. (1997)
		42–63	6–13	Newman (1996)
		3.6–29.4	Colello et al. (1998)	
	>47	8.0–8.4	Fjeld et al. (2001)	

Note: Values shown in bold indicate additional site-specific sorption studies not reported in the PA (DOE-ID 2003).

The additional site-specific studies for strontium sorption coefficients would not result in INL altering those chosen for use in the PA (DOE-ID 2003) analyses. The interbed sediment sorption coefficients for strontium chosen in the PA (i.e., 12–24 mL/g) are lower than those presented in Table RAI-15-1. The basalt sorption coefficients for strontium chosen in the PA (i.e., 1–13 mL/g) are also within the ranges provided in Table RAI-15-1. The conservative value used in the PA for strontium in basalts (i.e., 6 mL/g) is also lower than the values presented by Fjeld et al. (2001).

Beasley et al. (1998) state:

The scope of our study did not include ancillary measurements necessary to evaluate complex questions of speciation as it affects the subsurface mobility of the radionuclides measured. Rather, the intent was to provide ‘first-order’ comparisons of the relative rates of transport for ⁹⁹Tc, ¹²⁹I, ²³⁶U, and ²³⁷Np for contrast with those predicted in the waste repository study.

Therefore, Beasley et al. (1998) provide the relative rates of transport of radionuclides in the aquifer, but do not provide measured sorption coefficient values for use in modeling.

Beasley et al. (1998) found that a substantial fraction of ¹²⁹I is retained in the basalt rock and sedimentary deposits. Beasley et al. (1998) also note that ⁹⁹Tc has a greater mobility than ¹²⁹I in basalts. The sorption coefficients selected in the PA exhibit higher sorption coefficients for ¹²⁹I than those for ⁹⁹Tc, as indicated by Beasley et al. (1998).

Also, please provide the following references:

Del Debbio, J. A. and T. R. Thomas. "Transport Properties of Radionuclides and Hazardous Chemical Species in Soils at the Idaho Chemical Processing Plant." WINCO-1068. Idaho Falls, Idaho: Westinghouse Idaho Nuclear Company, Inc. 1989.

Rodriguez, R.R., A.L. Shafer, J. McCarthy, P. Martian, D.E. Burns, D.E. Raunig, N.A. Burch, and R.L. VanHorn. "Comprehensive RI/FS for the Idaho Chemical Processing Plant OU 3-13 at the INEEL—Part A, RI/BRA Report (Final)." DOE/ID-10534, Binders 1-3. Idaho Falls, Idaho: DOE. 1997.

These references are being provided in the response to NRC Clarifying Request 21.

References:

Beasley, T. M., P. R. Dixon, and L. J. Mann, 1998, "⁹⁹Tc, ²³⁶U, and ²³⁷Np in the Snake River Plain Aquifer at the Idaho National Engineering and Environmental Laboratory, Idaho Falls, Idaho," *Environmental Science and Technology*, Vol. 32. pp. 3,875–3,881.

Colello, J. J., J. J. Rosentreter, R. C. Bartholomay, and M. J. Liszewski, 1998, *Strontium Distribution Coefficients of Basalt Core Samples from the Idaho National Engineering and Environmental Laboratory, Idaho*, U.S. Geological Survey Water Resources Investigations Report 98-4256, 1998.

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Fjeld, R. A., T. A. DeVol, R. W. Goff, M. D. Blevins, D. D. Brown, S. M. Ince, and A. W. Elzerman, 2001, "Characterization of the Mobilities of Selected Actinides and Fission/activation Products in Laboratory Columns Containing Subsurface Material from the Snake River Plain," *Nuclear Technology*, Vol. 135. pp. 92–108.

Hawkins, D. B., and H. L. Short, 1965, *Equations for the Sorption of Cesium and Strontium on Soil and Clinoptilolite*, IDO-12046, U.S. Atomic Energy Commission, 1965.

Leecaster, M. K., and L. C. Hull, 2004, *Spatial Distribution of Neptunium and Uranium Partition Coefficients (Kd) for Interbed Sediments at a Radioactive Waste Subsurface Disposal Area*, ICP/EXT-03-00088, 2004.

Liszewski, M. J., J. J. Rosentreter, and K. E. Miller, 1997, *Strontium Distribution Coefficients of Surficial Sediment Samples from the Idaho National Engineering Laboratory, Idaho*, U.S. Geological Survey Water Resources Investigations Report 97-4044, May 1997.

Newman, M. E., 1996, *Evaluation of the Mobility of Am, Cs, Co, Pu, Sr, and U Through INEL Basalt and Interbed Materials: Summary Report of the INEL/Clemson University Laboratory Studies*, WAG7-82, INEL-95/282, 1996.

Rodriguez, R. R., A. L. Shafer, J. McCarthy, P. Martian, D. E. Burns, D. E. Raunig, N. A. Burch, and R. L. VanHorn, 1997, *Comprehensive RI/FS for the Idaho Chemical Processing Plant OU 3-13 at the INEEL – Part A, RI/BRA Report (Final)*, DOE/ID-10534, Binders 1–3, November 1997.

Schmalz, B. L., 1972. *Radionuclide Distribution in Soil Mantel of the Lithosphere as a Consequence of Waste Disposal at the National Reactor Testing Station*, IDO-10049. U.S. Atomic Energy Commission, 1972.

NRC

Comment 16:

The assumptions made in the performance assessment for the acute and chronic well intruder scenario should be analyzed quantitatively in a sensitivity study.

Basis:

The assumptions regarding the depth and area of contamination significantly impacts the resulting dose for the acute or chronic well intruder scenarios. The assumption that the waste is spread out over a large area (2200 square meters) effects the thickness of contamination, the proximity of the receptor to the contamination, and the resulting doses from the external dose pathway. The basis for the assumed 6" diameter (vs. 8" diameter) well for the chronic well intruder scenario, as well as the assumed area of contamination significantly impacts the concentrations and dose from the external dose pathway.

Equation 5-19 in the PA appears to be incorrect. Furthermore, uncertainty in the inventory for the sand pads should also be taken into consideration in these calculations.

Path Forward:

Please evaluate the sensitivity of modeling results to the thickness (based on well radius and area assumption) and concentration (based on well radius, area and mixing depth assumption) of the contaminated zone for the acute and chronic well intruder scenarios, respectively. Consideration of the uncertainty associated with the sand pad inventory should also be considered.

Correct Equation 5-19 and verify whether this equation was a typographical error, or if the equation was actually used in the PA as presented.

Response:

The response to NRC Comment 16 will be provided at a later date.

NRC

Comment 17:

DOE-ID needs to determine if the final end-state of residual contamination in grouted tanks, vaults, and auxiliary equipment at the TFF is Class C or greater as defined in 10 CFR 61.55.

Basis:

The Ronald W. Reagan National Defense Authorization Act for Fiscal Year 2005 (NDAA) provides criteria for determining whether certain waste resulting from the reprocessing of spent nuclear fuel is not high-level waste (HLW). Criteria 3(A) and 3(B) of Section 3116(a) of the NDAA require that the waste be disposed of in compliance with the performance objectives contained in NRC regulations at 10 CFR 61, Subpart C. The applicability of either 3(A) or 3(B) is dependent upon whether the waste exceeds Class C concentration limits, thus the classification of waste residuals must be determined in order to apply the NDAA criteria.

Path Forward:

DOE-ID should consult the interim concentration averaging guidance (70 FR 74846) for additional information regarding acceptable methods of estimating residual concentrations in Tank Farm Facility (TFF) tanks, vaults, and auxiliary equipment. DOE-ID needs to specify the class of residual waste at the TFF, as defined in 10 CFR 61.55. Assumptions used in the calculation of waste concentrations should be clearly stated and justification for these assumptions should be provided.

Response:

The response to NRC Comment 17 will be provided at a later date.

**NRC Clarifying
Request 1:**

Information on verification of the FORTRAN sand pad inventory code was not provided. Please provide any information on verification of the FORTRAN code used to calculate sand pad radionuclide inventories.

Response:

The response to NRC Clarifying Request 1 will be provided at a later date.

NRC Clarifying Request 2:

Clarify whether localized production of carbon dioxide from microbial activity in the subsurface has been considered in estimating the impacts of carbonation on concrete degradation.

Response:

As explained below, the localized production of carbon dioxide from microbial activity in the subsurface was not considered in the concrete degradation analysis because the INL Site is semi-arid and microbial activity would be insignificant. The calculation assumed a partial pressure of CO₂ equal to 10^{-3.8} atmospheres, an atmospheric concentration. The rate of carbonation increases with the square root of the concentration of carbon dioxide in the air.

Burton et al. (1997) give soil concentrations of carbon dioxide as being less than 1,500 parts per million by volume (ppmv). The INL Site is semi-arid and the low precipitation will be associated with higher rates of gas-phase diffusion (lower soil moisture content) and lower rates of respiration. Thus, the anticipated soil carbon dioxide concentration will be closer to atmospheric than soils typical of more humid climates.

For conservatism, the degradation calculations in Appendix E of the PA (DOE-ID 2003) assumed carbon dioxide transports through rubble. A less conservative, yet more realistic analysis would assume carbon dioxide transports through concrete rather than rubble. Bin Shafique et al. (1998) show that diffusion rates increased after carbonation, but not dramatically. The calculation was repeated with a lower diffusion coefficient, which is more appropriate for carbonated concrete, and with the correct calcium content (see response to NRC Clarifying Request 6). The soil carbon dioxide concentration was also increased to 1,500 ppmv. The results are provided in Figure CR-2-1.

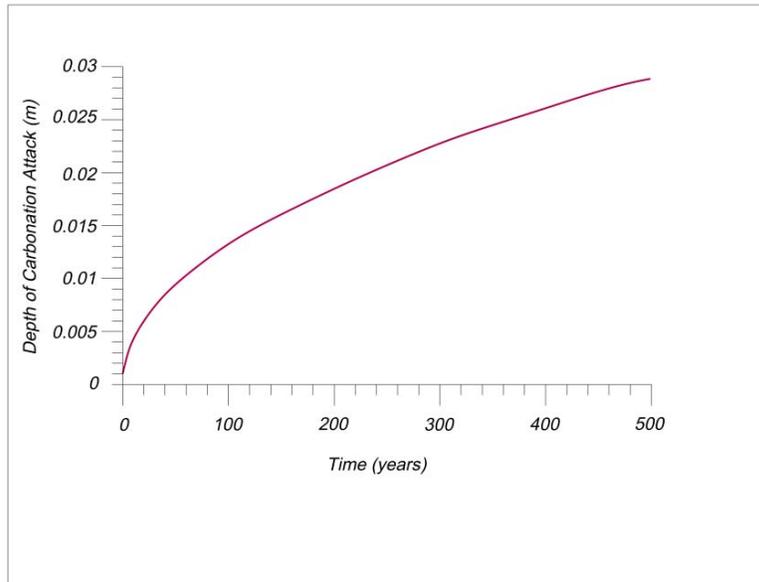


Figure CR-2-1. Carbonation attack for 1,500 ppmv carbon dioxide.

References:

Bin Shafique, Md. Sazzad, John C. Walton, Neyda Guitierrez, Robert W. Smith, and Anthony Tarquin, 1998, "Influence of Carbonation on Leaching of Cementitious Wasteforms," *Journal of Environmental Engineering*, Vol. 124, No. 5, May 1998.

Burton, A. J., G. P. Zogg, K. S. Pregitzer, and D. R. Zak, 1997, "Effect of Measurement CO₂ Concentration on Sugar Maple Root Respiration," *Tree Physiology*, 17:421-427, Heron Publishing: Victoria, Canada, 1997.

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

**NRC Clarifying
Request 3:**

The performance assessment calculations assumed that reducing conditions would prevail and used K_d values appropriate for reduced environments. No technical basis was provided for this assumption except a statement [page 7-11 of the performance assessment (DOE-ID, 2003b)] that “The concrete is expected to exhibit strong reducing conditions (Eh from !300 to !500 mV) as do most concrete systems.” This statement is incorrect. Measured values of Eh of Portland cement lie in the range +0 to +100 mV, and the redox potential is not well buffered and could easily change by the introduction of other electroactive species (Atkins and Glasser, 1992). Slag-rich cement blends, however, tend to have more reduced environments due to reduced sulfur species [from sulfides released on hydration of the blast furnace slag (BFS) glass], giving rise to strongly reducing conditions. For example, a mean Eh value of !305 mV was obtained on slag-cements (85-percent BFS) aged between 1 and 10 months (Atkins and Glasser, 1992). Reduced forms of iron and manganese (present in BFS) are unlikely to play a major role in establishing the redox level, because of their relative insolubility at high pH.

Although the performance assessment also states “the closure system will also consist of a mix of concrete and fly ash, slag, or other substances to ensure reducing conditions in the grout,” (page 7-11), it would be more appropriate to explicitly state that slag will be added to the concrete and grout to ensure the establishment of a reduced environment and mitigate the release of electroactive radionuclides, such as technetium (Tc)-99. Please clarify the percentage of slag to be used in the concrete or grout mixture to allow evaluation of whether a reduced environment will be supported.

Response:

The response to NRC Clarifying Request 3 will be provided at a later date.

**NRC Clarifying
Request 4:**

The performance assessment states, "the environment surrounding the vault is not aggressive and recent photos of the vault walls (i.e., 40- to 50-year old concrete) show no evidence of cracking" (DOE-ID, 2003b, p. E-20). The significance of this observation is not clear. When carbonation reaches the depth of the steel, accelerated steel corrosion may occur. The performance assessment does not indicate the thickness of the cover concrete, but it is likely carbonation has not yet reached the embedded steel. If that thickness were known, perhaps from the original specification, it could provide a maximum rate of carbonation that would be useful in bounding the calculations discussed in DOE-ID (2003b). Provide a more transparent application of the qualitative observation.

Response:

The initial blueprints for the vaults specify a 6- by 6-in. wire mesh with a diameter of 0.2043 in. and a minimum cover of 1.5 in. in some locations and deeper in others. Reinforcement of several sizes was specified, including #4 (0.5 in. diameter), #7 (0.875 in. diameter), and #9 (1.128 in. diameter) (INL Reference Drawings 105592, 106217, 106218, 106221).

Bin Shafique et al. (1998) show that diffusion rates increased after carbonation, but not dramatically. The carbonation calculations (discussed in detail in the response to NRC Clarifying Request 6) are shown below in Figure CR-4-1. At 50 years (the age of the vaults), the depth of carbonation attack is approximately 0.01 m, which is much less than the 0.038 m (1.5 in.) of minimum cover. It would take over 500 years for the depth of carbonation attack to reach the wire mesh at a depth of 0.038 m (1.5 in.). This is consistent with the current vault status of no visible degradation.

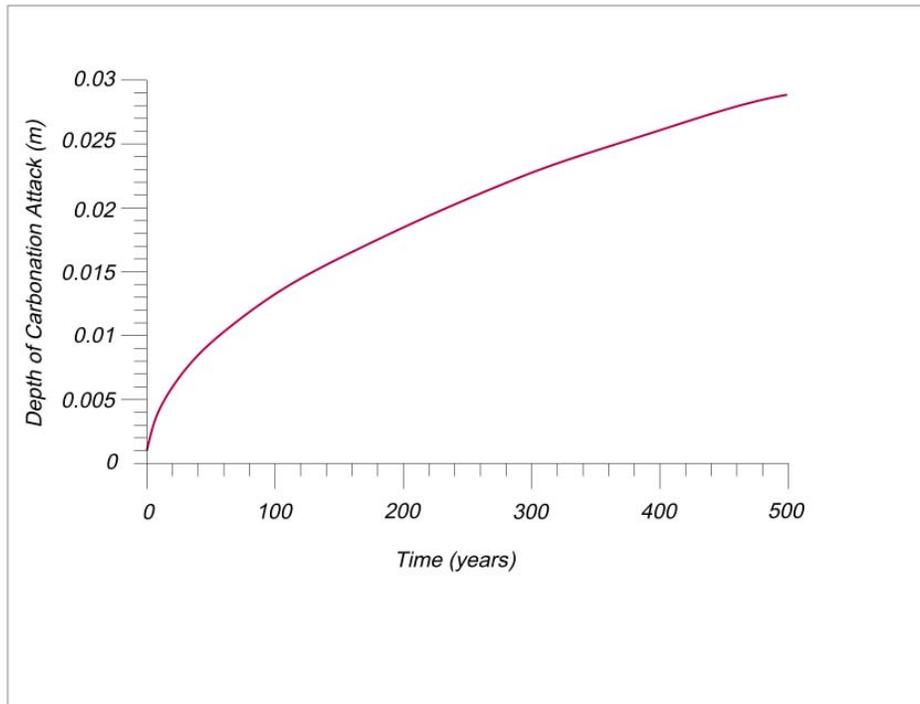


Figure CR-4-1. Carbonation attack for 1,500 ppmv carbon dioxide.

References:

105592, INL Reference Drawing, *CPP-300 WM-182 WM-183 WM-184 Tanks Structural Waste Storage Tank Base Slabs*, Alternate ID 4272-811-111, Rev. 3, June 24, 1954.

106217, INL Reference Drawing, *CPP Yard Struc. Waste Storage Tank Vaults WM-185 WM-186 Base Slab Plan Sections & Details*, Alternate ID 5773-CPP-WM-185-S-4, Rev. 4, June 24, 1954.

106218, INL Reference Drawing, *CPP Yard Struc. Waste Storage Tank Vaults WM-185 WM-186 Tank Enclosure Erection Diagram Plan Elevation Sections & Details*, Alternate ID 5773-CPP-WM-185-S-5, Rev. 3, June 24, 1954.

106221, INL Reference Drawing, *CPP Yard Struc. Waste Storage Tank Vaults WM-185 WM-186 Precast Ring Beams & Wall Panels Plans Elevations & Sections*, Alternate ID 5773-CPP-WM-185-S-8, Rev. 3, June 24, 1954.

Bin Shafique, Md. Sazzad, John C. Walton, Neyda Guitierrez, Robert W. Smith, and Anthony Tarquin, 1998, "Influence of Carbonation on Leaching of Cementitious Wasteforms," *Journal of Environmental Engineering*, Vol. 124, No. 5, May 1998.

**NRC Clarifying
Request 5:**

In DOE-ID (2003b, Appendix E), the point at which degradation of the vault to rubble occurs is taken as "when 50 percent of the reinforcement steel corrodes." It is not clear from the document why this value was selected and how it is defined, e.g., 50-percent loss of cross section or 50-percent loss of total mass. Clarify what is meant by 50 percent corrosion of the reinforcement steel.

Depending on the definition, the calculation may be sensitive to the physical dimensions of the rebar. On page E-22 (DOE-ID, 2003b, Appendix E), it is stated the reinforcement steel has a diameter of 0.25 inch. However, no basis is provided for this value. In a structure with the physical size of the tanks, larger diameter steel reinforcement would normally be specified. Because a constant rate of corrosion is applied in terms of thickness corroded per year, the calculation of the time required to reach 50 percent will be sensitive to the choice of rebar dimensions. Provide a reference for the reinforcement steel diameter.

Response:

The 50% corrosion referred to in Appendix E of the PA (DOE-ID 2003) is defined as 50% loss of the total mass. The initial blueprints for the vaults specify a 6- by 6-in. wire mesh with a diameter of 0.2043 in. and a minimum cover of 1.5 in. in some locations and deeper in others. Reinforcement of several sizes was specified, including #4 (0.5 in. diameter), #7 (0.875 in. diameter), and #9 (1.128 in. diameter) (INL Reference Drawings 105592, 106217, 106218, 106221).

The degradation calculations are based on the volume fraction of steel corroded. The time required to fail any fraction of the volume is proportional to the initial rebar diameter squared. Therefore, the assumption in the degradation analysis of small diameter rebar is conservative.

The assumption that corrosion of 50% of the rebar volume (or mass) causes failure is a professional opinion. Rebar corrosion causes failure of the vault by loss of tensile strength and by expansion. Volume per mole of iron is 7.2 cm^3 for steel and 15.3 cm^3 for hematite, a likely corrosion product. Studies have not been found that provide a solid basis for the degree of corrosion required to cause failure by rupture from internal pressure. One can perform an analysis of failure of a flexural beam from loss of tensile strength but this does not apply to a buried monolith where tensile strength is not required.

Damage from reinforcement corrosion is anticipated to begin with the shallowest steel (1.5 in.) followed by failure of more deeply buried steel. Because tensile strength is not required in a buried monolith, the importance of thickness of reinforcement cover by concrete is ambiguous. Less reinforcement cover means earlier failure from corrosion of the reinforcing steel. When tensile strength is required (e.g., prior to filling with grout), failure of reinforcement collapses the structure.

An important point of perspective is that although rebar corrosion is estimated and reported, no credit is taken for intact rebar in the other degradation or PA calculations. During operation, the concrete vault was responsible for structural

support. Subsequent to grouting as a monolith, the vault no longer provides structural support. The degradation calculations are based on a conservative assumption that transport of corrosive agents occurs through degraded concrete.

The degradation analysis in Appendix E of the PA provides a range of potential degradation times for each system component. The concrete/grout degradation analysis provided in Appendix E of the PA supports much longer degradation times than were assumed in the PA analyses. The vault and outer grout (between the stainless steel tank and vault) are assumed to degrade completely to rubble after 100 years. The tank and inner grout are assumed to degrade completely to rubble after 500 years. Therefore, the concrete/degradation analysis presented in Appendix E of the PA provides confirmation of the very conservative nature of the PA degradation assumptions (i.e., complete degradation times of 100 and 500 years).

References:

105592, INL Reference Drawing, *CPP-300 WM-182 WM-183 WM-184 Tanks Structural Waste Storage Tank Base Slabs*, Alternate ID 4272-811-111, Rev. 3, June 24, 1954.

106217, INL Reference Drawing, *CPP Yard Struc. Waste Storage Tank Vaults WM-185 WM-186 Base Slab Plan Sections & Details*, Alternate ID 5773-CPP-WM-185-S-4, Rev. 4, June 24, 1954.

106218, INL Reference Drawing, *CPP Yard Struc. Waste Storage Tank Vaults WM-185 WM-186 Tank Enclosure Erection Diagram Plan Elevation Sections & Details*, Alternate ID 5773-CPP-WM-185-S-5, Rev. 3, June 24, 1954.

106221, INL Reference Drawing, *CPP Yard Struc. Waste Storage Tank Vaults WM-185 WM-186 Precast Ring Beams & Wall Panels Plans Elevations & Sections*, Alternate ID 5773-CPP-WM-185-S-8, Rev. 3, June 24, 1954.

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

**NRC Clarifying
Request 6:**

DOE-ID (2003b, p. E-21) states that the value for the bulk concentration of Ca(OH)_2 in solid concrete used in Eq. E-14 is 27.5 kmol/m^3 . This value appears to be too high, overestimating the reserves of Ca(OH)_2 , and resulting in a lower rate of carbonation (dx/dt , Eq. E-14). Assuming a relatively cement-rich concrete is used, containing 400 kg of cement m^3 of concrete, modern Portland cement would hydrate to give about 20-percent Ca(OH)_2 . That is, the concrete will contain about $80\text{-kg Ca(OH)}_2/\text{m}^3$.

For a volume of 1 m^3 , this is equivalent to:

$$80,000 \text{ grams} / 74 \text{ grams / mole} = 1081 \text{ moles Ca(OH)}_2$$

The difference between 27.5 kmol/m^3 and 1.08 kmol/m^3 could be significant in terms of predicting the rate at which carbonation occurs.

Elsewhere in the text (DOE-ID, 2003b, p. E-22) a much lower value of Ca(OH)_2 content, 1.875 kmol/m^3 , is used for calculating Ca(OH)_2 leaching. Appendix E assumes that all the calcium is available. However, as calcium leaching progresses, the pH declines. For practical purposes, and using a conservative approach, only that fraction of calcium leached at high pH (above 12.4 at 20 EC) should be considered. Because much of the calcium is present in phases other than Ca(OH)_2 , it would be preferable to count only the contribution from Ca(OH)_2 . Hence the value 1.875 kmol/m^3 is too high if a conservative approach is to be maintained. Please clarify the discrepancies and technical basis for the assumed concentrations.

Response:

Carbonation results in a harder, denser grout with increased microcracking (Bin Shafique et al. 1998). Leaching of some constituents will increase after carbonation and release of others will decrease after carbonation. The major importance of carbonation is in protecting the steel reinforcement from corrosion. As the concrete and grout undergo carbonation, the pH is anticipated to pass through a series of stages (Atkinson et al. 1988) as different components undergo carbonation.

Stage I: pH is approximately 13 due to presence of alkali metal oxides and hydroxides. These are the first components to be leached.

Stage II: pH is 12.5 and is controlled by the solid Ca(OH)_2 following leaching of alkali metals.

Stage III: pH slowly moves to 10.5 as calcium silicate hydrate gel phases begin to dissolve incongruently following loss of calcium hydroxide. Calcium to silicon ratio drops to 0.85.

Stage IV: pH held at 10.5 by congruent dissolution of calcium silicate hydrate gel.

For this reason, carbonation calculations typically include all the calcium in the initial mixture; only including the calcium hydroxide phase would make the

calculations overly conservative. Calcium hydroxide leaching is specific to the high solubility Ca(OH)_2 phase and, thus, only should include the actual Ca(OH)_2 remaining after hydration.

It appears, from a review of the calculations contained in Appendix E of the PA (DOE-ID 2003), that the numbers used are appropriate for cement paste rather than grout. Given the grout specification of 200 lb of pozzolan per 100 lb of cement, the final concentration of calcium hydroxide should be near zero and the concentration of calcium is 0.67 kmol/m^3 . The concentration of calcium in the vault should be greater given its higher cement content.

For conservatism, the degradation calculations in Appendix E of the PA assumed carbon dioxide transports through rubble. A less conservative, yet more realistic analysis would assume carbon dioxide transports through concrete rather than rubble. Bin Shafique et al. (1998) show diffusion rates increase after carbonation, but not dramatically. Figure CR-6-1 presents the results for the calculation repeated with a lower diffusion coefficient, which is more appropriate for carbonated concrete, and with the correct calcium content. The soil carbon dioxide concentration is also increased to 1,500 ppmv.

Using diffusion rates appropriate for concrete, carbonation depth is small. If diffusion through rubble was assumed along with higher carbon dioxide concentrations and lower calcium content, carbonation would be very rapid.

Although reinforcement corrosion is estimated and reported in Appendix E of the PA, no credit is taken for intact rebar in the other degradation calculations. During operation, the concrete vaults were responsible for structural support. Subsequent to grouting as a monolith, the vaults will no longer need to provide structural support.

The degradation analysis in Appendix E of the PA provides a range of potential degradation times for each system component. The concrete/grout degradation analysis provided in Appendix E of the PA supports much longer degradation times than were assumed in the PA analyses. The vault and outer grout (between the stainless steel tank and vault) are assumed to degrade completely to rubble after 100 years. The tank and inner grout are assumed to degrade completely to rubble after 500 years. Therefore, the concrete/degradation analysis presented in Appendix E of the PA provides confirmation of the very conservative nature of the PA degradation assumptions (i.e., complete degradation times of 100 and 500 years).

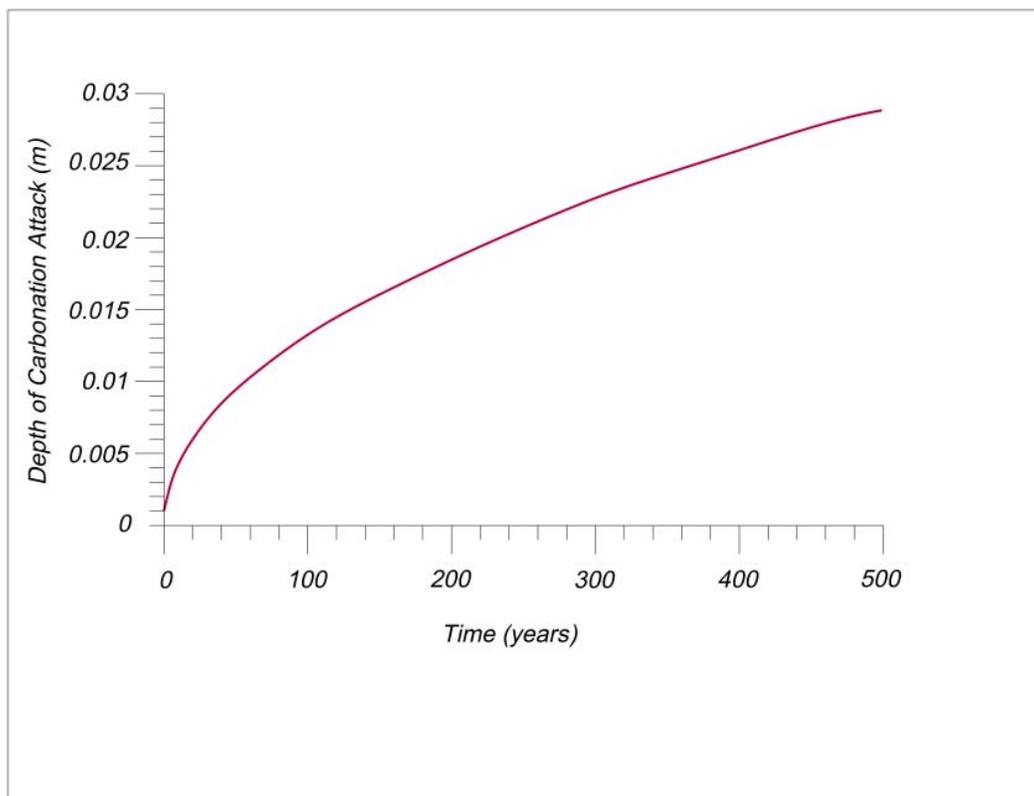


Figure CR-6-1. Depth of carbonation assuming upper range soil carbon dioxide concentrations, lower calcium content of concrete vault, and lower diffusion rates.

References:

Atkinson, A., F. T. Ewart, S. Y. R. Pugh, H. H. Rees, S. M. Sharland, P. W. Tasker, and J. D. Wilkins, 1988, "Experimental and Modeling Studies of the Near-field Chemistry for Nirex Repository Concepts," *Near-field Assessment of Repositories for Low and Medium Level Radioactive Waste*, Nuclear Energy Agency, OECD, Paris, pp. 143–157.

Bin Shafique, Md. Sazzad, John C. Walton, Neyda Guterrez, Robert W. Smith, and Anthony Tarquin, 1998, "Influence of Carbonation on Leaching of Cementitious Wasteforms," *Journal of Environmental Engineering*, Vol. 124, No. 5, May 1998.

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

**NRC Clarifying
Request 7:**

Figure 2-20 (DOE-ID, 2003b) presents relative permeability and moisture characteristic curves based on data presented in Rodriguez, et al. (1997). It is unclear whether the constitutive relationships in Figure 2-20 was used for the TFF performance assessment (2003b). If the relationships in Figure 2-20 were used in DOE-ID (2003b), small pores and absorbed water films are neglected (i.e., there is no residual water saturation using this relationship) and water drainage begins instantaneously with capillary pressure decrease (i.e., there is no air-entry pressure using this relationship). It also appears that only drainage (but not imbibition) is modeled. If a relationship by Magnuson (1995) was used, as cited on page 7-32 (DOE-ID, 2003b), it is unclear how the relationship by Magnuson differs from the one shown in Figure 2-20. Please clarify what relationship was used in the modeling.

Response:

The constitutive relationships in Figure 2-20 (attached) were used for the basalts in the PA (DOE-ID 2003) groundwater analyses. There is no residual moisture content using the basalt curves (i.e., porous medium representation of the fractured basalts) shown in Figure 2-20 and water drainage in the fractures begins instantaneously with capillary pressure decreases. This approach is intended to represent the rapid water flow through the fractures and to support the conceptual model where water released from sedimentary interbeds moves rapidly through the basalts with little or no residual water left in the fractures. This is a conservative approach in that most of the water is rapidly moved through the fractures and no credit is taken for capillary barriers at the pore/fracture interface, water lost to dead-end fractures, or water absorbed to the fracture walls.

Recent work by Magnuson (2004) compared unsaturated flow curves at a field test site and found that the Cory-based curves, similar to those used in the PA, resulted in a conservative approach to simulating flow in the unsaturated zone. Furthermore, Magnuson used a saturated hydraulic conductivity of approximately 0.1 m/d. The PA two-dimensional model used a minimum vertical hydraulic conductivity for the fractured basalts of 10 m/d, which is two orders of magnitude higher to model water moving rapidly through the basalts.

Finally, NRC's comment that imbibition is not modeled is correct. The same curve was used for both wetting and drying phases; consequently, hysteresis was not incorporated into the model. The flow model is based on steady-state flow conditions; therefore, changes in saturation are not occurring in the model domain, which would require consideration of hysteresis.

References:

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

Magnuson, S., 2004, "Regulatory Modeling for the INEEL's Subsurface Disposal Area and Conceptual Model Uncertainty Treatment," *Soil Science of America, Vadose Zone Journal*, 3:59-74.

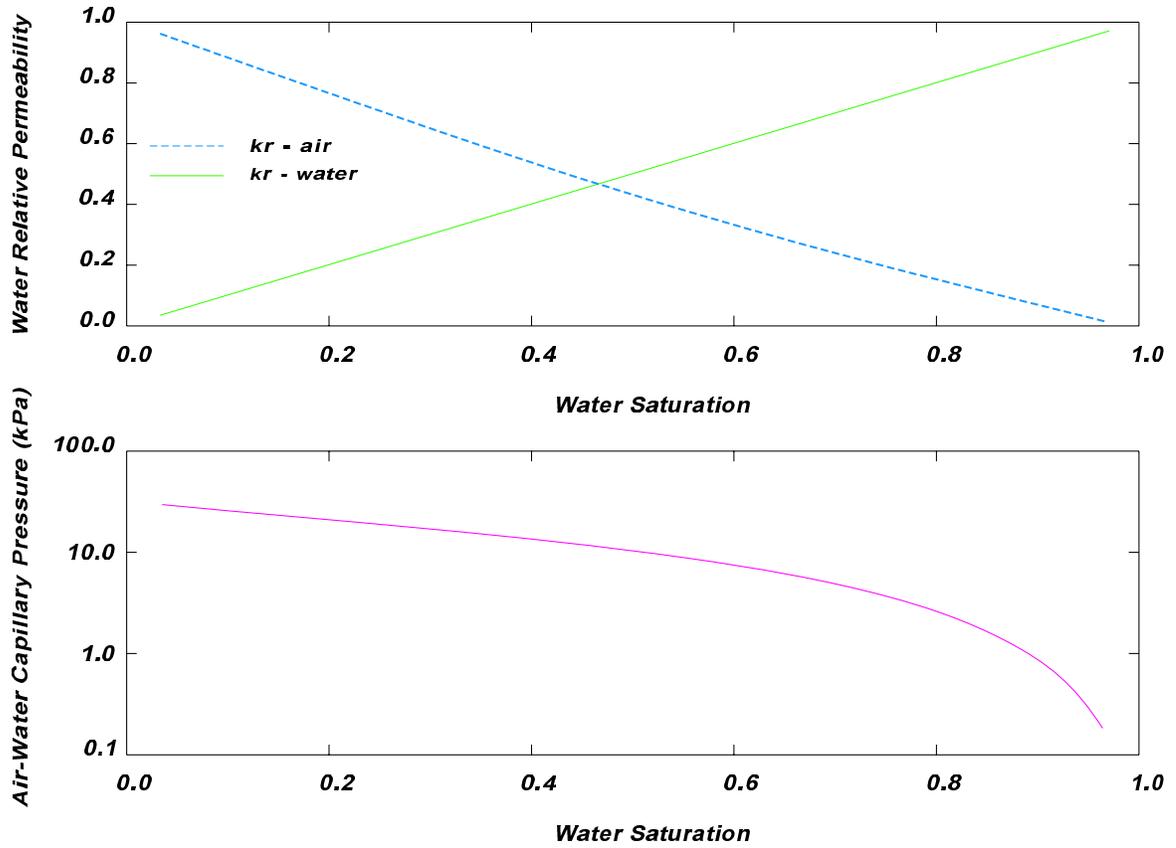


Figure 2-20. Basalt moisture characteristic curve and pressure versus saturation curve.^{aa}

aa. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

NRC Clarifying

Request 8:

Final calibrated hydraulic conductivities for major geologic layers should be provided, e.g., upper and lower perched zones.

Response:

For purposes of providing calibrated hydraulic conductivity values, it is important to clarify that the “upper and lower” perched zones consist of numerous stratigraphic units that, in some instances, combine to form a perched zone. The basalt units were assigned a vertical permeability that ranged from 10 to 100 m/d and were based on the thickness of the unit. In other words, thinner basalt flows were assumed to contain a higher density of permeable fractures due to cooling. Horizontal permeabilities in the basalts were set at a uniform value of 1.0 m/d. The assigned horizontal and vertical hydraulic conductivity values (in units of m/d) for individual interbed units in the PA (DOE-ID 2003) groundwater model were as shown in Table CR-8-1.

Table CR-8-1. The assigned horizontal and vertical hydraulic conductivity values for individual interbed units in the PA (DOE-ID 2003) (in units of m/d).

Interbed Unit Perched Water Zone	Hydraulic Conductivity (m/d)	
	Horizontal	Vertical
BC-C	3.2	3.2
C-CD	0.05	0.005
D-DE2	0.05	0.0025
DE2-DE3	2.4	0.35
DE3-DE4	0.28	0.028
DE4	0.027	0.0027
DE5	0.26	0.026
DE5-DE6	0.26	0.026
DE7-DE8	0.26	0.026

The geologic cross-section corresponding to the interbed unit designation is provided in the PA (see Figure 2-12; attached).

References:

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

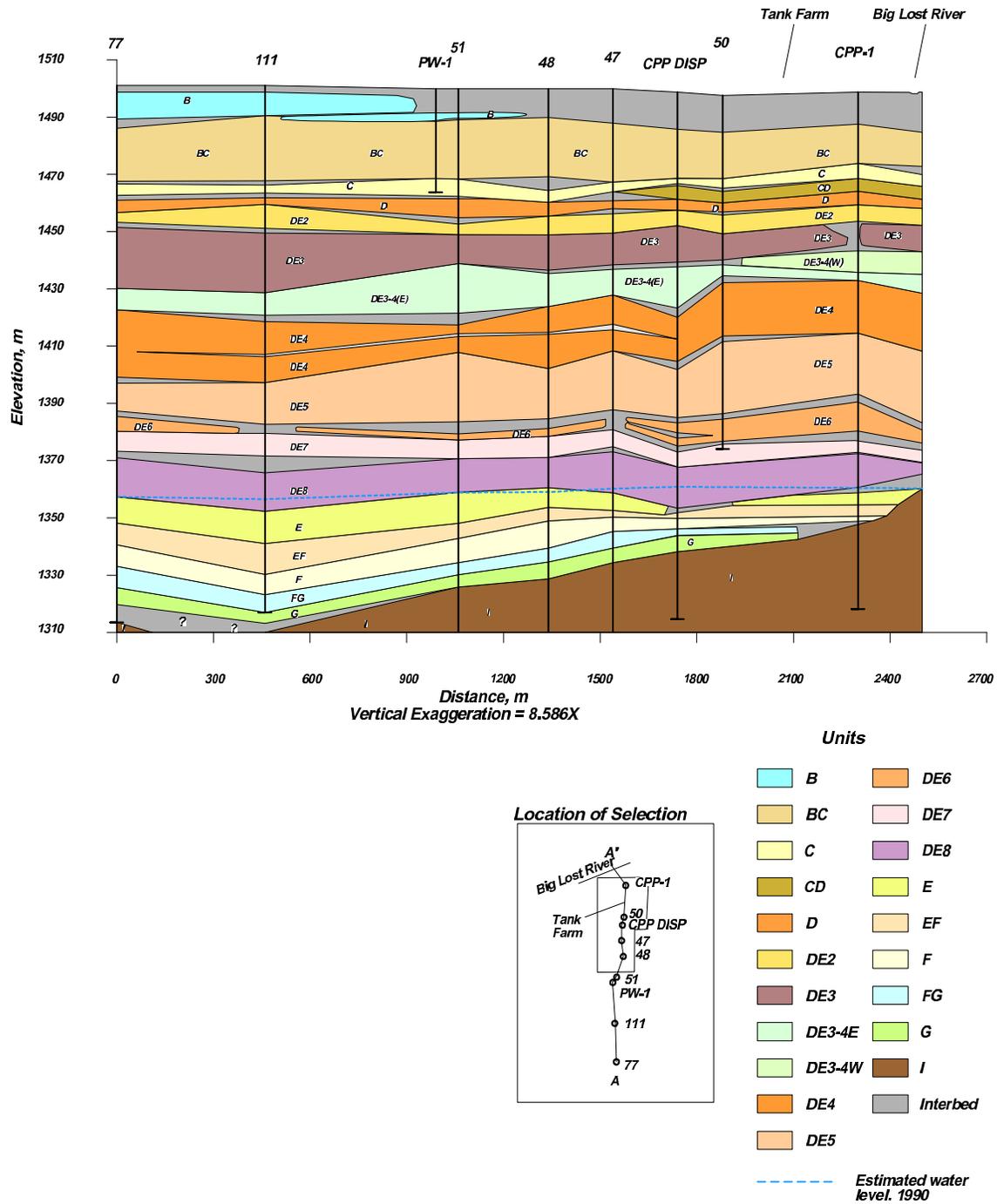


Figure 2-12. North-south geological cross-section.^{bb}

bb. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

NRC Clarifying Request 9:

Section 2.1.5.3.9 pertains to the lower perched zone (DOE-ID, 2003b). The nature and extent of the uncertainty in characterizing the lower perched zone is not clear. The statement made on page 2-44 "these data contain a high degree of uncertainty, since they consist of a combination of original driller's logs (some dating back 40 years), geophysical borehole logs, and monitoring wells that are completed in this zone," (DOE-ID, 2003b) requires more information. DOE-ID should provide more specific information regarding the source and implications of the subject uncertainty.

Response:

As discussed in Section 2.1.5.3.9 of the PA there are only four wells that are completed into the lower perched zones. The INL was able to simulate an extensive lower perched zone due to leakage from the percolation ponds (see Figure 3-18; attached) but, for conservative purposes, assigned permeability values that were in the upper range for sedimentary interbeds. For steady-state simulations, the extent of the lower perched system was based only on recharge from precipitation and the Big Lost River as shown in Figure CR-9-1. Using conservative hydraulic conductivities, the model predicts the minimum extent of the lower perched water zones. Since minimizing the extent of perched water in the lower zones provides a more direct route for the source term to reach the aquifer and does not account for any dilution of a potential contaminant plume by the lower perched water zones, the PA (DOE-ID 2003) model provides a bounding and conservative analysis based on the uncertainty of the lower perched water zones.

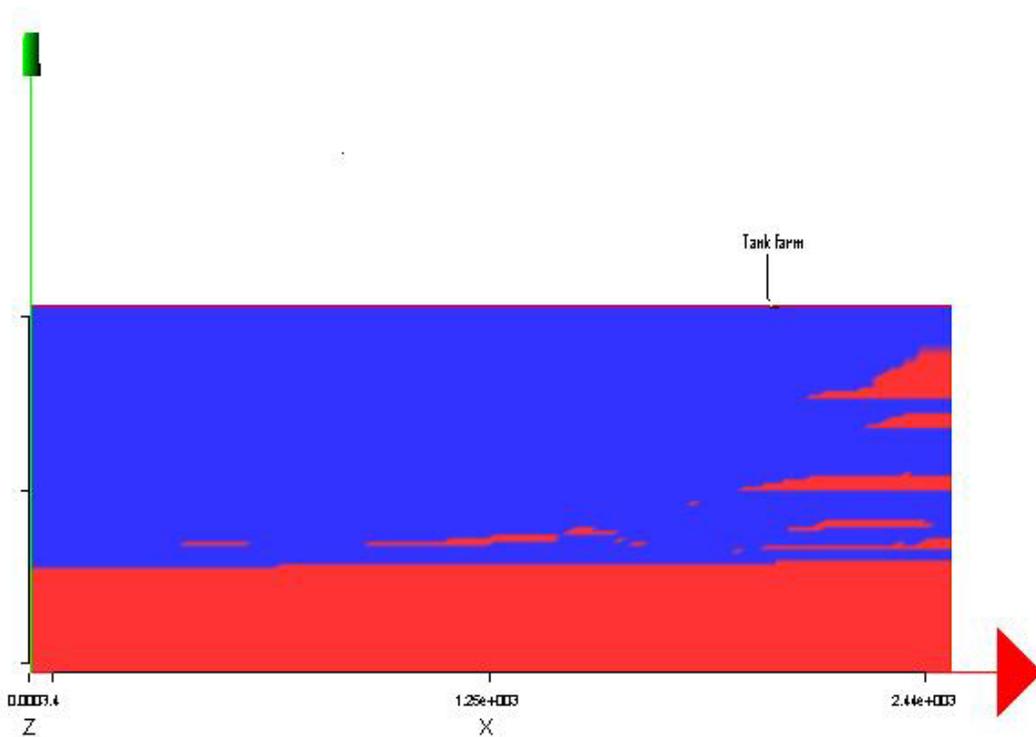


Figure CR-9-1. Final calibration perched water zones (saturated zones shown in red).

References:

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

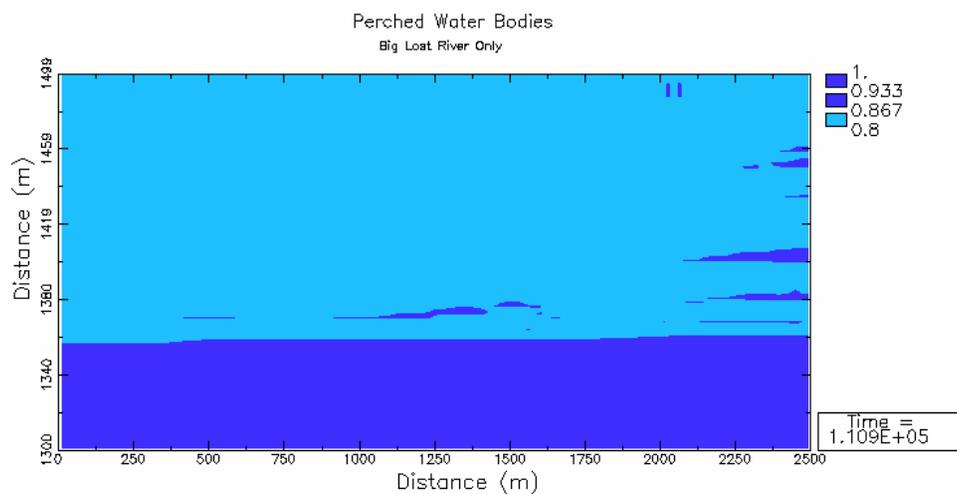


Figure 3-18. Model prediction of the hydrologic conditions illustrating the perched water zones resulting from the Big Lost River and percolation ponds seepage.^{cc}

cc. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

**NRC Clarifying
Request 10:**

The base of the groundwater model is set as a no flow boundary, partially based upon an assumption of "nonexistent future pumping of the aquifer for water supply." The appropriateness of this assumption is unclear. A more appropriate assumption might be that future pumping of the aquifer is expected to impart negligible impact on horizontal flow within the model domain, substantiated by a simple analysis given future expectations for water supply, pumping rates, and expected spatial intervals of water supply wells. DOE-ID should provide the technical basis for the no-flow boundary.

Response:

The methodology and assumptions used in the PA (DOE-ID 2003) for the receptor point and well location preclude the necessity to consider the impact of water supply wells. The PA methodology of using the highest concentrations entering the water table is conservative when compared to the dilution of the plume that would occur from a well being pumped. The base of the groundwater model was set as a no-flow boundary based on the assumption of horizontal flow. Only the upper portion (i.e., 60 m) of the aquifer was modeled for conservatism to limit the vertical dispersion in the model. The PA used an assumption that the receptor obtained water from a well located where the highest concentrations of radionuclides from the vadose zone enter the water table (see Figure 4-2; attached). Transport in the aquifer to the receptor well is not a factor using this methodology.

References:

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

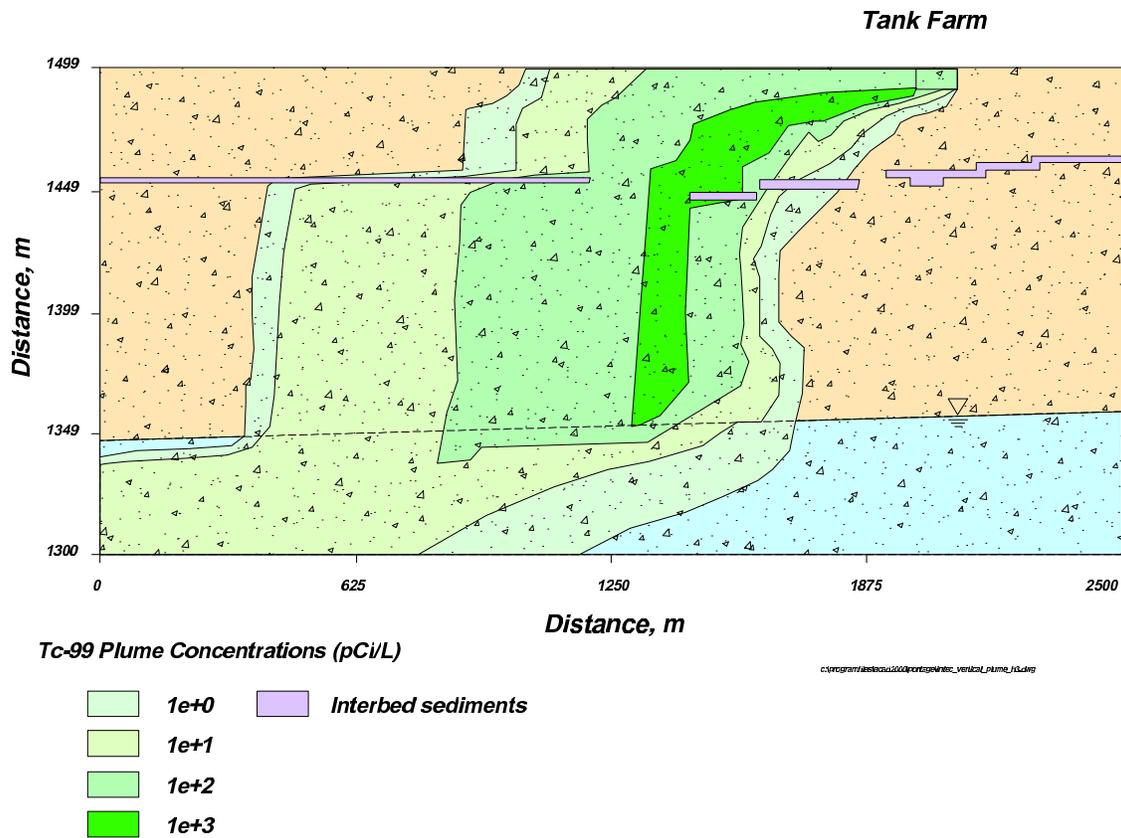


Figure 4-2. Groundwater modeling domain showing ⁹⁹Tc concentrations and location of maximum concentrations (all concentrations based on a unit source inventory).^{dd}

dd. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE-ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

**NRC Clarifying
Request 11:**

DOE-ID should provide a map showing the spatial relationship of the new Vadose Zone Research Park percolation ponds to the TFF, and indicate if there will be any influence from these percolation ponds on the formation of perched water in the study area and/or on contaminant flow and transport predictions. DOE-ID should clarify which figure in the PA shows the final distribution of perched water, as estimated via calibration, or provide this figure if it is not already included.

Response:

Scientific studies began September 2001 at the new Vadose Zone Research Park (VZRP) at the Idaho National Laboratory. The VZRP is located approximately 10 km southwest INTEC where the TFF is located (see Figures 1 and 2; attached). Waste percolation ponds for INTEC were relocated to the VZRP due to concerns that perched water within the vadose zone under the original infiltration ponds (located immediately south of INTEC) could contribute to migration of contaminants to the Snake River Plain Aquifer. The location of the new ponds at the VZRP does not influence the formation of perched water or contaminant transport in the INTEC area due to the large distance between the VZRP and INTEC (i.e., 10 km) (see Figures 1 and 2; attached). A figure of the final distribution of perched water, as estimated via calibration, was not included in the TFF PA (DOE-ID 2003). Therefore, the extent of the perched water bodies in the final calibrated flow model is provided in Figure CR-11-1 for the two-dimensional groundwater model.

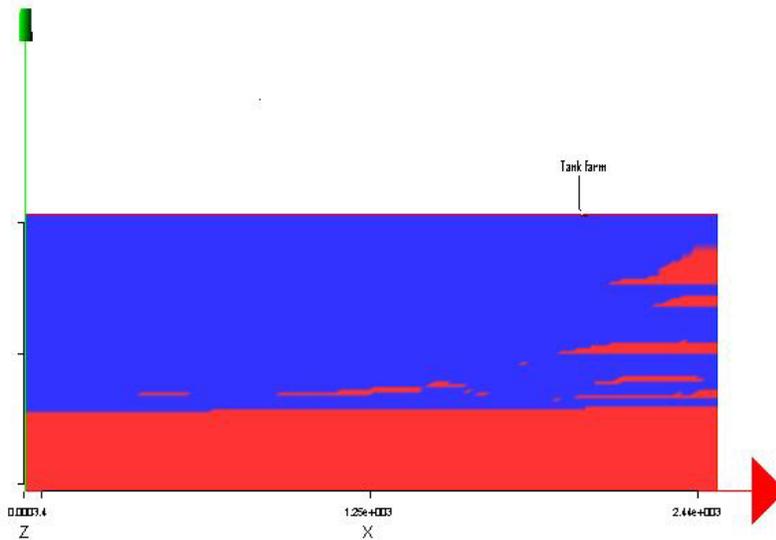


Figure CR-11-1. Final calibration perched water zones (saturated zones shown in red).

References:

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

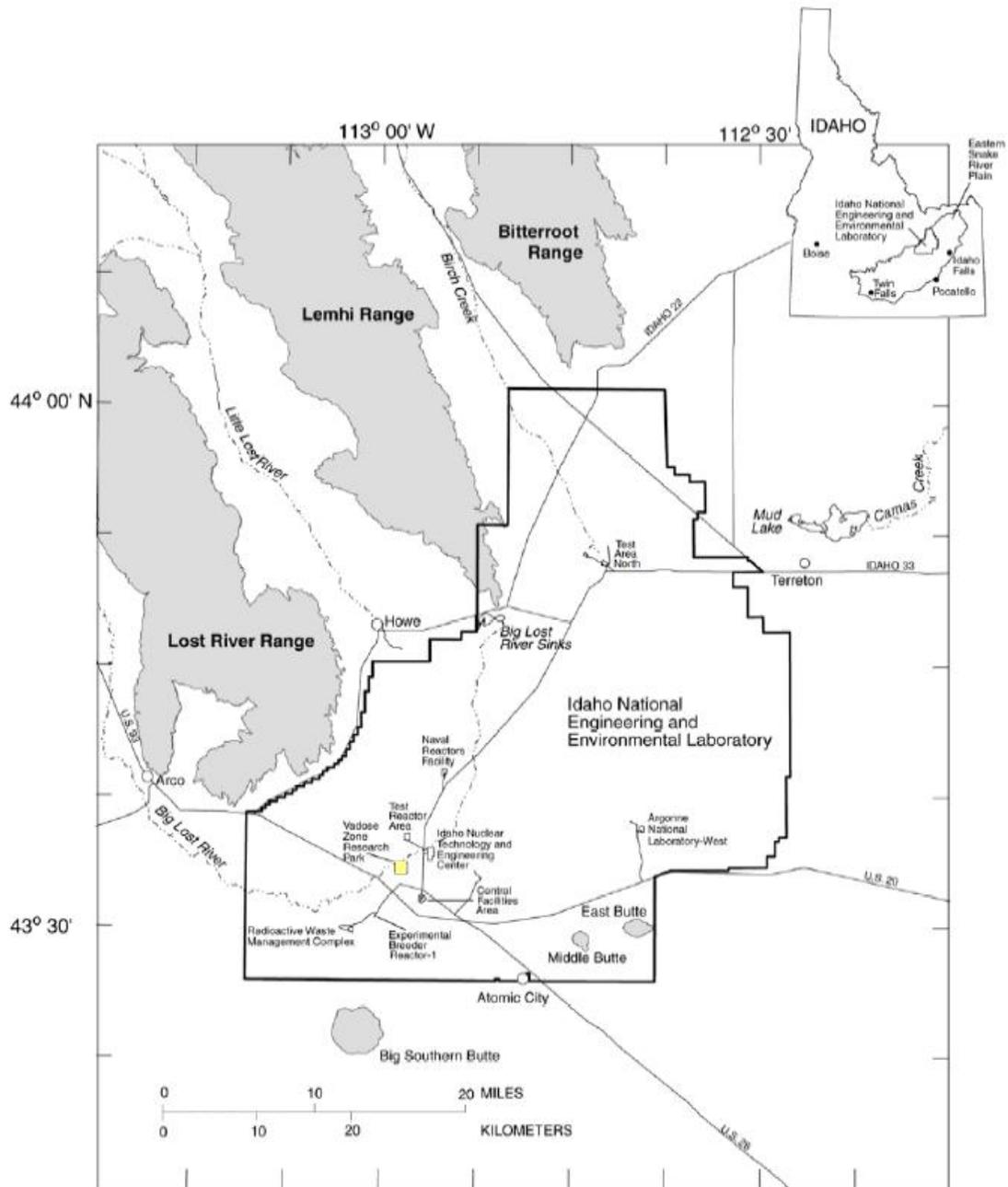


Figure 1. Location of the Idaho National Engineering and Environmental Laboratory, Idaho and selected facilities, including the Idaho Nuclear Technology and Engineering Center.^{ee}

ee. DOE-ID, 2003, *Spatial Variability of Sedimentary Interbed Properties Near the Idaho Nuclear Technology and Engineering Center at the Idaho National Engineering and Environmental Laboratory, Idaho*, DOE/ID-22187, June 2003.

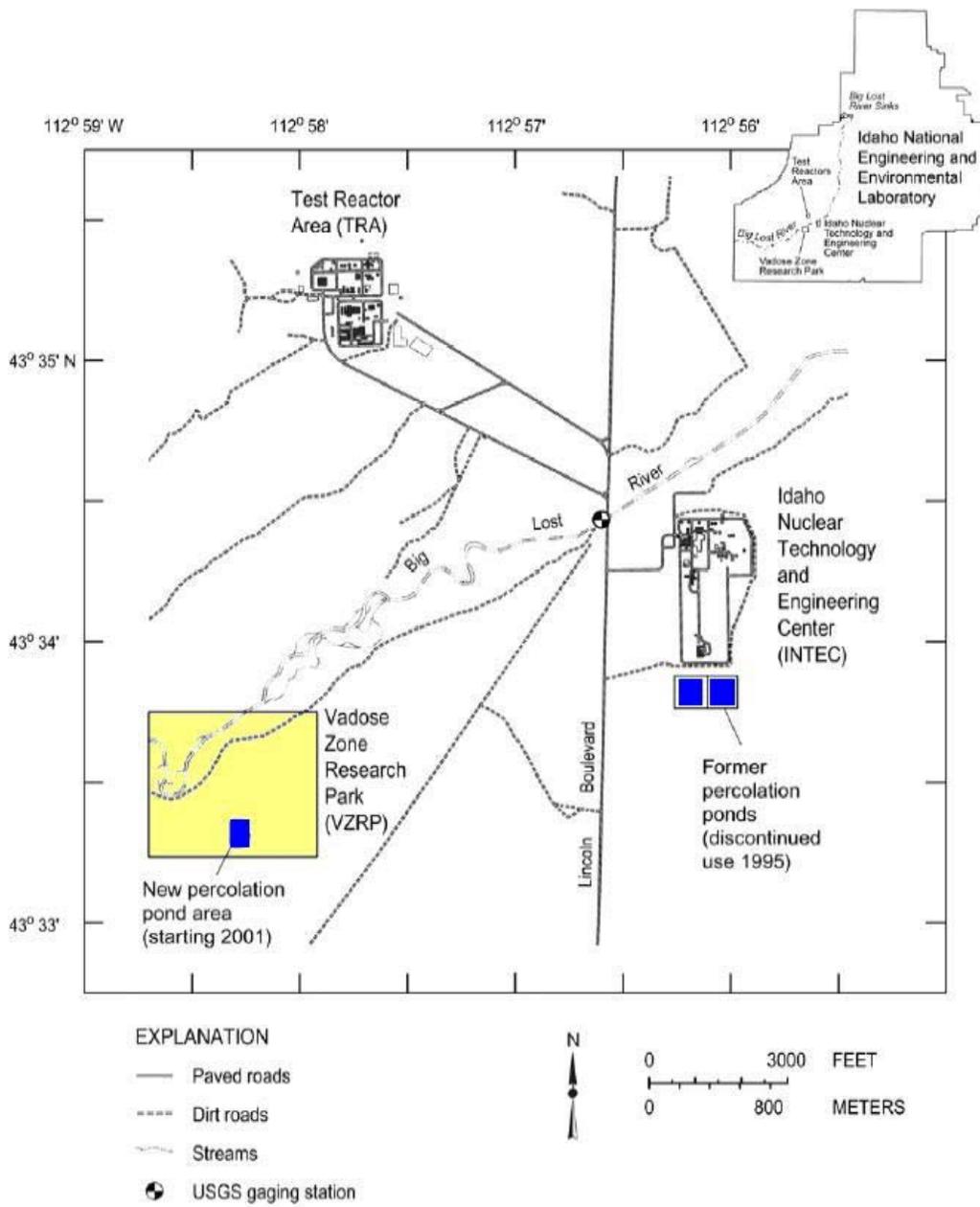


Figure 2. Location of the Vadose Zone Research Park (VZRP) relative to the Idaho Nuclear Technology and Engineering Center (INTEC) and the Big Lost River at the Idaho National Engineering and Environmental Laboratory, Idaho (adapted from the U.S. Geological Survey, 1:24,000 Circular Butte 3 SW quadrangle (1973)). Shown are the former INTEC percolation ponds, located adjacent to the facility, and the new INTEC percolation pond area within the VZRP.^{ff}

ff. DOE-ID, 2003, *Spatial Variability of Sedimentary Interbed Properties Near the Idaho Nuclear Technology and Engineering Center at the Idaho National Engineering and Environmental Laboratory, Idaho*, DOE/ID-22187, June 2003.

**NRC Clarifying
Request 12:**

With regard to the piping inventory, the waste determination states: *"The pipe encasements are similar to valve boxes in that they do not normally contain process solutions; they are designed so that any leakage into the pipe encasements would drain to sumps, which is then jetted to tanks; and the contamination in the secondary containment is addressed by using the safety factor of 500 established for the piping inventory. The safety factor of 500 is described in Subsection 2.3.1."* DOE should provide a better rationale for the use of this "500" factor. Subsection 2.3.1 only indicates the factor is used to create a conservative result.

Response:

As described in Section 2.4.5 of the draft 3116 Determination, a residual inventory estimate of 15.5 kg of SBW (30 Ci) was developed for the transfer piping at closure. Residual waste inventories at closure were not calculated for pipe encasements and valve boxes, as discussed in detail in Section 2.4.5 of the draft 3116 Determination. To account for the uncertainties associated with transfer piping, pipe encasements, and valve boxes, as described below, a safety factor of 500 was used. The safety factor was applied to an inventory that is insignificant to the PA (DOE-ID 2003) or the sum of the fractions calculations. The purpose of the inventory was to provide a bounding estimate for transfer piping, valve boxes, and pipe encasements.

The assignment of the safety factor is qualitative because a determination of a quantifiable safety factor would require more data than available. The piping system is a unique situation; therefore, there are no best acceptable practices or commonly used factors to evaluate the system. As described in Section 2.3.1 of the draft 3116 Determination, the conditions that need to be addressed by use of a safety factor include:

- Analysis for piping samples did not include radionuclides. Metals data were used and extrapolated to radionuclides.
- Bends in piping and areas near valves where turbulence occurs could have higher concentrations of contamination.
- Rinsate samples of the piping produce liquid samples. Fixed contamination was not included in the samples unless it was removed by using demineralized water. Since the piping transferred highly concentrated nitric acid, it is unlikely demineralized water would remove any additional metals.
- Samples of four 18-in. sections of piping were used to provide an inventory for 10,600 linear ft of piping and the same length of secondary containment. Samples were not collected from the secondary containment with the exception of valve boxes.

It is recognized that the safety factor of 500 is subjective and is based on engineering judgment. The safety factor of 500 could be altered to a lower number, but the inventory as proposed is intended to provide a conservative

estimate of radioactivity. A safety factor higher than 500 is not supported based on visual inspections and photographs of clean valve boxes and piping.

References:

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

**NRC Clarifying
Request 13:**

Figure 3-14 shows the conceptual model for the DUST-MS release modeling. The figure implies that release from the sand pad does not occur until the grouted tank fails at 500 years, i.e., there is a hydraulic barrier limiting infiltration to the sand pad until after 500 years. Please confirm that this conceptual model is for the tank release only.

Response:

Figure 3-14 (attached) of the PA (DOE-ID 2003) was presented only for conceptual purposes and shows the intact vault/tank system. This conceptual model is for the tank release only. No hydraulic barrier was assumed in the analysis for the sandpad releases. The sandpad releases were modeled beginning at 100 years post-closure in accordance with the degradation time assumption for the vault as described of the draft 3116 Determination. The sandpad releases were modeled in DUST-MS assuming that the material above the sandpad was not limiting the inflow of precipitation to the sandpad. In other words, the sandpad and the tank releases were modeled separately in DUST-MS. Assuming no limitation of infiltrating water to the sandpad at 100 years is considered conservative. In reality, the intact tank would provide some limitation of infiltration to the sandpad for 500 years.

References:

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

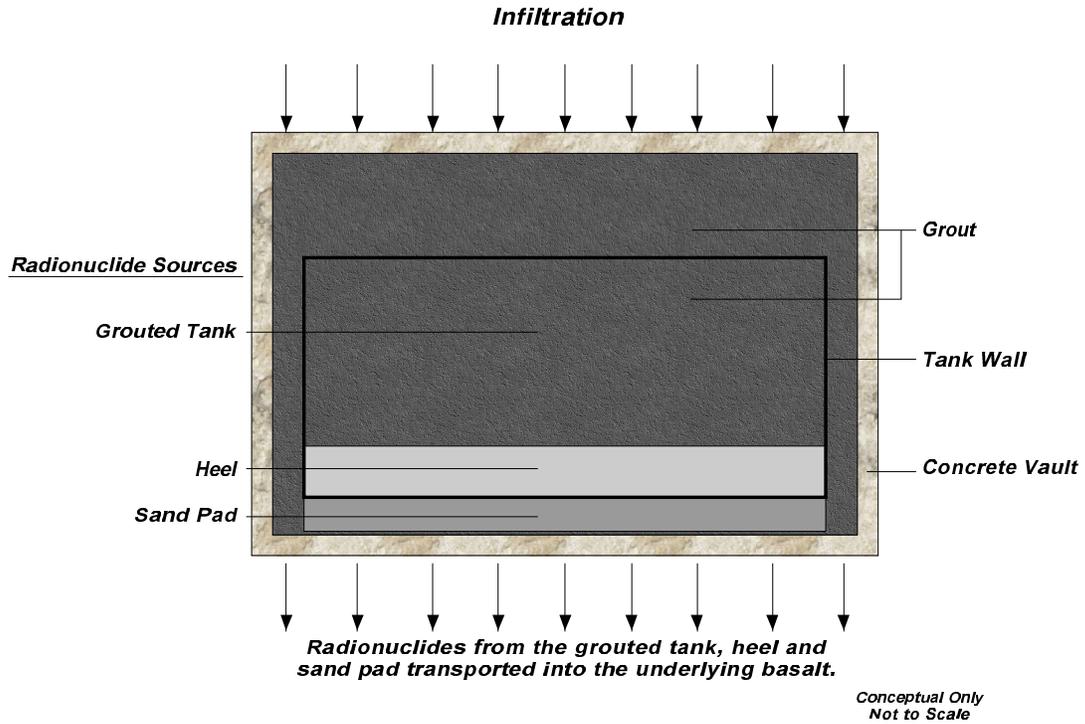


Figure 3-14. DUST-MS conceptual model for release from the TFF vaults and tanks.^{gg}

gg. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

**NRC Clarifying
Request 14:**

The DUST-MS release rates presented in Appendix F and Chapter 4 of the PA are significantly different (DOE-ID 2003b). For example, the Sr-90 peak release rate in Figure F-11 on page F-8 approaches 0.1 Ci/yr, while the release rate Figure 4-1 on page 4-2 approaches 0.001 Ci/yr. Discrepancies of similar magnitude exist for Tc and I as well. Please explain the differences in the release rates.

Response:

The information presented in Appendix F was originally developed and presented for review by the NRC in 2002 of an original draft TFF PA prepared to support a draft waste incidental to reprocessing determination under DOE Order 435.1 (Essig 2002). The NRC review of those original contaminant releases from the tanks in the PA was questioned by NRC.

The comment from NRC in 2002 was as follows:

The use of an arithmetic averaged K_d for grouted waste and a K_d approach for release modeling in the tanks need stronger technical basis.

The technical basis for the 2002 NRC comment was as follows:

In the release model, by reference to Kimmel (2000), solid waste release is calculated using an average of sand K_d (Sheppard and Thibault, 1990) and reducing concrete K_d (Bradbury and Sarott, 1995) (see Section 2.1.6 and Tables 3-8 and 3-9). This approach is based on the assumption that 50 percent of the heel waste is encapsulated in the grout (reducing grout K_d) and 50 percent is ungrouted (sand K_d). The necessity for making such assumptions in the absence of data (e.g., on the expected proportion of grouted to ungrouted waste) is clear, but the implications for release modeling are not discussed. Information on how this assumption affects performance would be useful.

In addition, there is no substantiation that an average K_d constitutes an appropriate parameter for the release modeling. The use of K_d values for calculating release inherently assumes that distribution of radionuclides between waste and solution can be adequately simulated as an equilibrium sorption relationship. In fact, release may be more dependent on dissolution/precipitation processes. This is particularly worth considering for radionuclides such as C-14 that may be stoichiometric components of waste solids and are more likely to participate in dissolution/precipitation reactions. An expansion of the discussion of solubility limits in Section 2.1.6 may be instructive. For example, it may discuss the effect on performance if radionuclides were released at their solubility limit.

Even if it could be argued that the K_d approach were valid for the release modeling, the use of an average may underestimate risk. Consider Tc-99 release. It appears (Section 3.3.1) that the release model calculates a liquid radionuclide concentration by dividing the solid waste concentration by K_d . Assuming, for simplicity, a unit of Tc-99 concentration in solid heel waste of

1 Ci/kg, use of the sand K_d ($0.0001 \text{ m}^3/\text{kg}$; Table 3-8) would yield a liquid concentration of 10^4 Ci/m^3 , and use of the reducing concrete value ($1 \text{ m}^3/\text{kg}$) yields 1 Ci/m^3 . The assumption that the waste is half one type and half another suggests the liquid results from a mixture of the liquids would be in equilibrium with these two materials (i.e., the liquid concentration would be the average of 10^4 Ci/m^3 and 1 Ci/m^3 , or 5000 Ci/m^3). The liquid concentration calculated using the average ($K_d = 0.5 \text{ m}^3/\text{kg}$), however, would be only 2 Ci/m^3 . These calculations are affected by the relative proportions of the two materials; a 50:50 mixture is assumed without basis. If the above calculation of liquid concentration is not consistent with the DUST-MS implementation, clarification is needed.

The averaging of K_d is also not appropriate for the retardation calculations inherent in DUST-MS (Section 3.3.1.2) for similar reasons, (i.e., transport times cannot be averaged). In addition to accounting for the chemical environment, justification for the average K_d must rely on characteristics of the material itself.

Based on discussions in 2002 with NRC, INL decided to abandon the K_d averaging method for the tank heel material that was contained in the original draft TFF PA. A new analysis for that PA was prepared to resolve the NRC's concerns (chapter 4 of the PA). The modeling results presented in Appendix F are from the original analyses and were maintained in the current PA to ensure that the material would be available to document the evolution of the modeling in the PA from the 2002 NRC review to present. Chapter 4 of the current PA (DOE-ID 2003), on the other hand, provides the analysis of radionuclide releases based on the concrete sorption coefficients in Bradbury and Sarott (1995) without averaging with the Sheppard and Thibault (1990) sand sorption coefficients. Therefore, the release rates provided in Appendix F would provide different results than those presented in the current PA analyses. For any future revision of the PA consideration will be given to removing Appendix F to eliminate confusion.

References:

Bradbury, M. H., and F. A. Sarott, 1995, *Sorption Databases for the Cementitious Near-Field of a LLW Repository for Performance Assessment*, Paul Scherrer Institute, 95-12, March 1995.

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

Essig, T. H., Office of Nuclear Material Safety and Safeguards, NRC, to J. Case, Director INTEC Waste Programs, DOE-ID, June 5, 2002, "Request for Additional Information on the Idaho National Engineering and Environmental Laboratory Draft Waste Incidental to Reprocessing Determination for Tank Farm Facility Residuals (DOE/ID-10777)."

Kimmel, R. J., DOE-ID, to S. Connor, Tetra Tech NUS, September 6, 2000, “ K_d Values and Physical Properties for Groundwater Modeling (EM-EIS-00-040).”

Sheppard, M. I., and D. H. Thibault, 1990, “Default Soil Solid/Liquid Partition Coefficients, K_{ds} , for Four Major Soil Types: A Compendium,” *Health Physics*, Vol. 59, pp. 471–482.

**NRC Clarifying
Request 15:**

Please clarify how the DUST-MS release rates were incorporated into the PORFLOW model. Detail any averaging assumptions in space and time of the source input that were necessary to facilitate differences in model scale, e.g., grid and time step size of the PORFLOW model.

Response:

The PORFLOW groundwater model is based on a two-dimensional slice through the major axis of two tanks (i.e., tank diameter of 15.24 m) that is 1 m wide, representing an area of 15.24 m² for each tank. The DUST-MS release rates; presented in the TFF PA (DOE-ID 2003), represent the releases from one tank with an area of 182.41 m². Therefore, the DUST-MS release rates required modification of the space component for input into the two-dimensional PORFLOW groundwater model. The DUST-MS releases were incorporated into the PORFLOW model by multiplying the ratio of the two-dimensional slice area for one tank to the area of an entire tank (i.e., 0.084) for input into the PORFLOW model. This adjusted release rate was input into PORFLOW at the grid nodes representing each tank location (i.e., two tanks in the model). Release rates from DUST-MS were also adjusted from units of Ci/yr to units of Ci/day for input into PORFLOW. Source adjustments in terms of solution time step were not required, since PORFLOW interpolates between the release rates and corresponding times input to PORFLOW from DUST-MS for a given solution iteration.

References:

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

**NRC Clarifying
Request 16:**

On page 3-54 and 3-55, DOE-ID discusses calibration of the transport model using tritium data from the percolation ponds (Orr and Cecil 1991). It is not clear that the most appropriate comparisons were made with the data. Please clarify the extent of tritium contamination versus the modeled predicted extent of contamination (only a comparison of concentration was made). Please provide the Orr and Cecil reference.

Response:

The response to NRC Comment 10 contains a more detailed discussion on the calibration of the PA groundwater model (DOE-ID 2003). The following is a summary:

Several of the perched water-series wells were used in the PA to compare predicted with actual tritium concentrations resulting from disposal operations in the percolation ponds. As noted in the PA, the model predicts concentrations that range from 10 to 80 pCi/mL compared with reported values by Cecil et al. (1991) of 5.5 to 36.7 pCi/mL. Considering the variations in the initial concentrations of tritium that were disposed of in the ponds and the yearly disposal volume variations, the relative magnitude of predicted versus actual concentrations is very good. The perched water-series wells are relatively shallow and only provide calibration for the upper portions of the unsaturated zone.

The purpose of the PA model is to describe conditions in the unsaturated zone and locate the highest concentrations in the regional aquifer where contaminants enter from the vadose zone. The disposal of other species of potential contaminants was evaluated to assess the effectiveness of the PA model to predict migration pathways. Chloride was disposed of in the Idaho Chemical Processing Plant (ICPP) (now INTEC) disposal well, located in the center of the ICPP facility, from 1952 to February 1984. After this time, chemical wastes were diverted to the infiltration ponds located south of the ICPP facility. Bartholomay et al. (1997) present water quality data for several regional groundwater wells surrounding the percolation ponds, provided in Table 7 (attached) and corresponding well locations in Figure 2 (attached). Since chloride flows at the same rate as the groundwater, similar to tritium, the tritium calibration study in the PA can be used to assess the transport of chloride as well. Wells 40 and 47, located approximately 1,000 m north of the percolation ponds but near the disposal well, show elevated chloride in late 1983, presumably due to the disposal well. However, water quality data from 1984 to 1995, after the disposal well was no longer in service, indicate chloride concentrations that are near background water quality. These data indicate that wastewater from the disposal well impacted Wells 40 and 47 in 1984, but after disposal operations switched to the percolation ponds, infiltrating water from the ponds had no impact on these wells. Comparing these actual water quality data with the extent of the contaminant plume predicted by the model calibration simulation illustrated in Figure 3-20 (attached) of the PA, the extent of the predicted plume is approximately 350 m north or upgradient of the percolation ponds. Well 40 and 47 are approximately 1,000 m north of the percolation ponds and, based on water-quality data, are beyond the influence of chloride infiltration from the ponds, in agreement with the PA model calibration predictions. Well 59, located

upgradient of the percolation ponds and within the 350-m upgradient distance predicted by the model, shows elevated chloride concentrations, once again matching the PA model calibration predictions. As expected, all of the downgradient wells, 37, 57, 111, and 113, show elevated concentration of chloride, consistent with the PA model calibration predictions.

In summary, the calibration of the chloride data with the PA model predictions of the plume extent due to seepage from the percolation ponds clearly illustrates the influence of the Big Lost River on the migration pathways in the unsaturated zone. The balance of hydraulic forces between infiltration from the percolation ponds and recharge from the Big Lost River results in groundwater concentrations that agree with existing groundwater-quality data from wells in the regional aquifer.

The PA model provides an accurate representation of hydrologic processes controlling water movement and contaminant transport in the vadose zone underlying the INTEC facility and is verified by existing site data.

References:

Bartholomay, R. C., B. J. Tucker, D. J. Ackerman, and M. J. Liszewski, 1997, *Hydrologic Conditions and Distribution of Selected Radiochemical and Chemical Constituents in Water, Snake River Plain Aquifer, Idaho National Engineering Laboratory, Idaho, 1992 through 1995*, DOE/ID-22137, U.S. Geological Survey Water Resources Investigations 97-4086, April 1997.

Cecil, L. D., B. R. Orr, T. J. Norton, and S. R. Anderson, 1991, *Formation of Perched Ground-water Zones and Concentrations of Selected Chemical Constituents in Water, Idaho National Engineering Laboratory, Idaho 1986-88*, DOE/ID-22100, U.S. Geological Survey Water-Resources Investigations Report 91-4166, November 1991.

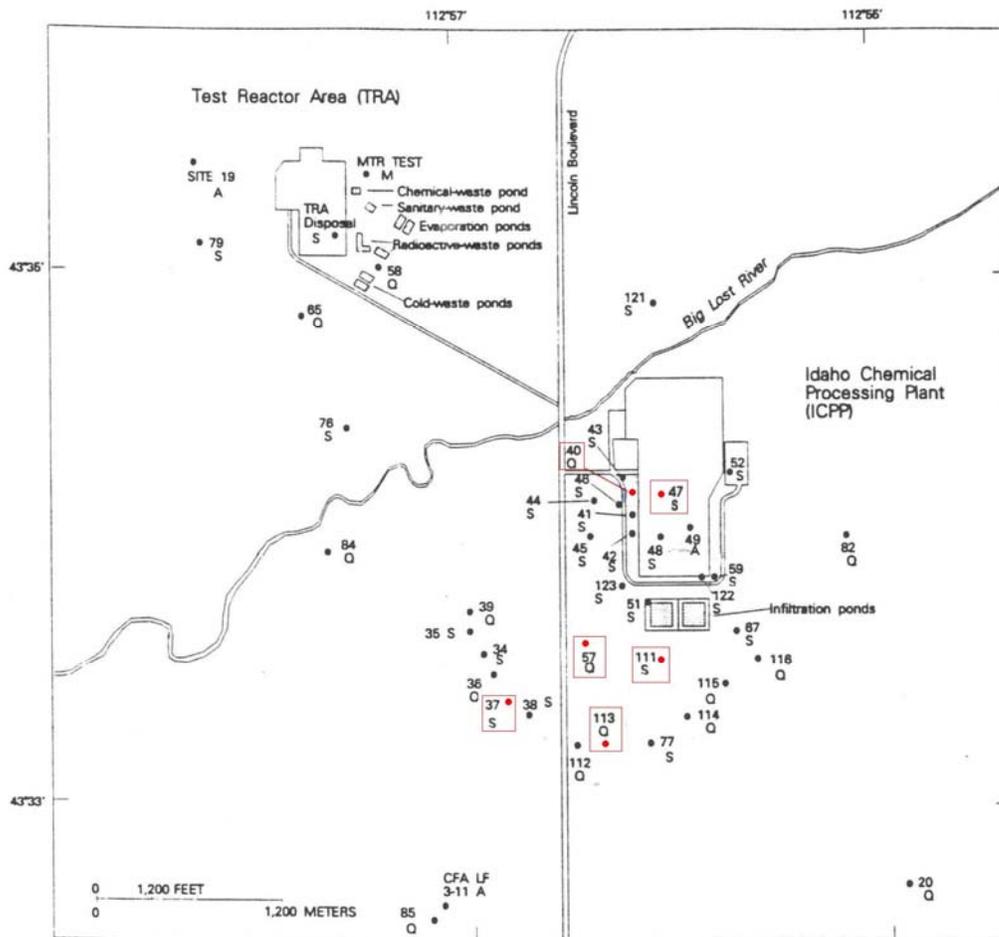
DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

Table 7. Chloride concentrations in water from selected wells at the Idaho National Engineering Laboratory [now INL], 1981-95 [concentrations in mg/L].^{hh}

Well CFA-1		Well 37		Well 40		Well 47	
Date	Concentration	Date	Concentration	Date	Concentration	Date	Concentration
10/13/81	82	10/07/81	66	10/12/81	150	10/81	NS
10/11/82	86	10/07/82	85	10/07/82	150	10/07/82	32
10/06/83	78	10/10/83	59	10/10/83	150	10/17/83	160
10/12/84	53	10/10/84	53	10/16/84	44	10/23/84	24
10/25/85	51	10/28/85	37	10/29/85	23	10/29/85	21
10/31/86	78	10/27/86	50	10/30/86	23	10/29/86	23
10/15/87	83	10/05/87	70	10/13/87	24	10/26/87	23
10/28/88	86	10/07/88	86	11/09/88	23	9/30/88	23
10/26/89	93	9/29/89	92	10/18/89	23	10/19/89	24
10/15/90	86	10/17/90	78	10/16/90	27	10/31/90	31
10/10/91	100	10/07/91	75	10/29/91	31	10/24/91	30
10/08/92	100	10/02/92	120	11/17/92	29	10/21/92	36
10/93	NS	10/21/93	140	10/08/93	30	10/26/93	31
10/94	NS	10/07/94	140	10/18/94	30	10/19/94	38
10/11/95	100	10/11/95	100	10/19/95	37	10/16/95	35

Well 57		Well 59		Well 111		Well 113	
Date	Concentration	Date	Concentration	Date	Concentration	Date	Concentration
10/81	NS	10/06/81	37		--		--
10/06/82	140	10/06/82	47		--		--
10/13/83	130	10/13/83	60		--		--
10/10/84	68	10/10/84	28		--		--
10/29/85	42	10/30/85	40	11/05/85	50	11/01/85	67
11/14/86	64	11/14/86	35	10/27/86	100	10/27/86	110
10/09/87	67	10/06/87	23	9/25/87	120	10/02/87	150
10/05/88	74	10/21/88	24	10/04/88	120	9/27/88	160
12/22/89	67	10/23/89	22	10/04/89	120	10/06/89	160
10/29/90	85	10/12/90	44	9/27/90	140	10/03/90	180
10/24/91	120	10/21/91	190	10/25/91	110	10/08/91	170
10/06/92	160	10/23/92	67	10/09/92	130	10/22/92	200
10/12/93	180	10/25/93	44	10/21/93	140	10/23/93	220
11/03/94	170	11/01/94	47	10/13/94	130	10/17/94	210
10/11/95	150	10/23/95	150	10/26/95	100	10/17/95	190

hh. Bartholomay, R. C., B. J. Tucker, D. J. Ackerman, and M. J. Liszewski, 1997, *Hydrologic Conditions and Distribution of Selected Radiochemical and Chemical Constituents in Water, Snake River Plain Aquifer, Idaho National Engineering Laboratory, Idaho, 1992 through 1995*, DOE/ID-22137, U.S. Geological Survey Water Resources Investigations 97-4086, April 1997.



EXPLANATION

- 79 S • WELL COMPLETED IN THE SNAKE RIVER PLAIN AQUIFER-Entry, 79, is the local well identifier (numbers are USGS wells) and S is the frequency at which the water level is measured: A, annually; S, semiannually; Q, quarterly; M, monthly

Figure 2. Location of wells and frequency of water-level measurements (as of December 1995) in the Snake River Plain aquifer, Idaho National Engineering Laboratory [now INL] and vicinity.ⁱⁱ

ii. Bartholomay, R. C., B. J. Tucker, D. J. Ackerman, and M. J. Liszewski, 1997, *Hydrologic Conditions and Distribution of Selected Radiochemical and Chemical Constituents in Water, Snake River Plain Aquifer, Idaho National Engineering Laboratory, Idaho, 1992 through 1995*, DOE/ID-22137, U.S. Geological Survey Water Resources Investigations 97-4086, April 1997.

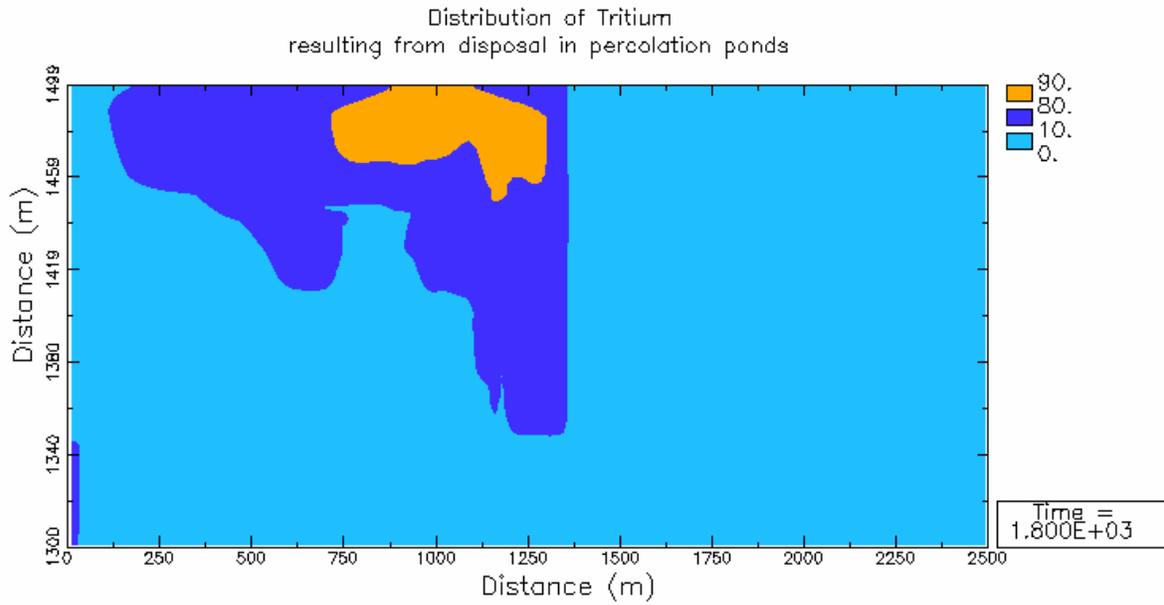


Figure 3-20. Distribution of tritium resulting from discharge into percolation ponds (units in pCi/mL).^{jj}

jj. DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

**NRC Clarifying
Request 17:**

In Portage (2005j), DOE justifies its choices of sorption coefficient values for the compliance or "conservative" case and for the other cases ("worst-case," "realistic," and "best") used in the sensitivity analysis presented in DOE-ID (2003b). The values are based on a combination of literature and site-specific data. As discussed below in more detail for each radioelement, there are three common problems in the Portage (2005j) discussions: (i) values for the "realistic" and "best" cases are identical and the label "realistic" is typically used for the upper bound values, rather than being based on anything demonstrably realistic; (ii) "conservative" values are sometimes arbitrarily chosen; and (iii) "worst-case" values typically should be more accurately termed "conservative." This discussion will focus on the "conservative" case values, because this case is used for comparing with performance objectives. Note that, with the exception of Tc, the grout discussions apply also to concrete, for which DOE uses the same K_d s.

Strontium—For basalt, site-specific laboratory studies yielded K_d s in the range 1.1 to 3.4 mL/g (Del Debbio and Thomas, 1989; Porro, et al., 2000), yet the "conservative" performance assessment value is 6 mL/g (chosen as a midpoint between low and high values). The "worst-case" value of 1 mL/g is more appropriately termed "conservative," because it bounds the range of laboratory data. Two INL-specific studies yielded Sr basalt values of 0.5 and 3.0 mL/g (Arnett, et al., 1990; Robertson, 1974); it would appear unreasonable to adopt a "conservative" K_d higher than these. For interbed sediments, the values overall appear reasonable, although the "conservative" value was arbitrarily chosen within a range (note also that the confirmatory Japan Nuclear Cycle data are for "mudstone," which may not be generally appropriate for INL interbeds). For grout, the Bradbury and Sarott (1995) compendium recommends 1 mL/g as a conservative value, yet DOE uses this value for their "worst-case." The "conservative" value of 3 mL/g was arbitrarily scaled between high and low values. Note that Atkinson and Nickerson (1988), cited in Bradbury and Sarott (1995), recommended 3 to 6 mL/g as a best estimate range; this terminology appears appropriate for a "realistic," rather than "conservative," case.

Technetium—For basalt and interbed sediments, the "conservative" K_d of 0.01 mL/g appears to be appropriate on the basis of the literature, but this choice needs to be better reconciled with the selection of 0 mL/g by Rodriguez, et al. (1997). For grout, Bradbury and Sarott (1995) recommend 1000 mL/g as a conservative value for reducing conditions. Clearly, the "worst-case" would be if Tc were oxidized, in which case Bradbury and Sarott recommend 1 mL/g. Assuming that DOE can justify maintenance of reducing conditions, an appropriate "conservative" value would be 1,000 mL/g, rather than the DOE value of 2,500 mL/g, arbitrarily scaled between 1,000 mL/g and the highest literature value of 5,000 mL/g. Note that the latter value would be more appropriately termed "best" rather than "realistic." The value of 1 m³/kg for technetium K_d used for the worst-case scenario in the groundwater pathway sensitivity analysis (DOE-ID, 2003b, Section 7.3) neglects the diffusion or advection of oxygen into the concrete and grout that could produce locally oxidized conditions where concrete and grout contact the external environment.

These oxidized zones could allow certain radionuclides, particularly Tc-99, to be more mobile and govern radionuclide release rates.

Iodine—The basalt "conservative" K_d of 0.1 mL/g needs to be reconciled with (i) the "site value" of 0 mL/g in Portage (2005, Table 5) and (ii) the lower NEA database value of 0.05 mL/g. For interbed sediments, no basis is provided for the apparently reasonable "conservative" value of 0.1 mL/g. For grout, DOE has chosen, without justification, to use the Bradbury and Sarott (1995) conservative K_d of 2 mL/g as the "worst-case" value. The DOE "conservative" value is arbitrarily scaled between 2 mL/g and the "best" (not "realistic") literature value of 30 mL/g; the Bradbury and Sarott (1995) conservative value would be more appropriate.

Carbon—Basalt K_d s are from Sheppard and Thibault (1990) sand soil values (based on three observations), with the lower end used for the "worst-case" and the mean used for the "conservative" case. The acknowledged uncertainty in these assignments (Portage, 2005, p. 13) would appear to suggest that the low end value be used for the conservative case. The same may be said for the sediment values; the lowest reported value for sand soil (1.7 mL/g) would appear to be an appropriate "conservative" choice. For grout, it would be useful to know if the investigators attempted to calculate a carbon K_d using the approach of Bradbury and Sarott (1995).

DOE should be judicious in choices of "conservative" K_d values. Such values, when used for the compliance demonstration, should be demonstrably bounding at the low end of reasonably expected values. In many cases, DOE has instead used such lower bounds for the "worst-case," with little explicit basis for "conservative" values. This approach does not build confidence in the applicability of the compliance demonstration doses. DOE should consider revising the "conservative" values so that there is greater confidence that their "conservative" values reasonably and defensibly bound the expected range. Likewise, DOE should use only the label "best," and not "realistic," for its high- K_d cases. In only some of the cases is there a basis for terming the values "realistic."

Response:

The initial comment is concerned with the four cases analyzed in the sensitivity/uncertainty analysis, "best," "realistic," "conservative" (i.e., compliance case), and "worst." The sensitivity/uncertainty analysis in Section 7 of the PA (DOE-ID 2003) provides for a matrix of possible scenarios using different parameter combinations. These combinations provide the reader with information to assess the impact of changes in the assumptions for sorption coefficients, infiltration, and radionuclide inventories. The data sets analyzed in Section 7 of the PA were developed in response to a 2002 NRC request for additional information (Essig 2002) as part of a previous review. In that request, the NRC requested "best case, expected behavior, reasonably conservative, and worst case" scenarios to be developed. The INL worked closely with NRC staff to develop the different parameters used in the analysis. In keeping with NRC recommendations, the "conservative case" is not analyzed as a bounding case, but as a "reasonably conservative case." Rather than arbitrarily selecting

parameter values for the conservative case, a consistent methodology was adopted as described below. In the PA, this results in 36 scenarios (Tables 7-10 and 7-11 [DOE-ID 2003]). The “worst-case” analysis would involve the use of parameters being changed to maximize the resulting doses. However, the various scenario results presented in the PA sensitivity/uncertainty analysis (DOE-ID 2003) use only one set of parameter changes (e.g., sorption coefficients). Therefore, the use of the terms “worst case,” “conservative case,” and “realistic/best case” is only applicable in the truest sense to four of the 36 scenario results provided in Tables 7-10 and 7-11 of the PA. The remaining cases presented in these PA tables are combinations of the “worst case,” “conservative case,” and “realistic/best case.” These intermediate cases provide information in which the impact of the sorption coefficients may be assessed.

For example, the results in Tables 7-10 and 7-11 of the PA indicate the use of the “worst-case” sorption coefficients, “conservative-case” scenario infiltration of 4.1 cm/yr, and the “conservative-case” inventory results in doses for drinking water (i.e., 3.42 mrem/yr) and all-pathways exposures (15.0 mrem/yr) that are below the performance objectives. These doses are for the “conservative” PA inventory and do not consider the fact that tank cleaning has significantly reduced the tank inventories. Therefore, consideration of the NRC argument that the lowest measured values should be applied to the “conservative” scenario has been evaluated in the uncertainty analysis, indicating that compliance is still maintained if these selection criteria were applied to the PA analyses.

The sorption coefficients chosen for strontium, technetium, iodine, and carbon are discussed individually in the following sections of this response.

Strontium

Site-specific sorption values for strontium are provided in Table CR-17-1.

Table CR-17-1. Summary of strontium sorption coefficient measurements at the INL.

Element	K _d Values (mL/g)			
	Alluvium	Interbed Sediment	Basalt	Reference
Sr	35–52	110–186	1.1–2.7	Del Debbio and Thomas (1989)
	23–26			Schmalz (1972)
	8.3–16.6			Hawkins and Short (1965)
	61–134			Liszewki et al. (1997)
		42–63	6–13	Newman (1996)
			3.6–29.4	Colello et al. (1998)
			>47	8.0–8.4 Fjeld et al. (2001)

The basalt value chosen in the PA of 6 mL/g for the conservative case was based on the ranges provided by Del Debbio and Thomas (1989) and Newman (1996). The conservative value is also less than the sorption coefficients measured by Fjeld et al. (2001). The use of the 1-mL/g value, which is termed “worst case” in the PA, is suggested by NRC to be better suited as a “conservative” value since it bounds the range of laboratory measurements. As discussed at the beginning of this response, it is not deemed appropriate to use the lowest measured sorption values for purposes of a compliance analysis. If the lowest measured values were applied to the compliance case, then the question arises as to what values would be applied to the worst case. The study by Colello et al. (1998) has provided sorption coefficients that range up to 29.4 mL/g. Considering the basalt sorption coefficients range from 1.1 to 29.4 mL/g, the selection of 6 mL/g for the compliance case is appropriate and toward the lower end of the measured ranges.

The strontium interbed sediment sorption coefficient of 18 mL/g for the PA conservative scenario (i.e., compliance case) is notably conservative, considering the range of measured values shown in Table RAI-17-1, which are all greater than the values chosen in the PA. The strontium interbed sediment sorption coefficient ranges were chosen to be lower than the measured site values based on the range used in Rodriguez et al. (1997) (i.e., 12–24 mL/g) and the lower values reported for soils by Sheppard and Thibault (1990) (i.e., 15–20 mL/g). Since ⁹⁰Sr has a short half-life, the impact of the interbed sediment K_d values is large due to decay during transport. Therefore, lower sorption coefficients were assigned to the interbed sediments for conservatism.

The strontium grout sorption coefficients selected ranged from 1 to 6 mL/g, as noted by the NRC comment. The PA “conservative” scenario used the midpoint value of 3 mL/g. The NRC suggests that the lowest sorption value of 1 mL/g should be applied to the “conservative” scenario. As discussed at the beginning of this response, it is not deemed appropriate to use the lowest measured sorption values for the purpose of a compliance analysis. The lowest measured values were applied to the worst case. The sensitivity/uncertainty analysis in Section 7 of the PA indicates that the selection of all the lowest reported strontium sorption values for each material of interest (i.e., grout [1 mL/g], interbed sediment [12 mL/g], and basalt [1 mL/g]) would increase the drinking water dose from 0.001 to 3.02 mrem/yr for the conservative case. The all-pathways dose would increase from 0.006 to 15.0 mrem/yr. These increased doses, based on the lowest reported technetium sorption values for all materials, would not exceed the performance objectives. The changes in sorption coefficients for strontium result in larger changes in dose compared to the other radionuclides of interest (i.e., iodine, technetium, and carbon) due to the short half-life of ⁹⁰Sr.

Technetium

The NRC states “For basalt and interbed sediments, the ‘conservative’ K_d of 0.01 mL/g appears to be appropriate on the basis of the literature, but this choice needs to be better reconciled with the selection of 0 mL/g by Rodriguez et al. (1997).” Rodriguez et al. (1997) only used 0 mL/g for the vadose zone basalts and not for the interbed sediments. A detailed discussion on the sorption

coefficients for the vadose zone basalts and the Rodriguez et al. (1997) use of zero sorption coefficients for vadose zone basalts is provided in the response to NRC Comment 14.

The technetium grout sorption coefficients were varied between 1,000 and 5,000 mL/g. The “conservative” scenario involved the use of a 2,500-mL/g sorption coefficient. As noted in PEI-EDF-1023 (2005), Bayliss et al. (1991) reported sorption coefficients in cement of 5,000 mL/g for technetium. The NRC states “Assuming that DOE can justify maintenance of reducing conditions, an appropriate ‘conservative’ value would be 1,000 mL/g, rather than the DOE value of 2,500 mL/g, arbitrarily scaled between 1,000 mL/g and the highest literature value of 5,000 mL/g.” The sensitivity/uncertainty analysis in Section 7 of the PA indicates that the selection of all the lowest reported technetium sorption values for each material of interest (i.e., grout [1,000 mL/g], interbed sediment [0 mL/g], and basalt [0 mL/g]) would increase the drinking water dose from 0.12 to 0.35 mrem/yr for the conservative case. The all-pathways dose would increase from 0.87 to 2.46 mrem/yr. These increased doses, based on the lowest report technetium sorption values for all materials, would not exceed the performance objectives.

The NRC states:

The value of 1 m³/kg for technetium K_d used for the worst-case scenario in the groundwater pathway sensitivity analysis (DOE-ID, 2003b, Section 7.3) neglects the diffusion or advection of oxygen into the concrete and grout that could produce locally oxidized conditions where concrete and grout contact the external environment. These oxidized zones could allow certain radionuclides, particularly Tc-99, to be more mobile and govern radionuclide release rates.

The release modeling for ⁹⁹Tc in the PA was conducted using reducing sorption coefficients for the grouted tank; however, an oxidizing sorption coefficient of 0.001 m³/kg was used for the concrete vault that contacts the external environment. The errata sheet to the PA (DOE-ID 2003) corrects the statement in the PA on page 2-55 (Section 2.1.6.2), “Concrete K_d values for reducing conditions have been chosen...”

Iodine

The basalt sorption coefficient value of 0 mL/g used for the worst case referenced in Table 5 of PEI-EDF-1023 (2005) is based on the Rodriguez et al. (1997) use of zero sorption coefficients for vadose zone basalts in their modeling work and is not a site-specific measured value. The conservative value of 0.1 is higher than the 0.05-mL/g value reported in the Nuclear Energy Agency (NEA) database (Ticknor and Ruegger 1989); however, consideration of the iron oxides in basalts at the INL Site was included when applying the 0.1-mL/g sorption coefficient, as discussed in the response to NRC Comment 14. A detailed discussion on the sorption coefficients for the vadose zone basalts and the

Rodriquez et al. (1997) use of zero sorption coefficients for vadose zone basalts is provided in the response to NRC Comment 14.

The iodine sorption coefficients in interbed sediments range from a high value of 5 mL/g for loam soil to a low value of 0.01 mL/g as reported in Rodriquez et al. (1997). These are literature values and not site-specific measurements. Therefore, the value of 0.01 mL/g from Rodriquez et al. (1997) was used for the worst case and 0.1 mL/g was used for the conservative case, based on the NEA database values.

Bradbury and Sarott (1995) selected a conservative value (i.e., lower value) of 2 mL/g for iodine in reducing concrete systems. Allard et al. (1985) recommended a sorption value of 30 mL/g. The PA conservative value was 8 mL/g, the worst case was assigned a value of 2 mL/g, while the best/realistic case was assigned a value of 30 mL/g. The NRC states, "The DOE 'conservative' value is arbitrarily scaled between 2 mL/g and the 'best' (not 'realistic') literature value of 30 mL/g; the Bradbury and Sarott (1995) conservative value would be more appropriate." The sensitivity/uncertainty analysis in Section 7 of the PA indicates the selection of all the lowest reported iodine sorption values for each material of interest (i.e., grout, interbed sediment, and basalt) would increase the drinking water dose from 0.77 to 3.42 mrem/yr for the conservative case. The all-pathways dose would increase from 1.35 to 5.97 mrem/yr. These increased doses, based on the lowest reported technetium sorption values for all materials, would not exceed the performance objectives.

Carbon

The NRC suggested that the lower end of the reported carbon sorption coefficients be used for the conservative case instead of the worst case. As discussed at the beginning of this response, it is not deemed appropriate to use the lowest measured sorption values for the purpose of a compliance analysis. The sensitivity/uncertainty analysis in Section 7 of the PA indicates that the selection of all the lowest reported carbon sorption values for each material of interest (i.e., grout, interbed sediment, and basalt) would increase the drinking water dose from 6.00E-12 to 5.00E-12 mrem/yr for the conservative case. The all-pathways dose would increase from 2.88E-11 to 2.48E-04 mrem/yr. These increased doses, based on the lowest report technetium sorption values for all materials, would not exceed the performance objectives. There was no attempt to calculate a carbon Kd for grout using the approach of Bradbury and Sarott (1995).

The NRC has commented on the selection of sorption coefficient in the PA for the worst and conservative cases, specifically noting that INL should use the lower bounds as the compliance case. As discussed at the beginning of this response, it is not deemed appropriate to use the lowest sorption values for purposes of a compliance analysis, but these values should be applied to the worst-case analysis. The additional cases presented in Section 7 of the PA are combinations of the "worst case," "conservative case," and "realistic/best case." These intermediate cases provide information in which the impact of the sorption

coefficients may be assessed. For example, the results in Tables 7-10 and 7-11 of the PA indicate the use of the “worst-case” sorption coefficients, “conservative-case” scenario infiltration of 4.1 cm/yr, and the “conservative-case” inventory results in doses for drinking water (i.e., 3.42 mrem/yr) and all-pathways exposures (15.0 mrem/yr) that are below the performance objectives. These doses use the “conservative” PA inventory and do not consider the fact that tank cleaning has significantly reduced the tank inventories. Therefore, consideration of the NRC suggestion that the lowest-measured sorption coefficient values should be applied to the “conservative” scenario has been evaluated in the uncertainty analysis. This analysis indicates that compliance is still maintained if the worst-case sorption coefficient values were applied to the conservative case in the PA analyses.

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Newman, M. E., 1996, *Evaluation of the Mobility of Am, Cs, Co, Pu, Sr, and U Through INEL Basalt and Interbed Materials: Summary Report of the INEL/Clemson University Laboratory Studies*, WAG7-82, INEL-95/282, 1996.

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Rodriguez, R. R., A. L. Shafer, J. McCarthy, P. Martian, D. E. Burns, D. E. Raunig, N. A. Burch, and R. L. VanHorn, 1997, *Comprehensive RI/FS for the Idaho Chemical Processing Plant OU 3-13 at the INEEL – Part A, RI/BRA Report (Final)*, DOE/ID-10534, Binders 1–3, November 1997.

Schmalz, B. L., 1972, *Radionuclide Distribution in Soil Mantel of the Lithosphere as a Consequence of Waste Disposal at the National Reactor Testing Station*, IDO-10049, U.S. Atomic Energy Commission, 1972.

Sheppard, M. I., and D. H. Thibault, 1990, "Default Soil Solid/Liquid Partition Coefficients, K_{ds} , for Four Major Soil Types: A Compendium," *Health Physics*, Vol. 59, pp. 471–482.

Ticknor, K. V., and B. Ruegger, 1989, *A Guide to the NEA's Sorption Data Base*, Version 2.0, September 1989.

**NRC Clarifying
Request 18:**

List key radionuclides for worker dose during closure and clarify if short-lived radionuclides were screened out during the analysis to identify these radionuclides, i.e., confirm whether short-lived radionuclides would contribute to worker dose during closure activities.

Response:

Worker doses are discussed in Subsection 7.12 of the draft 3116 Determination, as provided below:

The closure activities at the TFF are maintained under the DOE dose limits for the worker, public, and environment. The TFF PA does not address the operational (i.e., closure and disposal activities) of the facility, only the post-closure (i.e., post-disposal) aspects of the facility. However, during Tank WM-182 cleaning operations, the total radiation exposure to the 23 workers was about 650 mrem for an average exposure of about 30 mrem per individual. This information is based upon a review of dosimetry results from TFF radiation work permits for the January 2002 to June 2005 timeframe (Martin 2005)...

As noted in the above discussion, worker dose for tank cleaning is minimal because all cleaning is accomplished remotely. Worker exposures would be limited to removal of existing equipment, installation of cleaning equipment, and sampling and analysis activities. The radionuclides of interest for worker doses would be high-energy gamma emitters. The TFF PA (DOE-ID 2003) does not address the operational (i.e., closure and disposal activities) of the facility, only the post-closure (i.e., post-disposal) aspects of the facility. Therefore, the radionuclide screening presented in the PA is for the analysis of post-closure doses to members of the public and inadvertent intruders, not for worker dose evaluations. In summary, short-lived, gamma-emitting radionuclides (the key radionuclide being ¹³⁷Cs) would contribute to the worker dose during closure activities. As such, all work is planned and performed under the INL Radiation Protection Program to ensure doses to workers are low as reasonably achievable and within applicable limits.

References:

DOE-ID, 2003, *Performance Assessment for the Tank Farm Facility at the Idaho National Engineering and Environmental Laboratory*, DOE/ID-10966, Rev. 1, April 2003 (Errata December 2, 2003).

**NRC Clarifying
Request 19:**

Table 3 (page 39) presents the sand pad residual inventory at closure in Ci per sandpad. The value for Cesium (Cs)-137 is presented as 2.53E-06 Ci. Based on a number of other references to this inventory, including Table 14 (page 68) of this document, and the activity for Ba137m, the Cs-137 inventory in the sand pad is expected to be on the order of 1.6E+03 Ci. The value for Pu-238 in Table 3 (page 39) of the waste determination (DOE-ID, 2005) is 5.06E-06 Ci, while the value presented in the PA(DOE-ID, 2003b) is 2.0 Ci (see Table 2-17 on page 2-73). Additional radionuclide inventory estimates are discrepant between the waste determination and PA tables. Clarify the correct value for the sand pad inventory and confirm which values were used in the performance assessment.

Response:

The response to NRC Clarifying Request 19 will be provided at a later date.

**NRC Clarifying
Request 20:**

There are several inconsistent and redundant statement in the performance assessment (DOE-ID 2003b).

On page 2-68, it states that the center of the sand pad is 2 inches in one sentence and 3-4 inches in the next sentence.

On page 2-70, it states that in the "*absence of data*", 38 flushing events were assumed, while on page 2-68 it implies that records exist that suggest over 100 flushing events have occurred.

On page 4-5, it states that source release for two tanks is loaded at the location of the southern tank, while on page 3-37 it states that releases from one tank were simulated and the concentrations doubled at downgradient locations.

Indicate which value or statement is correct.

Response:

On page 2-68, it states that the center of the sand pad is 2 inches in one sentence and 3-4 inches in the next sentence.

The first statement on page 2-68 is correct. The correct statement is "Because the vault floor is conical and the tank bottom is flat, the thickness of the sandpad ranges from about 2 in. (5 cm) at the tank center to about 6 in. (15 cm) at the curb."

On page 2-70, it states that in the "absence of data", 38 flushing events were assumed, while on page 2-68 it implies that records exist that suggest over 100 flushing events have occurred.

The statement on page 2-70 is partially correct. A total of 38 flushings were assumed in the sandpad analysis for conservatism. The statement on page 2-70, "In the absence of data," is incorrect since flushing data are available as indicated on page 2-68. That statement on page 2-70 should read "For conservatism, it is assumed that the flushing events occurred once a year for 38 years." The statement on page 2-68 indicating that over 100 flushing events have occurred is correct.

On page 4-5, it states that source release for two tanks is loaded at the location of the southern tank, while on page 3-37 it states that releases from one tank were simulated and the concentrations doubled at downgradient locations.

Both statements are correct but the statement on page 3-37 is stated more clearly. One tank was modeled in the analysis and the resulting receptor concentrations were doubled.

**NRC Clarifying
Request 21:**

Please provide the following references:

Ackerman, D.J., 1991, "Transmissivity of the Snake River Plain Aquifer at the Idaho National Engineering Laboratory Site," Idaho, USGS Water-Resources Investigations Report 91-4058, DOE/ID-22097.

Anderson, S.R., M.A. Kuntz, and L.C. Davis, 1999, "Geologic Controls of Hydraulic Conductivity in the Snake River Plain Aquifer at and near the Idaho National Engineering and Environmental Laboratory, Idaho," U.S. Geological Survey Water-Resources Investigations Report 99-4033, DOE/ID-22155.

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Del Debbio, J. A. and T. R. Thomas. "Transport Properties of Radionuclides and Hazardous Chemical Species in Soils at the Idaho Chemical Processing Plant."

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Dunnivant, F. M., et al., 1998, "Water and Radioactive Tracer Flow in a Heterogeneous Field-Scale System," Ground Water, Vol. 36, No. 6, pp. 949-958.

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Orr, B.R., and L.D. Cecil, 1991, "Hydrologic Conditions and Distribution of Selected Chemical Constituents in Water, Snake River Plain Aquifer, INEL, Idaho, 1986-1988," USGS Water-Resources Investigation Report 89-4008, DOE/ID-22078.

Robertson, J.B. "Digital Modeling of Radioactive and Chemical Waste Transport in the Snake River Plain Aquifer at the National Reactor Testing Station, Idaho." U.S. Geological Survey Open-file Report IDO-22054. 1974.

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Walker, E. H., 1960, "Analysis of Aquifer Tests, January 1958-June 1959, at the National Reactor Testing Station, Idaho," Atomic Energy Commission.

Response: The requested references were provided via a reference disk mailed to NRC on January 17, 2006.