Historical Data Recovery for Accelerated Site Closure, Nevada Test Site, Nevada – 10336

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ABSTRACT

Atmospheric nuclear tests at the Nevada Test Site (NTS) resulted in widespread contamination. Underground tests also had the potential for releasing contamination to surface soils. Historical data available to characterize these sites were often collected with lower quality requirements than currently used and may be incomplete and/or difficult to retrieve. However, these data can often be recovered to develop cost-effective closure approaches for legacy sites and save time and effort by optimizing characterization efforts, accelerating site closures at the NTS, and allowing most likely final end states and approaches for remedial actions to be developed.

Two sources of historical data are available. One source includes log books and other paper files with information on contamination, contaminant migration, and initial mitigation measures. These data can be used for preliminary health and safety evaluations and to qualitatively guide investigations. Another source, developed by the Radionuclide Inventory and Distribution Program (RIDP) in the 1980s to determine the distribution and total inventory of radionuclides in surface soils at the NTS, includes aerial radiological surveys, soil sample results, and in-situ gamma spectroscopy.

In-situ measurements were collected at 3,850 locations. Man-made radionuclides were reported in nanocuries per square meter (nCi/m^2), and naturally occurring radionuclides were reported in picocuries/gram (pCi/g). Plutonium (Pu) was reported as total inventory across study areas. Non-traditional reporting units and lack of location-specific results for Pu limit data usability. Data originally archived on magnetic tape were transferred to a Microsoft Access database in 2006.

A phased approach was used to determine usability and limitations of RIDP data. The data were first compared to aerial survey data and evaluated against data quality indicators. Data were then converted to pCi/g to allow dose calculations. Hard copy RIDP reports were reviewed to apply correct Inverse Relaxation Lengths (IRLs) and plutonium-to-americium ratios. Results were obtained for all 3,850 locations (approximately 80,850 results). A dose assessment was then performed. A GIS interface was added to enhance usability and display total dose at each location. Locations exceeding the 25-millirem per year (mrem/yr) dose constraint were then mapped.

INTRODUCTION

This paper describes methods and activities to recover historical RIDP data. A phased approach allowed decisions to be made regarding the benefit of pursuing additional data recovery. The following tasks were conducted:

- Summarize the RIDP data set and evaluate the quality of the data
- Determine the current uses of RIDP data and cautions associated with its use
- Provide recommendations for enhancing data use through uncertainty and sensitivity analysis and additional field verification

A description and analysis of data collection methods and techniques, data storage systems, data quality evaluations and verification activities, cautions associated with the data, and guidelines for current data use are provided. This paper also provides recommendations for additional verification methods.

DATA ASSESSMENT

Precision, accuracy, representativeness, completeness, and comparability (PARCC) criteria were not formally used at the time of RIDP; however, these concepts can be used to assess RIDP data. Prior to fieldwork, calibrations and test measurements were performed to ensure accuracy. Calibration of each detector was also checked three times per day during operations. Consistency of these measurements ensured precision over time. Approximately 30 percent of the 8,550 spectra recorded were laboratory calibration runs, and 23 percent were field calibrations.

Laboratory quality assurance (QA) procedures during RIDP included inter-laboratory comparisons (comparability), analysis of blind reference samples (accuracy), and comparisons of hidden replicates (precision). A QA “referee” managed sampling protocols, reviewed data results, and provided summary statements of data quality. In addition, an assessment of pilot measurements collected prior to 1980 was conducted. For this paper, other data sets were evaluated and compared to RIDP data to further assess data quality.

Pilot Measurements and Project Improvements

Pilot studies prior to 1980 encountered difficulties with instrument calibration and storage. An assessment of methodology, results, and equipment was conducted. A data summary was developed that stated that soil samples prior to 1980 were questionable due to laboratory analytical problems [1]. In addition, obtaining soil samples to accurately reflect site conditions and associated potential dose rates is difficult. Estimating the total effective dose equivalent (TEDE) from soil samples is problematic due to the heterogeneous nature of soil contamination (the "hot particle problem"). The hot particle problem leads to variable dose rate estimates depending on whether a "hot particle" is captured in the sample. The distribution of Pu isotopes in a soil sample can vary by a factor of 10 between 1-gram aliquots [2]. In-situ methods are not subject to these errors because the sample size, or measurement field of view, is large relative to a soil sample. The large field of view integrates contributions from discrete particles and trinity glass and represents a more realistic exposure scenario than soil samples.

Collection of in-situ measurements began at the NTS in 1978. However, these measurements were of unknown validity or were unreported. Until 1980, very little of the data collected were considered reliable. Criteria such as accuracy and basic reporting standards were poorly defined. Problems with calibration procedures, spectral analysis software, and equipment configuration were noted. Protocols to address these issues were integrated into the RIDP. To continue the project, soil sampling supplemented the in-situ measurements to determine ratios of radionuclides and depth distributions and aerial radiological surveys were used to guide the ground-based measurements.

Calibration Analysis

Since the RIDP derived sensitive parameters, such as IRLs, through empirical methods and used assumed values only for relatively insensitive parameters, RIDP measurement errors were minimized. The IRL is a measure of how uniformly the contaminant is distributed with depth; therefore, this is a sensitive parameter for both RIDP and direct soil sampling methods. The RIDP found these distributions varied by radionuclide and location. A static soil sample depth will concentrate some values and dilute others. This is a potential source of error for both RIDP and soil sample data and thus a potential basis for non-comparable data.

Detection Capability Evaluation

Detection capabilities were not reported in the RIDP database and were estimated using a reporting protocol in the RIDP reports that identified measurements that were at or near detection capabilities as "upper-limit values." An average upper-limit value was established for each radionuclide. Table I provides the maximum upper-limit value for each radionuclide and compares them to the draft Derived Concentration Guides (DCGs) established for internal dose rates at Corrective Action Unit (CAU) 370. By summing the maximum upper limit-values for all radionuclides, a total dose rate of 0.02 mrem/yr is found, which is less than 0.1 percent of the 25-mrem/yr limit. As a result, it can be stated that the RIDP system can detect radionuclides below decision levels.

Table I. RIDP Sensitivity for Internal Dose Rate Calculations

Radionuclide	Derived Concentration Guide (pCi/g)	Maximum Upper-Limit Value (pCi/g)	Dose Rate (mrem/yr)
Am-241	5.63 E+03	5.15	9.14 E-04
Co-60	2.70 E+06	1.03	3.82 E-07
Cs-137	1.63 E+06	1.36	8.35 E-07
Eu-152	6.99 E+06	3.62	5.18 E-07
Eu-154	5.09 E+06	6.16	1.21 E-06
Eu-155	3.42 E+07	5.43	1.59 E-07
Pu-238	6.30 E+03	6.70	1.06 E-03
Pu-239	5.72 E+03	99.10	1.73 E-02
Sr-90	4.52 E+05	3.49	7.73 E-06
Th-232	1.86 E+03	2.31	1.24 E-03
U-235	2.45 E+04	2.89	1.18 E-04
U-238	2.55 E+04	11.33	4.44 E-04
		Total Dose Rate	2.11 E-02

As with other methods used to measure radioactivity, RIDP detection capability is related to background radiation in a given area. It is an indication of how well an instrument can discern contamination from background radiation. Generally, the higher the local background radiation levels, the higher the detection capability of a given instrument. Detection capabilities of the RIDP measurements for external dose rates were determined by calculating the average and the standard deviation of upper-limit values. Average external dose rates plus two standard deviations were then calculated for each radionuclide. These isotope-specific external dose rates were summed to determine a conservative sensitivity estimate. This determination indicates that the RIDP measurement sensitivity for external dose rate is approximately 4 mrem/yr. This was calculated from non-decay-corrected values. Over time, the value will decrease, and essentially, the RIDP data will become more sensitive.

The RIDP sensitivity estimate for determining external dose rates was compared with estimated thermoluminescent dosimeter (TLD) sensitivity. TLD sensitivity was estimated using results of TLD stations across the NTS. This is analogous to the RIDP estimate because the RIDP upper-limit values were also derived across the NTS, where background radiation varies. An average background TLD external dose rate plus two standard deviations was calculated. This resulted in an estimated TLD measurement sensitivity for external dose rate of approximately 16 mrem/yr, which is much closer to the 25-mrem/yr limit. Both the RIDP estimate and the TLD estimate are at a 95-percent confidence interval. In most areas, the actual values are lower than the estimate. This evaluation reflects on the accuracy, precision, and relative sensitivity of the RIDP data.

Duplicate Measurement Quality Analysis

McArthur and Kordas reported QA results for duplicate measurements [3]. The average deviation from the mean of duplicate sets was 8.6 percent. Maximum deviations were not presented. For this paper, 454 duplicate RIDP measurements were assessed to determine duplicate measurement quality. Only 35 of the duplicate measurements varied from each other by more than 25 percent. All of the less precise values were at or near the calculated detection capability; therefore, any errors at these levels have an insignificant impact on RIDP-calculated dose rates. The analysis of duplicate measurements indicates that higher values result in better agreement between values. This indicates that at levels that could impact total dose rate calculations, the RIDP data are precise.

Quality Control Measurement Analysis

The RIDP collected measurements in real time to evaluate data quality. These measurements were collected with instruments set 1 meter above the ground to directly measure external exposure rates. The RIDP also measured radionuclide-specific contaminant levels, used these measurements to calculate an expected exposure rate in real time, and compared these calculated rates to the measured rates. If an error was suspected, another measurement was taken, and if needed, the instrument was recalibrated. This process improved RIDP measurement accuracy.

Laboratory Quality Assurance

Each RIDP report, except the first, contains an appendix that addresses QA issues for soil samples, describes changes in procedure since the previous report, explains problems with the data sets and how they were addressed, and concludes with a summary statement. Each appendix also presents replicate, inter-laboratory, and other supporting QA data. The QA procedures in each report included an analysis of replicate aliquots from samples (a measure of precision), independently calibrated reference blinds for a related program (a measure of accuracy), and duplicate gamma spectroscopy measurements of samples (a measure of comparability). In addition, to resolve uncertainties and provide further assurance of data reliability and comparability, samples were typically analyzed by two independent labs, for a total of four labs.

While the raw laboratory QA data are not readily available, the summary discussions provided in the RIDP reports are adequate to determine data quality and usability for the limited purposes of estimating IRLs and ratios. The analyses indicate there are no specific problems with the data that would preclude further development of the RIDP data or the evaluation of RIDP data quality.

Comparison of RIDP Data to 1994 Aerial Radiological Survey Data

RIDP results for americium (Am)-241 and strong gamma-emitting radionuclides were compared to aerial radiological survey data collected in 1994 [4]. Am-241 aerial survey data are presented in units of counts per second, and strong gamma-emitting radionuclides are quantified as estimated exposure rates. Aerial radiological survey data, by nature, represent average contaminant levels over wide areas.

Aerial survey data are presented as ranges of results, or bins. There are six bins for Am-241 aerial survey data and ten bins for total man-made aerial survey data. RIDP results were grouped according to the ranges or bins in which they were located. GIS selection tools were used to determine which RIDP values were geographically located within each bin. The RIDP values within each bin were averaged and compared to the middle value of each aerial radiological survey range. Figures 1 and 2 illustrate these comparisons and demonstrate a linear relationship based on wide-area averages of RIDP data. The results of this evaluation indicate that the RIDP data are of reasonable quality to proceed with the additional quality assessments discussed later in this paper.

Additional assessments presented in this paper establish RIDP data quality more firmly than the initial positive indication provided by these wide-areas average comparisons. Additional data development may allow a coefficient to be applied to the aerial radiological survey data to estimate dose rates in areas where there is sparse RIDP measurement coverage. The result of such an estimate would only provide a wide-area average.

Since the actual variability of contaminants in soils is high, such a process would be developed with conservative assumptions. This limitation exists for application of a correction factor whether the factor is applied to RIDP data or to newly collected data. In either case, application of a correction factor to aerial survey data to make closure decisions requires a full understanding of the actual variability of contaminant levels within a site-specific survey bin and, based on that variability, calculating an appropriately conservative correction factor to estimate dose rates.

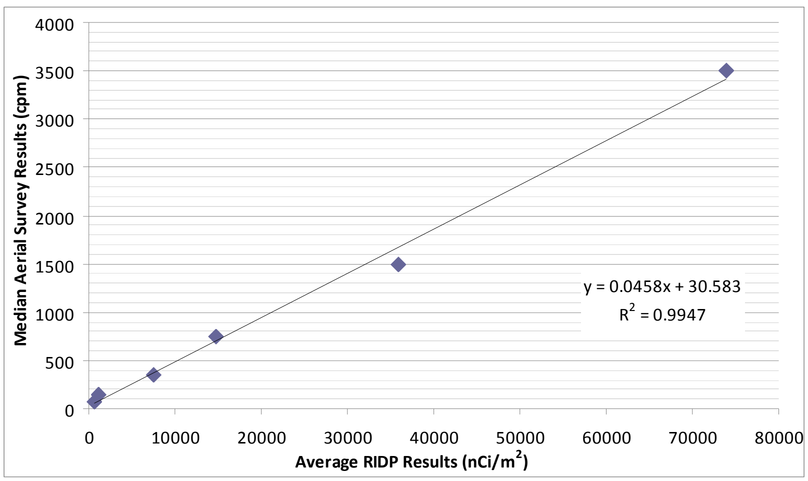

Fig. 1. 1994 Aerial Radiological Survey Data Compared to RIDP Results for Am-241.

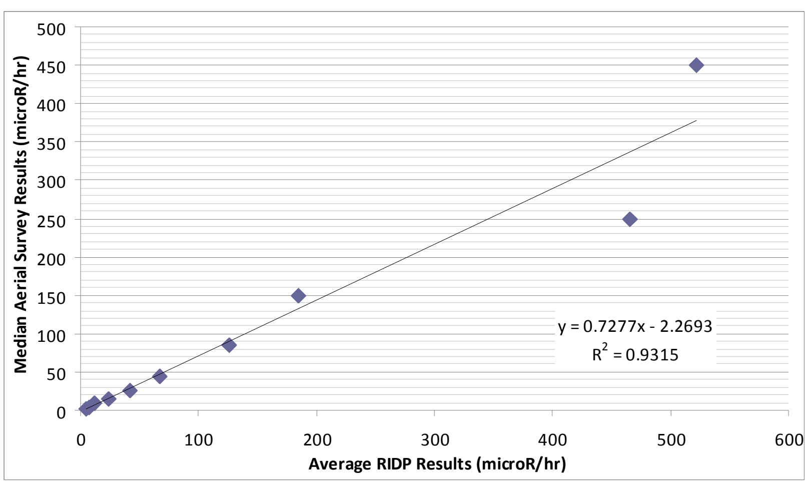

Fig. 2. 1994 Aerial Radiological Survey Data Compared to RIDP Results for Total Man-Made Radionuclides.

Comparison of RIDP Data to Corrective Action Unit 370 Investigation Results

No specific efforts to field verify the RIDP data have been made to date, however, the RIDP data have been compared to the CAU 370 corrective action investigation (CAI) results, and the data show excellent agreement. Evidence demonstrating the currency and accuracy of the RIDP data are presented below. Dose rate assessments for CAU 370 involved calculating internal and external dose rates separately; therefore, the following comparisons and discussion are also separated in this way.

Soil sample results for CAU 370 were used to assess internal dose rates. The Residual Radioactivity (RESRAD) code models dose rates based on the concentrations of radionuclides in soil was used to convert soil sample results in units of pCi/g to dose rates in units of mrem/yr. The RESRAD code was also used to determine draft DCGs for CAU 370. To compare the RIDP results to the CAU 370 soil sample results for the primary, internal dose-driving radionuclides, the RIDP data have been converted to units of pCi/g. Figures 3–5 compare the converted RIDP data to the CAU 370 soil sample results averaged across each CAU 370 soil sample plot for Am-241, Pu-238, and Pu-239.

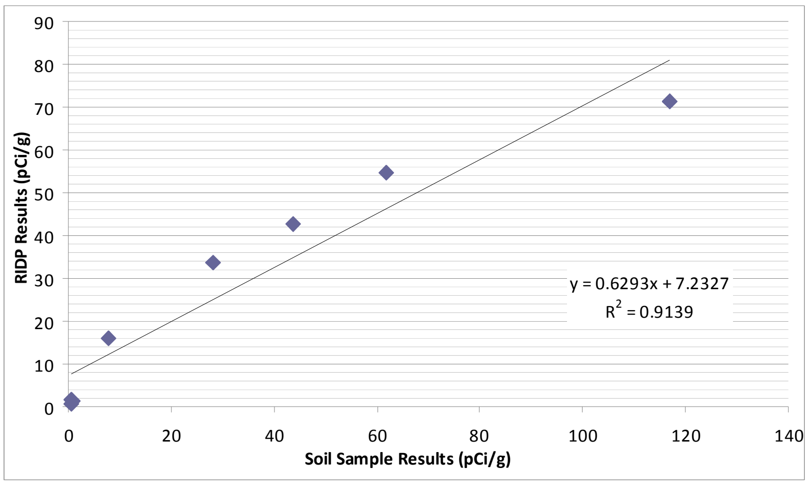

Fig. 3. CAU 370 CAI Results for Am-241 Compared to RIDP Data.

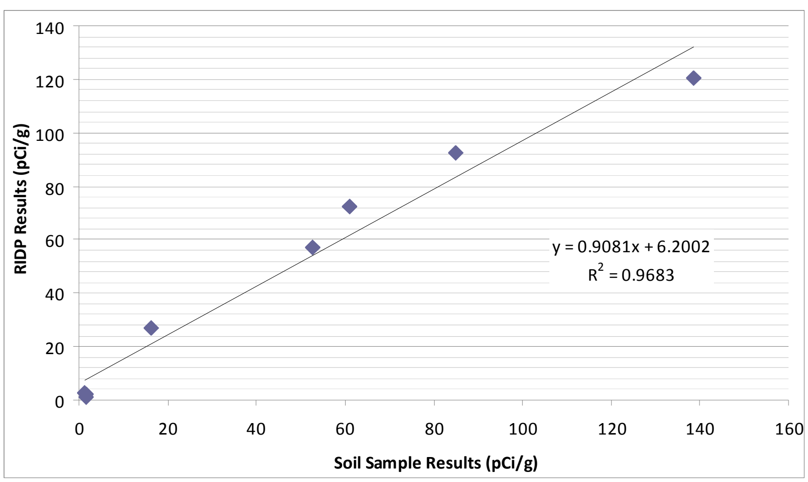

Fig. 4. CAU 370 CAI Results for Pu-238 Compared to RIDP Data.

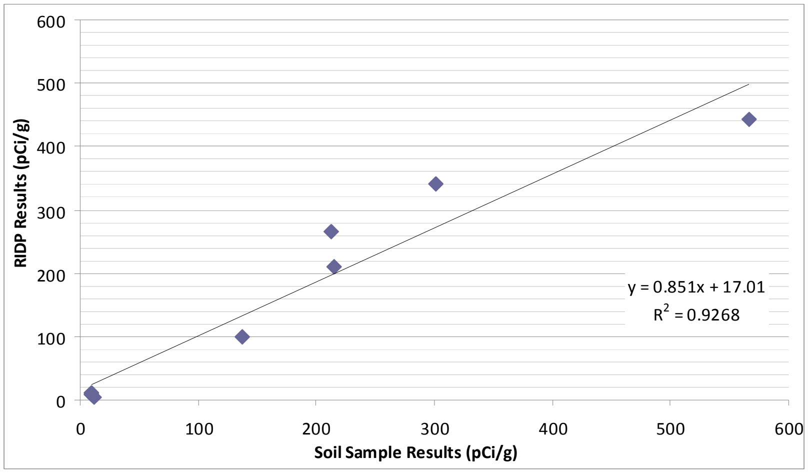

Fig. 5. CAU 370 CAI Results for Pu-239 Compared to RIDP Data.

The graphs in Figures 3–5 show that the data are well correlated; however, converted RIDP measurements expressed in units of pCi/g vary from the average CAU 370 soil sample results at each sample plot. The difference between the converted RIDP values and the average CAU 370 soil sample values potentially results from the following sources:

1) **RIDP Ratios:** Soil samples collected during the RIDP provided ratios to determine contaminant values for radionuclides that the RIDP did not directly measure. Ratios were determined through a limited number of soil samples collected by the RIDP at subject locations. Ratios were then applied across each entire subject area. Applying ratios that were calculated using only a few soil samples to many RIDP measurements may result in some error.
2) **IRLs:** This parameter is based on contaminant distribution with depth. This distribution is known to vary by radionuclide and by location, especially near ground zero locations. IRLs were determined through a limited number of soil samples collected by the RIDP at subject locations. IRLs were then applied to RIDP measurements across each entire subject area. Applying IRLs that were calculated from only a few soil samples to many RIDP measurements may result in some error.
3) **CAU 370 Plot Data:** Relatively high variability may result when a sample mean is calculated from multiple soil samples, even when the samples are collected from locations close to each other. In addition, variability in contaminant distribution with depth may cause variability in sample results, even across a single sample plot. A uniform sampling depth of 5 cm may concentrate values at some locations and dilute values at others. This variability can produce the differences in results between the two characterization methods.

The magnitude of the differences between converted RIDP results and CAU 370 soil sample results caused by each potential source listed above cannot be easily quantified with existing data. The converted RIDP data involve potential error; however, average soil sample values also have associated error. To provide context to the differences between the two data sets, internal dose rates based on converted RIDP data and internal dose rates based on the average CAU 370 soil sample results plus or minus two standard deviations were compared and plotted in Figure 6. Internal dose rates were calculated for both data sets using the draft DCGs established for CAU 370. The error bars shown in Figure 6 are associated with the four CAU 370 sample results collected in each sample plot and were calculated at the 95-percent confidence level.

The maximum difference between internal dose rates calculated from RIDP values and from average CAU 370 soil sample results is 0.8 mrem/yr. The difference was calculated at the 95-percent confidence level based on two results. This difference is insignificant at the relatively low contaminant levels found at CAU 370 and is more than accounted for by the error associated with the average soil sample value itself.

The deviation (95-percent confidence) of internal dose rate based on CAU 370 soil samples alone was also calculated. A maximum deviation of 2 mrem/yr occurred at sample plot M. While the uncertainty in the average internal dose rate based on CAU 370 soil samples is higher than the difference between internal dose rates calculated from RIDP values and from average CAU 370 soil samples, it cannot be concluded that RIDP provides a more precise measure of internal dose. It can only be concluded that using converted RIDP values to determine internal dose rates is as accurate as using average CAU 370 soil sample values across the sample plots at CAU 370.

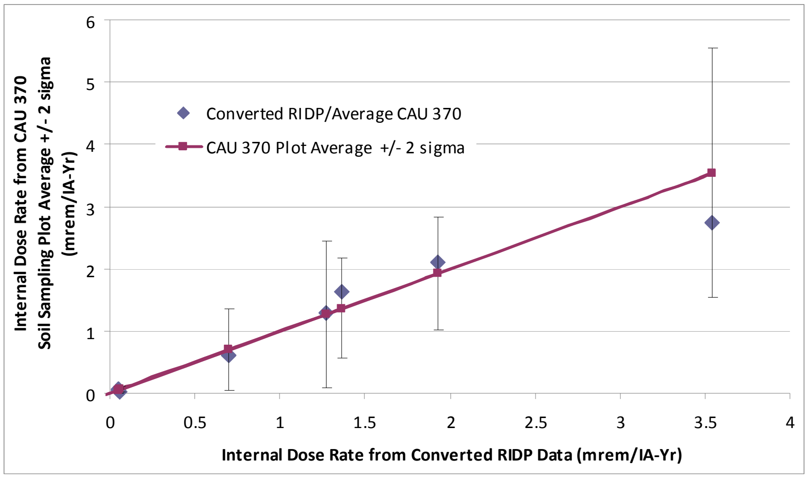

Fig. 6. Comparison of Internal Dose Rates.

The term missing from this evaluation is a propagated RIDP error based on uncertainties in RIDP measurements, IRLs, and RIDP ratios. Calculating such an error is outside the scope of this paper and may not be possible based on available data (i.e., historical RIDP data and CAU 370 data); however, such a calculation may not be needed to effectively use the RIDP data for guiding decision making.

Am-241, Pu-238, and Pu-239 comprise the greatest component of internal dose rate. Complete analysis of internal dose rate requires an evaluation of contribution from other radionuclides, such as europium (Eu)-152, Eu-154, strontium (Sr)-90 and cesium (Cs)-137. Direct comparisons between converted RIDP measurements and CAU 370 average soil sample results were completed. The results showed poor correlation for Sr-90, reasonable correlation for Cs-137, and some clear outliers for Eu-152 and Eu-154.

There was also variability in CAU 370 soil sample results across sample plots. The sources of these differences are the same as those noted above for Am-241, Pu-238, and Pu-239. Sr-90 differences also result from a technically invalid assumption on the part of the RIDP that a ratio between Cs-137 and Sr-90 would yield accurate Sr-90 concentration values.

Eu-152 is a soil activation product produced when neutrons from nuclear fission interact with soil. As such, Eu-152 contamination extends to greater depths than other radionuclides and in some cases increases with depth. Eu-152 contaminant distribution with depth is more variable than Am-241, Pu-238, and Pu-239, which also causes greater differences in the direct comparisons of converted RIDP data and CAU 370 soil sample data for these radionuclides.

The sources of these differences are related to uncertainty in both RIDP data and CAU 370 soil sample results. The differences between the two characterization methods are expected. However, errors in the data sets and differences between the two data sets are inconsequential to the calculation of internal dose rates for these radionuclides.

To support this conclusion, the following analysis was completed:

- The internal dose rate resulting from these radionuclides was calculated using converted RIDP data and CAU 370 average soil sample results. This calculation was based on the draft CAU 370 DCGs for internal dose.
- The maximum difference between these internal dose rate values was determined.
- The internal dose rate resulting from these radionuclides was determined for all RIDP measurements across the NTS, and the maximum was selected. This calculation was based on the draft CAU 370 DCGs for internal dose.
- It was assumed that the maximum percentage difference determined at CAU 370 would apply to the highest internal dose rate identified across the NTS.
- The resulting worst-case internal dose rate resulting from these radionuclides was 0.5 mrem/yr.

The differences between the converted RIDP values and the CAU 370 soil sample results could be investigated, and the sources of the differences could be identified and roughly quantified. However, the above analysis clearly demonstrates that additional research would not add value because these radionuclides do not contribute to internal dose rates. Therefore, accurate measurements are not needed for these radionuclides.

Internal and external dose rates were measured and calculated separately and by two different methods during the CAU 370 CAI. External dose rates were determined during the CAU 370 CAI using TLDs. TLDs measured the total exposure over the time period they were placed in the field. The data were then converted to units of mrem/yr to determine external dose rate for CAU 370. RIDP data were evaluated to determine accuracy in determining external dose rate through a number of methods that yielded varying results. Table II lists the CAU 370 TLD measurements and external dose rates calculated several ways using RIDP data and other CAU 370 data.

Table II demonstrates that each approach to calculating external dose rates has strengths and weaknesses. The following three comparisons were made to evaluate the effectiveness of various approaches to use the RIDP data:

1) The first set of comparisons uses calculated dose rates based on activity per unit mass (pCi/g). The CAU 370 soil sample data were converted to external dose rates in units of mrem/yr using the conversion factors in the RESRAD code. These values were compared to the CAU 370 TLD-measured dose rates, and the comparison is shown in Figure 7.

   The RIDP data were converted to units of pCi/g and then converted to external dose rates in units of mrem/yr using the conversion factors in the RESRAD code. These values were also compared to the CAU 370 TLD-measured dose rates, and the comparison is shown in Figure 8.

   Figures 7 and 8 show good comparison between the calculated and measured dose rates. The RIDP results show a slightly closer correlation to the TLD measurements than the CAU 370 soil sample results. The two data sets indicate that at least one TLD result may be biased high.

2) The second set of comparisons uses external dose rates calculated directly from RIDP data in units of activity per unit area (nCi/m^2) without first converting the RIDP data to units of pCi/g. Figures 9 and 10 compare the CAU 370 TLD-measured dose rates to the calculated external dose rates. This comparison was done using two approaches. The first approach used conversion factors from Beck calculated for exponentially distributed radionuclides in soil [5]. The Beck factors allow for a direct conversion from the RIDP data reported in nCi/m^2 to exposure rates without the use of the RESRAD code or other modeling codes. Exposure rates were converted to dose rates using the following conversion factors:

   - Exposure to dose-in-air: 0.87 rad/R [7]
   - Dose-in-air to effective dose equivalent: 0.7 rem/rad [7]

   Figure 9 illustrates the comparison of CAU 370 TLD-measured dose rates to the RIDP data converted using the Beck coefficients. The data are well correlated. This approach to external dose rate calculation also indicates that at least one TLD result may be biased high. While the external dose rates calculated using the Beck coefficients are not as strongly correlated as the values calculated using converted RIDP values and FGR 12 dose rate conversion factors, they yield closer absolute value comparisons. The data show that using the Beck coefficients is the better approach. This is because the Beck coefficients were specifically derived for soil contamination at nuclear weapons test sites and account for the exponential contaminant distribution with depth, which the FGR 12 values do not.

   The second approach used the dose-based limits derived by Anspaugh and Daniels [6]. Figure 10 compares the CAU 370 TLD-measured dose rates to the RIDP data using this approach. The data are highly correlated, but the calculated external dose rates are higher than the TLD-measured dose rates. This is due to the intentional conservatism in the Anspaugh and Daniels dose-based limits. In addition, this method did not easily allow the internal dose component from gamma-emitting radionuclides to be removed. These factors cause a high bias in the calculated data. Given this high bias, external dose rates based on the Anspaugh and Daniels limits that are currently available should be used as a screening and planning tool only.

3) The third set of comparisons made to evaluate the effectiveness of the RIDP data uses external dose rates calculated directly from RIDP data using the Beck coefficients. These values are compared to the results of direct-reading field surveys conducted during the CAU 370 CAI and to the 1994 aerial radiological survey data.

   Figure 11 compares the RIDP data to the CAU 370 direct-reading field survey data. Figure 12 compares the RIDP data to the 1994 aerial radiological survey data. Both the CAU 370 direct-reading field survey data and the aerial radiological survey data show good correlation to the external dose rate values calculated using RIDP measurements directly and applying Beck's exposure conversion factors.

Table II. Summary Comparison of External Dose Rate Results

			Calculated Dose Rates Based on Activity per Unit Mass		Calculated Dose Rates Based on RIDP Measurements Directly		Calculated Dose Rates Based on Direct Reading Instrumentation	
RIDP ID[a]	CAU 370 Sample Plot ID[b]	CAU 370 External Dose Rate TLD[c] (mrem/yr)	CAU 370 Soil Samples[d] (mrem/yr)	Converted RIDP[e] (mrem/yr)	Direct RIDP (Beck)[f] (mrem/yr)	Direct RIDP (A&D)[g] (mrem/yr)	1994 Aerial Survey[h] (mrem/yr)	CAU 370 Field Survey[i] (mrem/yr)
KE0003	A	**173**	257	239	188	447	343	149
KE0008	C	**132**	171	173	135	316	206	81
KE0009	E	**10**	16	14	13	33	62	14
KE0014	J	**10**	11	16	14	31	62	2
KE0020	P	**21**	12	15	13	32	62	14
KE0021	M	**153**	183	169	125	276	206	104
KE0052	F	**141**	206	153	120	283	343	104
KE0053	H	**166**	144	141	109	255	206	104

[a] RIDP ID: Location ID of the RIDP in-situ measurement
[b] CAU 370 Sample Plot ID: CAU 370 CAI soil sample plot. A large plot was used in an attempt to replicate the large field of view (sample size) acquired by the RIDP in-situ measurements.
[c] CAU 370 External Dose Rate TLD: The dose rate calculated using the TLDs hung at each sample plot. This value was used to calculate external dose rate for CAU 370.
[d] CAU 370 Soil Samples: The dose rate calculated using the CAU 370 soil sample results and the dose conversion factors in Federal Guidance Report (FGR) 12, the same factors used in The RESRAD code.
[e] Converted RIDP: The dose rate calculated using RIDP values converted to pCi/g and the dose conversion factors in FGR 12.
[f] Direct RIDP (Beck): The dose rate calculated using the RIDP measurements directly and the dose conversion factors developed by Beck [5].
[g] Direct RIDP (A&D): The dose rate calculated using the RIDP measurements directly and the dose-based limits in Anspaugh and Daniels [6].
[h] 1994 Aerial Survey: The dose rates calculated using the aerial survey values converted to and normalized to 2,250 hours for the industrial use scenario.
[i] CAU 370 Field Survey: The dose rate measured with field instrumentation during the CAU 370 CAI.

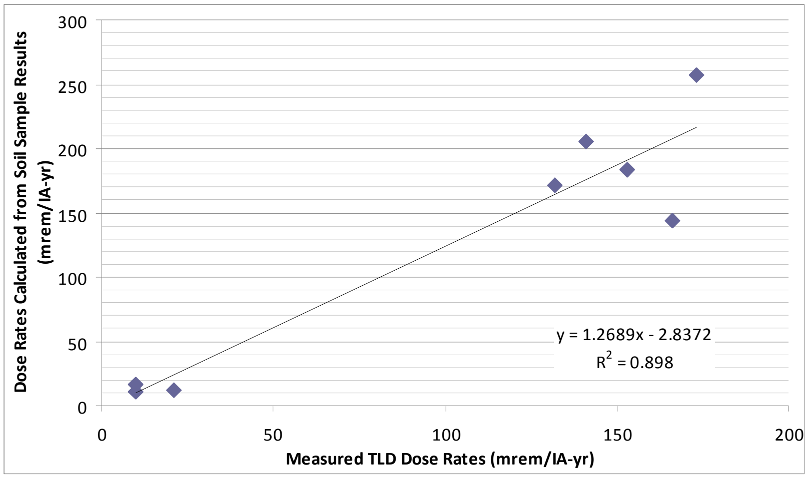

Fig. 7. CAU 370 TLD Results Compared to CAU 370 Soil Sample Results.

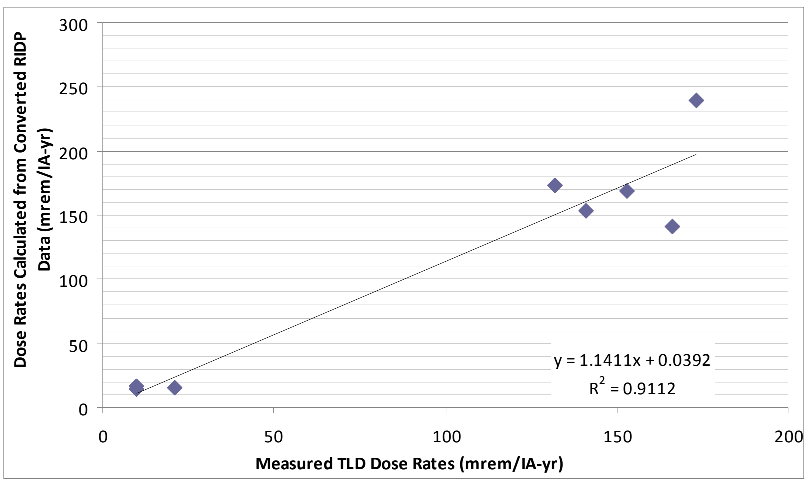

Fig. 8. CAU 370 TLD Results Compared to Converted RIDP Data.

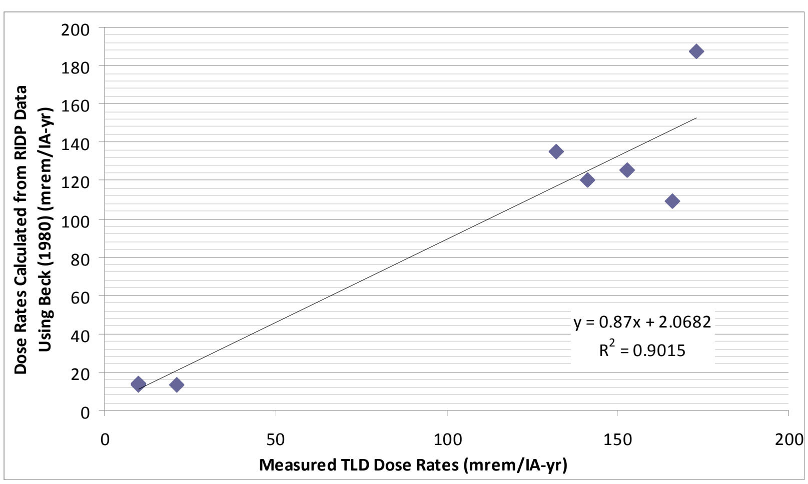

Fig. 9. CAU 370 TLD Results Compared to RIDP Data Using Beck Conversion Factors.

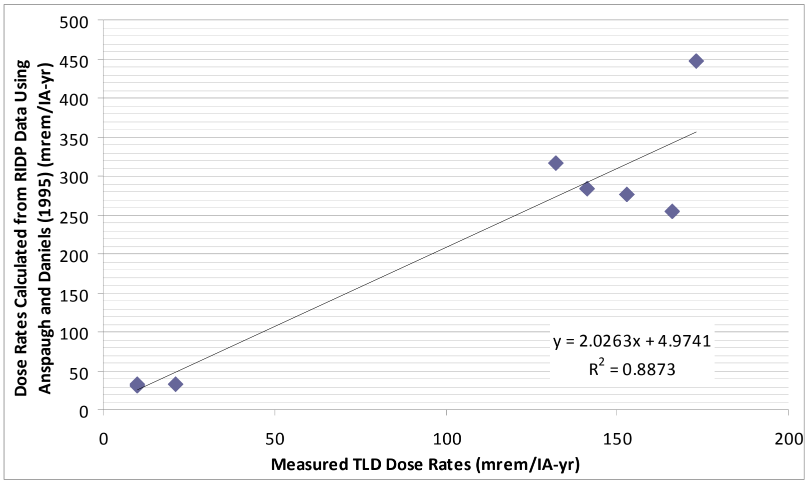

Fig. 10. CAU 370 TLD Results Compared to RIDP Data Using Anspaugh and Daniels Dose-Based Limits.

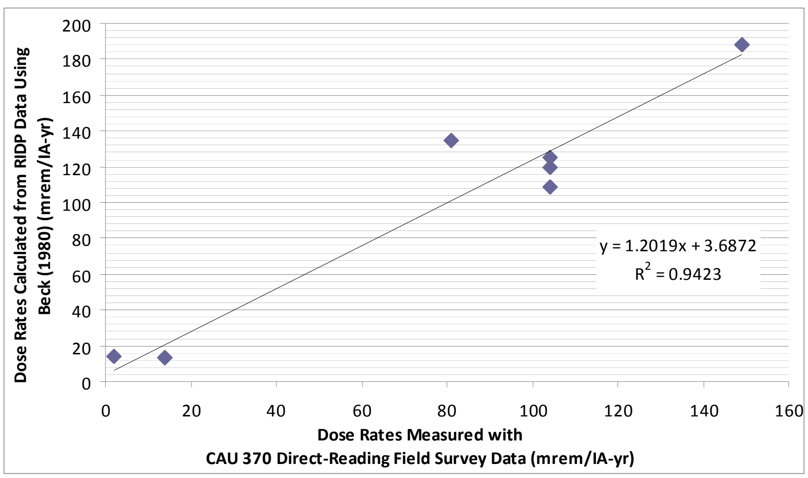

Fig. 11. CAU 370 Direct-Reading Field Survey Results Compared to RIDP Data Using Beck Conversion Factors.

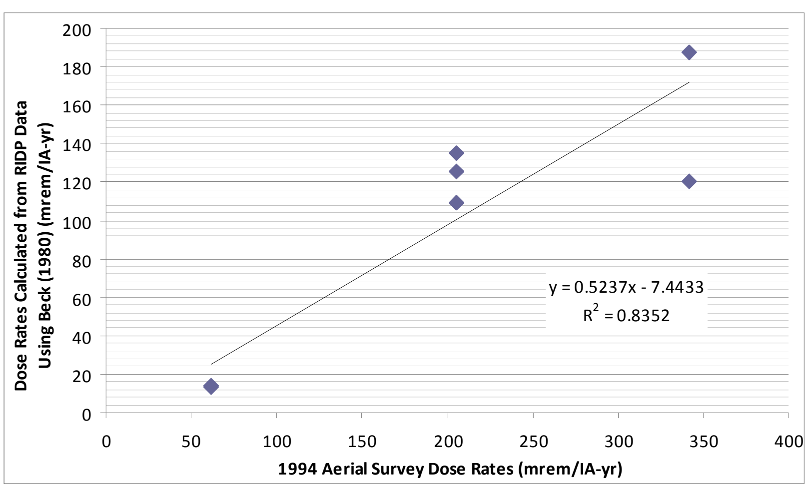

Fig. 12. 1994 Aerial Survey Results Compared to RIDP Data Using Beck Conversion Factors.

The three data comparisons presented above demonstrate that the RIDP data can be used to accurately determine external dose rates. Additional sampling at CAU 370 did not yield better results than the RIDP data. In fact, the RIDP data converted to pCi/g yielded a stronger comparison to the TLD data than the actual CAU 370 soil sample data; however, neither data set yielded equivalent results. Both sets of results were higher than the TLD values.

The RIDP values used directly to calculate external dose rates using Beck's coefficients provided the best correlation to the TLD data of any of the data comparisons and scored well on a paired t-test (a statistical test used to determine differences between data sets). The results presented above show that the use of RIDP data to determine external dose rates at CAU 370 yield reasonably accurate results.

The largest differences between external dose rates calculated with RIDP and TLD measurements occurred at locations where TLDs measured dose rates greater than 25 mrem/yr. The IRL at these locations is also 0.05 cm^{-1}. An IRL of 0.05 cm^{-1} represents the far end of the exponential depth distribution assumption, and deviations in actual depth distribution from calculated depth distribution produce larger errors in dose rates in this region of the curve.

To further investigate this error and its implications to use of the RIDP data for calculation of external dose rates, comparisons between TLD and RIDP data were made where RIDP measurements were relatively close to TLD monitoring locations. This evaluation indicated that most of the larger deviations between these two values occurred at locations where the applied IRL is 0.05 cm^{-1}. Understanding the source of RIDP error relative to TLD values will allow quantification of the error and a method to mitigate the risk associated with the error.

The final assessment completed was to determine the 25-mrem/yr dose rate boundary using RIDP values and compare it to the 25-mrem/yr boundary determined with the results of the CAU 370 CAI. The RIDP 25-mrem/yr boundary was established by using direct RIDP measurements and the Beck coefficients. The internal dose rate was determined using RIDP values converted to pCi/g and the draft DCGs established for CAU 370. As Figure 13 shows, the location of the 25-mrem/yr boundary is the same using either data set.

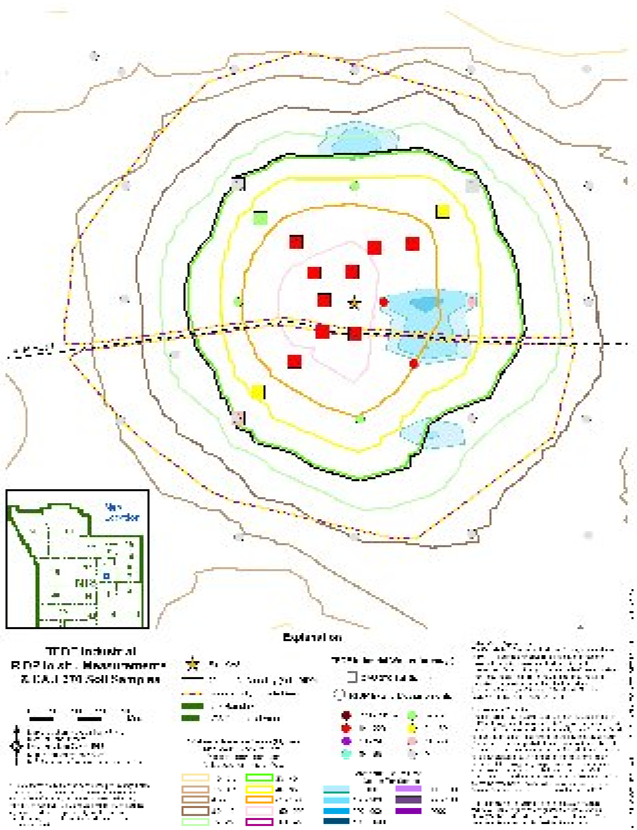

Fig. 13. RIDP 25-mrem/yr Dose Boundary for CAU 370.

DATA CAUTIONS AND USE

RIDP data are currently usable, but the following cautions on RIDP data use have been identified. These cautions can be managed through conservative assumptions and some additional research or limited characterization:

- Some ratios of Pu-238 and Pu-239/240 to Am-241 are higher than expected, such as for measurements collected in the WILSON area. However, these high ratios, if incorrect, would result in deriving higher than actual activities, so the results using the existing ratios would be conservative.
- Applying the ratios of Sr-90 to Cs-137 to calculate aged Sr-90 contamination is not reliable. The ratios of Pu-238 and Pu-239/240 to Am-241 are based on the physics of radioactive decay, but the ratios of Sr-90 to Cs-137 are not. These two radionuclides are produced independently of each other and are not in the same decay chain. This means that different ratios would be expected at different sites, with little if any process knowledge to evaluate the ratios. Additionally, Sr-90 is more mobile in the environment than Cs-137, resulting in additional potential changes in the ratio over time. However, Sr-90 is not a contributor to internal dose rate, even under the most conservative assumptions. Because Sr-90 does not contribute to internal dose at NTS sites, an accurate measurement is not needed.
- Some site cleanup carried out under the Waste Consolidation Project (WCP) was performed concurrently with the RIDP, and RIDP measurements were collected before and after cleanup activities at several sites. The RIDP did not identify appreciably different values before and after these cleanup activities were performed. RIDP may not have detected significant differences because the scope of the clean up only included minimal amounts of slightly contaminated debris or soil hot spots that were not near RIDP sampling locations. However, because clean up activities occurred after the RIDP, sites that were cleaned up under the WCP should be noted and the RIDP data evaluated. Sites where cleanup occurred after the RIDP can be flagged for additional data cautions.
- RIDP data cannot capture recent contaminant migration. Sites where migration might be an issue may require additional, focused characterization in the known migration channels.

RIDP data are currently usable for a number of applications given the level of QA reviews performed to date. Enhanced uses of RIDP data may be possible with additional calculations and verification.

Project Planning: Evaluations of RIDP data indicate it may be used for project planning without additional field verification. Planning activities may include estimating 25-mrem/yr boundaries, optimizing characterization efforts, projecting end states, and planning remedial actions. Figure 14 provides an example of the 25-mrem/yr boundary at the GALILEO site. This example highlights a secondary plume that does not follow the pattern observed at CAU 370 of decreasing dose rate with distance from ground zero and indicates that the secondary plume may require corrective actions. The 25-mrem/yr dose rate boundaries have been estimated for all areas of the NTS. Figure 15 provides an example of how RIDP data may be used to optimize characterization efforts.

Augmenting TLD Data: Approximately 90 percent of the dose rate at CAU 370 is due to external gamma exposure from radionuclides with short half-lives relative to Am-241 and Pu-239. Data collection at CAU 370 used TLDs to determine external dose rates. TLD data cannot be easily corrected for decay. RIDP data can be used to identify specific radionuclide distributions, thus augmenting the CAU 370 data set to allow for decay corrections and dose rate projections for any future date. Less risk will be involved for closure at sites where the dose rate and 25-mrem/yr boundary is steadily decreasing. This approach provides a way to show that the TLD or calculated RIDP dose rate errors steadily decrease over time.

Calculating Dose Rates and Estimating the 25-mrem/yr Boundaries: The RIDP data collected in the CAU 370 area provide reasonably accurate dose rate values for both internal and external dose rates. The RIDP data lead to the same decisions as the newly collected data for CAU 370; therefore, the RIDP data should be used to estimate the 25-mrem/yr boundaries during the DQO process. The estimated boundaries can potentially limit additional characterization requirements. Figure 15 outlines this characterization optimization.

Internal Dose Rate Estimates: There is error associated with internal dose rate estimates based on both RIDP data and soil samples. Internal dose rates calculated using converted RIDP values are likely accurate within the inherent deviation in soil sample plots. This is expected due to the variation between soil sample values. Using the process outlined in this paper, internal dose rates at all RIDP locations were calculated. A query was run to determine how many RIDP locations exceeded 25 mrem/yr with an internal dose component greater than 25 percent of the total dose, or 6.25 mrem/yr. Only 47 RIDP locations resulted from this query. The maximum difference between CAU 370 dose rates calculated with soil samples and calculated with converted RIDP values was 50 percent. This error was applied to the 47 values to gain a rough idea of the increase in the size of corrective action boundaries using the conservative application of potential error. The adjusted values only resulted in 16 additional points exceeding 25 mrem/yr with an internal dose component greater than 6.25 mrem/yr. This informal and brief evaluation highlights that conservative assumptions may be applied with little impact to the size of corrective action boundaries. Using this type of conservative estimate may allow these areas to be characterized with limited additional sampling. This approach would be presented during the DQO process. The outcome of using the RIDP data would reduce worker risk associated with sampling in contaminated areas, accelerate schedules, and reduce project costs.

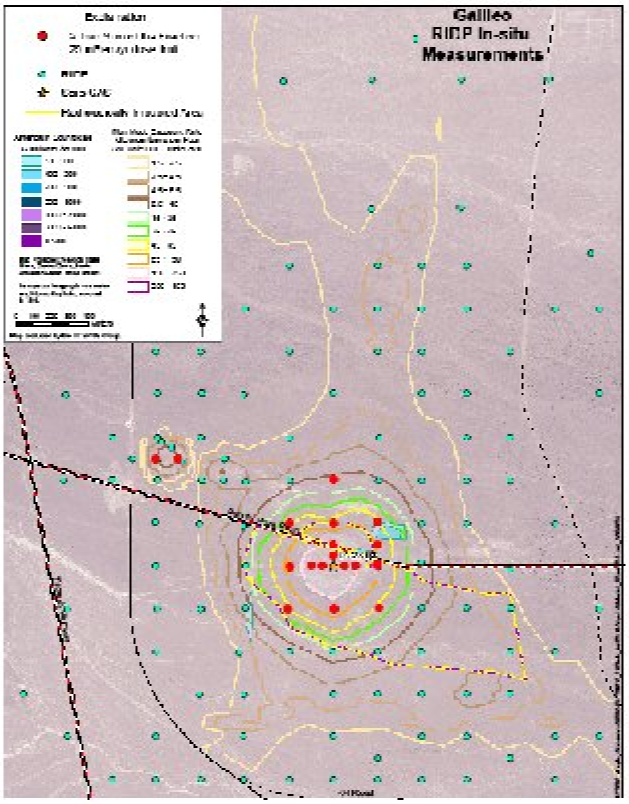

Fig. 14. Example of Estimated 25-mrem/yr Dose Rate Boundaries.

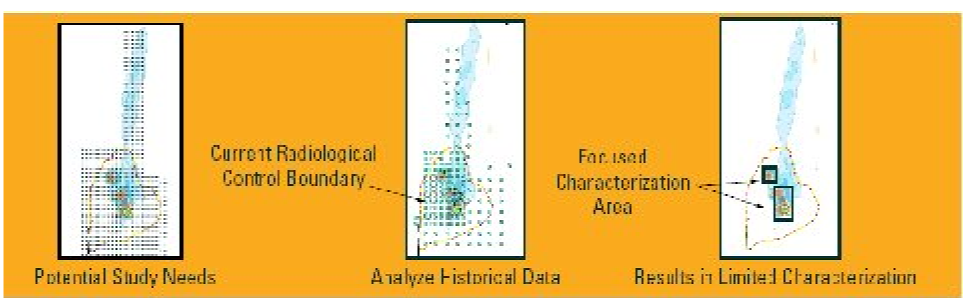

Fig. 15. Using Historical Data to Optimize Characterization.

External Dose Estimates: There is error associated with estimating external dose rates based on TLDs and the RIDP data. RIDP values associated with IRLs of 0.05 cm^{-1} can lead to differences between TLD measurements and RIDP-calculated dose rates in some areas. The risk associated with this potential error can be limited by selecting a more conservative Beck coefficient. Many of these locations are at or near ground zero locations, so using a more conservative coefficient to determine dose is unlikely to increase the 25-mrem/yr dose rate boundaries.

SAFER Plan Closure: Estimated dose rates for each RIDP measurement location have been calculated in support of this paper using the process described in this paper. This information was used to determine Corrective Action Site (CAS) areas that did not exceed a dose rate of 25 mrem/yr. Approximately 20 CAS areas do not have RIDP locations with calculated dose rates greater than 25 mrem/yr. This information can be used to select candidate sites that may be appropriate for closure under the SAFER process. The evaluation of these CASs for inclusion into a proposed SAFER closure will consider similarities to CAU 370, proximity to operating facilities, whether migration is likely, and existing fencing or posting. After the CASs are evaluated against these criteria, DQOs will be prepared that include an evaluation of the RIDP data and a determination of any additional data needs.

CONCLUSIONS AND RECOMMENDATIONS

The RIDP data can be used to estimate the 25 mrem/yr boundary. Uncertainties in the RIDP data are known and associated uncertainty values can be conservatively calculated to expand those boundaries thus ensuring the areas with the potential to cause an NTS worker dose of 25 mrem/yr are appropriately addressed. Conservatively calculating these boundaries does not adversely affect future land use. The conservatively estimated boundaries still fall within the aerial survey plumes and future facilities would be located outside these plumes regardless of the closure decisions made by the Soils Sub-project.

The NTS is an ideal site for historical data recovery efforts. This is due to the fact that long term institutional control is assumed (no future public land use) and there are no public receptors within close proximity of the sites. As such, it is appropriate to fully leverage historical data, apply uncertainty values to that data, perform minimal boundary verification when needed, characterize potential migration paths outside the estimated boundary and use the data for final site closure.

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