C O L U M B I A R I V E R K E E P E R
Working to Enhance Water Quality

HANFORD 100-BC REACTOR AREA CLEANUP

Will DOE Soil Cleanup Protect Groundwater?

Will DOE Restore Groundwater to Its Highest Beneficial Use?

John R. Brodeur, P.E., L.E.G.
Energy Sciences & Engineering
Kennewick, Wash.

Greg deBruler
Columbia Riverkeeper
Bingen, Wash.

September 2005

Funded by the Citizens Monitoring and Technical Assessment Fund

Introduction

This review was conducted as a part of a review of the cleanup work at several DOE facilities, including Hanford, Savannah River, Oak Ridge and Idaho National Engineering Laboratory.

The original intent of this multiple-site review was to assess whether the DOE's current cleanup strategy will protect groundwater, and whether the DOE will restore the groundwater to its highest beneficial use (in most cases, the Drinking Water Standard [DWS]). We sought to review and describe the groundwater contamination issues at these sites; assess the cleanup approach for the contamination; and critically examine, evaluate and explain the effectiveness of the cleanup, including a description of the contamination that is left behind and its potential impact or risk. Our intent was to provide an assessment of the effectiveness of DOE's overall approach and compare it to the stated cleanup goals or remedial-action objectives that inherently consider the projected future land-use and end-states of the sites.

During the review, we determined that to do an acceptably comprehensive job of reviewing the complex and multiple groundwater contamination plumes at each of the sites, we would require much more time and significantly greater effort than was budgeted. So, we changed the focus and scope of our study in order to perform a more comprehensive review of a few key facilities at each of the DOE sites.

For the Hanford Site, the focus of our study is the cleanup work conducted along the Columbia River, including the 100 Areas and the 300 Area. For a more intensive investigation, we chose to review the cleanup of the 100-BC Area because it was the first remedial action area of the 100 Areas. It basically set the pattern for remediation of the rest of the 100 Areas.

The following is a review of the principal documentation in the Administrative Record describing the cleanup work of the 100-BC Area at Hanford. This documentation is reviewed with a critical eye on determining just what was cleaned up and what was left behind.

This review is not conducted with any specific review criteria. Rather, we simply look at what is stated in the DOE documentation as the intent of the cleanup action and then assess the cleanup action against the stated goals to see if it can be documented that they have satisfied the goals.

Review of 100 BC Area Cleanup

The 100 BC Area, adjacent to the Columbia River, is the furthest upstream and the furthest to the west of all 100 Area reactor sites. This area contains the first plutonium production reactor at Hanford, the B Reactor, as well as the C reactor; hence, the "BC Area."

The cleanup of the BC Area, and all of the 100 Areas at Hanford, is managed under the same characterization and remediation strategy in which "emphasis is placed on initiating and completing waste site cleanup through interim actions" (DOE 1990), including "accelerating decision-making by maximizing the use of existing data consistent with data quality objectives." DOE has stated that "an important element of this strategy is the application of the observational approach, in which characterization data are collected concurrently with cleanup," and the use of limited field investigations on high priority sites. "Interim remedial measures are intended to achieve remedies that are likely to lead to a final Record of Decision (ROD)" (DOE/RL 1994a).

We interpret these statements to mean that the DOE intends the interim measures to be the final remedial measure, at least for the soil-contamination sites.

The 100 BC Area is divided up into three operable units (OUs): the 100-BC-1 OU encompasses the north portion, including the B reactor and all surface buildings, structures, pipelines and soil-contamination sites; the 100-BC-2 encompasses the southern portion, including the C reactor and its nearby surface facilities; the groundwater beneath the 100 BC Area is covered by a separate OU, designated the 100-BC-5 OU.

Cleanup of the surface operable units (100-BC-1 and -2) was primarily handled as an Interim Remedial Measure (IRM). The remedial investigation/feasibility study work plan of 1990 provides the strategy and rationale for the use of limited field investigations (LFI) of high-priority waste sites to allow application of the IRM approach. (DOE/RL, 1990). The LFI reports for 100-BC-1 & 2 OUs are DOE/RL-93-06 and DOE/RL-94-42.

There are three primary sources of characterization data for the waste sites within the BC Area Operable Units:
Previous characterization of select waste sites covered in Dorian and Richards (1978)
Limited field investigations (LFI) covering select waste sites (DOE/RL (1994a, b and c)
Characterization that occurred during remediation reported in cleanup verification packages
Cleanup verification packages (CVP) do not appear to be in the Administrative Record but some were obtained elsewhere, including those for the 116-B-1 and 116-C-1 trenches (Bechtel 1999a and b).

The Dorian and Richards report of 1978 (henceforth referred to as "D&R") was a major resource used in development of conceptual models and in the development of the LFI data needs. This was a rather extensive initial assessment of the radioactivity of the retired facilities in the 100 Areas. This study included a review and summary of facility process records and some of the most extensive sampling of the retention basins, burial grounds, cribs, reactors, fuel storage basins, ancillary facilities and the 118-B-1 burial ground. This study includes inventory estimates

of radionuclide quantities in several of the basin sludges and facilities, as well as some of the surrounding soils. Inventory estimates are based on gamma and beta-gamma activity surveys and limited sample assays. These estimates were not reviewed or assessed in our review.

The soil sampling reported in D&R occurred at select facilities to an average depth of approximately 25 feet. Soil samples were typically taken at 5-foot spacings and analyzed for a select group of radionuclides. Spatial data is shown at several facilities where they were able to place up to three boreholes along a trench facility and provide some assessment of the contamination distribution along the axis of the trenches. The D&R report also includes an assessment of groundwater contamination data for the BC area.

Relative to soil contamination, D&R conclude that "soil contamination usually drops off to essentially background levels within five to 20 feet below the crib, trench or retention basin bottoms." D&R also states that "the majority of radioactivity in the 107 liquid waste trenches is in the immediate vicinity of the trenches and is within 10 feet of the trench bottoms." These general conclusions are an important part of the conceptual model developed for the waste sites in the 100 Areas and used to make the interim ROD.

Not found in D&R is an assessment of the nature and extent of contamination in the region of the deeper vadose zone, from approximately 25 feet to the top of the water table (43 feet). There is also no development of an understanding of the spatial distribution of contamination causing us to question the general conclusions quoted above. Radionuclide analysis in D&R was also limited to a select list of radionuclides. Some of the contaminants of potential concern in the interim RI/FS work are not included in the D&R study, such as Tc-99. This was likely due to analytical capabilities and budget constraints at the time D&R was prepared (in the mid-1970s).

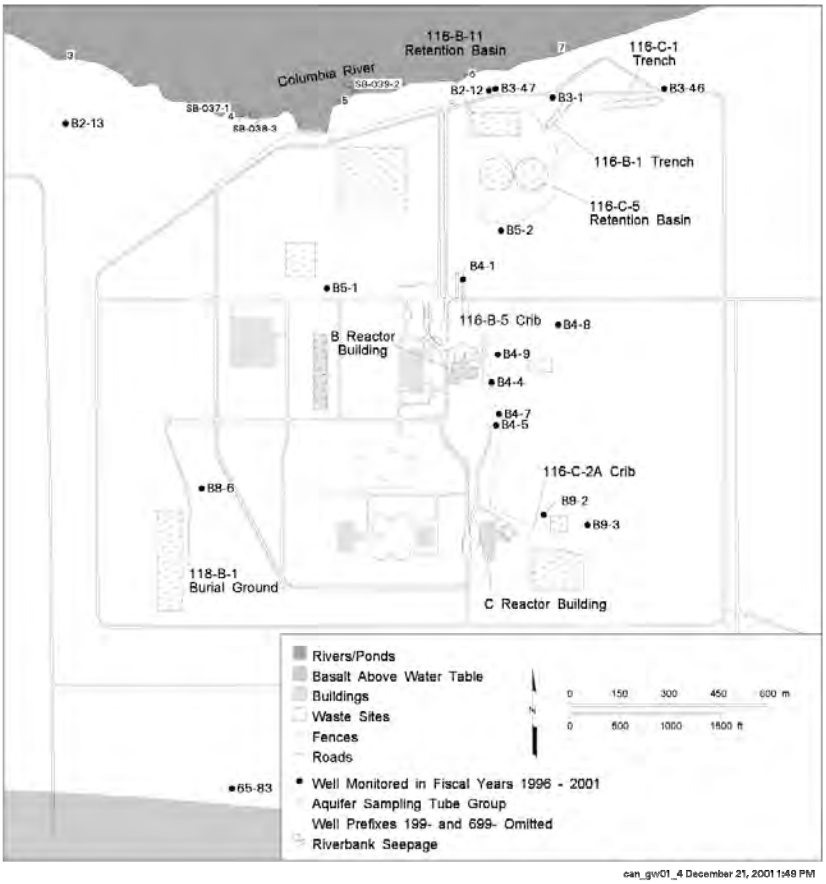
A secondary source of information used to make the IRM ROD is the analogous waste-site studies. In this report, we will look at the principal data that is used to create the analogous site model for the 100-BC Area waste sites, and only apply the data at the facility where it is obtained. Inferences made in the site(s) conceptual model about analogous waste-site relationships will be handled in the discussion, perhaps with the interjection of additional knowledge and information gained from other analogous areas at Hanford.

The LFIs preceded the plan for interim remedial measures (DOE/RL 1994d) and the plan for an excavation demonstration project (DOE/RL 1995a & b). LFI work at 100-BC-1 OU involved intrusive sampling methods including test pits, surface soil sampling and subsurface drilling/sampling, and spectral gamma geophysics in select boreholes. Non-intrusive surface geophysical methods were also used, primarily to identify subsurface structures and features. The surface geophysical methods were not used to assess the nature and extent of contamination, which is the focus of this review.

Intrusive borehole sampling or test pits occurred at five structures within 100-BC-1 OU, including 116-B-1, 116-B-2, 116-B-3, 116-B-5 and 116-C-5. All other waste sites used non-intrusive investigations including process knowledge, D&R and analogous waste site information to assemble information for the interim ROD.

Our review looks at the cleanup of two disposal trenches within the 100-BC-1 OU including 116-B-1 and 116-C-1. Information on these facilities and the associated groundwater is obtained from WHC (1994), from D&R (1978) and from the three LFI reports of the DOE/RL (1994a, b and c). 116-B-1 was remediated as an IRM, while trench 116-C-1 was remediated as an ERA, which was called an Excavation Demonstration Project.

The B reactor operated from 1944 to 1968; the C reactor operated from 1952 to 1969. Both used single-pass cooling systems. Cooling water was drawn from the Columbia River, treated to remove sediment; conditioned with, among other things, sodium dichromate; and sent into the reactor. After passing through the reactor, cooling effluent was sent to retention basins 116-B-11 and 116-C-5 (Figure 1), where it was retained for a short time to allow cooling and short-lived radionuclides to decay (typically four hours), and then pumped directly to the Columbia River via outfall structures. The 116-B-11 and 116-C-5 retention basins (Figures 1 and 2) received enormous quantities of cooling effluent, which contained radionuclides and metals. Leaks that occurred in these concrete basins caused both groundwater and extensive vadose-zone contamination. D&R report that leaks from basin structures and pipelines caused an increase in the temperature of groundwater in the area and created thermal seeps along the river.



(Fig.1)

The 116-B-1 and 116-C-1 trenches (Figure 1) were used as liquid-effluent disposal sites whenever there was a fuel element failure in the reactor. When it was apparent that a fuel cladding

failure had occurred, effluent in the retention basins was sent to these trenches for disposal instead of releasing it to the Columbia River. The intent of disposal in the trenches was to allow the effluent to infiltrate into the permeable soils and allow the radionuclide contamination to absorb into the sediment, as well as provide time for radioactive decay of the short-lived radionuclides. This created considerably more contamination of the vadose-zone sediments at these trench sites than most other sites within the BC Area. Because they are presumably the most contaminated sites, they are the focus of our review.

Trench 116-B-1

Trench 116-B-1 is approximately 200 feet long, 30 feet wide and 15 feet deep and located 400 feet east of the 116-B-11 retention basin (Figure 1). It was used from 1948 until 1955 whenever fuel-cladding failures occurred. After 1955, degradation of the concrete in the B-11 retention basin no longer allowed separation of effluent batches and cladding-failure effluent was sent to the Columbia River instead of to the trench. The B-1 trench received approximately 16M gal of cooling effluent that contained fission products, activation products, transuranics and sodium dichromate (2mg/L).

D&R drilled four boreholes along the axis of the trench, the exact location of which could not be determined from the document copy in the administrative record; we assume they were drilled along the trench axis. The maximum depth of the boreholes was 20 feet, or approximately five feet below the trench bottom. Eight soil samples were taken from depths of five to 20 feet and analyzed for the standard suite of the D&R study (Table 1).

116-B-1

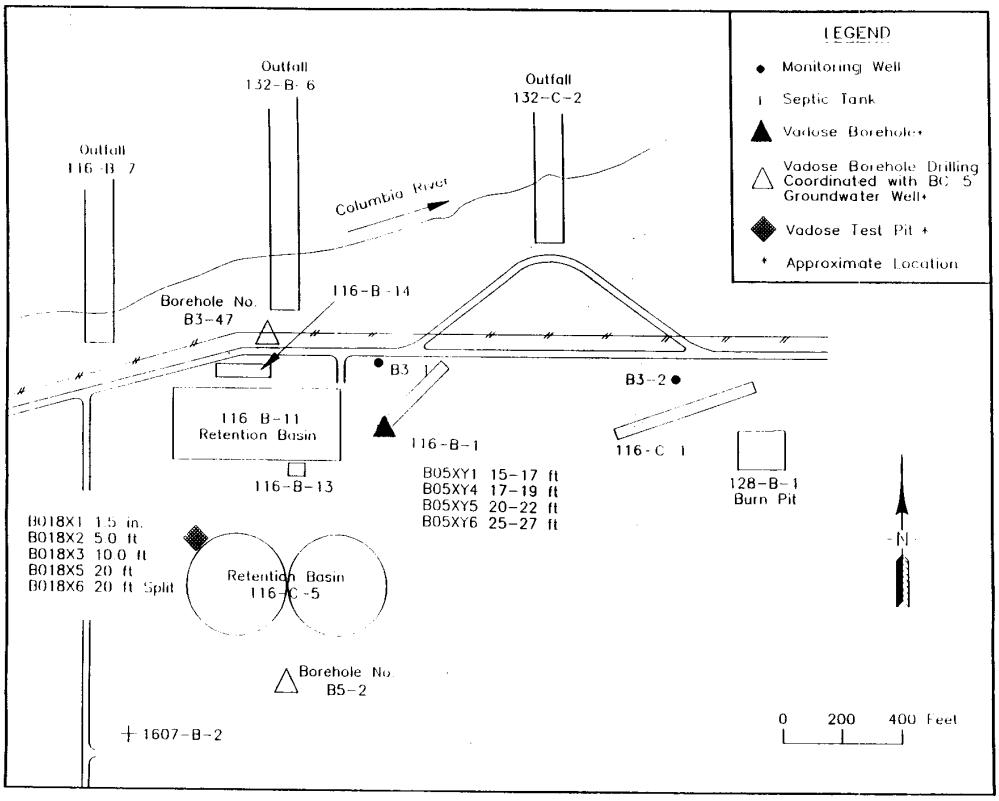
107-B LIQUID WASTE DISPOSAL TRENCH

	Concentration (pCi/g)											
Sample No.	Pu-238	Pu-239/240	Sr-90	H-3	P-11/Scaler c/m	Eu-152	Co-60	Eu-154	Cs-134	Cs-137	Eu-155	U
B 5	*	*	1.3×10^{-2}		<200/30	1.0×10^{0}	2.2×10^{-1}	*	6.8×10^{-2}	1.2×10^{-1}	1.7×10^{-1}	
B 15	*	*	3.8×10^{-2}		<200/80	1.5×10^{0}	2.8×10^{-1}	5.9×10^{-1}	*	2.6×10^{-1}	6.0×10^{-2}	
B 17	1.7×10^{-2}	9.9×10^{-1}	3.3×10^{0}	1.1×10^{0}	400	2.2×10^{2}	3.9×10^{1}	4.8×10^{1}	7.2×10^{-1}	3.6×10^{1}	1.2×10^{1}	2.8×10^{-1}
B 20	*	1.1×10^{-1}	6.2×10^{0}		300	2.5×10^{1}	1.3×10^{0}	3.4×10^{0}	2.2×10^{-1}	1.1×10^{1}	5.0×10^{-1}	2.5×10^{-1}
C 10	*	*	7.0×10^{-2}		<200/20	7.8×10^{-1}	1.1×10^{-1}	*	*	*	1.2×10^{-1}	
C 20	*	2.2×10^{-1}	4.6×10^{0}	2.7×10^{0}	<200/60	4.6×10^{1}	7.8×10^{0}	1.0×10^{1}	*	1.5×10^{1}	*	
D 20	*	*	1.9×10^{-1}		<200/20	*	5.7×10^{-2}	*	*	*	*	
E 20	*	*	9.1×10^{-2}		<200/20	3.0×10^{-1}	*	*	3.8×10^{-2}	*	2.3×10^{-1}	

The best spatial data was obtained from Borehole B, with four samples taken from depths of five5, 15, 17 and 20 feet. The highest radionuclide concentrations are found in the 17-foot depth samples, which is two feet below the trench bottom. A notable exception to this is the Sr-90 distribution data, which shows a high of 6.2 pCi/g at the bottom of the borehole (20 feet), with lower concentrations at shallow depths. For Sr-90, it is apparent from the data that the highest concentration may reside below the maximum depth of sampling (20 feet).

The general conclusion of D&R of decreasing concentration with depth is not supported by this sampling due to the limited depth extent of the boreholes. The maximum sampling depth of 20 feet is only five feet below the bottom of the trench, making it difficult to assess distribution trends beneath the trench.

The relative concentrations of Sr-90, Eu, Cs and uranium in Table 1 indicate the maximum depth of contamination may not have been reached with the total depth of these boreholes (20 feet). No other assessment about the horizontal or vertical extent of contamination can be made from the D&R data.



(Figure 2)

One borehole was drilled through the 100-BC-1 trench as a part of the LFI for the 100-BC-1 OU (Figure 2).. Four samples were taken from depths of 16, 18, 21 and 26 feet. Radionuclide and metals assay data is summarized in Tables 2 and 3. Data shows that from the trench bottom, radionuclide concentrations decrease with depth, to the maximum sampling depth of 26 feet. Unlike D&R, Sr-90 is not an exception to this general trend for the depth range of the data.

Analytes	Sample and Sample Interval (ft bls)			
	B05XY1 15 - 17	B05XY4 17 - 19	B05XY5 20 - 22	B05XY6 25 - 27
Gross Alpha	0*	8.89R	5.18R	1.9R
Gross Beta	201	76.7R	54.3	N/D
Carbon-14	3.77J	6.18J	3.76J	1.89J
Cobalt-60	4.167	1.589J	0.389	N/D
Strontium-90	13.2	6.38	5.08	1.54
Cesium-137	43.85	22.99J	10.36	1.394
Europium-152	121.9	59.15J	17.56	4.114
Europium-154	9.9	4.749J	1.195	N/D
Plutonium-238	0.108R	0.088R	N/D	N/D
Plutonium-239	3.6R	0.92R	0.269	N/D
Americium-241	0.482R	0.13R	0.05	0.002

* = Interpreted as 0, analysis reported negative concentrations
R = Value marked as rejected in validation because of missing calibration data
J = Value estimated, due to quality control deficiencies
N/D = Constituent not detected, data package includes detection limit

(Table 2)

Analytes	Sample and Sample Interval (ft bls)		Hanford Site Background 95% UTL
	B05XY1, 15 - 17	B05XY4, 17 - 19	
Chromium	33	A	27.9
Manganese	A	839	612
Zinc	128	A	79

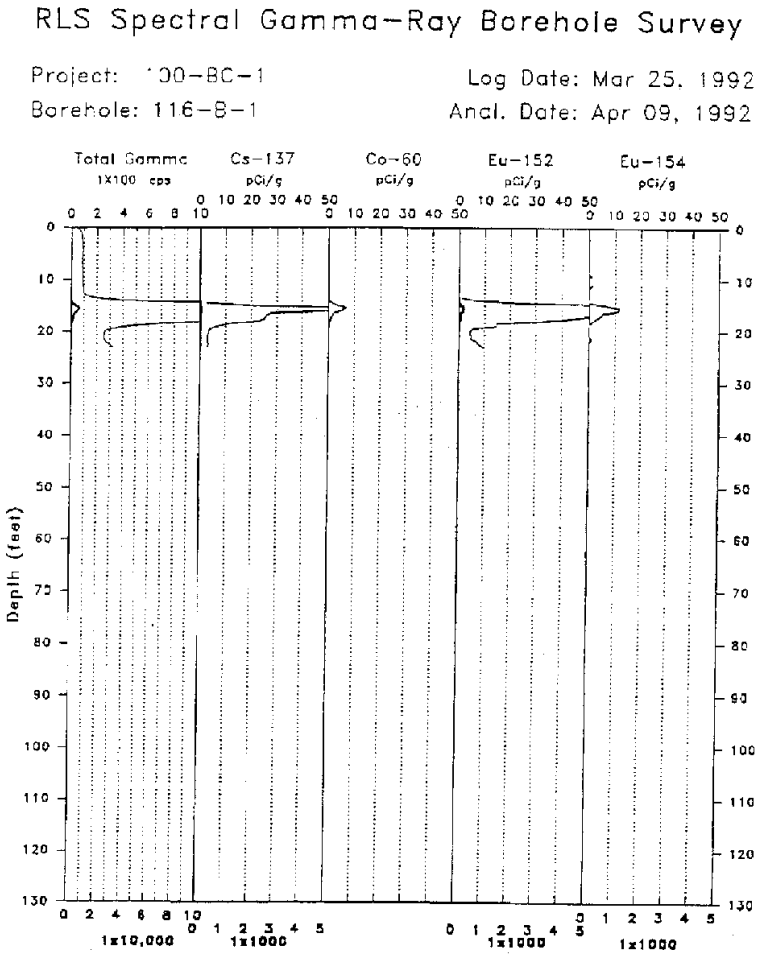
A = Concentration less than Hanford Site background 95% upper threshold limit (UTL)

(Table 3)

Tc-99 was assayed but not detected in any of the four soil samples from this borehole. Chromium was detected above background in only one sample.

Spectral gamma-ray logging of the LFI borehole (Figure 3) shows the highest Cs, Co and Eu concentrations near the 15 feet depth, which is the bottom of the trench structure. Spectral gamma logs also show the depth extent of cesium and europium has not been reached at 23 feet, the maximum depth of gamma log data.

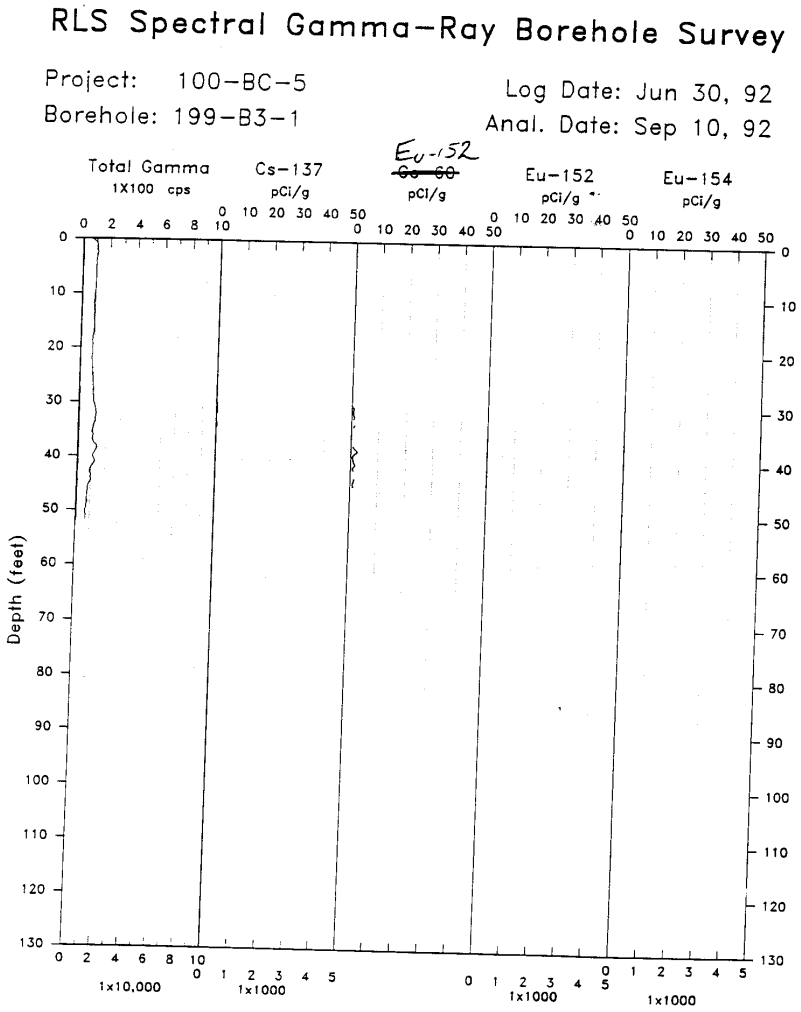
(Figure 3)



Groundwater beneath trench 100-BC-1 is assessed with one well (299-B3-1), which was drilled in 1953 as part of an original groundwater monitoring network in the BC Area. That well is located approximately 150 feet northwest of the trench (Figure 2).

An exceptionally poor plot of some of the spectral gamma ray log data from this groundwater well (Figure 4) shows a region between approximately 30 and 50 feet at which low concentrations of Cs-137 and Eu-152 are found. The concentration representation on the plot is not visible, but it is there. The cesium in particular, is a low-mobility radionuclide and poses a relatively low risk to human health and safety. This low-concentration cesium is interpreted to be the result of unsaturated moisture flow and the migration of contamination from the 116-B-1 trench to the location of this well, approximately 150 feet away. Alternately, it is the same from the C-11 retention basin. Qualified plots of the potassium, uranium and thorium concentrations in this borehole

might be useful for correlating lithology with contamination distribution, but they are not provided.



(Figure 4)

There is no soil-sample data from this older groundwater well, and no assessment of non-gamma-emitting radionuclides such as Sr-90 and Tc-99 in the vadose zone sediment.

Groundwater sample assays from this well indicate Sr-90 and Tritium levels that exceed the maximum contaminant level (MCL) for drinking water or the equivalent drinking water standard. In 1993, the Sr-90 concentration was 44 pCi/L (MCL = eight pCi/L) and the tritium level was 3,800 pCi/L, below the 20,000 pCi/L MCL. The 1993 data also show an elevated Tc-99 concentration of 90 to 95 pCi/L, which is approximately 1/10 of the MCL of 900 pCi/L. The 1995 groundwater data also shows a chromium level of 16 ug/L, which is very close to the assay-detection limit. This was one of three wells in the 100-BC Area with detectable chromium.

Annual monitoring of this well reported in the 2002 Hanford site groundwater monitoring report (PNNL 2003) provides temporal information about tritium and strontium concentrations (Figures 5a and b) These plots show a significant rise in both Sr-90 and Tritium in 1997, which was shortly after the site was disturbed with remediation. This rise doesn't show up very well in the plots; our preference would be for individual plots that show the temporal variance. However, the following qualify as "significant" increases: Forty-percent increase in Sr-90 concentration to 70 pCi/L Tritium's rise from under-MCL (3,800 pCi/L) to levels exceeding 60,000 pCi/L. Both increases are well above the MCL and correlate with other groundwater-monitoring data for the BC Area (PNNL 2003).

During remediation, the vegetation on most of the surface of the BC Area was stripped, which caused an increased precipitation infiltration. There also appears to have been a significant amount of dust-suppression moisture, and possibly compaction moisture, added to the soils. Records regarding the quantities of water added, as well as how, when and where it was added, were not found in the Administrative Record. Informal discussion with regulators suggests there may have been quite a large quantity of dust-suppression water added at the site.

In addition, the excavation of trench 116-B-1 to 15 feet created a local surface area depression with a temporary high-infiltration rate. A good picture of the extent of the surface disturbance in the 100-BC Area is found in the 2002 groundwater monitoring report (PNNL 2003, Page 2.2).

The increase in Sr-90 and tritium concentrations in the groundwater at well 299-B3-1 is consistent with similar increases in nearby groundwater wells; it correlates with the remedial excavation work in this area. If reliable estimates can be made of the water-addition data, it might be possible to correlate the groundwater data to see if there is an opportunity for model-calibration.

The increases in groundwater contamination could also be the result of seasonal changes in river/groundwater level, combined with regional changes in contamination plumes that originated from the 200 Area. However, this is unlikely, considering the uniform and smooth trends of the highly mobile tritium shown in multiple wells in the BC Area. The high temporal variability of Sr-90, which has a lower mobility than tritium, may be the result of seasonal changes in river level combined with the general trend of increased infiltration effects. Regardless of the release mechanism, we believe the magnitude and trend of the changes in groundwater contamination identify a situation that must be addressed and resolved before any additional RODs are issued for the 100 Areas in general.

The increase in Sr-90 and Tritium in the groundwater that results from increased infiltration show there are significant quantities of both radionuclides in the vadose zone just above the groundwater. It demonstrates what will happen to this contamination that likely resides in the deep vadose zone just above the groundwater, waiting for the first pulse of moisture to ferry it into the groundwater system. Sr-90 and tritium are short-lived radionuclides; institutional control of the sites will undoubtedly be involved in the final remedy of this problem. However, this problem must be looked at in terms of characterization and risk assessment, and must be factored into any ROD.

The response of Tc-99 to the apparent increased infiltration is not known, because Tc-99 was not monitored in the groundwater. Tc-99 is listed as a contaminant of potential concern in the 100-BC-5 OU LFI report because of its high mobility. It was monitored for a short period as a part of the groundwater LFI characterization effort, but appears to have been dropped from the list of analytes for monitoring during the development of the groundwater-monitoring program for the BC Area (Sweeney and Chou 2002). This was presumably the result of a data quality objective (DQO) meeting.

There is only one trench in the 100 Areas (H- Area) in which Tc-99 consistently exceeds the drinking water standard. However, that site may be the indicator of things to come, rather than an anomaly. This was case at the SX and BX tank farms in the 200 Areas at Hanford where Tc-99 was one of the first high-mobility radionuclides to reach groundwater.

Review of the cleanup verification package for the 116-B-1 trench (Bechtel 1999a) shows a cleanup verification model in which the deep zone is represented by the excavation floor contaminant concentration data. These data are the result of a sampling based on classical statistics (as opposed to a spatial statistic or geostatistics).

This conceptual model assumes the worst-case scenario to be that the excavation-floor concentrations extend to the groundwater at the same contaminant -concentration level. This model is considered to be conservative because of a general conclusion of D&R and the LFI that " … radionuclide distribution with depth beneath similar waste sites demonstrates decreasing radionuclide contamination with depth" (Bechtel 1999a). We have shown that this assumption is not supported by the data.

The soil contaminant concentration values from the statistical sampling at the pit bottom were used as input to a RESRAD model in which the one-dimensional RESRAD calculation was used to determine groundwater impact (and hence the impact to the Columbia River). The soil-concentration data from the CVP is provided in Table 4; the peak groundwater concentrations calculated with RESRAD are shown in Table 5.

Radionuclides	Hanford Site Background[a] (pCi/g)	Activity (pCi/g)		
		Overburden	Shallow Zone (Side Wall)	Deep Zone (Excavation Floor)
Am-241	N/A	0.04	0.047	0.042
Co-60	0.008	0.05	0.053	0.163
Cs-137	1.1	0 (<BG)	0.15	2.89
Eu152	N/A	0.48	0.69	6.35
Eu-154	0.033	0.037	0.11	0.45
Ni-63	N/A	6.42	4.0	8.6
Pu-238	0.004	0.02	0.039	0.056
Pu-239/240	0.025	0.015	0.037	0.218
Sr-90	0.18	0 (<BG)	0.16	1.23
U-238	1.1	0 (<BG)	0 (<BG)	0 (<BG)
Nonradionuclides	**(mg/kg)[b]**	**Concentrations (mg/kg)**		
Total chromium	18.5	14.9	12.4	16.1
Chromium (VI)	N/A	0.04	1.18	0.363
Hg	0.33	0.03	NC	NC
Pb	10.2	5.48	NC	NC

[a]Represents the 90th percentile of the lognormal distribution (DOE-RL 1995).
[b]Background for metals not subtracted.
BG = background
N/A = Not analyzed as part of the Hanford Site-wide background study (DOE-RL 1995).
NC = Not a contaminant of concern except for overburden.

(Table 4)

The description of the RESRAD model as a one-dimensional calculation is a bit gratifying. This model assumes that the deep vadose zone is one layer of specified physical properties (porosity, density and hydraulic conductivity, etc). For a specified distribution coefficient (sorption coefficient) it will calculate, using a simple linear equation, the release to the groundwater of a specified quantity of uniformly distributed contaminant within the uniform block of soil. That is about as simple as it gets in terms of modeling a contaminant-source in the soil: What is added as a soil contaminant and what is selected as a distribution coefficient determines what comes out in the groundwater and when.

This model is used to support the determination of whether or not the remedial action objectives (RAO) or Remedial Action Goals (RAG) are satisfied. Two primary RAOs are to: Control the sources of groundwater contamination to minimize impacts to groundwater resources and protect the Columbia River. Return soil concentrations to levels that allow for unlimited future use and exposure, to the extent practicable. A rural residential exposure scenario is used that includes a permanent residency and drinking of the groundwater. So, one RAG is to satisfy drinking-water standards (MCLs) in the groundwater.

Estimated Peak Radionuclide Groundwater Concentrations (Summed Over Shallow and Three Deep Zone Levels) Compared to RAGs.

Radionuclide	Peak Concentration (pCi/L)	Approximate Time of Peak Concentration (years)	RAG (pCi/L)
Am-241	0	0	1.2
Co-60	0.0012	10	147
Cs-137	0.118	43	120
Eu152	0	0	235
Eu154	0	0	59
Ni-63	4.53	100	46
Pu-238	0	0	1.6
Pu-239/240	0	0	1.2
Sr-90	0.355	43	8

RAG = remedial action goal

(Table 5)

The conclusion of the CVP is that the RESRAD model results indicate that radionuclides from the deep zone do not reach groundwater at levels above the four mrem/yr level (MCL) and thus they satisfy the RAG (Table 5).

However, the current one-dimensional absorption model discussed in the cleanup verification packages does not predict such a fast and large increase in groundwater strontium, as was found in groundwater-monitoring data. This raises questions about the sorption calculations and the validity of the model, particularly for low-mobility radionuclides.

In our review, we must first question the assumption of decreasing concentration with depth. This assumption can only be applied once it is apparent that one is below the region of maximum contamination. D&R and LFI data show that the depth of maximum concentration is beneath the depth of excavation, 15 feet, which is the depth at which statistical sampling provided the implied maximum concentration used for calculation in the modeling exercise.

Compare the data of Table 4 from the statistical sampling of the trench bottom (15-foot depth) with Table 1 from the borehole sampling of trench 116-B-1 in D&R and one finds there are higher radionuclide concentrations at depths of 17 and 20 feet than there are at 15 feet, which is the depth of statistical sampling for the cleanup verification model. The radionuclides that show higher concentrations deeper than 15 feet include Sr-90, Pu-239/240, Co-60, Eu-154 and Cs-137. Likewise, data from the LFI borehole (Table 3) shows much higher concentrations at 17 feet than those at 15 feet in Table 4. This means that the model used residual radionuclide concentrations that are too low. Based on the very limited site characterization data is available, we conclude that the model is not conservative.

Since D&R and the LFI sampled just a few feet below the bottom of the crib, little is known about the deeper vadose zone. The assumption of decreasing concentration with depth may be a valid general assumption for most radionuclides, but it does not take into consideration the manner of contamination transport and the way the high concentration regions are distributed in the soils beneath the site. It is likely that there are deep-layered regions or lenses of high contamination concentrations, all the way down to groundwater. The relatively complex path of contamination migration with higher concentrations of more mobile contaminants close to the ground water is not considered in the simplistic conceptual model of decreasing concentration with depth. The RESRAD calculation model associated with this conceptual model does not calculate the conservative values of potential groundwater concentrations.

This assumption that the 15-foot-deep layer is the layer beneath the site of highest contaminant concentration is rejected by the D&R and IRM data and the model itself is rejected by fact that there is simply not enough data to make the type of generalization or assumptions that are being made about the distribution of contamination.

The DOE notes that "an important element of this strategy is the application of the observational approach, in which characterization data are collected concurrently with cleanup" (DOE 1994a). So, we review the CVP sampling plan and we find a classical statistical sampling of a very large surface area. We have no spatial data on the contamination distribution, no understanding of the distribution variance or anything other than an estimation of a mean across a very large surface area at a common depth.

Knowing from waste sites studied at other locations at Hanford that the contamination distribution in the soil is non-uniform, it is an understatement to say that the characterization data collected concurrently with cleanup is nearly worthless for characterization purposes. The CVP sampling methodology assumes and is based entirely upon the supposition of a uniform distribution of contamination, and the sampling objective is to determine the general depth trend . Yet the contaminant-concentration variation with depth, as seen at many Hanford waste sites, is entirely different, which violates the basic assumptions of the sampling methodology. We conclude that the cleanup-verification characterization and sampling methodology have produced faulty residual-contamination data that is clearly not conservative and does not provide any information about the spatial variation of contamination distribution.

The risk to groundwater from the residual contamination will be found as small, isolated regions of limited area that show high-concentration regions of contamination close to the groundwater. It is not found as uniform distributions of contaminants at low concentrations high in the vadose zone. Yet, this was the only region of the subsurface that was sampled.

We believe the characterization data collected during cleanup produces a completely false representation of the actual contamination concentration and distribution by effectively averaging the samples over very large areas. This cleanup verification sampling/characterization produced a false cleanup verification and the model produces an inaccurate risk assessment. A better characterization during cleanup would have produced an understanding of the distribution of contami-

nation, with depth and across the facility. This would likely have led to the development of a model with regions of high contaminant concentration close to the groundwater.

Let us now consider the more mobile radionuclides that, according to the RESRAD model, reach groundwater within 1,000 years (Table 5). Sr-90 appears to be troublesome in that the concentration and time of peak concentration do not appear to be as significant and fast as the increase that we see in the groundwater that we attribute to an increase in infiltration. The 43-year peak concentration time and the peak concentration magnitude of Table 5 don't appear to be consistent with recent groundwater monitoring data. So, we have a problem with the conceptual model or the RESRAD modeling scheme, or both.

Again, we need to say that Sr-90 and tritium will probably both be managed for decay and the risk is likely to be managed with institutional controls but this should have been addressed in the interim ROD. The real concern, of course, is that we have little data about the deeper contamination, including Sr-90 and tritium, as well as chromium, Ni-63, Tc-99 and others.

Trench 116-B-1

The 116-B-1 trench was excavated to a depth of 15 feet. According to pre-excavation characterization data, it is unlikely that the soil with the highest contaminant concentrations was removed. No data were available on contamination in the deeper vadose zone prior to interim remediation.

One borehole was drilled through the trench for soil sampling and spectral-gamma logging. Sampling occurred within the upper 20 feet of the borehole; the spectral gamma log is to 23 feet.

The best spatial data, other than the short section of spectral gamma log, is a set of four samples with depth in one borehole (from D&R). There is little data on the horizontal extent of contamination; the depth is known only for the highest gross activity-concentration region, immediately below the trench bottom. This region was probably the portion of soil that was subjected to saturated flow during effluent disposal. Spectral gamma-ray log data shows the highest gamma activity at a depth of 20 feet.

The only information available on the deeper vadose zone from 20 feet to the groundwater table at approximately 45 feet comes from a spectral gamma-ray assay profile from a water well located 150 feet from the trench. That profile shows low-mobility radionuclide contamination (Cs-137 and Eu) from 30 to 50 feet. This contamination likely resulted from saturated flow, migration and adsorption.

The presence of low-mobility and generally low-risk contamination far from the source, causes us to question the concentration and distribution of high-mobility, non-gamma-emitting radionuclides in the deeper vadose zone, where they would be readily available for transport into the groundwater with a pulse of moisture.

No information or data is available on the distribution of Tc-99 in the soil other than four near-surface soil samples, which did not detect Tc-99. This is not a surprise, considering technetium's mobility and behavior at 200 Area soil sites. We generally do not find Tc in the bulk of the contamination at soil sites; we find it further from the source, in the deeper vadose-zone sediment. We know Tc-99 was released; it is found in the groundwater at this site at a level of approximately one-tenth of the MCL. However, we have no temporal monitoring data for this groundwater contaminant. It is very likely that the quantities and concentrations of Tc-99 in the deeper vadose-zone soils is low enough that it does not create a significant long-term risk to groundwater, but that remains to be determined.

Data on Sr-90 concentrations in the near-surface region show the highest Sr-90 concentration in the deepest soil sample. As with Tc-99, the horizontal and depth extent is not known for Sr-90; particularly for the deeper vadose-zone region. Evidence of a significant quantity of Sr-90 in the deep vadose zone is indicated by groundwater monitoring data from the well closest to this site and from other wells in the BC Area. Sr-90 has a 29 year half-life, making it a relatively short-lived risk that makes institutional controls very important.

There are also questions about the distribution of contaminants such as chromium, Ni-63, and even Co-60 and Cs-137 in the deep soil at 116-B-1. Chromium was detected during the apparent infiltration increase; it was found in excess of MCL at one BC Area location. It too is a long-term risk to groundwater and we know little about its distribution in the sediment, particularly in the deeper vadose zone.

Our assessment of this trench shows that the IRM contamination excavation and removal action may not have satisfied the intent of achieving a remedy that leads to a final ROD, as intended with this interim measure. We clearly show how a faulty conceptual model and inappropriate sampling methods lead to incorrect residual contamination estimates being used as input to a very simplistic contaminant transport model, which in turn led to the false verification of achieving remedial action goals.

The question of whether or how well the IRM satisfied the remedial action goals and objectives will not be known until characterization data are available on the deeper vadose zone sediment. The data gaps should have been resolved in the initial remedial investigation that led to the interim ROD, or at least during the development of the data-quality objectives. Applying additional remediation measures, such as removing additional contamination or even soil flushing, would have been an option when the site was excavated. The final RI/FS and ROD process must now consider and resolve the characterization and risk-assessment problems identified above.

The main reason the verification of IRM fails is because of an inappropriate conceptual model and lack of data to support it. There is little characterization data to support that basic assumption of decreasing concentration with depth in the manner that the assumption was used in the ROD and applied to create this arbitrary excavation and sampling depth of 15 feet. The rejection of the assumption makes the RESRAD calculation non-conservative. Clear evidence that the calculation is non-conservative, and the interim remedial action goals are not satisfied, comes from

groundwater-monitoring data showing how increased infiltration causes a violation of groundwater standards and remedial action goals.

Trench 116-C-1

Trench 116-C-1, which is 500 feet long, 50 feet wide and 25 feet deep, is located to the northeast of the 116-C-5 retention basins that it served (Figure 1). It was used for the same purpose as Trench 116-B-1, to receive cooling effluent from the 116-C-5 retention basins in the event of a fuel-cladding failure. This trench operation is similar to 116-B-1 except that it was used a bit

(Figure 1)

longer, from 1952 to 1968, because the 116-C-5 basins were constructed as steel tanks that did not degrade like the concrete basins. The trench received approximately 180M gallons of highly radioactive cooling water resulting from ruptured fuel-cladding events. In 1967, this trench was used for an infiltration test, at which time 10 billion gallons of hot cooling water were added over a 150-day period. This likely drove previously released contaminants deeper into the soil.

The 116-C-1 trench remediation was handled as an expedited response action demonstration project in which this site was used to demonstrate the remediation process and the remediation verification process. This was then used as an analogous site to implement and verify other 100 Area waste sites.

D&R drilled fifteen boreholes in and around the trench to obtain subsurface-contamination data. Boreholes were distributed along its length and soil samples were taken at discrete depths. The deepest samples were five taken from 30 to 35 feet in separate boreholes. The radionuclide analysis data from D&R borehole samples is provided in Table 6a and b.

107-C LIQUID WASTE DISPOSAL TRENCH

Sample No.	Pu-238	Pu-239/240	Sr-90	H-3	P-11/Scaler c/m	Eu-152	Co-60	Eu-154	Cs-134	Cs-137	Eu-155	U
		Concentration (pCi/g)										
A 22-1/2	*	1.8×10^{0}	9.8×10^{1}	8.4×10^{0}	750	7.2×10^{1}	4.7×10^{2}	9.7×10^{-1}	*	4.8×10^{2}	*	2.5×10^{-1}
A 30	*	5.3×10^{0}	6.6×10^{0}		250	6.6×10^{-1}	2.3×10^{-1}	3.7×10^{-2}	*	2.9×10^{1}	3.6×10^{-1}	
B 5	*	*	2.5×10^{-2}		<200/20	*	*	*	5.8×10^{-2}	6.5×10^{-2}	7.9×10^{-2}	
B 12	*	*	6.1×10^{-2}		300	7.3×10^{0}	1.2×10^{1}	2.6×10^{0}	1.1×10^{-1}	1.9×10^{1}	*	
B 30	*	5.3×10^{0}	8.4×10^{0}	4.0×10^{1}	2000	6.4×10^{2}	3.9×10^{2}	2.1×10^{2}	*	2.1×10^{2}	2.8×10^{1}	1.6×10^{1}
C 5	*	*	1.7×10^{-1}		<200/20	8.4×10^{-1}	1.3×10^{-1}	*	5.1×10^{-2}	2.4×10^{-1}	*	
C 12	*	7.0×10^{-1}	5.2×10^{-1}	1.1×10^{0}	1000	2.7×10^{2}	5.2×10^{2}	1.0×10^{2}	3.6×10^{-1}	3.1×10^{1}	2.1×10^{1}	2.2×10^{-1}
D 7-1/2	*	*	2.2×10^{-1}	5.5×10^{-1}	300	1.5×10^{1}	2.2×10^{1}	7.5×10^{0}	1.8×10^{-1}	1.7×10^{1}	1.7×10^{0}	2.8×10^{-1}
D 10	*	*	4.1×10^{-1}		<200/40	2.9×10^{0}	2.3×10^{0}	1.3×10^{1}	6.1×10^{-2}	8.9×10^{0}	3.5×10^{-1}	
D 30	*	*	9.0×10^{-1}		<200/20	1.7×10^{-1}	3.0×10^{-1}	*	*	5.6×10^{-1}	*	
E 15	*	*	1.8×10^{-1}	5.1×10^{-1}	<200/15	2.7×10^{-1}	3.4×10^{-1}	*	*	8.7×10^{-2}	*	9.8×10^{-2}
F 20	*	*	2.8×10^{-1}	2.9×10^{-1}	<200/30	1.1×10^{0}	9.7×10^{-1}	7.8×10^{-1}	4.5×10^{-2}	1.0×10^{1}	2.8×10^{-1}	1.2×10^{-1}
G 20	*	2.1×10^{0}	3.1×10^{-1}		1000	3.9×10^{2}	5.0×10^{2}	1.5×10^{2}	3.3×10^{0}	8.0×10^{1}	2.1×10^{1}	
G 23	*	1.5×10^{0}	4.1×10^{-1}		750	2.2×10^{2}	3.7×10^{2}	7.7×10^{1}	2.7×10^{-1}	6.2×10^{1}	9.6×10^{0}	
H 5			6.7×10^{-2}		<200/30	*	*	*	5.3×10^{-2}	*	2.8×10^{-1}	
H 12	*	2.5×10^{-1}	3.5×10^{-1}	*	1000	3.0×10^{2}	4.8×10^{2}	1.1×10^{2}	7.3×10^{-2}	5.8×10^{0}	2.1×10^{1}	9.9×10^{-2}
H 20	*	1.3×10^{-1}	2.8×10^{-2}		500	4.3×10^{1}	8.3×10^{1}	2.1×10^{1}	4.6×10^{-1}	1.5×10^{1}	3.1×10^{0}	
I 5	*	*	3.9×10^{-1}		<200	1.1×10^{0}	3.0×10^{-1}	5.5×10^{-1}	3.1×10^{-2}	3.5×10^{-1}	*	
I 11	*	7.5×10^{-1}	7.9×10^{-1}		2000	4.8×10^{2}	9.5×10^{1}	6.0×10^{2}	2.1×10^{0}	5.2×10^{1}	*	
I 17	*	1.5×10^{0}	7.7×10^{-1}	7.5×10^{-1}	5500	9.1×10^{2}	1.8×10^{3}	3.7×10^{2}	6.8×10^{0}	2.7×10^{1}	6.1×10^{1}	3.2×10^{-1}
I 25	*	2.7×10^{-1}	5.4×10^{-1}		850	1.0×10^{2}	2.2×10^{2}	3.2×10^{1}	2.4×10^{0}	1.1×10^{1}	7.0×10^{0}	
J 15	*	*	2.3×10^{-2}		<200/15	*	1.7×10^{-1}	*	*	9.5×10^{-2}	2.3×10^{-1}	
J 36	*	*	5.0×10^{-1}	2.1×10^{1}	<200/50	2.3×10^{1}	9.6×10^{0}	1.2×10^{1}	4.5×10^{1}	2.0×10^{1}	5.2×10^{0}	2.1×10^{-1}
K 15	*	*	2.4×10^{-1}		<200/10	1.9×10^{-1}	*	*	*	*	*	
K 25	*	*	3.1×10^{-1}		<200/20	1.7×10^{-1}	8.4×10^{-2}	*	*	8.7×10^{-2}	1.1×10^{-1}	
K 35	*	*	3.1×10^{-1}		<200/8kg	1.8×10^{-1}	*	*	*	*	*	
L 18	*	2.7×10^{-1}	2.8×10^{-1}	2.4×10^{0}	600	1.0×10^{2}	2.0×10^{2}	4.2×10^{1}	8.9×10^{-1}	3.4×10^{1}	9.5×10^{0}	2.8×10^{-1}
L 22	*	*	4.6×10^{-1}		<200/15	*	1.9×10^{-1}	*	*	*	*	

(Table 6a)

116-C-1

107-C LIQUID WASTE DISPOSAL TRENCH (CONT'D)

Sample No.	Pu-238	Pu-239/240	Sr-90	H-3	P-11/Scaler c/m	Eu-152	Co-60	Eu-154	Cs-134	Cs-137	Eu-155	U
		Concentration (pCi/g)										
N 10	*	*	2.1×10^{-2}		<200/20	*	4.7×10^{-2}	3.8×10^{-1}	*	*	*	3.1×10^{-1}
N 20	*	*	2.2×10^{-1}		<200/5	*	*	*	*	*	*	
N 10	*	*	9.4×10^{-2}	4.2×10^{0}	<200/20	1.7×10^{-1}	4.7×10^{-2}	*	*	*	*	9.7×10^{-2}
N 20	*	*	*		<200/10	*	*	*	3.6×10^{-2}	*	*	
O 5	*	*	1.0×10^{-1}	8.2×10^{-1}	<200/20	8.5×10^{-1}	4.0×10^{-2}	3.0×10^{-1}	*	1.5×10^{-1}	*	7.5×10^{-2}
O 20	*	*	*		<200/10	*	*	*	*	*	1.1×10^{-1}	

Test holes N and O are between 107-C and 116-C1 and are around expansion boxes in effluent discharge lines to 1904-C.

For sample debris inside of boxes (See 107-B data).

(Table 6b)

D&Rs general conclusions for this trench were that underground contamination extends along its entire length and that the maximum contamination levels were found at a 20-foot depth. However, they show that the highest concentration of some radionuclides was found in the deepest samples, suggesting that the highest concentrations of strontium, europium, cobalt and cesium may not been reached in their sampling..

The 100-BC-1 LFI did not include a characterization of 116-C-1 trench. The LFI assumes the geology is the same as that shown by the borehole drilled in the 116-B-1 trench and that " analytical data from the LFI sampling of the 116-B-1 trench are considered analogous …" and "contamination levels are assumed to be analogous to 116-B-1 (DOE 1994a)."

However, the geology of 116-B-1 was not truly assessed, because that single borehole was only 23 feet deep. Also, we demonstrate above that the contamination distribution of 116-B-1 was never truly assessed. It is clear that there is a paucity of geology and lithology data in this area, as well as data on the contamination distribution.

The analogue correlation between C-1 and B-1 trenches is assumed despite the clear differences in the D&R data from the sites, and despite the differences in quantities of contaminants, effluent volumes and release histories of the two sites. There is no assessment in any of the remedial investigation reports regarding the differences between the sites, based on the limited data available.

Two groundwater-monitoring wells close to Trench 116-C-1 are 199-B3-46 and 199-B3-2. No data is provided on 199-B3-2, which was drilled in 1953 to a depth of 790 feet and undoubtedly terminates in an aquifer within the Miocene basalt flows.

Borehole 199-B3-46 was drilled as a part of the 100-BC-5 OU (groundwater) investigation for monitoring the unconfined aquifer downstream of the trench. Little data from this borehole are provided in the 100-BC-1 OU LFI report or in the 100-BC-5 OU LFI report. Two soil samples were obtained from this well from 30 to 32 feet and from 35 to 38 feet (Table 7). Note: The only radionuclide detected above background is Sr-90 at approximately eight pCi/g at a 36-foot depth; Tc-99 was not assayed.

Analytes	Sample and Sample Interval (ft bls)	
	B05XS4 30.0 - 32.0	B05XS5 35.0 - 38.0
Gross Alpha	7.8[R]	4.4[R]
Gross Beta	32[R]	53[R]
Strontium-90	0.4[J]	7.8[J]
Cesium-137	N/D	0.154[J]
Radium-226	0.723[J]	0.786[J]
Thorium-228	0.641[J]	0.5[J]
Uranium-235	0.007	0.006
Uranium-238	0.15	0.15
Americium-241	N/D	0.01

Water Table Depth: 48.7 ft below land surface

[R] = Value marked as rejected in validation because of missing calibration data
[J] = Value estimated, due to quality control deficiencies
N/D = Constituent not detected, data package detection limit

(Table 7)

A gross gamma-log survey was performed in 1992 on B3-46; the profile data are not provided. Gross gamma-log data indicates an anomaly at the depths of which Sr-90 was found. However, the gross-gamma anomaly is unlikely to have been caused by bremsstrahlung radiation from only eight pCi/g of this purely beta emitter, indicating that other gamma-emitting radionuclides are likely present. This borehole was not logged with the spectral gamma-ray assay system, so the cause of the gross-gamma anomaly and the soil-contamination profile at this location are not known.

Groundwater assay at this well shows elevated levels of Sr-90, Tc-99 and tritium, and the 100-BC-1 LFI indicates that "it appears that 116-C-1 trench is impacting groundwater."

The Tc-99 concentration is the highest detected in the 100-BC Area. Conclusive assessment of the source of the Tc-99, or any other radionuclide, has not been made. Other wells located outside of the BC Area also detect Tc-99 but, for this well, it is not likely that the source of Tc-99 is anything but the 116-C-1 trench.

Sr-90 data from the 2002 groundwater-monitoring report shows a rather large but erratic increase in groundwater concentration just after the site was remediated. The contamination levels increased from approximately 50 pCi/L (MCL = 8 pCi/L) to more than 160 pCi/L. This is similar to other groundwater-monitoring data in the BC Area in the 1998 time frame. However, the previous increase in this particular well in 1993, and the erratic nature of the 1998 increase, do not provide conclusive evidence of a correlation between increased infiltration due to site remediation and the increase in groundwater contamination. At the same time, we must criticize the lack of any assessment of the temporal changes in groundwater contamination in general, and the lack of any assessments of this Sr-90 data specifically. We believe this data shows evidence of infiltration-induced groundwater contamination . This conclusion is supported by Sr-90 and tritium-monitoring data from other BC Area groundwater wells.

The tritium levels in well B3-46, collected from various annual groundwater monitoring reports, shows that tritium reached a level of 8,000 pCi/L in 1999, which is less than half of the MCL. No temporal data or assessments of the tritium levels in this well are available. Likewise, monitoring data on the chromium and Tc-99 levels are not provided.

Based on the above information, a decision was made to remediate the 116-C-1 trench as an interim action; the site was remediated as an expedited response demonstration project. One objective of the interim remediation was to allow for unrestricted future use of the site, but, use of the groundwater was not included in the initial exposure scenario and the cleanup levels to be evaluated were to address the top 15 feet of soil only.

The decision to remediate this site was made with no confirmation of the similarity of this site to its analogous site and with no data about the deeper zone of contamination, other than some limited sampling conducted by D&R. That limited sampling showed that the contamination at 116-

C-1 was actually much deeper, while inappropriately concluding that the total depth of the contamination had been reached.

After the site was remediated by excavating to a depth of 16 feet, a test pit was dug to provide data on the deeper vadose-zone sediment as a part of the remediation-verification process. This test pit was located near the inlet to the trench and was excavated to groundwater. The test-pit excavation occurred in eight lifts of three-foot depth each, from the bottom of the excavation to the groundwater table (eight lifts x three feet/lift = 24 feet). Soil samples were taken from four quadrants of the test pit and composited together to create one sample for analysis per lift or depth. Samples were analyzed for a select list of radionuclides (Figure 6). Notably absent from these plots is an assay for Tc-99, which was apparently dropped as a contaminant of concern in the 1998 version of the sampling and analysis plan (DOE 2001); the justification for this is not provided in the document.

Figure 9. 116-C-1 Test Pit Data -^{137}Cs, ^{152}Eu, ^{63}Ni

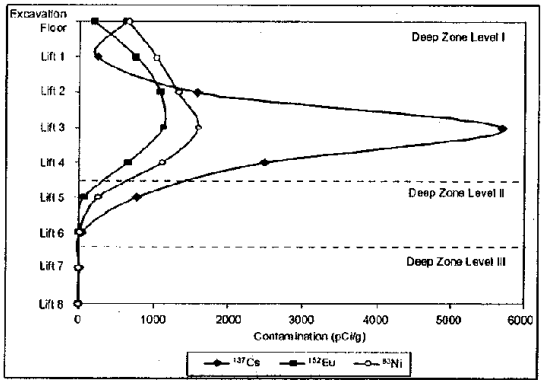


Figure 10. 116-C-1 Test Pit Data – ^{90}Sr, ^{238}Pu, $^{239/240}$Pu, ^{238}U

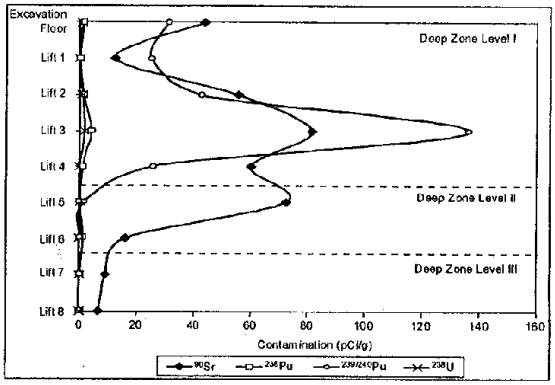


Figure 11. 116-C-1 Test Pit Data – ^{241}Am, ^{60}Co, ^{154}Eu, ^{155}Eu

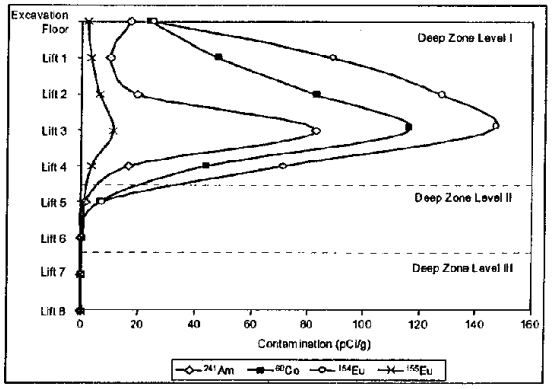
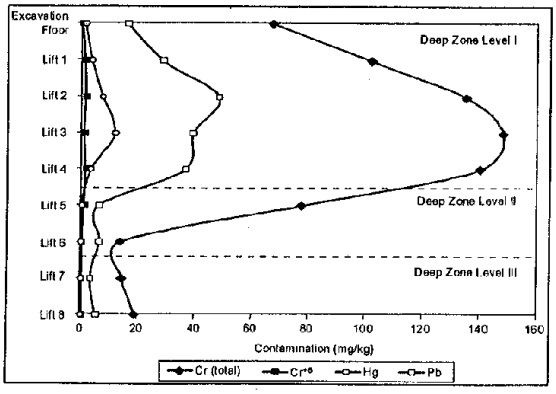


Figure 12. 116-C-1 Test Pit Data – Nonradionuclides Cr(total), Cr^{+6}, Hg, Pb



(Figure 6)

The test-pit data generally confirms the results of D&R (Table 6a); the bulk of the contamination is at or below a depth of approximately 20 feet, which is below the depth of remediation excavation. It is clear from the test-pit data that the limited sampling in D&R missed the bulk of the contamination, as D&R appropriately concluded. The test-pit data provides the only data on the

deeper vadose-zone contamination, with the exception of a single data point from the sample taken from a groundwater monitoring well 150 feet away.

The test-pit data, combined with the sampling data from the verification sampling, were used to produce a cleanup verification model that was used in RESRAD to determine groundwater concentrations and ultimately to estimate exposure.

The model for 116-C-1 has three residual contamination zones, including the overburden, shallow and deep zones. The overburden and shallow zone residual contamination levels are derived from remediation survey statistical sampling similar to that of 116-B-1. This data shows the overburden and shallow zones to not contain appreciable quantities of contamination. Unlike the 116-B-1 verification model, the 116-C-1 model uses three layers of residual contamination in the deep zone. The residual contamination levels of each layer were obtained by averaging the data from the verification test pit (Table 8).

		OVERBURDEN	SHALLOW ZONE (Side Wall)	DEEP ZONE (Excavation Floor and Test Pit)		
				Level I	Level II	Level III
Radionuclides	Hanford Site Background[a] (pCi/g)	Activity Above Background[b] (pCi/g)	Activity (pCi/g)	Activity (pCi/g)	Activity (pCi/g)	Activity (pCi/g)
^{241}Am	N/A	0.041	0.024	29.3	0.663	0.035
^{60}Co	0.00842	0.076	0.074	62.7	3.34	0.015
^{137}Cs	1.05	0	0.803	2120	407	3.91
^{152}Eu	N/A	0.755	0.643	759	33.4	0.032
^{154}Eu	0.0334	0.011	0.258	91.1	1.84	0.047
^{155}Eu	0.0539	0	0.084	3.32	0.286	0.060
^{63}Ni	N/A	4.96	1.54	1140	131	0.905
^{238}Pu	0.0038	0.005	0.017	1.40	0.535	0.082
$^{239/240}Pu$	0.0248	0.041	0.215	52.0	0.998	0.011
^{90}Sr	0.178	0.181	0.345	50.8	44.6	7.98
^{238}U	1.06	0[c]	0[c]	0[c]	0[c]	0[c]
Nonradionuclides	(mg/kg)	Statistical Value (mg/kg)	Statistical Value (mg/kg)	Concentrations (mg/kg)		
Chromium (Total)	18.5	12.1	12.3	119	45.8	16.9
Chromium (VI)	N/A	0.210	0.058	1.15	0.073	0.034
Mercury	0.33	0.030	0.030	5.53	0.255	0.025
Lead	10.2	4.03	4.41	33.9	6.45	3.63

N/A = Not analyzed as part of the Hanford Site-wide background study (DOE-RL 1995).
[a]Represents the 90th percentile of the lognormal distribution (DOE-RL 1995).
[b]Both anthropogenic and natural radionuclide background subtracted from overburden statistical values.
[c]The background value for ^{238}U is subtracted for the shallow and deep zones.

(Table 8)

The RESRAD calculation model uses a homogeneous, isotropic soil model with constant physical properties. It assigns a distribution coefficient for each radionuclide and considers each residual contamination layer with a separate calculation, and then adds the result to produce the effect of all layers.

The estimated peak radionuclide concentrations in groundwater that results from the RESRAD calculation are provided as (Table 9) (Bechtel 1999b). Note that the only calculated groundwater radionuclide concentration that comes close to the groundwater remedial action goal is Sr-90,

Estimated Peak Radionuclide Groundwater Concentrations (Summed over Shallow and Three Deep Zone Levels) Compared to RAGs.

Radionuclide	Peak Concentration (pCi/L)	Time of Peak Concentration (Years)	RAG (pCi/L)
^{241}Am	0	0	1.2
^{60}Co	1.0E-04	10	147
^{137}Cs	1.6E-01	41	120
^{63}Ni	5.1E-01	157	46
^{152}Eu	0	0	235
^{154}Eu	0	0	59
^{155}Eu	0	0	587
^{238}Pu	0	0	1.6
^{239}Pu	0	0	1.2
^{240}Pu	0	0	1.2
^{90}Sr	2.3	41	8

(Table 9)

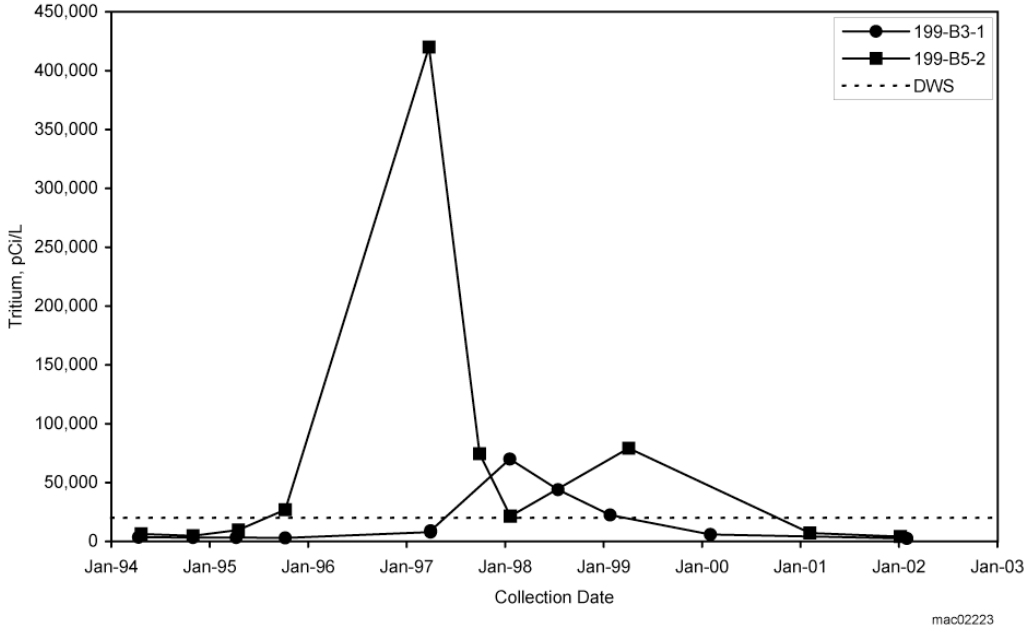
which is approximately ¼ of the RAG. This peak concentration value is calculated to occur in 41 years. Review of the calculation data (Bechtel 1999b appendices) shows that the entire contribution of contamination to groundwater comes from the deepest layer in the deep zone or below approximately 25 feet. This is essentially true for all of the contaminants that are shown to reach groundwater. This means that as far as Sr-90 contamination of the groundwater is concerned, it does not matter what levels of residual contamination is left in anything but the deepest zone. Or, put another way, the deeper zone contamination level is a sensitive parameter in the model.

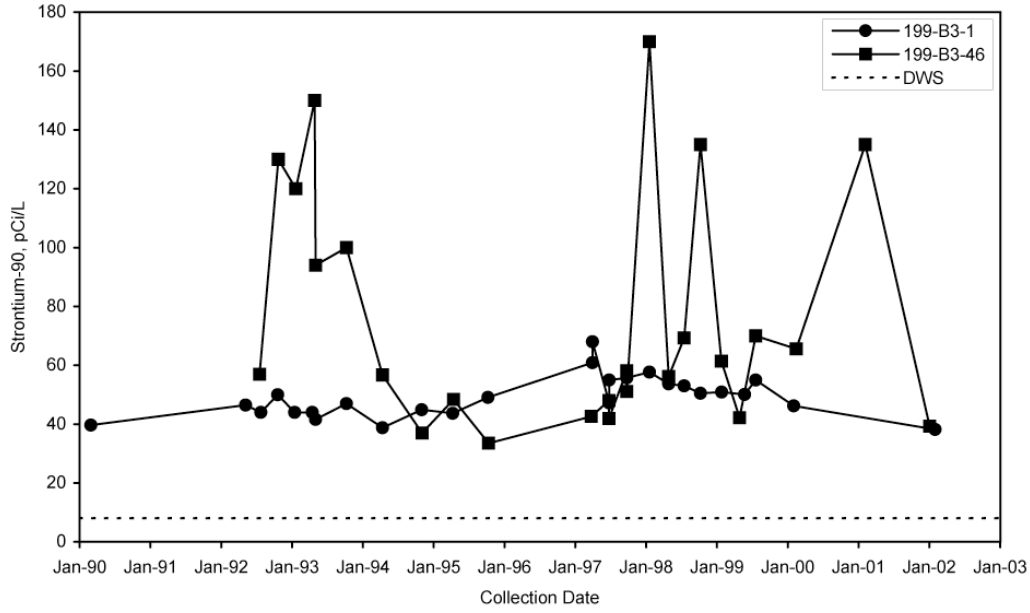
This also increases the importance of having good site characterization data on this sensitive parameter, and it also means that for cleanup verification, there is total reliance on the data from the one test pit. That single data point, combined with the simple RESRAD sorption calculation, provides the only verification that the groundwater beneath this site will be safe to drink.

Because the Sr-90 groundwater concentration is predicted to be ¼ of the RAG, we are not satisfied with this cleanup verification process. No information is provided about the sensitivity of the various parameters that went into the calculation that verified cleanup. Specifically, we do not know how sensitive the residual concentration of the deep layer is to meeting the RAG. If the residual Sr-90 concentration is 10 pCi/g instead of eight, does that cause a failure of the RAG? Considering that only one sample from one deep location went into the verification model, are we justified in relying on that one sample to show what will happen over the entire site?

Consider that the model used 8 pCi/g Sr-90 to calculate the corresponding future groundwater concentration levels. This same eight pCi/g Sr-90 level in the deep zone is also found at the 299-B3-46 groundwater well, approximately 150 feet away from the trench. If the eight pCi/g level calculated over the waste site dimensions produces a large enough groundwater contamination level that causes it to be close to the RAG, what happens when the true horizontal extent of deep

contamination is determined and considered in a more accurate calculation of groundwater contamination levels? Does the remediation still meet the RAG? Looking at the results of the RESRAD modeling exercise, we doubt that it will. And, if one looks at the groundwater monitoring data for 199-B3-46 (Figure 5b), it is clear what the data suggests.





Looking at the levels of Sr-90 in Level II, just above Level III in the deep zone, we see that a small change in the depth extent of the contamination distribution will greatly affect the groundwater contamination levels, at least according to the RESRAD model. The critical question here

is whether or not there are regions along the trench in which the contamination has moved just a bit deeper. This is clearly a sensitive parameter and, with no data on the spatial extent of contamination, the verification model is not appropriate.

One must also consider the distribution coefficient assigned in the RESRAD calculation for Sr-90. The basis for selecting that value and the sensitivity of that selection on the net calculation is not discussed in the CVP. We believe there is an improper basis for assigning distribution coefficients in the model, to the point that the calculated groundwater concentrations are incorrect; and ultimately, the verification showing compliance with the groundwater RAG is not supported. The assignment of distribution coefficients is a critical parameter of this model; there is no calibration of the model or the coefficient assignments.

We do not believe that the characterization of the residuals of the deep layer, the cleanup verification model itself and the RESRAD calculations are accurate or precise enough to reliably predict groundwater concentrations; we also do not believe that the calculated values can be used for cleanup verification. In effect, we need more and better proof. This conclusion considers the fact that the groundwater monitoring data from the BC Area shows increases in Sr-90 specifically, as the result of increased infiltration, and thereby refutes the RESRAD model calculation.

Because there was no real characterization prior to site remediation, and no subsequent detailed characterization during or since remediation, the complex and anisotropic nature of the contamination distribution is neither evident nor recognized. Therefore, we must reject the use of only a single data point as *the* characterization of deep residual contamination. Judging from the verification RESRAD model findings, the value of the deep residual contamination level is probably the most sensitive input parameter of the model. Yet, it is the value for which there is the least amount of information.

We must also reject the use of the simple homogeneous, isotropic assumptions of the verification model unless there is some form of a model calibration to support it. Judging from the few studies of contamination distribution at other waste sites, and particularly at the 200 Area waste sites, there is reason to believe that similarly complex contamination distributions will be found at the 100 Area waste sites. But, we don't know this yet: studies involving a spatial assessment of the contamination distribution have not been completed for any of the 100 Area waste sites. Therefore, there is no basis for applying an homogeneous, isotropic soil profile model.

The RESRAD model is helpful in indicating radionuclides that might be troublesome (i.e. The model indicates that if the distribution of Sr-90 is not homogeneous and isotropic, deep regions of higher concentration will be a problem.) What that model and the groundwater-monitoring data suggest is that with increased infiltration or an increase in deep moisture flux through the system, there is a problem with contaminants that have a distribution coefficient similar to Sr-90. This is the real utility of the RESRAD model; this is the approximate accuracy of the model: it

suggests that we must have appropriate data on the deep vadose zone in order to ensure that higher-mobility radionuclides do not present a problem in the future.

Consider the unrestricted use of the groundwater in a rural residential scenario: One of the first things required for a commercial building permit in Washington State is a site-drainage plan. This usually involves the installation of a dry well, which is a point-source of water that infiltrates deeply (approximately 15 feet) into the vadose zone. This will mobilize any contamination found in the deeper vadose zone, causing groundwater contamination that can then be accessed downstream for domestic use.

Granted, a commercial building is not a "rural residential scenario." but it is close. Point-sources of water in a rural setting come from street drainage dry wells, domestic septic systems, building drainages, downspouts and irrigation runoff. It is not uncommon to have multiple point-sources of water in a rural residential setting.

For unrestricted use of the site and groundwater, every site should have an assessment of the effect of single or multiple point-sources of water. The scenario noted above is not an unlikely scenario, but it is probably the most likely scenario for any kind of real exposure by way of the groundwater pathway.

The current conceptual model and corresponding RESRAD model consider a uniform irrigation rate of 30 inches per year, but also includes an evapo-transpiration rate that keeps the moisture flux through the deep vadose zone at a low level. A more realistic infiltration scheme from a point-source of water with minimal evapo-transpiration will cause increased groundwater contamination. This is why removal of the vegetation with subsequent addition of water after excavation caused large increases in mobile contaminants such as tritium, Sr-90, Cr and probably Tc-99.

For any final remediation action and any additional interim actions, we believe an analysis is required of the effects of a point-source of water to a well-characterized vadose zone contamination profile.

Our review of the 116-C-1 trench interim remedial action shows that there was little characterization of this site prior to remediation. This data gap has a significant influence on the selection of the remedial action and the verification that demonstrates the remedial action goals have been satisfied.

The only data on the contamination distribution beneath the trench prior to excavation is from D&R. The limitations of their data in terms of depth of investigation and even in terms of areal extent, were not appropriately resolved through the data-quality-objectives process before the interim remedial action was taken. Additionally, there was no assessment of the differences in what little is known about the contamination distribution between this site and the site analog, 116-B-1. There is no direct data comparison between the two sites. If that had been done, the

study would have identified a need for additional information about the deep contamination distribution at both sites.

The site-cleanup verification model, as simplistic as it is, shows that the only residual vadose-zone contamination that presents a problem for future groundwater is found in the deepest region of the vadose zone. Yet, this is the region for which there is the least amount of data. The assignment of the deep-zone residual contamination level in the RESRAD cleanup verification calculation comes from a single data point, the test pit that was dug after site excavation. In all likelihood, this single data point does not represent the highest concentration levels nor the deepest contamination at this trench; therefore, the verification model is non-conservative. There is also little information on mobile contaminants such as Tc-99, tritium and chromium, particularly from the deep zone.

We believe there is an inadequate basis for adopting the site conceptual model and the corresponding RESRAD verification model. Considering what little we do know about the distribution of contamination at other sites at Hanford, and the variability of the soil stratigraphy, it is clear that an homogeneous site model is not appropriate and additional site characterization data are needed.

There is also no assessment or explanation of the sensitivity of the model to the input parameters and no basis for the assignment of distribution coefficients. If the distribution coefficient of Sr-90 is off by 10 percent, for instance, we have no idea how that will effect the calculation of contamination-concentration in the groundwater. This uncertainty does not produce an appropriate verification of the cleanup.

Finally, the model considers only an homogeneous infiltration with evapotranspiration. It does not consider a point source of water which is the most likely cause of future mobilization of residual contamination; this has a much greater potential to cause future groundwater contamination.

In summary, we find very limited site characterization data, inappropriate adoption of general assumptions about the contamination distribution, no basis for the site conceptual model, no basis for the use of the homogeneous RESRAD model, inadequate basis for the residual contamination levels used as input to the model, and no verification of groundwater protection by the interim action. This occurs in a situation where the groundwater monitoring data clearly shows the effect of increased infiltration on groundwater contamination levels, demonstrating that the RESRAD model is wrong and the remedial action goal of protecting the groundwater did not occur.

Summary and Conclusions

Our review of the interim action cleanup at the 100-BC Area focused on the two trenches that are likely to have the highest levels and the most contamination of the soil and groundwater. In the remediation process, these sites are considered to be analogous in terms of contamination distribution and potential future risk. Although not specifically stated in the documentation, it is assumed in the cleanup that these sites represent the highest risk and greatest cleanup challenge in the BC Area. Therefore, DOE assumes that if it is shown that the cleanup of these sites satisfies the remedial action goals, then the rest of the BC Area waste sites will do the same.

In our review, we first assessed the site characterization data for these two sites and revealed that there was little characterization of these sites prior to remediation. This is as intended by the interim process, which will make maximum use of existing data, consistent with data quality objectives.

We find that general assumptions about the contamination distribution from past site characterization work (D&R) is generally adopted without assessing the validity of those assumptions by either reviewing and evaluating the original data, or by correlating it with new data obtained from recent limited field investigations.

Our review of the data shows that the generally held conclusion that contaminant concentration decreases with depth is supported by neither the older data nor the newest data (test pit). We demonstrate that for some radionuclides, the depth extent of the contamination has not been reached by the limited sampling and assays. The contamination from the deepest portion of the vadose zone sediment beneath both sites is obtained from one sampling event at one location.

This type of site characterization provides an inadequate basis for developing a site conceptual model. There is no basis for the assumption of a uniform distribution of contamination in the soil; there is no basis for the assumption of homogeneous and isotropic soil properties, and there is no basis for the assumption of a uniform moisture flux through the system.

The site conceptual model is used to create a RESRAD calculation model is in turn used to verify that the cleanup action will meet the remedial action goals and specifically will comply with groundwater standards (MCL). As a result, we find an inadequate basis for adoption of the RESRAD calculation model. This is due to the fact that the homogeneous nature of the model is not supported by site characterization data nor by data from other facilities and sites at Hanford.

We also found that the RESRAD calculation model is non-conservative. Residual soil contamination levels used as input to the verification model for the deep vadose zone sediment originates from one sampling data point, a single test pit at one trench. There is little-to-no knowledge or understanding of the concentration or distribution of contamination in the deep vadose zone. Yet this is the region that, according to the RESRAD model, is most likely to cause failure of groundwater MCL. It is also the region most likely to hold the highest concentration of the more mobile, higher-risk radionuclides such as Sr-90, tritium and Tc-99.

In other words, **the region of soil contamination that is of greatest concern from a risk standpoint is the region about which there is the least amount of data and the poorest understanding of contamination distribution**.

We find an inadequate basis for adoption of the residual contamination levels used in the RESRAD calculation. As a result, verification of groundwater protection by the cleanup process is not demonstrated.

In fact, groundwater monitoring data demonstrates that compliance with drinking water standards is problematic. The recent groundwater monitoring for select radionuclide concentrations shows a rise in the levels of Sr-90, tritium and Tc-99. This is due to either increased infiltration from water application during excavation or to an increase in river level. Either way, this increase is exactly what should be expected; this scenario is not represented by the RESRAD model.

We find that the source of many of the 100 BC Area the cleanup problems is inadequate site-characterization data.

- There is little information about pre-remediation site conditions
- Characterization data collected during remediation is sparce, inadequate and never assessed or analyzed.
- Characterization data collected after excavation (one test pit) does not provide any spatial information. Yet, that single data point is used as the basis for verification of a highly sensitive parameter of the verification model.

For the two sites that we reviewed in the 100 BC Area, our conclusion is that we cannot assess the effectiveness of the cleanup because we don't know what is left behind and we don't know what risk is associated with the remaining contamination.

We are certain that our assessment of these two sites applies to the rest of the 100 BC Area sites that were remediated. However, we are reluctant at this point to conclude that the cleanup at the other 100 Area sites has the same failures, even though the other 100 Areas received the same level of characterization and used the same modeling and cleanup verification procedures. Instead, we recommend that the key sites in each of the 100 Areas be subjected to an extensive review, similar to this one, in order to determine with certainty whether or not our assessment of the 100 BC Area applies to those other sites as well.

100-BC Area References

Bechtel, 2003 “Cleanup Verification Package for the 100-C-3 French Drain” CVP-2003-00009. Bechtel Hanford Inc., Richland, Wash.

Bechtel, 1999a “Cleanup Verification Package for the 116-B-1 Process Effluent Trench” CVP-98-00006 December 1999, Bechtel Hanford Inc., Richland, Wash.

Bechtel, 1999b “Cleanup Verification Package for the 116-C-1 Process Effluent Trench” CVP-98-00006 January 1999, Bechtel Hanford Inc., Richland, Wash.

DOE/RL, 1990 “Remedial Investigation/Feasibility Study Work Plan for the 100-BC-1 Operable Unit, Hanford Site, Richland, Wash.” DOE/RL-90-07, U.S. Department of Energy, Richland Wash.

DOE/RL, 1994a “Limited Field Investigation Report for the 100-BC-1 Operable Unit” DOE/RL-93-06

DOE/RL, 1994b “Limited Field Investigation Report for the 100-BC-5 Operable Unit” DOE/RL-93-37, U.S. Department of Energy, Richland Wash.

DOE/RL, 1994c “Limited Field Investigation Report for the 100-BC-2 Operable Unit” DOE/RL-94-42, U.S. Department of Energy, Richland Wash.

DOE/RL, 1994d “Proposed Plan for Interim Remedial Measures at 100-BC-1 Operable Unit” DOE/RL-94-99, U.S. Department of Energy, Richland Wash.

DOE/RL, 1995a “100-BC-1 Demonstration Project Expedited Response Action Proposal” DOE/RL-95-51, U.S. Department of Energy, Richland Wash.

DOE/RL, 1995b “100-BC-1 Excavation Demonstration Project Plan” DOE/RL-95-56, U.S. Department of Energy, Richland Wash.

DOE/RL 1998 “Remedial Design Report/Remedial Action Work Plan for the 100 Area” DOE/RL-96-17 Rev1 U.S. Department of Energy, Richland Wash.

DOE/RL 2001 “100 Area Remedial Action Sampling and Analysis Plan” DOE/RL-96-22 Rev.3 U.S. Department of Energy, Richland Wash.

DOE/RL, 2003 “100-BC-5 Operable Unit Sampling and Analysis Plan” DOE/RL-2003-38, U.S. Department of Energy, Richland Wash.

Dorian, J.J. and V.R. Richards, 1978 “Radiological characterization of the retired 100 Areas” UNI-946 United Nuclear Inc., Richland, Wash.

Peterson, R.E., R.F. Raidl and C.W.Denslaw, 1996, “Conceptual Site Models for Groundwater Contamination at 100-BC-5, 100-KR-4, 100-HR-3 and 100-FR-3 Operable Units” BHI-00917, Bechtel Hanford Inc., Richland, Wash.

Price, R.K. 1993 “Spectral Gamma-Ray Log Report for the 100 Area Borehole Surveys” WHC-SD-EN-TI-123 Westinghouse Hanford Company, Richland, Wash.

PNNL 2003 “Hanford Site Groundwater Monitoring for FY 2002” PNNL-14187, Pacific Northwest National Laboratory, Richland, Wash.

Sweeney, M.D. and C.J. Chou, 2002 “DQO Summary Report – Designing a groundwater Monitoring and Assessment Network for the 100-BC-5 and 100-FR-3 Operable Units” PNNL-14287, Pacific Northwest National Laboratory, Richland, Wash.

WHC 1994 “100-B/C Technical Baseline Report” R.W. Carpenter, S.L. Cote, D.H. Deford, and M.W. Einan, Westinghouse Hanford Company, Richland, Wash.