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RF/ER-96-0048 UN

**SUMMARY OF
EXISTING DATA ON
ACTINIDE MIGRATION AT THE
ROCKY FLATS ENVIRONMENTAL
TECHNOLOGY SITE**

DRAFT

August, 1997

**Prepared by
Rocky Mountain Remediation Services L L C
Environmental Restoration**

for

**U S Department of Energy
Rocky Flats Environmental Technology Site
Golden Colorado**

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ACRONYMS

ALF	Action Level and Framework
Am	americium
BGCR	Background Geochemical Characterization Report
BSCP	Background Soils Characterization Program
BZ	Buffer Zone
CDPHE	Colorado Department of Public Health and the Environment
CEC	cation-exchange capacity
cfm	cubic feet per minute
COC	chemicals of concern
D&D	Decontamination and Decommissioning
DCG	derived concentration guide
DOE	Department of Energy
GIS	geographical information system
HPGe	High Purity Germanium
HSPF	Hydrological Simulation Program FORTRAN
IA	Industrial Area
IM/IRA	Interim Measure/Interim Remedial Action
IMP	Integrated Monitoring Plan
ITS	interceptor trench system
LHSU	Lower Hydrostratigraphic Unit
MUSCLE	Modified USLE Model
NEA	European Nuclear Energy Agency
NORM	natural organic matter
NSD	New Source Detection
NWC	North Walnut Creek
POE	Point of Evaluation
Pu	plutonium
RAAMP	Radioactive Ambient Air Monitoring Program
RFCA	Rocky Flats Cleanup Agreement
RFEDS	Rocky Flats Environmental Database System
RFETS	Rocky Flats Environmental Technology Site
RFFO	Rocky Flats Field Office
RFI/RI	RCRA Facility Investigation/Remedial Investigation
RUSLE	Revised USLE Model
SCSCR	Soil Conservation Service Curve Number
SID	South Interceptor Ditch
SWC	South Walnut Creek
SWMM	Storm Water Management Model
SRM	Sediment Removal Model
TIMS	thermal ionization mass spectrometry

TSP	total suspended particulates
TSS	Total Suspended Solids
U	uranium
UHSU	Upper Hydrostratigraphic Unit
USLE	Universal Soil Loss Equation
WWTP	Waste Water Treatment Plan

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1 0 INTRODUCTION

The evaluation of remedial alternatives for actinides (herein considered as plutonium [Pu] americium [Am] and uranium [U] isotopes) at the Rocky Flats Environmental Technology Site (RFETS) (the "Site") must consider migration and mobility (i.e. transport) along environmental pathways. Transport pathways include soil (surficial and subsurface) groundwater surface water and air. During 1996 an Actinide Migration Expert Panel was convened by the Department of Energy Rocky Flats Field Office (DOE RFFO). The Panel which consisted of nationally recognized specialists on actinide chemical behavior and migration/mobility in the environment reviewed existing data on actinide migration at RFETS and made recommendations for further work. Their recommendations included 1) develop a defensible conceptual model for actinide transport based on a thorough understanding of chemical and physical processes 2) investigate the long term impacts of actinide geochemical mobility on remedial requirements and 3) evaluate the long term protectiveness of the Rocky Flats Cleanup Agreement (RFCA) Attachment 5 Action Levels and Standards Framework for Surface Water Groundwater and Soils (DOE 1996a) soil action levels on surface water quality.

This report presents a summary of existing data that has been initiated to further the understanding of the conceptual model as recommended by the Panel. An update to the *Development of Conceptual Model of Potential Pathways for Migration of Actinides Uranium Plutonium and Americium at RFETS* (DOE 1997a) is developed in this report to better understand the relationship among the transport pathways and the potential effects of actinides on surface water and air quality.

1 1 PURPOSE AND OBJECTIVES

Over the past two decades routine monitoring and studies of environmental media at RFETS have shown that site derived actinides have been released and are present in the environmental media at RFETS. DOE RFFO and the Kaiser Hill Team have initiated a

multi year Actinide Migration Study to review and summarize the diverse body of existing data and provide additional data on actinides in the Site's environment as required for remedial action. This work supports the Site's closure plan *Accelerating Cleanup Focus on 2006* (DOE 1997b) that was developed to achieve accelerated risk reduction and significant cleanup of the Site by calendar year 2006.

The purpose of this report is to develop a better understanding of the chemical and physical mechanisms of actinide mobility and potential impacts to surface water by identifying existing knowledge on actinide behavior and distribution in soils, groundwater, and surface water to gain a better understanding of the actinide migration conceptual model (DOE 1997a) and identify additional data required for remedial action. This information will better the understanding of the conceptual model and will assist in designing remedial actions.

1.2 SCOPE OF THE REPORT

This report supports the mission and goals of the Actinide Migration Study by presenting actinide data compiled from the Rocky Flats Environmental Database System (RFEDS), RCRA Facility Investigation/Remedial Investigation (RFI/RI) reports, Interim Measure/Interim Remedial Action (IM/IRA) decision documents, and the results of other reports and studies on the behavior of actinides in the RFETS environment. The current state of knowledge is summarized using available information with respect to the occurrence and migration of actinides in surface soil, subsurface soil, sediment, groundwater, surface water, and air at RFETS. The data deficiencies are identified and recommendations for future actions are made.

1.3 REGULATORY FRAMEWORK

The RFCA Attachment 5 Action Levels and Standards Framework for Surface Water, Groundwater, and Soils (DOE 1996a) sets forth standards and action levels for

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environmental media. These standards and action levels incorporate the RFCA Vision and land and water use controls in the RFETS cleanup. Actinide surface water standards (0.15 pCi/L each for plutonium and americium) are based upon human health risk as a consequence of human ingestion of surface water and are assigned at compliance points as water leaves the site so that the surface water is of acceptable quality for all designated uses. The Site's water quality standard for uranium is currently set at 10 pCi/L based on ambient levels in Site surface waters. The State of Colorado has suspended consideration of a statewide health based standard until the Environmental Protection Agency promulgates a maximum contaminant level for uranium in drinking water. The soil action levels for radionuclides are based upon radiation dose limits for designated land use scenarios for RFETS. The soil action levels did not consider the transport of soil containing actinides to surface water because no consumption of groundwater or surface water was assumed when the soil action levels were calculated. The soil radionuclide action levels must be assessed to be sure they are protective of surface water.

The RFCA states that "protection of surface water uses with respect to the long term Site condition will be the basis for making soil and groundwater remediation and management decisions and that additional groundwater or soil remediation or management may be required for the protection of surface water quality or ecological resources." This report reviews and summarizes the diverse body of existing data on actinides in the RFETS environment and identifies existing deficiencies in knowledge of actinide behavior in soils, groundwater, and surface water. The *Proposed Path Forward for Actinide Migration* (DOE 1997b) proposes actions that will facilitate the development of the conceptual model providing a better understanding of the relationship of actinide levels in soil and the effects of remedial activities on the long term protection of surface water.

Development of a conceptual model is an iterative process and the model is used to define further data gaps including additional analysis of existing data. This report presents data summaries for actinides in surface soil, subsurface soil, surface water and groundwater. These data summaries are incorporated into developing a basic conceptual model of

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actinide contamination events the resulting distribution and subsequent transport of actinides at RFETS Further refinement of the conceptual model will occur in fiscal years 1998 and 1999 Much of this further refinement will be performed by maximizing the use of existing analytical data For example the association of actinides with other elements and with other measured parameters such as pH will be evaluated Data will be evaluated temporally to examine changes in actinide content and associated parameters over time as related to specific sources

1 4 REFERENCES

DOE 1996a *Rocky Flats Cleanup Agreement* June 27 1996

DOE 1997a *Accelerating Cleanup Focus on 2006* June 1997

DOE 1997b *Proposed Path Forward for the Actinide Migration Studies* June 1997

2 0 CLIMATOLOGIC GEOLOGIC AND HYDROLOGIC SETTING OF RFETS

The geologic hydrologic and climatologic setting at RFETS is briefly described in the following sections relative to the occurrence and transport pathways of actinides in the environment. Environmental conditions influence the mode, rate, and direction of contaminant transport and, to a large extent, determine the chemical fate of contaminants in the environment. Chemical and physical interaction of various environmental media can play an important role in mobilizing or demobilizing the spread of contamination, which must be understood in order to evaluate and design appropriate remedial actions. For additional discussion of these media, the reader should consult the 1994 RFETS *Environmental Report* (Kaiser Hill 1995) and the 1995 RFETS *Sitewide Geoscience Characterization Study* series of reports, including the *Geologic Characterization Report* (EG&G 1995b), *Hydrogeologic Characterization Report* (EG&G 1995c), and *Groundwater Geochemistry Report* (EG&G 1995a). A discussion of potential actinide transport pathways and their interaction is further described in Section 9.

2 1 CLIMATOLOGY

Basic climatological information (including general descriptions of the climate, precipitation, temperatures, and wind patterns) is pertinent to understanding the hydrologic setting and actinide migration potential of environmental media at RFETS. The local climate at RFETS exerts a strong influence on soil, groundwater, and surface water conditions. For example, precipitation amount, frequency, intensity, and seasonality (combined with air temperatures, humidity, and wind conditions) influence the potential for wind and soil erosion, groundwater recharge, and evapotranspiration. These parameters will be discussed briefly to provide background information for the hydrology sections of the report. A more detailed summary of the climatology and meteorology at the Site is provided in *RFETS Environmental Report* (Kaiser Hill 1995).

The RFETS area has a semi arid climate that is characteristic of much of the central Rocky Mountain region. Table 2.1 summarizes climatic data for the Site collected in 1994. Historically, annual precipitation at Rocky Flats averages nearly 15.5 inches, with about 42% of the annual precipitation falling during the spring season (April through June) (EG&G 1995c). Precipitation falls primarily as snow from late October through early April and as rain during the remaining months. The Site is less prone to severe weather activity than the plains to the east because of its locality close to the foothills. Thunderstorms initiated over the foothills usually intensify as they move eastward into the more humid and unstable air mass over the eastern plains of Colorado. Tornado formation over RFETS is very unlikely (Kaiser Hill 1995).

The thin, dry atmosphere results in wide diurnal temperature ranges, with strong daytime warming and nighttime cooling. Temperatures are generally moderate, but short periods of exceptionally warm or cold weather resulting from incursions of hot southwestern desert or frigid Canadian air masses are possible during the mid summer and mid winter months. Temperature ranges, averages, and extremes for the Site in 1994 are presented in Table 2.1 (Kaiser Hill 1995).

High wind events, known as chinooks, are common along the Front Range during the winter and early spring months. Wind gusts will typically exceed 70 miles per hour (mph) several times in a normal year. In the strongest gusts, winds have exceeded 100 mph. Wind directions most frequently are from the west southwest through northerly directions, with the strongest gusts originating from the west northwest and west. Wind speeds above 18 mph occur primarily with westerly winds and, to a lesser extent, northerly winds. A summary of 1994 wind directions and wind speed frequencies measured at a 10 meter height at the Site is provided in Table 2.2 and is shown graphically by a wind rose diagram in Figure 2.1 (Kaiser Hill 1995).

Table 2 1 Summary of Meteorological Data for 1994

1994 Annual Climatic Summary									
Temperatures (°F)									
Month	Means			Extremes			Mean Dew Point (°F)	Mean. Rel Humidity (%)	
	High	Low	Average	High	Date	Low			Date
January	44.6	18.5	31.6	60.1	23	5.4	31	10.4	45.2
February	44.0	14.7	29.4	61.0	16	2.6	8	12.8	55.0
March	54.3	25.4	39.9	68.7	14	5.7	9	19.8	51.3
April	56.7	31.3	44.0	78.3	23	18.1	5	28.0	61.1
May	72.4	43.7	58.1	87.3	30	32.0	2	40.1	54.8
June	84.7	51.8	68.3	101.0	26	38.7	9	41.1	45.1
July	85.3	52.8	69.0	94.1	10	42.4	7.8	43.6	45.8
August	84.8	55.3	70.1	95.0	6	49.1	20	47.7	51.4
September	78.1	48.2	63.1	90.1	10	24.4	22	34.8	34.1
October	61.1	36.1	48.6	75.3	11	26.0	30	30.4	50.8
November	46.7	22.4	34.6	68.8	7	8.7	19	16.8	50.2
December	47.0	21.1	34.0	66.0	1	-4.0	31	11.7	39.7
Annual	63.3	35.1	49	101.0	6/26	5.4	1/31	28.1	48.7

Month	Wind Speed (mph)		Atmos. Pressure Mean (mb)	Solar Total kW/h/m
	Mean	Peak		
January	12.8	85.4	810.9	81.1
February	10.7	94.8	807.8	101.3
March	8.9	72.0	811.4	145.2
April	9.1	67.3	810.8	137.8
May	8.9	57.0	812.9	201.2
June	8.4	42.7	814.4	224.9
July		76.9	816.8	201.5
August	8.6	62.2	817.7	171.9
September	8.7	56.4	816.7	152.7
October	9.4	61.4	811.7	118.6
November	10.0	75.5	808.1	93.0
December	9.5	93.4	811.2	71.0
Annual	9.5	94.8	812.5	141.7

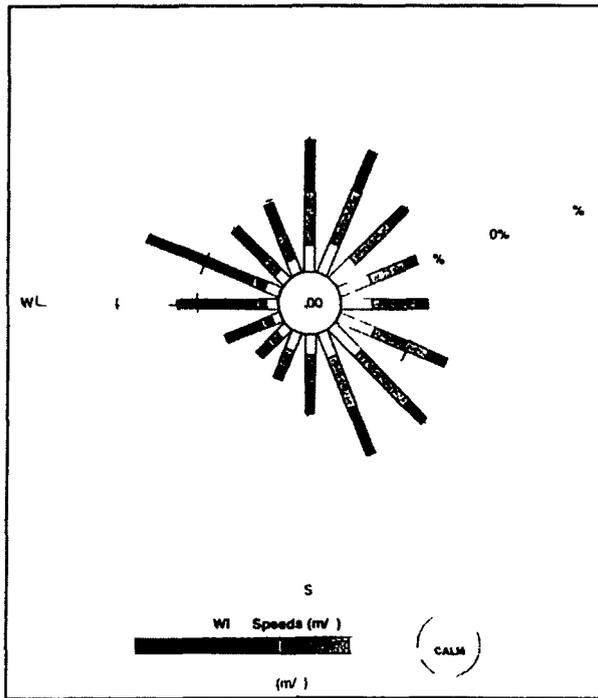
Month	Precipitation (inches)					Number of Days		
	Total	Daily Max	Date	15 Min Max	Snowfall Inches	Precip >0.10	Temp >90 F	Temp <32 F
January	0.45	0.2	26	0.0	7.5	0.0	0.0	30.0
February	0.76	0.4	28	0.0	9.4	3.0	0.0	27.0
March	1.05	0.4	28	0.0	11.5	4.0	0.0	24.0
April	2.46	0.5	25	0.1	24.5	7.0	0.0	18.0
May	1.37	0.4	13	0.2		5.0	0.0	1.0
June	1.12	0.4	22	0.1		4.0	9.0	0.0
July	0.4	0.1	31	0.0		1.0	9.0	0.0
August	1.51	0.6	10	0.3		5.0	8.0	0.0
September	0.68	0.5	21	0.1	5.0	2.0	2.0	2.0
October	0.96	0.4	17	0.1	3.0	2.0	0.0	7.0
November	1.08	0.4	8/13	0.2	18.0	4.0	0.0	27.0
December	0.16	0.0	6	0.0	11.5	0.0	0.0	29.0
Annual	12.0	0.6	8/10	0.3	90.4	37.0	28.0	165.0

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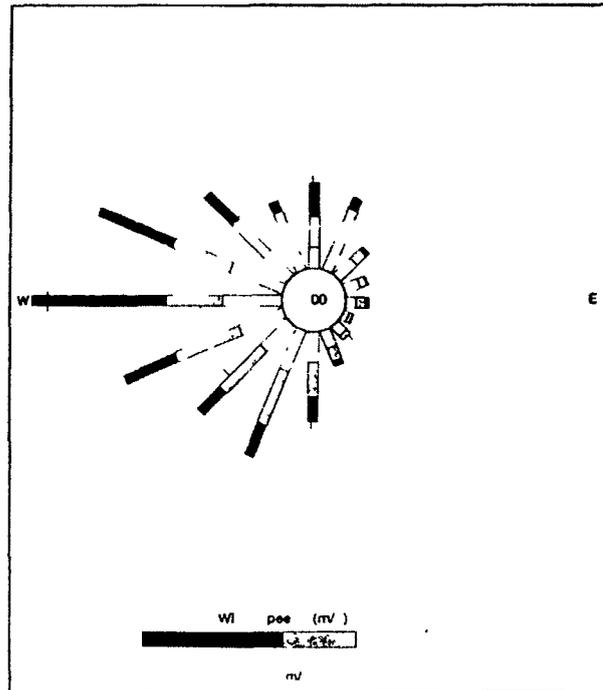
Table 2 2 Wind Data Summary for 1994

Site Wind Direction Frequency (Percent) by Four Wind Speed Classes						
(15 Minute Averages Annual 1994)						
	Calm <0.5 m/s (<1.1 mph)	0.5 2.5 m/s (1.1 5.6 mph)	2.5 4.0 m/s (5.6 9.0 mph)	4.0 8.0 m/s (9.0 18 mph)	>8.0 m/s (>18 mph)	Total %
	2.2					
N		1.55	2.60	2.57	0.15	6.87
NNE		1.63	2.21	1.55	0.14	5.53
NE		1.36	1.85	0.87	0.02	4.10
ENE		1.42	1.41	0.48	0.01	3.32
E		1.27	1.62	0.49	0.02	3.40
ESE		1.42	1.94	0.68	0.01	4.05
SE		1.66	2.3	0.82	0.02	4.82
SSE		1.40	2.11	1.70	0.22	5.43
S		1.59	2.03	1.54	0.13	5.29
SSW		1.67	2.08	1.52	0.10	5.37
SW		1.50	2.03	2.25	0.22	6.00
WSW		1.79	2.94	3.32	0.95	9.00
W		2.00	2.46	2.69	3.00	10.15
WNW		1.88	2.33	4.00	3.82	12.03
NW		1.49	2.00	2.35	0.53	6.37
NNW		1.35	2.21	2.51	0.10	6.17
TOTALS	2.20	24.98	34.14	29.34	9.44	100.00

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Rocky Flats 1994 Windrose Day



Rocky Flats 1994 Windrose Night

Figure 2 1 1994 Wind Speed and Direction

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2 2 GEOLOGY

At an elevation of 6 000 feet above mean sea level RFETS is located on the Colorado High Plains approximately two to six miles east of the Front Range mountain front. Geologic units at RFETS can be grouped into two general categories: unconsolidated Quaternary surficial deposits and underlying consolidated Cretaceous bedrock (EG&G 1995b). Figure 2 2 is a generalized geologic cross section that illustrates the surficial and bedrock materials within each group.

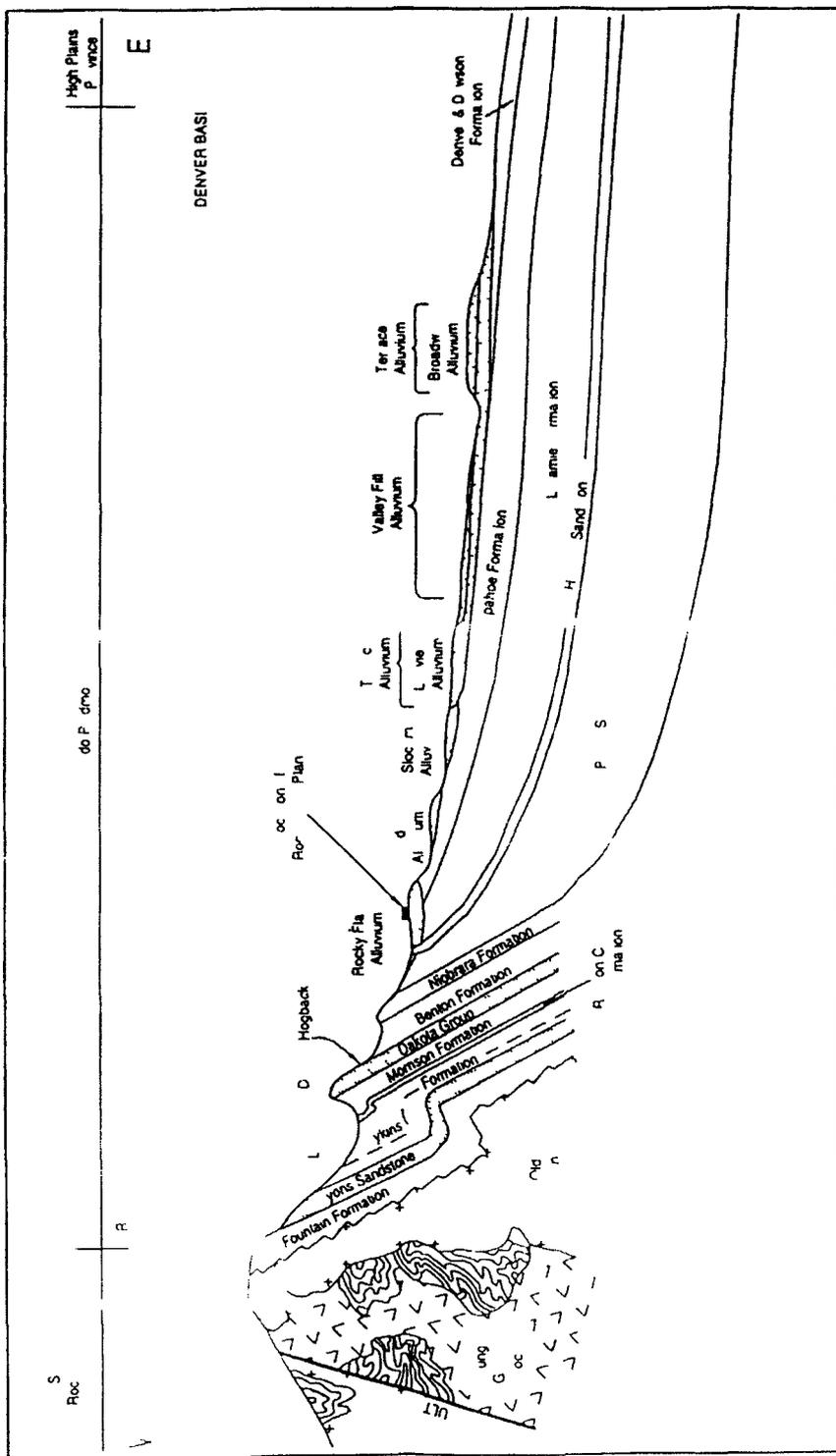
Prior to the deposition of Quaternary surficial deposits, the gently eastward-dipping Cretaceous age bedrock in the Rocky Flats area was subjected to erosion that produced a broad, flat erosional surface (a peneplain). During the Pleistocene, this bedrock surface was subsequently overlain by a blanket of alluvial fan deposits known as the Rocky Flats Alluvium that were derived from Coal Creek Canyon (EG&G 1995b). Headward erosion by westward progressing drainages has since incised both the Rocky Flats Alluvium and the underlying bedrock peneplain. Approximately half of the surface area covered by RFETS has been incised, removing the Rocky Flats Alluvium. In most areas, these erosional surfaces have been subsequently covered by stream sediments or colluvium. A summary of the stratigraphic profile is presented in the following sections. Younger units are described first, followed by progressively older, deeper units.

2 2 1 Unconsolidated Deposits

The surface of RFETS is covered almost entirely by an extensive layer of unconsolidated Quaternary deposits consisting of Holocene colluvium and valley fill alluvium and Pleistocene Rocky Flats Alluvium. Holocene slump and landslide material are also present locally on valley slopes and indicate that mass movement is an important, albeit slow, erosional agent in areas of RFETS.

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RF/ER 96-0048 UN
 Draft Summary of Existing Data on Actinide Migration
 at the Rocky Flats Environmental Technology Site



PREPARED FOR U.S. DEPARTMENT OF ENERGY OC FLATS ENVIRONMENTAL TECHNOLOGY SITE GOLDEN, COLORADO
FIGURE NORMALIZED EAST-WEST CROSS-SECTION FRONT RANGE TO DENVER

S B U C g or 883 W S 960

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Colluvial deposits which are present on the valley slopes in the central portion of RFETS were derived from geologic material exposed on the steep slopes and topographic highs and were formed by slope wash and downslope creep. The colluvium ranges in thickness from 0 to 20 feet with the thickest sequences occurring at the base of the valley slopes. The colluvium is composed of clay, clayey gravels, and lesser amounts of sand and silt. Slump and landslide deposits were derived from the colluvium and Rocky Flats Alluvium and are most common on valley slopes along the Rock Creek and Walnut Creek drainages in the northern portion of RFETS. Valley fill deposits were fluvially derived from upstream materials and consist of clay, silt, and sand with lenses of gravel. These deposits occur along the drainage bottoms in and adjacent to stream beds and are most common in the eastern portions of RFETS. Thicknesses range from 0 to 25 feet (EG&G 1995b).

Pleistocene deposits consist primarily of the Rocky Flats Alluvium, which is the most prevalent unconsolidated surficial deposit at RFETS. The Rocky Flats Alluvium ranges in thickness from 0 to 100 feet and forms a broad layer that extends across most of the western portion of RFETS (EG&G 1995b). The deposit is comprised of poorly sorted, clayey gravels and sands with abundant cobble and boulder sized material and discontinuous lenses of clay, silt, and sand. Caliche, a pedogenic deposit of calcium carbonate, is found discontinuously in the Rocky Flats Alluvium but tends to be better developed and more abundant from the western end of the Industrial Area (IA) eastward to the margin of the Rocky Flats Alluvial Fan. Significant quantities (up to 25 to 80% by volume) of caliche are present in some stratigraphic intervals of the Rocky Flats Alluvium (EG&G 1995b). Caliche horizons are most commonly encountered within 10 feet of the ground surface.

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2 2 2 Consolidated Bedrock Deposits

The unconsolidated surficial deposits unconformably overlie the claystone siltstone and sandstone bedrock of the Upper Cretaceous Arapahoe and Laramie Formations. The Arapahoe Formation ranges from 0 to 50 feet thick at RFETS and has been shown to contain a mappable but discontinuous fine to medium grained basal sandstone unit. This unit has been designated the Arapahoe (or Number 1) Sandstone (EG&G 1995b) and is known to subcrop beneath the Rocky Flats Alluvium and Colluvium in the 903 Pad East Trench and other areas of the eastern Industrial Area. It consists primarily of an argillaceous fine to medium grained locally conglomeratic moderately to poorly sorted sandstone that ranges in thickness from 0 to less than 50 feet (EG&G 1995b). Calcite cement found in some deeper Laramie Formation sandstone layers is absent in the Arapahoe Formation Sandstone. The Arapahoe Formation Sandstone is the uppermost sandstone of significant lateral extent and has been shown to be a pathway for lateral transport of contaminated groundwater.

The Laramie Formation conformably underlies the Arapahoe Formation and is approximately 600 to 800 feet thick. The Laramie Formation is subdivided into two members. The upper member of the Laramie Formation is 500 to 600 feet thick and consists primarily of massive claystone and siltstone. The lower member is about 300 feet thick and is composed of sandstones claystones and coal beds. The upper member is generally much finer grained than the lower member but contains several separate and discontinuous clayey sandstone units historically designated as the No. 2 through No. 5 Sandstones (EG&G 1995b). Unlike the Arapahoe Formation sandstone these sandstone units exhibit lithologic and hydrologic characteristics (i.e. high matrix clay content and low permeability) that are not indicative of groundwater flow pathways. These lenticular Laramie Formation sandstones are texturally distinct from the Arapahoe Formation sandstone by virtue of their high silt and clay content (EG&G 1995b).

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The Upper Cretaceous Fox Hills Sandstone conformably underlies the Laramie Formation and ranges from 90 to 140 feet in thickness. In general, the Fox Hills Sandstone is a very fine to medium grained, angular to subrounded, well sorted silty sandstone. The Fox Hills Sandstone is an aquifer of regional significance which lies at a depth of 700 to 800 feet below ground surface at RFETS. Underlying the Fox Hills Sandstone are several thousand feet of the Lower Cretaceous Pierre Shale and older units (EG&G 1995b).

Lithologic logs from boreholes drilled into bedrock indicate the presence of a weathered zone in the upper portion of the bedrock. Fracturing and weathering increase the permeability of bedrock material. The weathered zone is commonly less than 15 feet thick, but may be as thick as 60 feet. The thickness of the weathered bedrock zone is dependent on factors such as relative abundance of fractures, lithology, elevation relative to the water table, and proximity to valley bottoms.

2.2.3 Structural Features

The bedrock strata exposed immediately west of RFETS has been folded into steeply eastward-dipping exposures of the Fox Hills Sandstone and Laramie Formations. These units receive recharge from precipitation along the exposed hogbacks northwest and southwest of RFETS. The formations also receive recharge from the overlying Rocky Flats Alluvium and Arapahoe Formation.

Small scale structural features such as joints and fractures are present in the bedrock units. Joint and fracture surfaces are commonly coated with secondary iron and manganese oxides and hydroxides in the weathered portion of the bedrock units. Slickensides are also present on some fracture surfaces. The presence of such features increases secondary porosity and permeability and indicate that groundwater flow is probably enhanced in the weathered portion of the bedrock. Several bedrock faults have been identified in the RFETS Industrial Area, however none have been observed to offset alluvial materials (EG&G 1995b). The

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available hydrogeologic and geochemical data indicate that fractures and faults are not significant conduits for downward vertical groundwater flow to the Laramie Fox Hills deep aquifer (RMRS 1996)

2 3 HYDROGEOLOGY

2 3 1 Hydrostratigraphy

Shallow groundwater flow systems at RFETS have been categorized into two hydrostratigraphic units based on contrasts observed between groundwater geochemistry core logging and hydraulic conductivity determinations (EG&G 1995c). This division basically reflects the relative degree of hydrologic activity experienced by the units, the uppermost unit being the most active and transmissive of the two. The upper hydrostratigraphic unit (UHSU) consists of the distinct lithologic units of the Rocky Flats alluvium, colluvium, valley fill, alluvium, landslide deposits, weathered Arapahoe and Laramie Formation bedrock, and any sandstone units within the Arapahoe and Laramie Formations that are in hydraulic connection with the overlying unconsolidated surficial deposits or with the ground surface. The lower hydrostratigraphic unit (LHSU) is composed of the unweathered bedrock of the Arapahoe and Laramie Formations. This unit forms a thick (several hundred feet) regionally extensive confining layer that serves to isolate shallow groundwater from the underlying Laramie Fox Hills aquifer (RMRS 1996).

2 3 2 Groundwater Occurrence and Flow Conditions

Groundwater is found in all geologic units present at RFETS, although not always in predictable amounts and availability. In UHSU deposits, the quantity of water in storage at any particular locality varies widely and is a function of bedrock topography, proximity to drainages, seasonality, and local sources of recharge. The saturated thickness of UHSU alluvium is greatest in the western portion of RFETS (> 40 ft) and thinnest along hillsides.

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and in the eastern portion of the site. Areas of thin saturation commonly contain regions of discontinuous seasonally unsaturated alluvium due to irregularities in bedrock topography and water table fluctuations. Depth to shallow groundwater varies anywhere from 0 feet at flowing seepage areas to 70 feet at the west end of the site. The depth to water is generally shallowest in areas of thinner surficial deposits such as the Industrial area margins of the Rocky Flats Alluvium and creek drainages. Average depth to water and saturated alluvial thickness at the 903 Pad and Lip areas is on the order of 10 and 5 feet respectively (EG&G 1995c).

Seasonal fluctuations in shallow groundwater levels are observed in monitoring wells across the Site and normally range from three to six feet (EG&G 1995c). Larger annual fluctuations have been recorded at several locations in and adjacent to the IA with some wells exhibiting water level rises of over 10 feet. Well hydrographs indicate that water levels normally peak in May or June shortly following the spring rain season. Hydrographs of many older Rocky Flats Alluvium wells (installed 1986 and 1987) indicate that base water levels over the past decade have generally declined 1 to 5 feet possibly as a result of changes in water management practice at the plant.

In a broad sense the dramatic response of groundwater levels measured in many UHSU wells during the spring and early summer of 1995 was as noteworthy though less apparent as that observed for surface flow conditions. Record high water levels were recorded in many wells during and following the exceptionally wet spring of 1995. Water level rises in the range of 15 to 20 feet were observed in some wells located in the IA East Trenches and eastern Buffer Zone (BZ) Areas. Flowing artesian well conditions were noted at several well points that monitor seeps at the Solar Pond and southern IA. The rapid rate of decline observed in some well hydrographs was almost equal to the rate of rise suggesting that lateral subsurface drainage was rapid and probably more extensive than expected in normal years.

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The magnitude of these rises when compared to ground surface elevations indicate that a condition of temporary complete to near-complete saturation (groundwater flooding) probably formed locally in areas of thin to moderately thick surficial deposits. In areas of recharge such as all upland areas occupied by the Rocky Flats Alluvium and the majority of colluvium-covered hillslopes, the presence of saturated conditions indicate that the capacity of the geologic material to accept and transmit recharge water has been exceeded. The direction of soil water movement recharged from incident precipitation and runoff is initially downward until it reaches the water table and is transported laterally to a point of discharge. In comparison, groundwater discharge areas such as hillside and stream channel seeps are typified by saturated conditions that result in groundwater contact with surface soils. The potential significance of these two very different flow regimes on surficial and shallow subsurface soil actinide remobilization is reviewed in Section 2.3.4 and the Conceptual Model (Section 9).

The geology of the area exerts a controlling influence on groundwater flow due to the existence of sharp lithologic contrasts in the vertical distribution of geological materials (coarse grained surficial deposits underlain by fine grained bedrock) and the resulting bedrock topographic surface configuration. The predominantly claystone bedrock impedes the downward vertical migration of groundwater and directs flow laterally through the unconsolidated surficial and weathered bedrock materials. Groundwater flow tends to follow the bedrock surface and emerge as seeps drain into the hillside colluvium or migrate vertically into permeable subcropping sandstone units. In thinly saturated areas, the bedrock surface plays a critical role in directing groundwater flow and, where locally high, in the development of unsaturated zones in unconsolidated surficial deposits. Groundwater in the UHSU regionally flows eastward from broad areas of recharge (located upgradient and on nearby topographic highs) toward the erosional limit of the Rocky Flats Alluvium and then directly toward creeks in the drainage bottoms. After groundwater enters the valley fill alluvium from the hillslopes, it flows eastward in the direction of stream flow and exits the site at the RFETS east boundary (Indiana Street).

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2 3 3 Hydraulic Conductivities

In general the UHSU at RFETS has a low to moderate hydraulic conductivity and typically yields small amounts of water to monitoring wells. Hydraulic testing of the wells indicates that the UHSU exhibits a wide range of hydraulic conductivities because of the diverse nature of the individual geologic materials that comprise the unit. Values of hydraulic conductivity range from as high as 5×10^{-2} centimeters per second (cm/sec) in localized areas of the valley fill and Rocky Flats Alluvium to as low as 7×10^{-8} cm/sec in the clay lenses of the Rocky Flats Alluvium (EG&G 1995c). Hydraulic conductivities in the weathered bedrock portion of the UHSU range from 9.2×10^{-3} to 3×10^{-8} cm/sec, the highest values being associated with the Arapahoe Formation sandstone and the lowest values being associated with weathered claystone bedrock. Colluvial deposits indicate a range of 9.3×10^{-4} to 4.0×10^{-6} cm/sec. Average values calculated as geometric means are 4.3×10^{-5} cm/sec for the colluvium, 2.1×10^{-4} cm/sec for the Rocky Flats Alluvium, and 2.5×10^{-3} cm/sec for the valley fill alluvium in Woman Creek and Walnut Creek, respectively (EG&G 1995c).

Hydraulic conductivities of LHSU materials are significantly lower than those of the overlying unit, with values ranging from 10^{-6} to 10^{-8} cm/sec in the unweathered claystone bedrock. Discussion of LHSU hydraulic conductivities is presented in more detail in RMRS (1996).

2 3 4 Groundwater Interaction with Surface Water and Soils

As described later in Section 5.0, the importance of groundwater as an interactive actinide transport medium is limited to areas of contact with soils and surface water that contain actinide levels which exceed RFCA action levels. Groundwater is in direct connection with these media only immediately downgradient of areas where the water table intersects the

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ground surface. This requirement is met at hillside seeps and along stream beds containing valley fill alluvial deposits such as depicted in Figure 2.3

The pattern of seep distribution confirms that seep occurrence is controlled by local geologic conditions. Hillside seeps at RFETS are common along the eastern extent of the Rocky Flats Alluvium where the contact between the Rocky Flats Alluvium and underlying claystone subcrops along the upper margin of stream drainage valleys. In general, seeps occur in greater number and areal extent along the north side of the pediment ridges, as observed along South Walnut Creek and Rock Creek. Most seeps are ephemeral in nature and only discharge at the ground surface in the spring. Perennial seeps are relatively rare, with most located in the Rock Creek drainage. Groundwater seepage also occurs along segments of the Woman Creek stream channel, particularly above the Woman Creek stream diversion structure at Pond C-2, as determined from a stream gain/loss study (Fedors and Warner 1993). The stream channels of North and South Walnut Creeks are so extensively interrupted by impoundments that channel seepage measurements have not been attempted nor are they expected to yield meaningful data.

Seep flow data are generally unavailable due to difficulties inherent with measuring broad diffuse sources of discharge. The results of an incomplete seepage inventory conducted after the 1995 spring recharge event in portions of the Woman, Walnut, and Rock Creek watersheds revealed that of over 200 potential seep areas indicated by wetland vegetation, only 32 had a measurable flow and of these, 14 had flows of one gallon per minute or less. The remaining sites were moist to wet at the ground surface with little or no evidence of surface flow. Given the magnitude of the spring recharge event as reflected by sitewide high water table conditions, it is likely that seep flows measured during this time were at or near maximum levels. It was commonly observed during this survey that surface flow from many ephemeral hillside seeps percolate back into the soil below the discharge point before entering a surface water body. Direct contact with surface water may occur during exceptionally wet

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periods as a result of increased seep flow caused by abnormal water table rises or by mixing with surface runoff

The most common type of seep develops at the contact between the Rocky Flats Alluvium and underlying bedrock claystones. These seeps are thought to be related to preferential flow channels in bedrock surface topography and/or alluvial stratigraphy (high hydraulic conductivity zones) (EG&G 1995c). In the 903 Pad and East Trenches areas, some seep occurrences have been attributed to discharge from the subcropping Arapahoe Formation sandstone which receives recharge from the overlying surficial deposits (EG&G 1995c). The most notable sandstone seeps in this area include a grouping of seeps situated above the B 1 pond in the South Walnut Creek drainage and the 903 Pad hillside seep located at the soil study area in the Woman Creek drainage.

Examination of the most recent sitewide seep location map (EG&G 1995c Plate 9) indicates that there are fewer seeps in the 903 Pad and east BZ Areas relative to adjacent hillslopes located to the west and north. Along the north slope of Woman Creek from the 903 Pad eastward to Indiana Street, evidence of present-day seep activity is limited primarily to the 903 Pad hillside seep and potentially a few scattered small seeps. Periodic activation of a series of presumably old seepage sites located east of the 903 Pad have occurred from historic spray evaporation operations conducted at the South Spray field; however, these sites have since returned to a dry state following cessation of spray field operations. As shown on Figure 2-3, sites for groundwater interaction with surficial soils and surface water are both limited in extent and predictable based on the high degree of hydrologic control exerted by the local geology.

Groundwater discharge to surface water is presumed to occur along major stream channels although relatively little information is available to evaluate the significance of this interaction. Stream gain/loss studies conducted along Woman Creek (EG&G 1995c) have indicated that the flow regime in the upper reaches of the creek (west of confluence with

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Antelope Springs Creek) tend to be predominantly gaining while the lower reaches (east of Antelope Springs Creek) tend to be predominantly losing. The presence of gaining segments found just upstream of Ponds C 1 and C 2 suggest that these impoundments exert a local influence on groundwater discharge to surface water in the alluvium. Similar relationships are suspected to occur in the Walnut Creek drainage but to an unknown degree because gain/loss stream flow data is lacking in this area. It can be assumed however that groundwater/surface water interactions are potentially more complex in North and South Walnut Creeks owing to the influence of plant discharges and a more extensive system of ponds and other stream channel modifications.

2 4 SURFACE WATER HYDROLOGY

Streams and seeps at RFETS are largely ephemeral with stream reaches gaining or losing flow depending on the season and precipitation amounts. Surface water flow across RFETS is primarily from west to east with three major drainages traversing the site. Fourteen detention ponds (plus several small stock ponds) collect surface water runoff although only ten ponds require active management. Descriptions of the Site drainages and detention ponds including their respective interest to this report are contained below and shown in Figure 2-4.

2 4 1 Rock Creek

The Rock Creek drainage covers the northern portion of the RFETS BZ. The drainage channel is characterized by flat areas to the west several small stock ponds within the creek bed and multiple steep gullies and stream channels to the east. This basin is topographically isolated from the developed areas it receives no runoff from IA and contaminant transport by surface (or subsurface) processes is not suspected. Analytical data for Rock Creek is not presented in this report.

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2 4 2 Walnut Creek

Walnut Creek drains the central third of RFETS including the majority of the IA. It consists of several tributaries (McKay Ditch, No Name Gulch, North Walnut Creek, and South Walnut Creek) which join prior to leaving RFETS at the eastern boundary (Indiana Street). East of Indiana Street, Walnut Creek is diverted by the Broomfield Diversion Ditch around Great Western Reservoir and into Big Dry Creek. The Walnut Creek tributaries from north to south are described below.

McKay Ditch The headgates for this ditch are located at Coal Creek approximately three miles west of the IA. Flow is typically present only during spring runoff and during City of Broomfield water transfers to Great Western Reservoir. McKay Ditch does not receive runoff from the IA and analytical data from this drainage are not discussed in this report.

No Name Gulch This drainage is located downstream from the RFETS sanitary landfill. Runoff from the IA does not flow into this basin; analytical data from this area are not discussed in this report.

North Walnut Creek Runoff from the northern portion of the IA flows into this drainage, which has four detention ponds (Ponds A 1, A 2, A 3, and A-4). Ponds A 1 and A 2 are kept off line and maintained for emergency spill control; water levels in these ponds are controlled by evaporation or transfer. Pond A 1 also receives water pumped from the Landfill Pond roughly once per year. North Walnut Creek flow is diverted around Ponds A 1 and A 2 to Pond A 3, where water is held and settling of solids occurs. Pond A 3 is transferred in batches to the A series terminal pond, Pond A-4, which also receives water transferred from Pond B 5. Once filled to a predesignated level, Pond A 4 water is isolated, sampled, and released if water quality standards are met.

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These offsite discharges each averaging approximately 14 million gallons (43 acre feet) typically occur 8 to 10 times per year

The average annual discharge to North Walnut Creek is approximately 119 acre feet (39 million gallons) The average mean daily flow rate in North Walnut Creek (at station SW093) from October 1992 through April 1997 was 0.16 cubic feet per second (cfs) and the maximum mean daily flow rate during this period was approximately 9 cfs The combined capacity of the North Walnut Creek detention ponds (A series ponds) is approximately 52 million gallons (160 acre feet)

South Walnut Creek Runoff from the central portion of the IA flows into this drainage which has five detention ponds (Ponds B 1 B 2 B 3 B-4 and B 5) Ponds B 1 and B 2 are kept off line and maintained for emergency spill control water levels in these ponds are controlled by evaporation or transfer Pond B 3 receives effluent from the RFETS Waste Water Treatment Plant (WWTP) and flows into Pond B 4 South Walnut Creek flow is diverted around Ponds B 1 B 2 and B 3 into Pond B-4 which flows continuously into terminal pond Pond B 5 After filling to a predesignated level Pond B 5 is pump transferred in batches of approximately 9 million gallons to Pond A-4 (8 to 10 times per year) which is discharged offsite as described above

The average annual discharge to South Walnut Creek including effluent from the site WWTP is approximately 258 acre feet (84 million gallons) The average mean daily flow rate measured in South Walnut Creek (at station GS10) from October 1992 through April 1997 was 0.12 cfs and the maximum mean daily flow rate during this period was approximately 5 cfs The combined capacity of the South Walnut Creek detention ponds (B series ponds) is approximately 27 million gallons (83 acre feet)

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2 4 3 Woman Creek / South Interceptor Ditch (SID)

South of the IA is the Woman Creek / South Interceptor Ditch (SID) system. Descriptions of these drainages are provided below.

SID Surface water runoff from the southern portion of the IA is captured by the SID which flows from west to east into Pond C 2. Once filled to a predesignated level, Pond C 2 water is sampled and, if water quality standards are met, pump discharged into Woman Creek which flows east off the site. These offsite discharges, averaging approximately 11 million gallons (33 acre feet), typically occur once per year during the winter.

There is frequently no flow in the SID. The average mean daily flow rate (at station SW027) from October 1994 through April 1997 and including the periods of no flow was 0.05 cfs and the maximum mean daily flow rate during this period was approximately 6 cfs.

Woman Creek South of the SID is Woman Creek which flows through Pond C 1 and offsite at Indiana Street. The Woman Creek drainage basin extends eastward from the base of the foothills near Coal Creek Canyon to Standley Lake. The average annual yield of the basin is approximately 341 acre feet (111 million gallons). The average mean daily flow rate in Woman Creek (at Indiana Street) is 0.47 cfs and the maximum mean daily flow rate during this period was approximately 76 cfs.

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2 4 4 Other Drainages

The D series Ponds (D 1 and D 2) are located on the Smart Ditch located south of Woman Creek. Smart Ditch is hydrogeologically isolated from the RFETS IA, however, Smart Ditch does collect surface water runoff from surface soils with low levels of wind-blown radionuclide contamination. This drainage and these ponds are not discussed in this report.

2 5 ACTINIDE SOURCES AND CONTAMINATION EVENTS AT RFETS

Plutonium and Americium in the 903 Pad Area

The main source of plutonium and americium in the environment at RFETS is the former drum storage area known as the 903 Pad, with lesser contributions from the 1957 and 1969 fires (Meyer et al. 1996). The majority of plutonium and americium is associated with surface soils (0 to 6 inches {0 to 15 cm}). The distribution of plutonium and americium in surface soils is shown in Figures 4.1 and 4.2. The plutonium and americium contamination of the 903 Pad and Lip area and the soils to the east and southeast of the 903 Pad, extending to east of Indiana Street, were generated during a combination of high wind events and earth moving activities in the period from 1967 through 1969. Cleanup operations in the 903 area began in January of 1967 and continued until mid 1968. The asphalt pad was not placed until November 1969.

The RAC 1996 final report (Meyer et al. 1996) evaluated and summarized the history of the 903 area and correlated air monitoring results with wind events and physical disturbances of the 903 Pad soils. They concluded that the significant airborne plutonium releases from the 903 area appear to be associated with a relatively few events, most likely high winds following mechanical disturbances (Meyer et al. 1996). Covering the 903 Pad with fill and asphalt did not completely eliminate airborne contamination, although air concentrations were greatly reduced. Areas of soil outside of the pad were contaminated to a lesser extent than the primary 903 area, but remained subject to wind erosion. This

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“secondary resuspension has been a significant relative source of offsite air contamination since 1970 and is discussed in some detail in Rope et al (1997)

Plutonium in surface soil has shown little redistribution over the years since the initial contamination in 1968 1969 but evidence has not ruled out that some local redistribution has re occurred Webb et al (1996) demonstrated that plutonium concentrations and total deposition (from RFETS) decrease rapidly with distance from the 903 Pad and with deviation in direction from due east of the 903 Pad (Rope et al 1997)

IHSS 141 the sludge dispersal area is located just west of the original east perimeter road and consists of the former sewage sludge drying beds and related surface soil contamination The beds were not lined and excess liquid from the sludge could come in direct contact with the underlying soil The potential for contamination of the soil and groundwater beneath the beds is considered high (DOE 1992) During the OU6 investigation elevated activities of americium and plutonium were frequently detected above background screening levels at both IHSS 141 and 165 (EG&G 1996) IHSS 165 is known as the Triangle Area The area was used to store drums containing dilute nitric acid and fire waste from the May 1969 plutonium fire The contents of the drums were recoverable plutonium bearing wastes and residues (DOE 1992)

Solar Ponds

Uranium was handled at RFETS beginning in 1953 and continues to be handled as the site is being decommissioned Liquids containing uranium nitrate and other salts of uranium from processing were pumped to the Solar Ponds beginning in 1953 and continuing until 1986 Increasing levels of uranium and nitrate in groundwater around the Solar Ponds were noted in seeps at the pond edge in 1953 By 1970 nitrate contamination was noted in North Walnut creek downgradient of the Solar Ponds A number of small trenches were installed in the 1970s in an attempt to capture the contamination seeping from the ponds

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An interceptor trench system (ITS) keyed into bedrock (in most areas) was installed in 1981 to collect groundwater downgradient from the Solar Ponds. The uranium rich evaporation sludges were removed from the Solar Ponds in 1986 through 1995. The ponds are now empty except for occasional rainfall capture.

Old Landfill and Ash Pits

Relatively high concentrations of uranium are found in discrete samples from the old landfill and ash pits to the south of the IA (OU 5). This landfill and ash includes incinerator ash from the RFETS incinerator previously used to oxidize pyrophoric depleted uranium turnings. Uranium machine turnings have also been noted on the surface of the landfill (Jerry Anderson personal communication 1997).

East Trenches

Approximately 125 drums of depleted uranium machine turnings were disposed in Trench 1 (T 1) from 1954 until 1962. The trench was covered with 2 feet of soil as the drums were placed. There is no evidence that T 1 is a source of surface soil (wind blown) or groundwater uranium contamination (RMRS 1997). Lesser amounts of depleted uranium were likely disposed in other trenches on the east side of the plant however there is no evidence that these trenches are a source of wind blown or groundwater uranium contamination.

2.6 REFERENCES

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3 0 THE GEOCHEMISTRY OF URANIUM, PLUTONIUM, AND AMERICIUM

This section reviews specific chemical reactions which determine the behavior of actinides in the environment and discusses the origin occurrence and behaviors of actinides at RFETS

Actinides are transition elements with partially filled 5f electronic orbitals. The 5f orbitals are screened from the chemical environment by higher lying s and p electrons and this screening produces strong similarities among the chemical properties of elements in the group. The variation in chemical properties within the series is systematic and predictable and some properties of the heavier members of the series were predicted prior to discovery (Seaborg and Loveland 1990)

Chemical reactions which determine the behavior of actinides in the environment include reduction/oxidation precipitation and coprecipitation formation of aqueous complexes sorption and formation of finely divided particles (true colloids and pseudocolloids) (Allard and Rydberg 1983 Choppin 1988 Dozol and Hagemann 1993 and Silva and Nitsche 1995). The major salient points related to the actinides of concern at RFETS (uranium plutonium and americium) will be discussed below

3 0 1 Reduction/Oxidation

When a current passes through an aqueous ionic solution electrons enter the solution at the cathode and leave the solution at the anode. The electrons entering and leaving a solution can cause reactions. Reactions occurring at the anode are oxidation reactions while those occurring at the cathode are reduction reactions and more generally any chemical reaction involving loss of electrons is known as an oxidation reaction while any reaction involving gain of electrons is known as a reduction reaction. Oxidizing agents are materials that are reduced while taking on electrons driving oxidation reactions. Reducing agents are materials that are oxidized while donating electrons to drive reduction reactions

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Reduction and oxidation strongly influences the environmental behavior of actinides. The actinides assume a number of valences (Table 3 1) but redox conditions in natural systems are commonly buffered by reactions involving major constituents such as the degradation of water, reduction/oxidization of iron, or reduction of naturally occurring organic matter.

Under environmental conditions americium tends to occur primarily as Am(III) but uranium and plutonium partition among IV and VI valences and III, IV, V, and VI valences respectively (Allard et al. 1983). The single major valence of americium simplifies the study of its environmental behavior, but the more complicated behavior of plutonium and uranium commonly drives researchers to extrapolate behavior into the system from single valent actinides and lanthanum group elements (lanthanides).

Table 3 1 Actinide Elements and Their Oxidation States

89	90	91	92	93	94	95	96	97	98	99	100	101	102	103
Ac	Th	Pa	U	Np	Pu	Am	Cm	Bk	Cf	Es	Fm	Md	No	Lr
Oxidation States														
						(2)	(2)		(2)	(2)	2	2	<u>2</u>	
<u>3</u>	(3)	(3)	3	3	3	<u>3</u>	3	<u>3</u>						
	<u>4</u>	4	4	4	<u>4</u>	4	4	4	4	(4)	4?			
		<u>5</u>	5	<u>5</u>	5	5	5?		5?					
			<u>6</u>	6	6	6	6?							
				7	(7)	7?								

From Katz et al. (1986). Bold & underline valences are the most stable oxidation states. Bracketed valences are unstable.

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Actinides in the III IV V and VI valences form M^{+3} M^{+4} MO_2^+ and MO_2^{2+} cations respectively (Cleveland 1979) The simple cations (M^{+3} and M^{+4}) inter-convert rapidly and the oxygenated actinyl ions (MO_2^+ and MO_2^{2+}) inter-convert rapidly but conversions between simple and oxygenated cations are relatively slow because they require extensive changes to the linear dioxo structure before electron transfer (Cleveland 1979 Choppin 1983)

This slow conversion can produce significant disequilibria in systems involving both types of cations Experiments in ponds at Oak Ridge for example showed that dissolved Pu(V) could be reduced to Pu (III+IV) by adding glucose to force the water anaerobic but when the system was reaerated Pu(V) returned to solution slowly and at three months Pu(V) remained at less than one fifth of the initial value (Watters 1983)

Laboratory oxidizing and reducing agents also affect the redox state of actinides and experiments which use redox reagents to destroy major soil phases to establish the location of the actinides must take these effects into account Both Bunzl et al (1995) and Litaor and Ibrahim (1997) for example attempted to establish the relationship between plutonium and major soil phases using strong reducing and oxidizing agents to destroy sesquioxides and organic carbon respectively but did not consider the effects of these agents on the redox state of plutonium itself This oversight renders their conclusions that organic matter and iron sesquioxides are largely responsible for the affinity of soils for actinides of little value (Marty et al 1997a) Moreover their conclusions contradict earlier work showing little effect on the affinity of soils for actinides from the removal of organic matter and sesquioxides (except for the pentavalent valence Table 3 2)

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Table 3 2 Effect of Clay Treatment on Adsorption of Actinide Elements to Miami Silt Loam Clay

Treatment	CEC meq/100 g	Percent adsorbed			
		Th 234 (IV)	Cm 244 (III)	U 234 (IV)	Np-237 (V)
Intact soil	17	99 7	98 9	95 6	61 8
Organic matter removed	11	99 8	99 6	96 4	49 7
Fe and organic matter removed	9 9	99 7	95 6	99 1	18 2

From Bondietti and Tamura (1980)

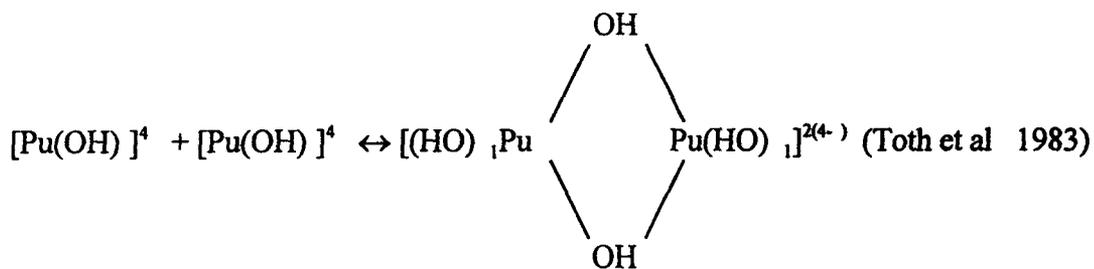
3 0 2 Coprecipitation and Precipitation

Actinides entered the RFETS environment primarily as reactive metals or in solution rather than as slow reacting thermodynamically stable or metastable phases. The behavior of the RFETS actinides therefore contrast in many respects with that of actinides deposited following thermonuclear explosions (high fired actinides) deposited following explosions of high explosives (low fired actinides) or deposited in the form of refractory wastes resistant to nitric hydrofluoric and hydrochloric acid digestion. None of the major phases found in RFETS soils are undergoing rapid recrystallization and the actinides appear to be concentrated outside major soil phases (Marty et al 1997a). The presence of the actinides outside the major phases allows them to enter into reactions in solution without the dissolution of the soil phases.

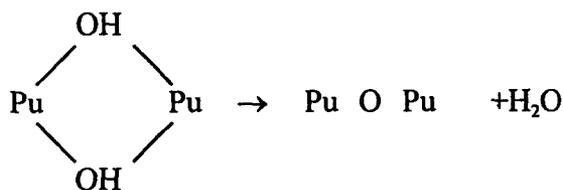
Actinide hydroxide/oxide have been observed to form discrete particles in some areas of RFETS but the observed abundances are not sufficient to account for the entire plutonium inventory (McDowell and Whucker 1978). Actinide hydroxides and oxides precipitate from solution through a well characterized process which involves hydrolysis of actinide ions in solution (Cleveland 1979). The hydrolyzed actinides then form polymers of various lengths (Cleveland 1979). The extent of polymerization depends on the concentration of actinides in solution and the reaction may not be easily reversible. Polymerization of plutonium for example forms networks of Pu(OH)₄ linkages but as the polymers age oxygen bridges increase

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in abundance and the material begins to resist dissolution even in concentrated nitric acid (Toth et al 1983 Choppin, 1983) This resistance may render simple solubility constants (K_{sp}) calculated for solid actinide hydroxides invalid (Choppin, 1983) and can affect apparent distribution coefficients (K_d) for systems containing actinide hydroxides



Upon aging the hydroxyl bridges convert to oxo bridges



3 0 3 Sorption

Sorption on geological materials is an important control on the environmental behavior of actinides and sorption reactions are one of the chief causes for the limited mobility of most actinides in the environment After 30 years of radioactive releases from the waste water treatment facility at Los Alamos for example plutonium moved less than 3 meters (m) into the tuff (Stoker et al 1991) and similar examples are common at other sites Physical adsorption chemisorption and electrostatic adsorption all appear to effect the environmental behavior of actinides (Benes and Majer 1980 Allard and Rydberg 1983) but prediction of sorption behavior remains a major weakness in determining the environmental behavior of actinides Sorption studies remain empirical and the K_d s are only valid under very limited conditions (Dozol and Hagemann 1993)

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One problem is the partially irreversible nature of sorption/desorption reactions. Some of this behavior may be due to artifacts, but experiments on the sorption of plutonium by red clay show that sorption/desorption reactions remain irreversible even after possible artifacts are accounted for (Higgo and Rees 1986).

3 0 3 1 Physical adsorption

Physical adsorption occurs by non-specific forces which attract solute to sorbent and binds solute species in consecutive layers to the exposed solid surface. This mechanism is rapid, non-selective, reversible, fairly independent of concentration in solution, and only slightly dependent on ion exchange capacity.

3 0 3 2 Chemisorption (or Specific Adsorption)

Chemisorption (or specific adsorption) results from specific chemical forces between sorbent and solute. This process can be regarded as complex formation and is commonly specific and selective, concentration dependent, and may be slow and only partially reversible.

Figure 3.1 shows typical sorption behavior for actinide ions on oxide surfaces. Actinides and other transition metals typically exhibit a sharp sorption shoulder which coincides with formation of the first hydrolysis complex in solution (Farley et al. 1985). Hydrolysis complexes of the form $M(OH)^{n-1+}$ appear responsible for the shoulders. These complexes are less charged than the parent (M^{+}) ions, and the sharp increase in sorption associated with their formation shows that the interaction is "non-coulombic" and does not involve simple electrostatic attraction which would be greater for the more highly charged uncomplexed actinides (it is "non-coloumbic") (Stumm and Morgan 1981).

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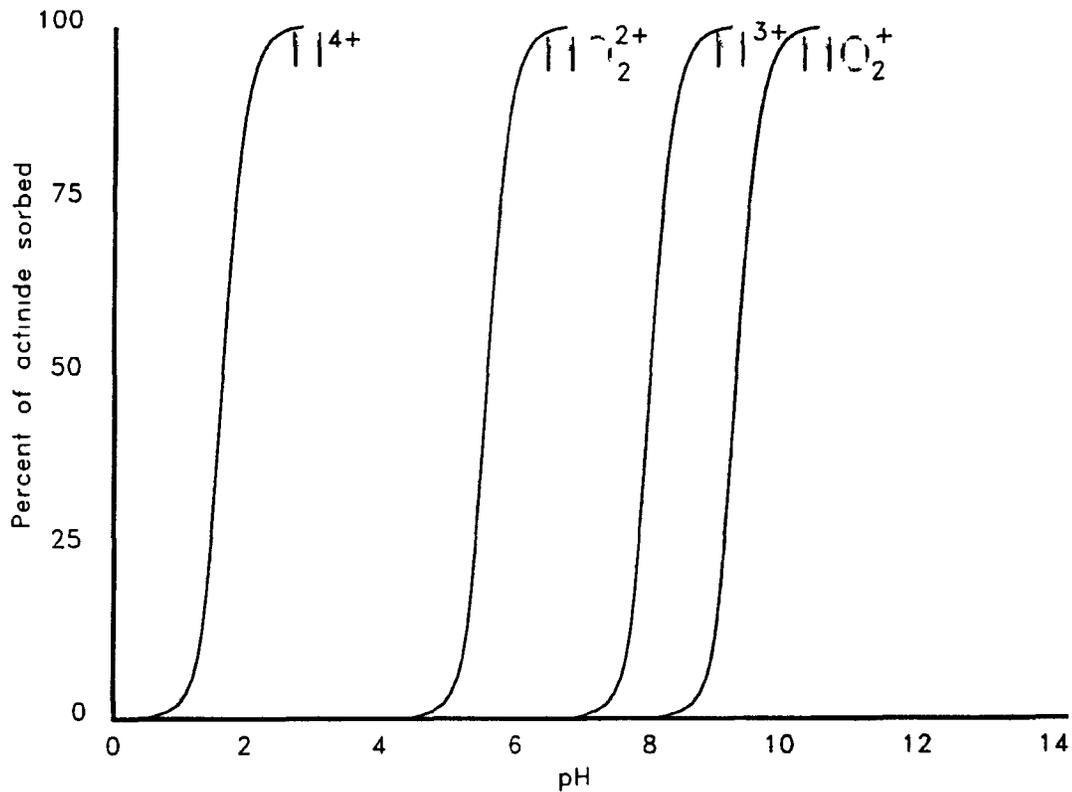


Figure 3 1 Typical Curves for Specific Sorption of Actinides on Solid Oxides

There is a considerable body of evidence suggesting that specific sorption is important with actinides. At Los Alamos, for example, plutonium was strongly held by a variety of materials including tuff which has an ion exchange capacity of only 0.5 to 3 meq/100 g (Christenson et al. 1958). Organic matter free soils from Hanford showed similar behavior and a large percentage of plutonium was adsorbed by soil almost immediately (Rhodes 1957).

The mechanism of specific adsorption has not been totally elucidated but appears to have much in common with that of hydroxide precipitation.

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3 0 3 3 Electrostatic Adsorption/Ion Exchange

Electrostatic adsorption/ion exchange is a common interaction between solid and species in solution and is due to coulomb forces which attract species from solution to solid surfaces. The process is dependent upon concentration.

Electrostatic adsorption also affects the behavior of actinides in the environment. Actinides which entered the environment as solutions or metals tend to be concentrated in the fine fraction of soil and sediments which also has the highest cation exchange capacity (Kingsley 1947, Hakonson and Nyhan 1980, Purtymun et al 1983, Watters et al 1983, Little and Whicker 1978). In Mortandad Canyon at Los Alamos, for example, plutonium concentrations in the silt and clay fraction are seven times the concentration in the coarser fractions of alluvium (Dahlman et al 1980, Hakonson and Nyhan 1980).

A number of studies, including recent work by Litaor and Ibrahim (1997), moreover, showed that a small fraction of the plutonium can be removed from soil with simple salt solutions. A small fraction of actinides are held in the exchange sites under normal conditions, but the fraction held by electrostatic sorption will increase if plutonium becomes less strongly held by specific sorption. It has been suggested, for example, that reduction of soils under natural conditions could destroy iron oxides and decrease specific sorption of actinides. Under such conditions, ion exchange reactions should increase in importance.

Electrostatic adsorption onto soils and sediments depends on the charge of the stationary phase, and this charge changes with solution pH. Soil particles are positively charged at low pHs but shift to a negative charge at higher pH values. The point at which the particles have no net-charge is the isoelectric point, and this corresponds to the point of zero proton condition (pH_{zpc}) when OH^- and H^+ are the only ions in solution (Table 3-3). The positively charged surfaces formed at low pH will retard movement of anions, and the negatively

charged surfaces formed at high pH will retard movement of cations (Stumm and Morgan 1981) Under environmental conditions most surfaces in soil will be negatively charged and attract cations and even positively charged minerals tend to be coated with natural organic matter (NORM) which provides a negative charge irrespective of the original surface

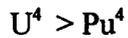
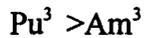
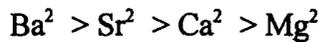
Table 3 3 Zero Point of Charge for Soil Constituents of Possible Significance at RFETS

Material	pH _{zpc}
α Al ₂ O ₃	9.1
α Al(OH) ₃	5.0
γ AlOOH	8.2
Fe ₃ O ₄	6.5
α FeOOH	7.8
γ Fe ₂ O ₃	6.7
"Fe(OH) ₃ (amorph)	8.5
δ MnO ₂	2.8
β MnO ₂	7.2
SiO ₂	2.0
feldspars	2.2-4
kaolinite	4.6
montmorillonite	2.5

From Stumm and Morgan (1981)

Ions in solution follow a general order of affinity during ion exchange reactions. This affinity is caused by coulombic interactions among counter ions and the exchanger and ion-dipole and induce dipole interactions (Eisenmann, 1962). Normally the first factor is weak compared to the second and the affinity sequence follows the Hofmeister series where ions of small hydrated radius are preferred over ions of large hydrated radius (Stumm and Morgan 1981)

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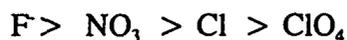


Ions of smaller hydrated radius (such as Pu^{3+}) displace those of higher radius (such as Am^{3+}) and high charge ions (such as Pu^{3+}) are preferred over low charge ions (such as Ca^{2+} and Na^+)

3 0 4 Aqueous Complexes

Actinides form strong complexes with a range of inorganic anions organic compounds with negatively charged functional groups and synthetic complexing agents These reactions influence the solubility of actinides in solution and greatly increase the apparent redox stability of valences which form the strongest complexes (Choppin 1983) Figure 3 3 shows the effects of complexing agents on sorption reactions In general formation of actinide anion complexes reduces the quantities of free ions available for hydrolysis and shifts both sorption and precipitation curves to higher pHs

The strength of complexes with inorganic anions follows the order



The thermodynamic constants for these actinide complexes are being critically reviewed and compiled as part of a European Nuclear Energy Agency (NEA) initiative (Dozol and

Hagemann 1993) Data for uranium currently are available but data for plutonium and americium are not complete. Plutonium is the most complicated system among the actinides because of the large number of possible valences involved and Table 3-4 provides a compilation of stability constants for important complexation reactions in this system.

Table 3-4 Complexes of Trivalent and Tetravalent Plutonium

	Reaction	Equilibrium Constant	Constant
EDTA	$\text{Pu}^4 + \text{H}_2\text{Y}^2 \rightleftharpoons 2\text{H}^+ + \text{PuY}^2$	$[\text{PuY}][\text{H}^+]^2/[\text{Pu}^4][\text{H}_2\text{Y}^2]$	$=4.57 \times 10^{17}$ *
Citrate	$\text{Pu}^4 + \text{C}_6\text{H}_5\text{O}_7^{3-} \rightleftharpoons \text{Pu}(\text{C}_6\text{H}_5\text{O}_7)^1$	$[\text{Pu}(\text{C}_6\text{H}_5\text{O}_7)^1]/[\text{Pu}^4][\text{C}_6\text{H}_5\text{O}_7^{3-}]$	$=3.0 \times 10^{15}$
Citrate	$\text{Pu}(\text{C}_6\text{H}_5\text{O}_7)^{1+} + \text{C}_6\text{H}_5\text{O}_7^{3-} \rightleftharpoons \text{Pu}(\text{C}_6\text{H}_5\text{O}_7)_2^2$	$[\text{Pu}(\text{C}_6\text{H}_5\text{O}_7)_2^2]/[\text{Pu}(\text{C}_6\text{H}_5\text{O}_7)^1][\text{C}_6\text{H}_5\text{O}_7^{3-}]$	$=1 \times 10^{30}$
EDTA	$\text{Pu}^3 + \text{H}_2\text{Y}^2 \rightleftharpoons 2\text{H}^+ + \text{PuY}^-$	$[\text{PuY}][\text{H}^+]^2/[\text{Pu}^3][\text{H}_2\text{Y}^2]$	$=1.3 \times 10^{18}$
Citrate	$\text{Pu}^{3+} + \text{C}_6\text{H}_5\text{O}_7^{3-} \rightleftharpoons \text{Pu}(\text{C}_6\text{H}_5\text{O}_7)^0$	$[\text{Pu}(\text{C}_6\text{H}_5\text{O}_7)^0]/[\text{Pu}^{3+}][\text{C}_6\text{H}_5\text{O}_7^{3-}]$	$=7.3 \times 10^8$

* most accurate Cleveland (1979)

3.0.5 Pseudo-Colloids and True Colloids

McCarthy and Zachara claimed that small negatively charged particles move in solution under special conditions and that these complexes can transport actinides quickly through groundwater across large distances (McCarthy and Zachara 1989). Actinides complexed with high molecular weight organic molecules sorbed on clays sorbed on hydrated iron oxides and hydrated aluminum oxides (*pseudo colloids*) and actinides trapped within finely divided actinide hydroxides and oxides (*true colloids*) have been suggested as mobile phases (Puls and Powell 1992, Robertson 1984, Ryan and Gschwend 1990, Nightengale and Bianchi 1977, Kim et al 1984).

The chief example of the potential significance of such colloids in transporting radionuclides has been the appearance of plutonium in monitoring wells 3 km from the source of contamination in Mortandad Canyon at the Los Alamos National Laboratory (McCarthy and Zachara 1989, Penrose et al 1990).

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Recently however it has been demonstrated that plutonium moved down this canyon in surface runoff and not through groundwater (Marty et al 1997b)

Such transport of actinides on colloids has been claimed to be active at RFETS (Harnish et al 1994 Harnish et al 1996) The results of studies however are suspect because of a combination of pathologic science (the data on which interpretations are based are less than detection limits) and potential artifacts (monitoring equipment was installed through surface actinide contamination without adequate safeguards)

Theoretical work on extremely fine particles in water moreover suggests that colloids should be effectively screened from groundwater over short distances through the action of natural collectors (Yao et al 1971) Natural analogs cast further doubt on the significance of this mechanism Natural Th^{4+} for example forms strong complexes with natural organic matter but remains largely immobile in soils presumably as a result of competing reactions

Facilitated transport of plutonium complexed by synthetic complexing agents however has been unequivocally demonstrated for ethylene diamine tetra acetic acid (EDTA) plutonium complexes at the MaxiFlats disposal site (Cleveland and Rees 1981) and NORM and other colloidal materials form aqueous phase complexes with actinides and stabilize plutonium in lake water (Nash and Choppin 1980 Watters 1983 McCarthy and Zachara 1989) It therefore seems possible that complexes of actinides with NORM and other colloids could increase the rate of actinide transport even though clear evidence has not emerged to date

The general processes described above control the environmental behavior of actinides but the chemical properties of uranium plutonium and americium also differ in many respects Environmental conditions also vary from site to site influencing the behavior of actinides The following sections contain a more detailed discussion of the effects of the specific chemistry of the individual actinides specifically uranium plutonium and americium under the soil and groundwater conditions found at RFETS

3 1 URANIUM

3 1 1 Origin and Occurrence of Uranium

Uranium (element 92) occurs naturally in the earth's crust with an average concentration of 2.7 mg/kg. Uranium is a lithophilic element which is most abundant in granites (averaging 5 parts per million [ppm]) and shales (averaging 3.5 ppm Krauskopf 1979). Myrick et al (1983) evaluated the concentrations of U-238 in surface soils across the United States to determine background levels. Surface soils of Colorado were found to range from 0.47 to 3.0 pCi/g with a mean and standard deviation of 1.2 pCi/g and 0.91 pCi/g respectively.

Natural uranium is 99.273% U-238 but also contains small amounts of U-234 and U-235 (0.005 and 0.72% respectively in most cases). The mass differences among the uranium isotopes is minor and the isotopes do not normally fractionate through natural physical or chemical processes (Faure 1977). U-238 has a long half life (4.51×10^9 years) and decays to Pb-206 through a chain which includes U-234 (Eisenbud 1987). U-234 has a relatively short 248,000 year half life but once secular equilibrium is established the radioactivity of U-234 exactly equals the radioactivity of U-238. U-235 decays through a separate chain to Pb-207 with a half life of 7.13×10^8 years (Eisenbud 1987).

3 1 2 Geochemistry of Uranium

Uranium assumes IV and VI valences in soil. As with all actinides, the IV valence hydrolyzes, precipitates easily, undergoes strong specific sorption reactions, and is preferred over other cations in ion exchange reactions. These reactions render U(IV) largely immobile in the environment despite highly stable complexes with inorganic and organic ligands. The more oxidized U(VI) is more soluble, undergoes weaker specific sorption, and tends to be more mobile (Salomons and Foerstner 1984).

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Anions such as carbonate nitrate chloride fulvic acid humic acid and EDTA form complexes with U(IV) and U(VI) and increase the amount of uranium which can remain in solution This in turn increases the overall mobility of the uranium Uranium generally is least mobile in reducing (anaerobic) environments free of complexing anions and most mobile in oxidizing (aerobic) environments with high concentrations of complexing anions

Natural uranium is ubiquitous in the Front Range of Colorado and complicates studies of uranium at RFETS High uranium granites occur throughout the front range and uranium ore (the Schwartzwalder mine) is located near RFETS The natural alkaline and oxidizing environment in near subsurface water mobilize uranium in groundwater and higher uranium concentrations in water samples are probably due to leaching of uraniumiferous strata in the Pierre and Laramie formations The South Platte River is anomalously rich in uranium compared to most other rivers of its size (Bolivar et al 1978)

The isotopic abundances (by weight) in some of the uranium used at RFETS differed significantly from natural values and this may be useful in determining which uranium represents RFETS contamination (Table 3 5) Both U 234 U 235 enriched and U 234 U 235 depleted nuclear weapons components were manufactured at RFETS (EG&G 1988) and the isotopic signatures of both types of contamination could theoretically be differentiated from natural uranium Unfortunately most samples collected from RFETS have been analyzed by alpha spectroscopy and this technique provides only an estimation of U 235 activity

Table 3 5 Isotopic Ratios in Potential Sources of Uranium at RFETS

	Natural Uranium*	Depleted Uranium**	Power Plant Uranium	Enriched Uranium**
U 238	99 273%	99 75%	97%	5%
U 235	0 72%	0 25%	3%	95%
U 233/234	0 005%	0 0005%	0 03%	TBD
U 234/U 238	1 06	0 09	5 74	> > 5 74

* Naturally found at RFETS ** Contributed by industrial use of RFETS

Efurd et al (1993) used thermal ionization mass spectrometry (TIMS) to measure U 234 235 236 and 238 in RFETS sediment and water samples. The technique is more accurate than alpha spectroscopy and provides more certain isotopic ratios. On the basis of this data the authors concluded that the largest source of radioactivity in the terminal ponds was naturally occurring uranium and its decay product radium and that the largest source of anthropogenic radioactivity in the terminal ponds was depleted uranium. Approximately half the uranium present in Ponds A-4 and C 2 and approximately 20% of the uranium present in Pond B 5 apparently originated as depleted uranium. These results are significant because they allow uranium contributed by industrial to be differentiated from uranium which occurs naturally.

U 236 is produced by neutron capture on U 235 in nuclear reactors or atomic explosions and does not occur in nature or in uranium which has not been through a reactor (Efurd et al 1993). U 236 therefore would not be expected in the uranium used at RFETS. Efurd et al (1993) however detected U 236 in some samples and concluded that the presence of U 236 in the surface water samples collected at RFP (RFETS) and the variable U 238/U 235 atom ratios detected in water samples collected from the holding ponds prove that anthropogenic uranium is present. This probably represents uranium subject to neutron flux during an explosion and may be atmospheric fallout from atomic testing.

U 238/ U 235 ratios for dissolved uranium in RFETS groundwater from which non-detects have been excluded show a wide scatter. This could be due to lack of a systematic treatment.

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of sampling and analytical error or real variability in isotopic ratios. The most likely explanation is unpropagated error because background wells show high variability in isotopic ratios which cannot be otherwise explained.

U 234/ U 238 activity ratios could theoretically be used to distinguish between natural enriched and depleted uranium (Table 3.5). The ratio should be 1.0 at secular equilibrium but the ratios ranged from <1.0 to >2.5 for background areas (DOE 1993a). If analytical errors were correctly handled in this study, this would indicate that unexpected factors affected U 234/ U 238 activity ratios, limiting its usefulness in distinguishing between natural and RFETS uranium.

DOE (1993a) reported a range of 1.19 to 2.43 for ratios of uranium isotopes in filtered background groundwater and stream water, and DOE (1995b) reported U 234/ U 238 ratios ranging from 0.34 to 18.5 for UHSU groundwater at the Solar Ponds (formerly known as Operable Unit 4 [OU4]). Neither report systematically dealt with analytical errors and the results therefore are of limited usefulness.

In summary, the results from most of the studies of isotopic ratios undertaken to date have not systematically dealt with error. TIMS analysis shows promise as a means to determine the source of uranium because its analytical errors are smaller and isotopic ratios can be determined more precisely. TIMS could allow natural uranium and anthropogenic uranium to be better determined. This may be important because background levels set natural limits on the effectiveness of cleanup, and the differentiation of contaminated areas from uncontaminated areas could allow cleanup efforts to be focused.

August 1997

3 2 PLUTONIUM

3 2 1 Origin and Occurrence of Plutonium

Plutonium was the second transuranic element discovered and the first transuranic element to be produced in macroscopic quantities (Seaborg 1958). It has been produced in far greater quantities than any other transuranic element because of its usefulness in producing atomic weapons and power. The plutonium used at RFETS was weapons grade and consisted primarily of Pu 239 (half life of 2.44×10^4 years) with lesser amounts of Pu 240 (half life of 6 580 years) and higher isotopes.

Plutonium does not occur in nature but an anthropogenic background of Pu 239, Pu 240, and Pu 241 exists worldwide due to fallout from atmospheric nuclear weapons tests. A considerable background of heat source plutonium (Pu 238) also occurs due to the atmospheric burnup of a nuclear powered satellite (SNAP 9A) over the Southern Hemisphere in 1964 (Hardy et al 1973).

The distribution of fallout plutonium across the earth's surface is not uniform. Geographic, orographic, and meteorologic effects produced spatial variations in fallout. Hardy et al (1973) studied the distribution of fallout plutonium and measured the isotopic ratios in soil samples collected from around the world. They determined that the heaviest fallout of Pu 239 and Pu 240 lies in the temperate latitudes in the Northern Hemisphere, whereas concentrations of Pu 238 are greatest in the temperate latitudes of the Southern Hemisphere (Hardy et al 1973). Purtymun et al (1990) studied the deposition and distribution of plutonium from worldwide fallout and concluded that the differences in plutonium concentrations and ratios can be attributed to regional and local weather patterns and to distribution by physical transport, and that variability in plutonium particle size also contributes to the inconsistency of plutonium concentrations and ratios found in soils and sediments.

The plutonium used at RFETS was exclusively "weapons grade" and was primarily in a metallic form. This plutonium is predominantly Pu 239 with small amounts of Pu 240 and traces of heavier isotopes. Some low pH plutonium containing solutions also were used at the plant as part of processes to recover waste plutonium.

3 2 2 Geochemical Behavior of Plutonium

The usefulness of plutonium in producing atomic weapons and as a power source has resulted in extensive studies of the solution chemistry of plutonium including the interaction of plutonium solutions and metals with soil (Seaborg and Loveland 1990). The environmental chemistry of plutonium is complicated by the large number of reactions involved, but despite the relatively recent discovery of the element its major reactions are well characterized (Seaborg and Loveland 1990; Choppin 1983).

As discussed above, the major reactions influencing the environmental fate of plutonium are reduction/oxidation (Eh controlled), formation of complexes with anions and NORM (controlled by pH and strength of complexing anions), precipitation (controlled by solution composition) and sorption (controlled by solution composition and the nature of solid phases). Formation and transport of colloids also may be a mechanism for the movement of plutonium (Essington and Fowler 1976; Nevissi et al 1976) even though the importance of this as a groundwater mechanism appears overstated. Harnish et al (1994) attempted to examine plutonium colloids at RFETS but used data which were less than detection limits to support their conclusions and did not consider the potential artifacts produced by emplacement of groundwater monitoring wells through plutonium contaminated soil. These problems also affected later work by these authors (Harnish et al 1996).

A number of studies have suggested that plutonium is not absolutely chemically immobile in the environment under all conditions. Noshkin et al (1976) for example showed that at the

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Enewetak Atoll plutonium had migrated 80 m downward to groundwater. This environment contrasts with that of RFETS: the actinides were initially trapped in calcium oxides, hydroxides, and carbonates; the seawater is high in complexing anions; and the soils and sediments lack both silicate and metal sesquioxides to sorb plutonium. It therefore is not too surprising that the authors found Pu 239+240 to be very mobile throughout the water saturated coral sand environments (Essington and Fowler 1976).

Litaor and Ibrahim (1996) claimed that plutonium might be chemically mobilized under environmental conditions at RFETS, but the mechanism by which movement would occur is not defined. The Actinide Migration Panel will evaluate the data.

Rusin et al (1994) demonstrated that iron reducing bacteria in combination with strong artificial complexing agents mediate the solubilization of hydrous $\text{PuO}_2(\text{s})$ under anaerobic conditions. As much as 90% of PuO_2 was "biosolubilized" experimentally in six to seven days. As will be discussed in subsequent sections, even relatively weak artificial complexing agents such as citrate can move PuO_2 into solution when used in conjunction with strong reducing agents such as dithionite. Strong artificial complexing agents such as EDTA or the nitrilotriacetic acid (NTA) used by Rusin et al (1994) can solubilize $\text{PuO}_2(\text{s})$ under weakly reducing conditions such as those induced by iron reducing bacteria. It is unclear whether reduction improves the removal efficiency with strong synthetic complexants in all cases since experiment results are mixed (Refer to Treatability Studies Appendix A).

3.3 AMERICIUM

3.3.1 Origin and Geochemical Behavior of Americium

Americium at RFETS primarily forms as a result of the decay of Pu 241 (half life = 14.4 years) and ingrows within weapons plutonium as the relatively short lived Pu 241 decays.

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As with plutonium americium is not found in nature but a world wide background occurs as a result of bomb tests

3 3 2 Geochemistry of Americium

Americium can exist in multiple oxidation states (III IV V VI) but is expected to be in the III state in aerated waters in the absence of oxidants other than atmospheric oxygen The geochemical behavior of americium in the environment is similar to that of plutonium both actinides tend to be strongly adsorbed to the solid phase under neutral to alkaline oxidizing conditions

As discussed above the major reactions influencing the environmental fate of americium are formation of complexes with anions and natural organic matter (controlled by pH and strength of complexing anions) precipitation (controlled by solution composition) and sorption (controlled by solution composition and the nature of solid phases) Americium also may form psuedo and true colloids which could potentially migrate through groundwater although evidence of such migration is equivocal (Silva and Nitsche 1995)

Am^{3+} should behave in much the same manner as Pu^{3+} and the behavior of Am^{3+} may help elucidate the environmental behavior of trivalent actinides The distribution of Am^{3+} and Pu^{4+} at RFETS generally are similar This suggests that differences in the environmental chemistry of Am^{3+} and Pu^{4+} may not significantly affect their environmental behavior over a period of decades and that reduction of Pu^{4+} to Pu^{3+} under natural conditions may not quickly affect the distribution of plutonium

3 4 DETERMINING PARTITION COEFFICIENTS FOR ACTINIDES

As stated above sorption studies remain empirical and distribution coefficients which are measured are only valid under very limited conditions (Dozol and Hagemann 1993) One

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problem is the partially irreversible nature of sorption/desorption reactions. Some of this behavior is due to artifacts, but experiments on the sorption of plutonium by red clay show that sorption/desorption reactions remain irreversible even after possible artifacts are accounted for (Higgo and Rees 1986). The problem of determining actinide K_d s therefore becomes one of determining the limited conditions most likely to represent natural conditions.

Prediction of aqueous actinide transport in soils and groundwater is dependent on an understanding of contaminant mobility in the soil/water environment. Estimates of contaminant mobility are normally obtained using a parameter known as the K_d , which is a quantitative measure of the degree of sorption for an individual contaminant that exists between a solution and a solid phase under specific geochemical conditions. K_d values are used in contaminant transport equations to calculate migration rates for long term predictions of contaminant movement and behavior in solution.

The value of K_d for each compound or element is a function of the geochemical behavior of that compound or element, as well as the composition and characteristics of the sorbent. In turn, the geochemical behavior of a compound is controlled by conditions such as redox potential (Eh), pH, and solution composition. Low K_d values indicate that contaminants are weakly sorbed to the solid phase, resulting in a relatively high aqueous mobility through soils and sediments. Conversely, high K_d values indicate that contaminants are strongly sorbed to the solid phase, resulting in a relatively low aqueous mobility through soils and sediments. Due to variable conditions, the range of K_d values reported in the literature spans several orders of magnitude for the actinides. As reported in the RFI/RI Report for OU3 at RFETS (DOE 1996), the ranges of published values for uranium, plutonium, and americium are 0 to 4,400, 0.4 to 87,000,000, and 0 to 47,230, respectively (DOE 1996). Representative K_d values given in the OU3 RFI/RI Report are 1,550 for U-234, 4,500 for plutonium, and 700 for americium (DOE 1996).

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Jakubick (1976) estimated a K_d of 5 000 for plutonium in partially saturated loamy soil with pH=6. The estimated vertical transport was 0.8 cm/yr. Jakubick (1976) suggested that PuO_2 migrated in the form of small discrete particles. If this is the case, then the migration rate of the plutonium would correspond to physical properties of a given soil. A study by Routson et al (1975) experimentally determined K_d values for americium. For soils in eastern Washington State, K_d values were greater than 1 200 and were not influenced by different concentrations of calcium or sodium ions. Glover et al (1976) conducted equilibrium sorption measurements on different soils for plutonium and americium nitrates. K_d values for plutonium nitrates ranged from 35 to 14 000, whereas those for americium ranged from 82 to 10 000. Clay and sand content, along with cation-exchange capacity (CEC), were determined to be the most important factors in plutonium and americium immobilization or retardation in the soils, suggesting that conventional ion-exchange processes were responsible.

The OU3 RFI/RI Report (DOE 1996) notes that the presence of multiple oxidation states and irreversible reactions between them makes the prediction of long term behavior of plutonium in aquatic systems difficult. The report also claims that environmental behavior of plutonium is complicated by the existence of ionic, particulate, and colloidal and pseudo-colloidal plutonium in the water column and notes that the adsorption of plutonium on sediments is not fully reversible "due to colloid formation" and changes in the oxidation state. DOE (1996) also states that the K_d of plutonium may be lower under reducing conditions than under oxidizing conditions.

One method of determining a K_d involves the calculation of apparent K_d values using data for soils and associated interstitial waters. The raw data to perform such calculations for RFETS have been evaluated for groundwater and surface water, unfortunately, dissolved results are close to or less than detection limits, rendering the values highly uncertain.

K_d s for plutonium and americium in groundwater and surface water also may be calculated on the basis of total plutonium versus total suspended solids (TSS) and total americium versus TSS correlations. These determinations assume that (in the case of plutonium)

$$Pu_{(tot)} (pCi / L) = Pu_{(diss)} (pCi / L) + Pu_{(part)} (pCi / g) * TSS (g / L)$$

and

$$K_d = Pu_{(part)} / Pu_{(diss)}$$

The K_d value is then calculated by dividing the slope of the total Pu TSS regression line (i.e. $Pu_{(part)}$) to the intercept value (i.e. $Pu_{(diss)}$) and correcting the result for proper units. These calculations result in estimated K_d values for plutonium and americium that are in the 10^3 to 10^5 L/kg range. The results are not presented here because further work is necessary to document the conditions under which the coefficients are valid and to systematically determine the errors associated with the determination.

Hursthouse and Livens (1993) used soil and interstitial water data to estimate K_d values for plutonium and americium in sediment profiles. Their K_d values ranged from 1.9×10^5 to 9.0×10^5 for plutonium and 11×10^5 to 39×10^5 for americium (Hursthouse and Livens 1993).

In summary, the range of plutonium K_d values reported in the literature spans as much as eight orders of magnitude, and reliable site specific K_d values have yet to be empirically determined for RFETS soils. The heterogeneous nature of geologic materials makes it difficult to extrapolate from one site to another, although sites with similar geological, geochemical, and climatic conditions may be anticipated to have similar K_d values.

As part of the 1997 Actinide Migration Studies, apparent K_d values will be determined for RFETS soils and sediments. The soils will be from the 903 Pad Area and the SID, and the sediments will be from Ponds B 1, B 5, and C 2. These sorption studies are intended to supply empirical measurement for K_d s for conditions which are directly applicable to RFETS.

The soil and sediment samples are chosen to represent the soil types most affected by plutonium. The water for laboratory experiments was obtained from monitoring wells at the Site and will contain anion and dissolved organic matter at concentrations representative of conditions at the site. Pu 242 will be used as a surrogate for the behavior of Pu 239 in soil. Proper precautions will be taken to ensure that Pu 242 is in correct valence during experiments and to prevent the problems with valence induced artifacts which affected earlier work. The importance of aging of actinides in soils on reversibility of sorption also Pu 239 should be investigated. The study should include a study of sorption and desorption kinetics of Pu 242 solutions onto and off of RFETS solid phases.

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4 0 ACTINIDE DISTRIBUTION IN SOILS AT RFETS

4 1 PLUTONIUM AND AMERICIUM IN BACKGROUND SOILS

Fallout from nuclear tests has contributed plutonium and americium to soils worldwide. Small amounts of plutonium and americium are found as background in all surface soils. These background levels are used as a benchmark at RFETS to determine which increased activity concentrations of americium or plutonium are due to Site activities.

4 1 1 Plutonium and Americium in Background Surface Soils Near RFETS

Surface soil is defined as the upper 6 inches (15 cm) of soil by the RFCA (DOE 1996b). Two background surface soil sampling programs have been completed at RFETS. Background surface soils samples were collected from the Rock Creek area of the Buffer Zone to the north of the RFETS IA. Results were reported in the *Background Geochemical Characterization Report (BGCR)* (DOE 1993). The Background Soils Characterization Program was undertaken in 1994 (DOE 1994) due to on going concern whether the Rock Creek soils are truly not influenced by plant activities and have representative background concentrations of radionuclides and other inorganic elements. Samples were collected in the Boulder County Open Space just north of RFETS and from 20 locations with soils similar to those of RFETS. Results of the program are presented in the *Geochemical Characterization of Background Surface Soils Background Soils Characterization Program (BSCP)* (DOE 1995a).

Results for the two sampling efforts were similar. The reported means and standard deviations for plutonium and Americium are given in Table 4 1. Levels of plutonium in surface soils varied from 0.026 to 0.1 pCi/g in the BGCR (Rock Creek) data. In the BSCP data surface soil plutonium levels ranged from 0.017 to 0.072 pCi/g.

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Americium surface soil levels ranged from 0.01 to 0.036 pCi/g in the Rock Creek data. In the BSCP data surface soil americium levels ranged from 0.001 to 0.025 pCi/g.

Table 4-1 Means and Standard Deviations for Plutonium and Americium in Background Surface Soil Data

Isotope	BGCR		BSCP	
	Mean	Standard Deviation	Mean	Standard Deviation
	pCi/g			
Americium 241	0.02	0.007	0.011	0.006
Plutonium 239/240	0.055	0.014	0.038	0.014

4.1.2 Plutonium and Americium in Background Subsurface Soils at RFETS

The BGCR also presents results of background subsurface soil sampling. Samples were taken to the north and south of the IA well out of the zone of influence of plant activities. Plutonium and americium are virtually at the limit of detection in the samples of background subsurface soils with the error terms as large or larger than the results (DOE 1993). For statistical and reporting purposes all results are considered detections although they may be below the detection limit. Reported activity-concentrations for plutonium in subsurface soils ranged from 0.01 to 0.03 pCi/g with a mean of 0.004 (standard deviation=0.007 pCi/g). Americium activity/concentrations in subsurface soils ranged from 0.015 to 0.01 with a mean of 0.002 pCi/g (standard deviation=0.007 pCi/g).

4.1.3 Distribution of Plutonium and Americium in Surface Soils at RFETS

Plutonium and americium are generally not as chemically mobile as uranium in the soil environment of RFETS (See Section 3.0). The primary mechanism of migration is physical transport of plutonium and americium-contaminated surface soils by wind or water which has distributed these two actinides across the downwind and downslope soilscape (See Section

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7) The spatial distribution of RFETS plutonium was estimated by two early studies (Krey and Hardy 1970 and Seed et al 1971) An aerial radiological survey was conducted in 1989 to measure both natural and anthropogenic gamma radiation from the terrain surface in and around the Site (EG&G 1990) In 1990 an in situ high purity germanium (HPGe) survey was conducted to the East of the 903 Pad Lip Area (EG&G 1991) A second HPGe survey of the 903 Pad and Lip Area was performed in 1994 (RMRS 1996) Data were collected in support of remedial investigations for most former OUs and these data are included in the data set used for the discussion that follows Many other surface soil samples have also been collected at RFETS as part of various projects All surface soil data from the RFEDS have been compiled for the figures and discussion in this section

Plutonium activities in surface soils around RFETS range from 7300 pCi/g near the 903 Pad to background in many areas Americium activities in surface soils range from background to 295 pCi/g also in the 903 Pad Area About 82 percent of the surface soil results for plutonium 239/240 and about 90 percent of the results for Am 241 are less than 1.0 pCi/g Americium activities and spatial distribution show a high correlation to those of plutonium (DOE 1995b)

Figures 4.1 and 4.2 show areas at RFETS with plutonium and americium activities estimated to be above 1 pCi/g The contours were produced by kriging grids produced using data for all surface soil locations shown in Figure 4.3 The reported analytical results for over 1800 sampling locations were used The contours were then examined and edited using professional judgment The figures show that surface soils with elevated activities for plutonium and americium occur to the east of the IA and within the Protected Area The highest activities are associated with the 903 Pad and Lip Area

Figure 4.3 presents the results of screening all surface soil and sediment data available from the RFEDS against the radionuclide action levels developed for the RFCA (DOE 1996) Table 4.1 shows the surface soil action levels for each radionuclide The maximum analytical

result for each radionuclide at each location was divided by the appropriate action level. The resultant ratios were then summed across radionuclides for each location to obtain the action level sum of ratios as described in the *Action Levels for Radionuclides in Soils for the Rocky Flats Cleanup Agreement* (DOE 1996) and shown below

$$\text{Sum of Ratios} = \text{Pu}_r/\text{Pu}_{al} + \text{Am}_r/\text{Am}_{al} + \text{U234}_r/\text{U234}_{al} + \text{U235}_r/\text{U235}_{al} + \text{U238}_r/\text{U238}_{al}$$

Where subscript *r* denotes an analytical result and subscript *al* denotes the appropriate surface soil action level (Table 4-2). A sum of ratios was calculated for the ratio of a result to both the Tier I and Tier II action levels. If the sum of ratios calculated using the Tier I action levels for the five actinides was greater than or equal to one, then the location was plotted as above Tier I (Figure 4-3). If the Tier I sum was less than one, then the sum for the Tier II ratios was used. When the Tier II sum of ratios was greater than one, the location was plotted as above Tier II but below Tier I, and when the Tier II sum was below one, the location was plotted as below Tier II.

Table 4-2 Radionuclide Surface Soil Action Levels (pCi/g)

Radionuclide	Tier I Action Level Buffer Zone	Tier I Action Level IA	Tier II Action Level Site-Wide
Americium 241	215	209	38
Plutonium 239/240	1429	1088	252
Uranium 234	1738	1627	307
Uranium 235	135	113	24
Uranium 238	586	506	103

By comparing Figure 4-3 to the contour maps (Figures 4-1, 4-2, 4-9, 4-10, and 4-11) it can be determined which Tier I exceedances are mainly due to plutonium and americium. The exceedances of Tier I and Tier II action levels due mainly to plutonium and americium are in

the area influenced by the 903 Pad. Exceedances in the other areas are mainly due to uranium isotopes which are discussed in Section 4.2

4.1.4 Distribution of Plutonium and Americium in the Subsurface Soils at RFETS

Subsurface soil samples have been collected from boreholes drilled for many projects at RFETS. The samples are well distributed across the site (Figure 4.5). Samples have also been collected from trenches and pits specifically excavated for sampling purposes (DOE 1995b). All subsurface soil data from RFEDS were used in this section.

Figure 4.5 shows borehole and well locations at RFETS where subsurface soil samples were collected. Plutonium activities in subsurface soils around RFETS range from 1486 pCi/g in the East Trenches Area to background in many areas. Americium activities in subsurface soils range from background to 208 pCi/g in the East Trenches Area.

Locations with concentration activities above Tier II or Tier I radionuclide action levels are shown in Figure 4.5. The same action levels are currently applied to subsurface soils for evaluation as for surface soils (Table 4.1). The method used to produce the ratios for subsurface soils depicted in Figure 4.5 was the same as explained above for surface soils (Figure 4.3). Results show that the areas with the highest concentration/activities are located in the East Trenches Area (near IHSS 110 [trench T 3] and in IHSS 111.4 [trench T 7]) to the west of the old landfill in and around IHSSs 133.1, 133.3, and 133.4, and in the area of the present landfill (IHSS 114).

Results from the sampling of the trenches excavated in support of the former OU 2 RFI/RI (DOE 1995b) make it possible to examine the distribution of americium and plutonium in surface and subsurface soils in the 903 Pad and Lip Area. Trench locations are shown in Figure 4.6. Profiles for three representative trenches with greatly varying concentration

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/activities of americium and plutonium are shown in Figures 4-7 and 4-8. The distributions are very similar for americium and plutonium at all concentration activities. Krey and Hardy (1970) also noted an apparent similarity in the downward transport mechanism at all sites.

The actinide contamination in the 903 Pad Area originated from releases in the 903 drum storage area (the present 903 Pad) from 1958 to 1967 when the drums were removed. Contamination from the releases was then distributed to the south and east of the storage area by wind and surface water runoff. The soil profile data show the results of all combined migration factors that have been active since the plutonium was deposited in the surface soil. There has been little movement of plutonium and americium below a depth of 20 cm (about 8 in) during the 25 years since the release occurred even with very high surface soil concentration activities. This agrees well with the findings of Krey et al (1976) and Litaor et al (1994) that 90 percent of the total plutonium is contained within the upper 10 to 12 cm of soils downwind of the 903 Pad. Krey et al (1976) suggested that for depths below 6 cm diffusion transport better describes plutonium distributions than particle transport via infiltration flow.

4.2 URANIUM IN BACKGROUND SOILS

Bolivar et al (1978) evaluated uranium distributions in waters and sediments of the Front Range and speculated that higher uranium concentrations in water samples are probably due to leaching of uraniumiferous strata in the Pierre and Laramie formations. This same study noted that the granites of the Front Range are known to be rich in uranium and that the South Platte River is anomalously rich in uranium compared to most other rivers of its size. The types of rocks in the RFETS area (claystones of the Laramie Formation and Precambrian granites) the presence of nearby ore grade uranium deposits (i.e. the Schwartzwalder mine) and a generally alkaline and oxidizing environment in the near

Figure 4-7 Depth Distribution of Americium 241 in Soils Near the 903 Pad

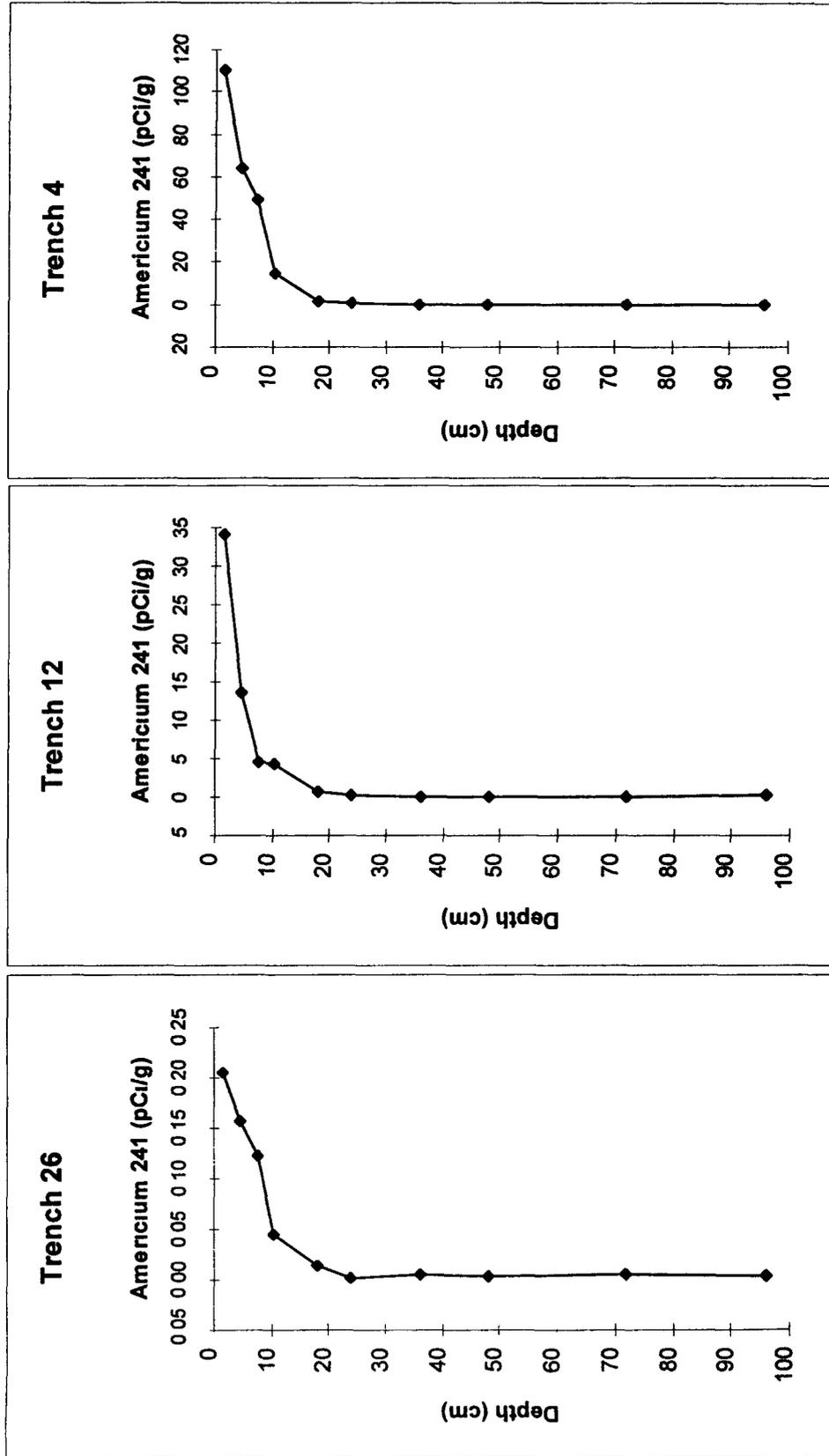
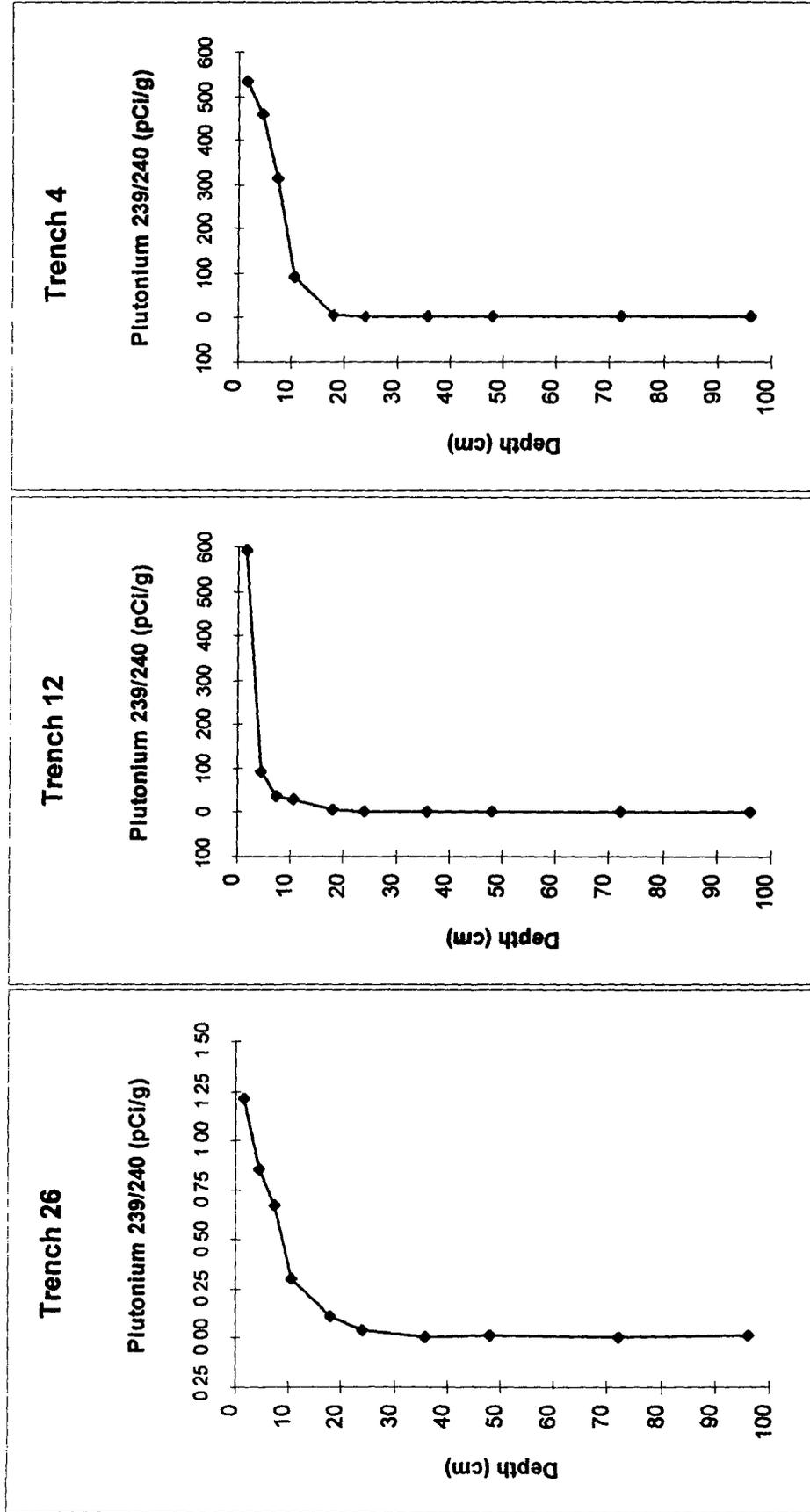


Figure 4-8 Depth Distribution of Plutonium 239/240 in Soils Near the 903 Pad



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subsurface soils contribute to the likelihood of high and variable concentrations of uranium existing in soils in the RFETS area

4 2 1 Uranium in Background Surface Soils at RFETS

Data compiled for the BGCR (DOE 1993) and the BSCP (DOE 1995a) gave very similar results for the mean activities for uranium isotopes in surface soil samples collected from background areas near RFETS. The means and standard deviations from the two data sets for uranium 233/234, 235, and 238 are presented in Table 4 2. The distributions for all the uranium isotopes were determined to be lognormally distributed.

Table 4-3 Means and Standard Deviations for Uranium Isotopes in Background Surface Soil Data

Isotope	BGCR		BSCP	
	Mean	Standard Deviation	Mean	Standard Deviation
	pCi/g			
Uranium 233/234	1.145	0.156	1.097	0.578
Uranium 235	0.053	0.033	0.054	0.020
Uranium 238	1.183	0.188	1.090	0.455

4 2 2 Uranium in Background Subsurface Soils at RFETS

Subsurface soils include soils and geologic materials collected from the drilling of boreholes at depths below 6 inches. Subsurface materials include alluvium, colluvium, weathered bedrock, and sandstones. For subsurface soils (boreholes) at RFETS, means and standard deviations for uranium isotopes were calculated from data presented in the BGCR (DOE 1993). The background means and standard deviations for U 233/234, 235, 238 are shown in Table 4-4.

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Table 4-4 Means and Standard Deviations for Uranium Isotopes in Background Subsurface Soil Data

Isotope	BGCR	
	Mean	Standard Deviation
	pCi/g	
Uranium 233/234	0 779	0 932
Uranium 235	0 022	0 046
Uranium 238	0 733	0 376

4 2 3 Uranium in Surface Soils at RFETS

Analytical data for uranium in surface soil samples collected across RFETS show a wide range of uranium isotope activities uranium 233/234 activities in surface soils at RFETS range from 0 218 pCi/g to 2800 pCi/g activities for uranium 235 range from -0 02 pCi/g to 670 pCi/g and uranium 238 activities ranged from 0 25 pCi/g to 38000 pCi/g Figures 4-9 4 10 and 4 11 show the distribution of the uranium isotopes in surface soils at RFETS Elevated concentration activities for uranium are found around the Old Landfill (IHSS 115) and the Solar Ponds The exceedances of Tier I and Tier II action levels in these areas (Figure 4 3) are due to the elevated uranium isotope activities

Litaor (1995) studied the soils east of the RFETS IA downwind of the 903 Pad and concluded that Proposed wind-dispersal mechanisms were not consistent with the spatial distribution of U isotopes in surface soils east of the 903 Pad and that there was no clear relationship between known uranium burial and spill sites and the present distribution of U 235 in the soils He noted that the lack of similarity in spatial distribution between Pu 239+240 and U isotopes probably resulted from the higher solubility and leachability of U isotopes compared with Pu 239+240 in the soil system

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Isopleth maps of the uranium isotopes activities in surface soils do not support all of the above conclusions (Figures 4 9 4 10 and 4 11) It is true no elevated uranium 235 activities occur in surface soils to the east of the 903 Pad (Figure 4 10) however there is an area of elevated U 238 activity to the east of the 903 Pad that correlates with the areas of highest americium and plutonium activities (Figures 4 1 4 2 and 4 11) There are elevated activities of uranium isotopes in other known source areas such as the Old Landfill the Ash Pits and the Solar Ponds The lack of a well defined spacial distribution of the uranium isotopes to the east of the 903 Pad compared to plutonium and americium is probably due to their initially lower activities in the materials released as well as to their higher solubility This has allowed the uranium isotopes to leach deeper into the soil profile

4 2 4 Uranium in Subsurface Soils at RFETS

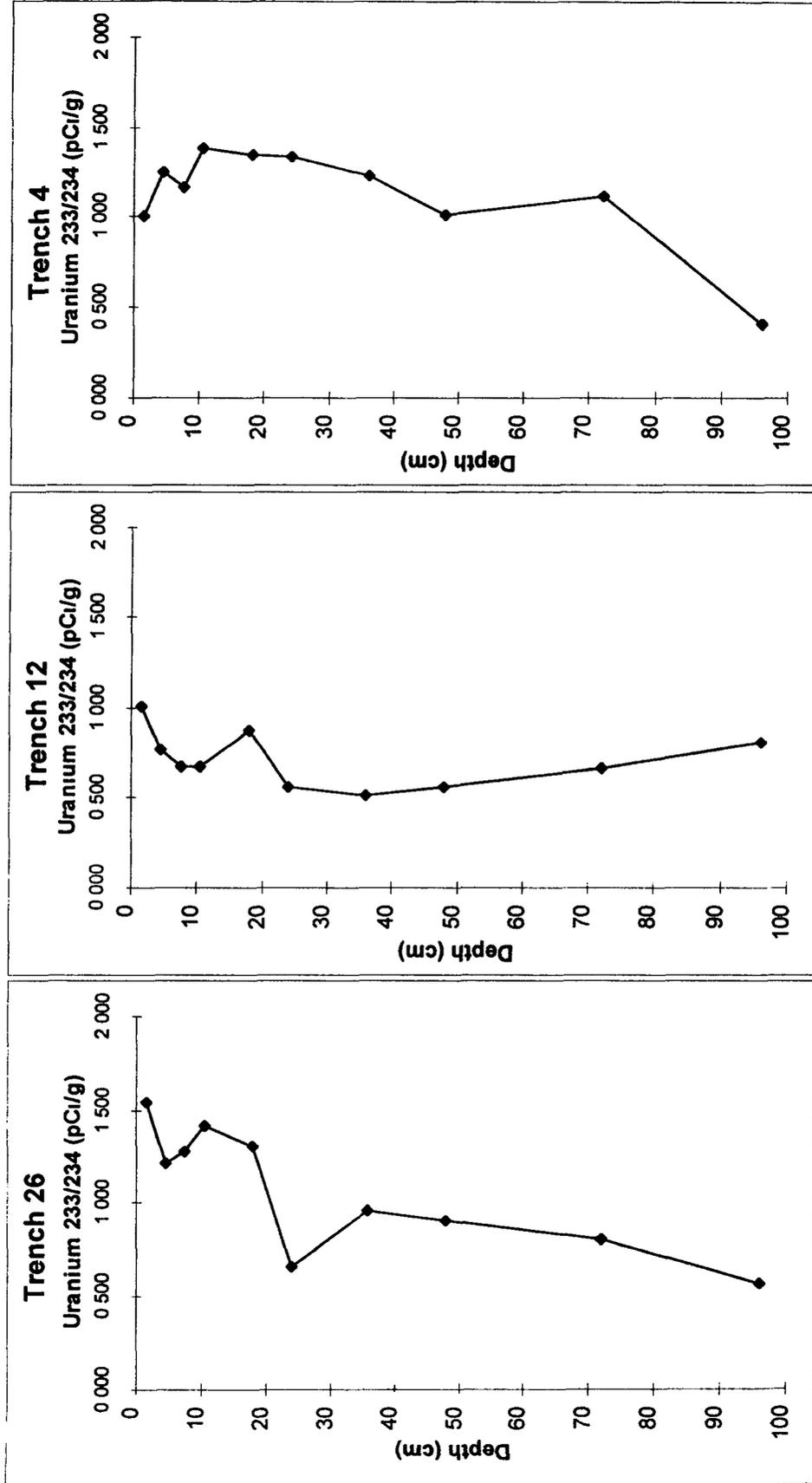
Minimum and maximum values for uranium isotopes in samples of subsurface soils (subsurface materials from below 6 inches) at RFETS are displayed in Table 4-4 The maximum values are considerably higher than those obtained for samples collected from background areas on Site (DOE 1993a)

**Table 4-5 Minimum and Maximum Activities for Uranium Isotopes in RFETS
Subsurface Soil Data**

Isotope	minimum	maximum
	pCi/g	
Uranium 233/234	0 21	2875
Uranium 235	0 023	288
Uranium 238	0	24790

Figure 4 5 shows the subsurface soil sampling locations and where radionuclide action levels are exceeded The exceedances in the areas of the Old Landfill (IHSS 115) the New Landfill (IHSS 114) and some locations in the East Trenches Area are due to elevated uranium

Figure 4-11 Depth Distribution of Uranium $^{233/234}$ in Soils Near the 903 Pad



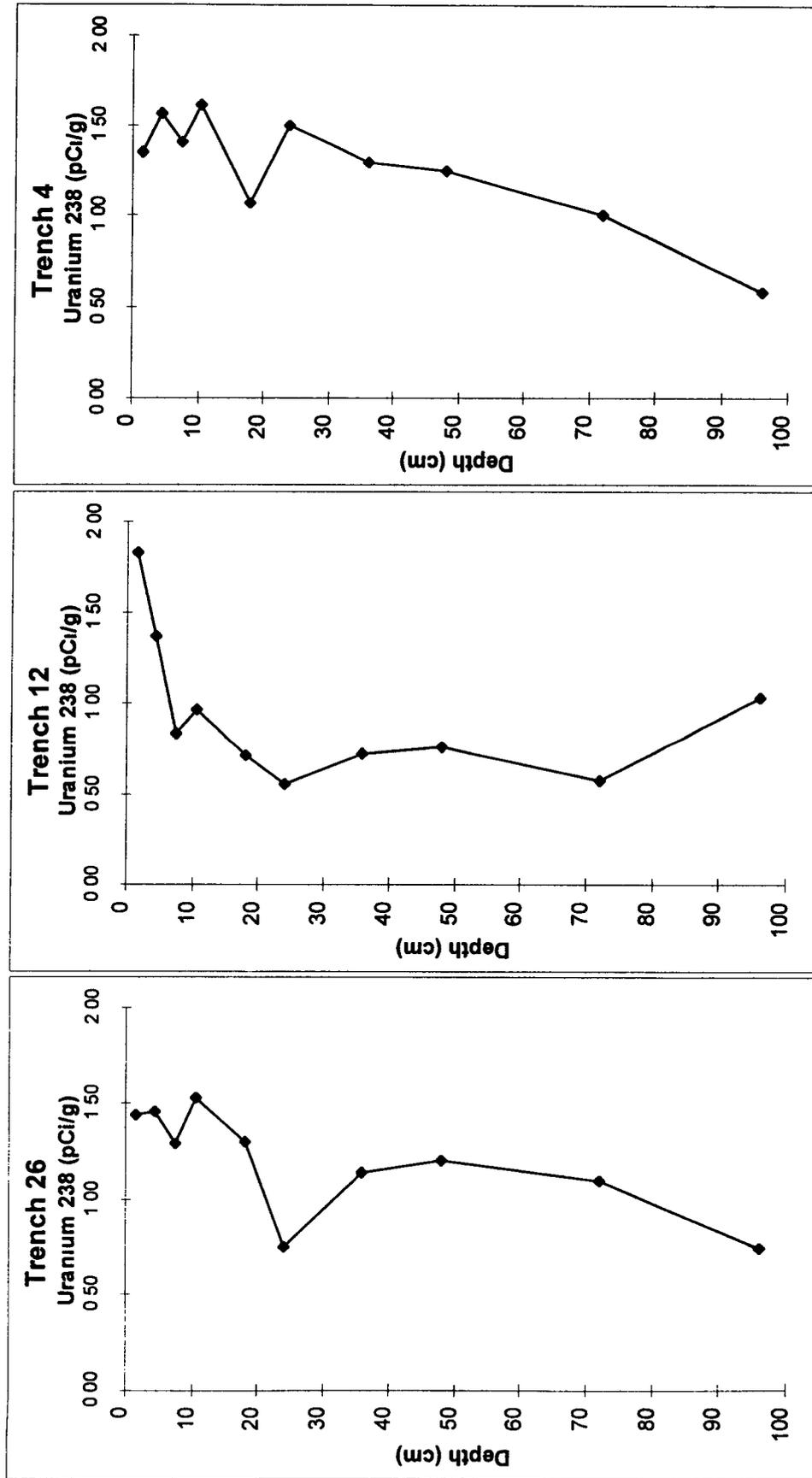
80

activities The high activity/concentrations of uranium in these areas are likely due to buried contamination rather than the leaching of surface deposited uranium into the subsurface soil. However, uranium is subject to dissolution and leaching in the near neutral, generally oxidizing environment found at RFETS and is more geochemically mobile than plutonium and americium (as previously discussed in Section 3). This will be discussed in relation to elevated concentrations of uranium in the former OU4 area in Section 5.

Soil profile studies performed in support of the OU2 RFI/RI indicated uranium leached downward under the RFETS 903 Pad (Litaor 1995). Figures 4.11 and 4.12 show the vertical distribution of U-233/234 and U-238 in soils near the 903 Pad. The uranium activities are very close to background levels (Section 4.2.2). Trenches 4 and 12 had high concentration/activities of plutonium and americium in the surface soil (Figures 4.7 and 4.8) but do not show a clear relationship of uranium concentration/activities to depth. This may be indicative of naturally occurring uranium.

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Figure 4-12 Depth Distribution of Uranium 238 in Soils Near the 903 Pad



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5 0 ACTINIDE DISTRIBUTION IN GROUNDWATER AT RFETS

This section presents data on the distribution of actinides in both background and non background monitoring wells at RFETS. Two background studies at RFETS have evaluated actinides in groundwater: the BGCR (DOE 1993) and the draft Comparison for Radionuclides in Groundwater (DOE 1997). Areas west, north, and south of the IA that are hydrologically upgradient or sidegradient to the IA are considered to be background areas (Figure 5.1). The background values reported by DOE (1993) included results from wells screened into the weathered bedrock as well as in the alluvial and colluvial materials that lay above it. The 1997 study was initiated to develop background values that are representative of the unconsolidated materials overlying the bedrock. The 1997 study also includes a larger number of wells sampled over a longer period of time than the 1993 study.

5 1 PLUTONIUM AND AMERICIUM IN BACKGROUND GROUNDWATER AT RFETS

The 1993 study (DOE 1993) reported only three records for americium and one record for plutonium in filtered (0.45 µm) samples of background groundwater. The 1997 study (DOE 1997) contains 24 and 28 filtered analyses for americium and plutonium, respectively (Table 5.1). The reported activities for filtered groundwater are very low. Many more analyses are available for unfiltered or total samples. The maximum reported activities for both americium and plutonium in unfiltered samples of background groundwater in the 1993 study are greater than the 0.15 pCi/L RFCA groundwater action level. However, the great majority of samples have very low reported values.

5 1 1 Plutonium and Americium in Groundwater at RFETS

Site wide groundwater at RFETS shows a narrow range of activities in filtered samples for americium-241 and plutonium-239/240 with maximums of 0.47 and 2 pCi/L, respectively.

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Table 5 1 Americium 241 and Plutonium 239/240 in Background Groundwater

	Obs	Mean	Stand Dev	BG Bench mark	Min	Max	Obs	Mean	Stand Dev	Min	Max	
	DOE 1993						DOE, 1997					
	Filtered (pCi/L)											
Americium 241	24	0 003	0 005	0 013	-0 007	0 018	2	0 01	0 01	0 0	0 02	
Plutonium 239/240	25	0 002	0 004	0 01	-0 004	0 014	1	0 01	--	0 01	0 01	
Unfiltered (pCi/L)												
Americium 241	275	0 006	0 015	0 036	-0 02	0 19	183	0 01	0 01	-0 01	0 10	
Plutonium 239/240	289	0 005	0 021	0 047	-0 05	0 224	194	0 004	0 02	-0 01	0 22	

The maximums for unfiltered americium and plutonium samples are 46 5 and 13 4 pCi/L respectively. The highest activities for both plutonium and americium were measured in samples collected from the vicinity of the 903 Pad as shown in figures 5 2 and 5 3.

The maximums, minimums, means, and standard deviations for groundwater data collected from 1991 through 1996 for each location with filtered or unfiltered americium or plutonium activities above the groundwater action levels (DOE 1996) are shown in Table 5 2 and are depicted in Figures 5 2 and 5 3. The figure shows a group of wells in the 903 Pad Area with elevated americium and plutonium activities in ground water samples. Activities in the sampled wells are generally below the background benchmarks (background mean plus two standard deviations) of 0 05 pCi/L for plutonium and 0 04 pCi/L for americium. Two wells (locations 09091 and 06991) have significantly higher average activities (Table 5 2). The contamination may be due to the movement of contaminated soil into the well hole during drilling and localized around each well which has been documented in other wells (EG&G 1995a). Since these wells were drilled, new technology has been developed at RFETS that eliminates the downward movement of contaminants in soil during drilling (EG&G 1995a). A method for determining the source of the contamination and whether it is more widely distributed in the groundwater is to develop new wells using the RFETS aseptic method.

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Soils data (Section 4 0) knowledge of actinide behavior in the environment (Section 3 0) and groundwater data do not indicate that large amounts of plutonium or americium are moving into and with the ground water at RFETS. Before any remediation decisions are made with respect to plutonium and americium in specific groundwater wells, one wells should be drilled using aseptic methods to determine the source and extent of the groundwater contamination.

5 2 URANIUM ISOTOPES IN BACKGROUND GROUNDWATER AT RFETS

Uranium isotopes in samples of filtered (0 45 µm) and unfiltered groundwater collected from the UHSU exhibit a wide range of reported activities (Table 5 3). The maximum values are from well B205589 which lies along the Rock Creek drainage just south of Highway 128 (Figure 5 1). There is no evidence that this well has any anthropogenic contamination. This is confirmed by both the uranium atom and activity ratios (Table 5 5 and 5-6) which indicate the uranium is naturally occurring. These ratios are explained and their significance discussed in Section 5 2 2 and 5 2 3. This well along with all other background wells is located outside the area in which groundwater has been impacted by releases from RFETS.

Table 5 3 Uranium Isotopes in Background Groundwater

	Obs	Mean	Std Dev	BG Bench mark	Min	Max	Obs	Mean	Std Dev	Min	Max	
	DOE 1993						DOE 1997					
	Filtered (pCi/L)											
Uranium 233/234	287	6 55	27 1	60 8	-0 078	199 5	207	6 91	25 4	-0 02	199 5	
Uranium 235	288	0 023	0 78	1 58	-0 035	5 35	207	0 195	0 635	-0 04	4 80	
Uranium 238	286	4 60	18 6	41 8	-0 04	135 6	177	4 83	17 7	-0 04	135 6	
Unfiltered (pCi/L)												
Uranium 233/234	39	14 7	35 3	85 3	0 0	164	35	15 6	38 75	0 0	164	
Uranium 235	39	0 69	1 54	3 77	-0 01	6 48	35	0 617	1 38	-0 02	6 29	
Uranium 238	41	10 5	24 9	60 3	0 0	108	22	10 8	27 7	0 0	108	

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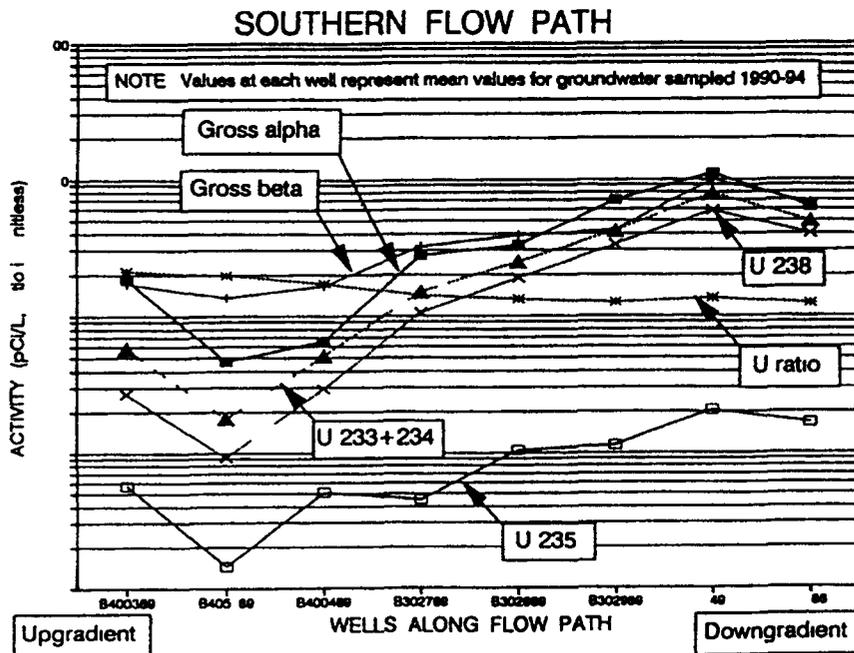
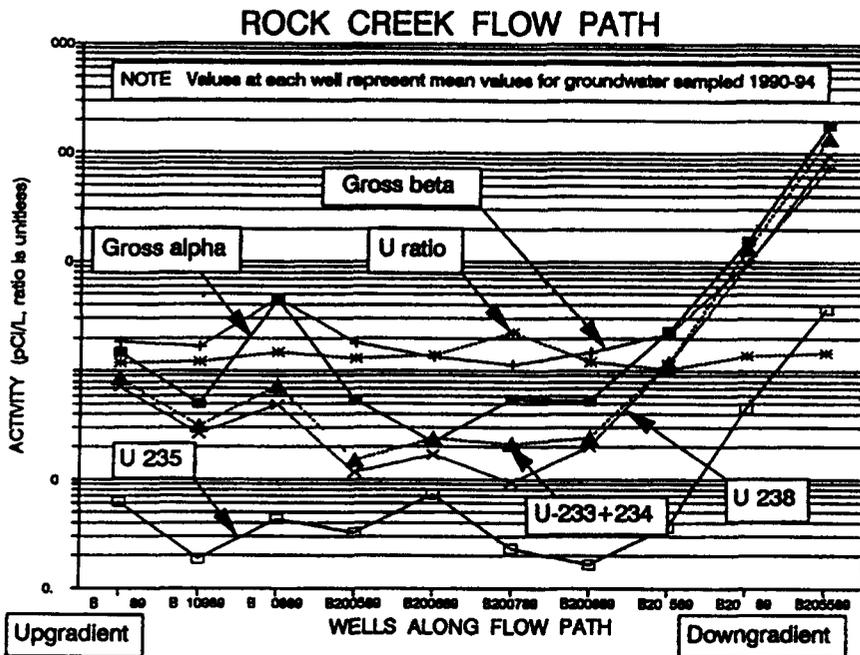
In addition to data from the 1993 and 1997 studies evaluations were performed in the *Groundwater Geochemistry Report* (EG&G 1995b) that show the geochemical evolution in the composition of shallow groundwater along flow paths at RFETS For the Rock Creek and Southern flow paths (Figure 5 1) concentrations of dissolved uranium isotopes show an increase as water moves west to east along the flow path (Figure 5-4) The increase in dissolved uranium observed in the report (EG&G 1995b) may be related to increasing levels of dissolved carbonate (which complexes with uranium to increase the solubility of uranium) or to naturally occurring accumulations of uranium The ratio of uranium 238 to uranium 235 was shown to be consistently in the range of naturally occurring uranium

The large variability shown for levels of uranium in background groundwater is expected considering the inherent heterogeneity of geologic materials and the presence of ore grade uranium deposits within 10 miles of RFETS (Schwartzwalder mine near Ralston Reservoir) A recent study performed by the Jefferson County Health Department (Moody and Morse 1992) found high levels of uranium in the groundwater of Coal Creek Canyon which is upgrade of RFETS The Jefferson County study compiled data for groundwater samples collected from 33 domestic wells in Coal Creek Canyon Uranium (total) ranged from 13 to 1200 pCi/L with a mean and standard deviation of 1749 and 3391 pCi/L respectively

5 2 1 Uranium in Groundwater at RFETS

Analytical data for uranium in groundwater samples collected across RFETS show a wide range of uranium isotope activities The maximum activities in groundwater samples during the years 1991 through 1996 for filtered and unfiltered samples are uranium 233/234 492 and 922 pCi/L uranium 235 7 and 648 pCi/L and uranium 238 325 and 101 pCi/L The RFETS background mean plus two standard deviations which is used as a screening benchmark for ground water evaluations at RFETS is equal to 608 and 853 pCi/L 178

and 3.77 pCi/L and 49 and 60.3 pCi/L for filtered and unfiltered uranium 233/234 235



U ratio = u238 / u235

Figure 5-4 Background Dissolved Uranium Activities in Groundwater Along the Two Flow Paths at RFETS

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 AH

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and 238 respectively (Table 5-3 and DOE 1997). The maximum values for each isotope are all above the background benchmarks. Table 5-4 shows all locations with filtered (dissolved) and unfiltered (total) uranium isotope activities in groundwater above the background benchmarks. The maximums, minimums, means, and standard deviations are given for data collected from each location from 1991 through 1996.

Figures 5-5 through 5-7 show the distribution of dissolved (filtered) uranium isotope activity concentrations across the Site. The groundwater monitoring data collected over the past seven years contain a preponderance of dissolved uranium analyses rather than total (unfiltered) analyses. The reason for this is that DOE and the Colorado Department of Public Health and the Environment (CDPHE) have agreed that groundwater standards for uranium apply to dissolved uranium isotopes. The mean activity for each sampling location over the period from January 1991 through December 1995 was used for plotting. This data is being used as a baseline for groundwater evaluations under the Integrated Monitoring Plan (DOE 1997b).

Uranium 233/234 activity-concentrations above the action level of 1.07 pCi/L have been found across RFETS (Figure 5-5). However, the background benchmark mean plus two standard deviations is 60.7 pCi/L. Values above the benchmark only occur in the Solar Ponds area and down gradient towards North Walnut Creek. This is the area with a known nitrate plume extending from the Solar Ponds toward North Walnut Creek. The presence of high nitrate concentrations in the groundwater may increase the mobility of the uranium (See Section 3.0).

Uranium 235 activity-concentrations above the action level of 1.01 pCi/L are not common but are found in background wells and also in areas affected by Site activities (Figure 5-6). Uranium 235 shows a similar distribution to uranium 233/234. Wells with the highest activity-concentrations are clustered in the Solar Ponds area and down gradient towards North Walnut Creek.

Uranium 238 activity-concentrations above the action level of 0.768 pCi/L have been found across RFETS (Figure 5.7). Wells with activity-concentrations above the background benchmark are located in the Solar Ponds area and to the north and south near the Site boundary. Other wells with activities of uranium 238 near the background benchmark are clustered in the Solar Ponds Area and extend downgradient toward North Walnut Creek. Wells with activity-concentrations above the action level but below the background benchmark are also located in the 881 Hillside area in the South Walnut Creek drainage and near the present landfill.

In summary, Figures 5.5 through 5.7 show that locations having activity-concentrations of uranium isotopes above the action levels are distributed across the Site in both background areas and in areas affected by Site activities. The reasons for the high background levels are discussed in Section 5.2. There are four areas with wells having activity-concentrations above the action levels for the uranium isotopes (but rarely above the background benchmark): the Solar Ponds area, South Walnut Creek drainage, the 881 Hillside area, and the present landfill area. In the following section, $^{238}\text{U}/^{235}\text{U}$ atom ratios and $^{234}\text{U}/^{238}\text{U}$ activity ratios are calculated and will be used to discuss the likelihood that the uranium is anthropogenic or naturally occurring.

5.2.2 Use of $^{238}\text{U}/^{235}\text{U}$ Atom Ratios to Determine Origin of Uranium

The atom ratio of uranium 238 to uranium 235 in naturally occurring uranium is a constant of 137.8. This atom ratio can be used to separate the components of anthropogenic (i.e., enriched or depleted) uranium and naturally occurring uranium. Efurud et al. (1993) used the following equation to transform activities in pCi/L into atom ratios:

$$A = N/\lambda$$

where A = activity N = number of atoms of the isotope λ = decay constant (0.693/half life) λ for uranium 238 = 9.84375×10^{-10} λ for uranium 235 = 1.55103×10^{-10}

Mean atom ratios were calculated for each well location with results above the background benchmark using the mean activity-concentration from 1991 through 1996 (Table 5.5). The ratios vary from 54.5 to 251. Background wells appear with both high and low ratios. Unfortunately, groundwater samples at RFETS have been analyzed using alpha spectroscopy which only estimates the ^{235}U activity. These ratios also do not treat sampling and analytical error and are of limited use. See Section 3.6 for further discussion of atom ratios.

5.2.3 Use of $^{234}\text{U}/^{238}\text{U}$ Activity Ratios to Determine Origin of Uranium

The $^{234}\text{U}/^{238}\text{U}$ activity ratio (as opposed to the $^{238}\text{U}/^{235}\text{U}$ atom ratio) has also been used to distinguish between natural and anthropogenic uranium. Uranium 234 is a product of the uranium 238 decay chain and its abundance is determined by the abundance of uranium 238. Table 5-6 shows activity ratios for filtered uranium 238 and uranium 235 in groundwater calculated for locations with average activities above the background benchmark.

The activity ratios of uranium 234 to uranium 238 are approximately 0.09 in depleted uranium, 1.06 in natural uranium, 5.74 in power reactor fuel, and higher for weapons grade uranium (EG&G 1988). The $^{234}\text{U}/^{238}\text{U}$ activity ratio in for uranium in natural waters usually ranges from one to three (Hess et al. 1985). Ratios that are above 3.0 or below 1.0 suggest the presence of artificially enriched or depleted uranium. The BGCR (DOE 1993a) reported a range of 1.19 to 2.43 for ratios of uranium isotopes in filtered background groundwater well within the expected range for natural waters. The current data shown in Table 5-6 have a range from 0.46 to 2.20. Only two wells have low activity ratios: Well 07391 located about 300 feet south of the 903 Pad and Well 61093 located near the Old Landfill. The analytical method used for RFETS samples does not resolve uranium 233 from uranium 234.

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they are reported together and this may lessen the usefulness of the activity ratio Refer to
Section 3 7 for a discussion of activity ratios

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Table 5 2 Groundwater Sampling Locations with Maximum Filtered or Unfiltered Americium or Plutonium Isotope Activity Concentrations Greater Than the Tier II Action Levels

Location	Max	Min	Mean	Std Dev
Filtered (pCi/L)				
Americium 241				
09091	21 310	0 012	10 661	15 060
1286	0 471	0 471	0 471	
08891	0 435	0 435	0 435	
07191	0 148	0 148	0 148	
06991	0 264	0 008	0 136	0 181
Plutonium 239/240				
08891	1 999	1 999	1 999	
09091	0 813	0 149	0 481	0 469
11791	0 230	0 022	0 126	0 147
72393	0 330	-0 002	0 050	0 124
Unfiltered (pCi/L)				
Americium 241				
09091	46 540	1 400	11 177	14 662
06991	9 730	0 190	1 358	2 947
1286	1 087	0 272	0 680	0 576
P208989	5 289	0 000	0 546	1 667
07191	2 270	0 030	0 503	0 988
P115489	0 380	0 380	0 380	
2286	0 950	0 002	0 272	0 319
06691	0 580	0 160	0 266	0 176
11791	1 321	0 010	0 253	0 359
P313489	0 250	0 250	0 250	
1587	0 650	0 015	0 246	0 229
05193	0 650	0 031	0 233	0 233
0271	0 381	0 066	0 223	0 223
0460	0 508	0 041	0 209	0 211
41691	3 200	0 001	0 196	0 587
08891	0 550	0 010	0 168	0 170
3686	0 160	0 160	0 160	
72093	0 340	0 002	0 114	0 115
06591	0 270	0 022	0 113	0 077
13191	0 597	0 012	0 112	0 157
B204189	0 470	0 003	0 099	0 207
72393	0 221	0 003	0 086	0 079

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Table 5 2 (continued)

12091	1 090	0 000	0 083	0 290
24993	0 150	0 003	0 077	0 104
34791	0 530	-0 015	0 050	0 159
59493	0 200	0 003	0 050	0 075
0987	0 450	0 000	0 045	0 116
09691	0 160	0 000	0 042	0 048
07391	0 340	0 001	0 041	0 105
0487	0 324	-0 004	0 041	0 106
01991	0 375	0 001	0 041	0 105
01491	0 230	0 000	0 038	0 068
7187	0 393	0 000	0 031	0 104
B400289	0 190	0 002	0 030	0 070
B110989	0 161	0 000	0 016	0 044
Plutonium 239/240				
09091	354 6	12 0	94 6	21 8
06991	71 7	1 20	9 82	6 71
11791	13 360	0 003	2 494	3 488
1286	3 650	0 699	2 174	2 087
B204189	10 320	0 000	2 065	4 615
P313489	1 600	1 600	1 600	
06691	3 361	0 832	1 527	1 052
1587	4 300	0 510	1 510	1 218
06591	2 900	0 778	1 505	0 728
2286	4 820	0 000	1 327	1 724
13191	5 024	0 058	1 247	1 320
08891	3 400	0 034	1 052	1 064
0366	0 900	0 900	0 900	
0271	1 261	0 388	0 825	0 617
5671	0 470	0 470	0 470	
72093	1 200	0 008	0 402	0 393
41691	2 204	0 000	0 378	0 487
13491	0 870	0 130	0 343	0 264
72393	0 799	0 005	0 332	0 292
P209189	0 510	0 083	0 325	0 153
09691	1 100	0 048	0 298	0 295
0290	2 510	0 002	0 285	0 834
59493	1 037	0 001	0 261	0 385
00291	0 780	0 042	0 241	0 196
6387	0 386	0 121	0 233	0 097
00191	1 300	0 020	0 204	0 367

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Table 5-2 (continued)

46792	0 530	0 000	0 189	0 296
3686	0 166	0 166	0 166	
0171	0 494	0 023	0 164	0 193
06891	0 290	0 084	0 154	0 083
02991	0 800	0 002	0 112	0 245
24993	0 210	0 009	0 110	0 142
00391	0 570	0 001	0 109	0 169
4286	0 269	0 002	0 106	0 075
25093	0 200	0 008	0 104	0 136
46692	0 970	0 000	0 103	0 289
46892	0 480	0 001	0 097	0 190
1687	0 420	0 007	0 087	0 117
03091	0 350	0 027	0 073	0 111
0374	0 199	0 000	0 067	0 114
05193	0 200	-0 009	0 065	0 068
0486	0 179	0 000	0 064	0 060
11891	0 208	0 007	0 058	0 069
20591	0 230	0 001	0 058	0 115
04591	0 580	0 001	0 052	0 147
22193	0 415	-0 001	0 049	0 137
01991	0 310	0 001	0 042	0 086
70093	0 215	0 000	0 037	0 073
0460	0 072	0 000	0 035	0 027
05191	0 170	0 003	0 030	0 047
07391	0 179	0 004	0 028	0 058
3287	0 171	0 000	0 028	0 063
10991	0 232	0 002	0 027	0 060
03791	0 164	0 000	0 024	0 042
5186	0 347	0 003	0 021	0 081
1490	0 182	0 005	0 021	0 057
46192	0 235	0 002	0 019	0 065
1786	0 210	0 002	0 018	0 045
5086	0 254	0 019	0 012	0 059

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Table 5-4 Groundwater Sampling Locations with Unfiltered and Filtered Uranium Isotope Activity Concentrations Greater Than the Background Benchmark¹

Location	Max	Min	Mean	Std Dev
Unfiltered (pCi/L)				
Uranium 235				
10294	6.48	1.40	3.21	2.24
59993	4.00	0.306	2.15	2.61
Uranium 238				
61093	101	0.026	47.3	40.0
10294	69.5	25.0	43.5	20.7
P114989	68.0	68.0	680	
Filtered (pCi/L)				
Uranium 233/234				
05093	492	56.0	221	131
2886	280	74.5	141	65.8
05193	258	136	199	34.3
B205589	180	2.45	121	51.8
B303089	150	120	135	21.2
3086	139	37.1	98.2	20.5
B305389	134	2.70	22.1	49.3
2689	95.1	95.1	95.1	
B208689	83.5	69.0	72.9	4.32
05393	81.1	81.1	81.1	
B210389	80.9	80.9	80.9	
P208989	73.0	1.73	59.1	19.4
0586	63.0	19.1	39.5	22.1
Uranium 235				
05193	35.7	5.40	10.5	9.12
05093	27.7	1.70	9.53	8.00
2886	11.0	2.32	4.59	2.92
5287	9.12	0.54	1.73	2.09
3086	8.03	2.47	4.31	1.65
B303089	6.90	4.00	5.45	2.05
B210489	5.61	0.508	1.36	1.26
B205589	5.54	0.035	3.66	1.76
2686	5.04	0.538	1.44	1.28
10294	4.25	0.770	2.20	1.32
B208689	4.10	1.00	2.19	0.890
05393	3.64	3.64	3.64	
B305389	3.21	-0.011	0.504	1.19
1586	2.91	0.360	0.833	0.606
35691	2.51	0.290	0.730	0.578
P209889	2.50	0.737	1.35	0.519
P208989	2.40	0.016	1.57	0.609
37791	2.30	0.007	0.861	0.550
10692	2.25	0.279	0.68	0.560

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[Handwritten mark]

Table 5-4 (continued)

B210389	2.19	2.19	2.19	
0586	2.10	0.578	1.35	0.761
P209489	2.09	0.835	1.37	0.392
1786	2.02	0.744	1.32	0.336
01391	1.99	0.120	0.532	0.560
2187	1.99	0.629	1.17	0.622
06491	1.95	0.630	1.13	0.515
P210289	1.94	1.94	1.94	--
B206589	1.90	0.530	1.01	0.416
B208589	1.90	1.03	1.37	0.463
37191	1.83	0.047	0.483	0.432
Uranium 238				
05093	325	39.0	144	87.3
2886	200	46.3	95.6	48.8
05193	126	62.2	90.4	19.6
B205589	121	2.23	84.3	35.1
B303089	120	110	115	7.1
3086	91.5	24.4	63.9	12.6
B305389	89.3	1.60	14.6	32.9
61093	80.8	9.80	39.2	37.1
07391	75.7	18.9	33.9	14.3
05393	66.4	66.4	66.4	
B208689	54.0	42.0	47.7	3.49
B210389	49.9	49.9	49.9	
P208989	48.0	0.50	37.7	12.8

- 1 The background bench marks for unfiltered and filtered samples are uranium 233/234 = 85.3 and 60.7 pCi/L, uranium 235 = 3.76 and 1.79 pCi/L, and uranium 238 = 60.3 and 41.8 pCi/L. No unfiltered uranium 233/234 results were above the benchmark.

**Table 5 5 Uranium Isotope Atom Ratios for Filtered Groundwater Locations
 Above Background**

Location	Mean U 238	U 238 N No of Atoms	Mean U 235	U 235 N No of Atoms	U 238 N/U 235 N
61093	39 2	3 85875E-08	0 991	1 53707E 10	251
07391	33 9	3 33991E-08	0 868	1 34657E 10	248
B305389	14 7	1 44181E-08	0 504	7 82187E 11	184
P208989	37 7	3 71190E-08	1 57	2 42728E 10	153
2689	76 9	7 57378E 08	3 33	5 16028E 10	147
B205589	84 3	8 30155E-08	3 66	5 66977E 10	146
B210389	49 9	4 91498E-08	2 19	3 39676E 10	145
4689	106	1 0395E-07	4 73	7 33482E 10	142
B208689	47 7	4 69755E-08	2 20	3 40830E 10	138
1786	28 5	2 80030E 08	1 32	2 05358E 10	136
P209889	28 9	2 84041E 08	1 35	2 10021E 10	135
P210289	41 2	4 05464E-08	1 94	3 00900E 10	135
B303089	115	1 13203E-06	5 45	8 45311E 10	134
2886	95 6	9 41133E-08	4 59	7 11125E 10	132
B206589	20 5	2 01591E-08	1 01	1 56806E 10	129
06491	22 5	2 21041E-08	1 13	1 74724E 10	127
B208589	26 9	2 64863E-08	1 37	2 13163E 10	124
P209489	26 5	2 61065E-08	1 37	2 12107E 10	123
1586	15 4	1 51662E-08	0 833	1 29251E 10	117
35691	13 4	1 32088E-08	0 730	1 13213E 10	117
05393	66 4	6 54019E-08	3 64	5 64730E 10	116
37791	15 1	1 48754E 08	0 861	1 33486E 10	111
0586	23 4	2 30487E 08	1 35	2 08668E 10	111
10692	10 9	1 07735E 08	0 679	1 05238E 10	102
2187	18 8	1 84529E 08	1 17	1 81390E 10	101
B210489	21 7	2 13546E-08	1 36	2 10805E 10	101
05093	144	1 41856E 07	9 53	1 47793E-09	96 0
3086	63 9	6 29358E-08	4 31	6 68737E 10	94 1
2686	21 1	2 07350E-08	1 44	2 23832E 10	92 6
10294	31 0	3 04729E-08	2 20	3 40990E 10	89 4
01391	7 47	7 35223E 09	0 532	8 25874E 11	89 0
5287	23 4	2 29937E-08	1 73	2 68753E-09	85 6
37191	6 51	6 40846E 09	0 483	7 49102E 11	85 6
05193	90 4	8 89479E 08	10 5	1 63106E 09	54 5

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**Table 5 6 Uranium Isotope Activity Ratios for Filtered Groundwater Locations
 Above Background**

Location	Mean U 233/234	Mean U 238	234 238
05193	198	90 4	2 20
0586	39 5	23 4	1 69
B206589	33 1	20 5	1 62
B210389	80 9	49 9	1 62
P208989	59 1	37 7	1 57
3086	98 2	63 9	1 54
05093	220	144	1 53
B208689	72 9	47 7	1 53
B305389	22 1	14 7	1 51
4689	158	106	1 50
2886	141	95 6	1 47
06491	32 5	22 5	1 45
B205589	121	84 3	1 43
2187	26 5	18 8	1 41
5287	32 7	23 4	1 40
P209889	40 0	28 9	1 39
10692	14 9	10 9	1 36
35691	18 0	13 4	1 34
37191	8 72	6 51	1 34
2686	28 0	21 1	1 33
1786	37 6	28 5	1 32
B208589	35 6	26 9	1 32
B210489	28 0	21 7	1 29
			1 24
05393	81 1	66 4	1 22
10294	37 3	31	1 21
P209489	31 8	26 5	1 20
P210289	49 3	41 2	1 20
1586	18 0	15 4	1 17
37791	17 7	15 1	1 17
B303089	135	115	1 17
01391	7 76	7 47	1 04
07391	16 0	33 9	0 47
61093	16 5	39 2	0 42

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6 0 ACTINIDE TRANSPORT IN SURFACE WATERS

Analyzing the transport of actinides in surface water is complicated by multiple factors that influence the investigation. These variables and their relation to actinide activity measured in surface water are addressed in sections 6 1 through 6 3 and include

- Sampling locations and their relationship to Site physical features
- Sampling methodologies employed
- Flow measurement
- Suspended solids measurement and
- Changes in the watershed

Following the discussion of these influential factors, data summaries are presented in Section 6 4 for surface water monitoring locations from throughout the Site. Section 6 5 contains a summary of conclusions regarding actinide transport in surface water.

6 1 SURFACE WATER SAMPLING AND FLOW MEASUREMENT

6 1 1 General Description

Protocols for collecting samples and measuring the flow of surface water runoff from the Site are designed to yield information about the migration of actinides via surface water. Results are used to facilitate management decisions to minimize the risk of off site discharge of radionuclides in excess of established limits.

Stormwater runoff from five primary Industrial Area flow paths is directed into three main drainages and associated pond systems previously described in Section 2 3. These drainages are

- North Walnut Creek (A series ponds)
- South Walnut Creek (B series ponds) and
- South Interceptor Ditch (SID and Pond C 2)

The detention ponds serve as settling basins for removing constituents including radionuclides from the water column prior to the water being discharged off the Site. The quality of the water is largely influenced by the location where a water sample is collected (e.g. upstream versus downstream from a detention pond). Therefore, for the purposes of this actinide transport analysis, surface water was categorized in the following three ways based on relative location in the drainages (listed in order from upstream to downstream):

- Industrial Area Runoff (surface water flowing within the Industrial Area fence)
- Detention Pond Influent (downstream from (1) but upstream from the ponds) and
- Detention Pond Effluent (downstream from the ponds)

Details on specific sampling locations and sampling protocols for these three location categories are provided in Section 6.1.2.

6.1.2 Surface Water Sampling Locations

6.1.2.1 Industrial Area Runoff Sampling Locations

Six stations discussed in this report are used or were used in the past to monitor surface water runoff within the Industrial Area fence boundary. These stations are listed in Table 6.1 and shown in Figure 6.1.

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Table 6 1 Summary Information for Surface Water Sampling Locations

Station	Drainage	Location	Year	Sampling	Sampling	Station
Industrial Area Runoff Sample Locations						
GS27	SWC	ditch NW of Building 889	1995 present	IM/IRA Tier II RFCA Perf	Flow paced Storm Event	0 28
GS28	SWC	Ditch NE of Building 889	1995 present	IM/IRA Tier II RFCA Perf	Flow paced Storm Event	1 13
GS21	SID	small culvert SE of Bldg 664	1995 1996	IM/IRA Tier II	Flow paced Storm Event	2 13
GS22	SID	outfall at SID of 400 Area culvert	1995 1996	IM/IRA Tier II	Flow paced Storm Event	37 50
GS24	SID	small culvert S of Bldg 881	1995 1996	IM/IRA Tier II	Flow paced Storm Event	0 84
GS25	SID	ditch draining SE of Bldg 881	1995 1996	IM/IRA Tier II	Flow paced Storm Event	5 41
Detention Pond Influent Sample Locations						
SW093	NWC	N Walnut Cr Upstream from the A 1 bypass	1991 present	Event Related IM/IRA Tier I RFCA NSD ALF	Flow paced Storm Event, Continuous	118 7
SW091	NWC	Gully NE of Solar Ponds tributary to N Walnut Creek	1995 present	IM/IRA Tier I RFCA NSD	Flow paced Storm Event	1 38
SW022	SWC	Central Ave Ditch at Inner East Fence	1995 present	IM/IRA Tier I RFCA NSD	Flow paced Storm Event	29 40
GS10	SWC	S Walnut Creek upstream from the B 1 bypass	1991 present	Event Related IM/IRA Tier I RFCA NSD ALF	Flow paced Storm Event Continuous	85 40
SW027	SID	South Interceptor Ditch at Pond C 2	1991 present	Event Related IM/IRA Tier I RFCA NSD ALF	Flow paced Storm Event Continuous	32 80

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**Table 6 1 Summary Information for Surface Water Sampling Locations
 (continued)**

Station	Sub-Drainage	Location Description	Operations Period	Project	Sampling Frequency	Number of Samples
Detention Pond Effluent Sample Locations						
Pond A 4	NWC	Terminal pond in N Walnut Ck	N/A (see Note 2)	NPDES RFCA	Composited grabs during discharge	387 (See Notes)
Pond B 5	SWC	Terminal pond in S Walnut Ck	N/A (see Note 2)	NPDES RFCA	Composited grabs during discharge	258
Pond C 2	SID	Terminal pond in SID/Woman Ck	N/A (see Note 2)	NPDES RFCA	Composited grabs during discharge	33

Not

- 1) All values are based on record from water year 1995 through March 1997 (if applicable) and are preliminary and subject to revision
- 2) Sampling periods for these ponds discharge data are Pond A-4 1/91 to 6/96 Pond B 5 3/94 to 6/95 and Pond C 2 3/92 to 6/95

During Water Years 1995 and 1996¹ the Industrial Area runoff locations were operated as Tier II Industrial Area Interim Measures/Interim Remedial Action (Industrial Area IM/IRA) monitoring stations and were therefore sited to monitor selected tributary Industrial Area sub drainages. Stations GS27 and GS28 remain active as Rocky Flats Cleanup Agreement (RFCA) Performance Monitoring locations.

6 1 2 2 Detention Pond Influent Sampling Locations

¹ Water Year is defined as October 1 through September 30. For instance Water Year 1995 is defined as October 1 1994 through September 30 1995.

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Five stations discussed in this report are used to monitor surface water influent to the detention ponds. These stations are listed in Table 6.1 and shown in Figure 6.1.

Originally, automated sampling was performed at these locations as part of the Event Related Surface Water Monitoring program conducted from the early 1990s through 1994. Under this program, the normal sampling protocol was flow-paced storm event sampling for radionuclides, metals, and water quality parameters. Data reports, some with data analysis, were produced for Water Years 1991-92, 1993, and 1994 (EG&G 1993, 1994, and 1995).

During Water Years 1995 and 1996, these locations were operated as Tier I Industrial Area Interim Measures/Interim Remedial Action (Industrial Area IM/IRA) monitoring stations located on the five main drainage pathways to the ponds.

In compliance with RFCA and in accordance with the monitoring objectives of the Draft Site Integrated Monitoring Plan (IMP), Performance, New Source Detection (NSD), and Point of Evaluation (POE) monitoring is currently performed at locations within the Industrial Area and along the five main drainages between the Industrial Area and the pond systems. Either storm-event or continuous flow-paced samples are collected at each location. Samples are analyzed for radionuclides, metals, volatile organic compounds (VOCs), and TSS, depending on location. Water quality probes have recently been deployed to collect a continuous 15-minute record of pH, turbidity, specific conductivity, temperature, and nitrate at NSD locations.

6.1.2.3 Detention Pond Effluent Sampling Locations

The three detention ponds furthest downstream in their respective drainages, Ponds A-4, B-5, and C-2, commonly referred to as the terminal ponds, are discussed in this report and

referenced in terms of detention pond effluent water quality. Surface water runoff from the Industrial Area ultimately is collected in these terminal detention ponds where it is sampled prior to batch discharge from the Site. As discussed in Section 2.3, the majority of water discharged off site from the North and South Walnut Creek drainages is routed through Pond A-4. Direct off site discharges from Pond B-5 occur occasionally (typically once every one to two years) during periods of high precipitation when the pond system is filled to the limit of safe capacity and an emergency discharge is required. Scheduled off site discharges from Pond C-2 also occur on average once a year.

6.1.3 Sampling Methodologies

6.1.3.1 Industrial Area Runoff and Detention Pond Influent Sampling

Two distinct automated sample collection methods, continuous and storm-event sampling, are employed per the RFCA Integrated Monitoring Plan to monitor actinide activity of surface water runoff from the Industrial Area into the detention ponds (Kaiser Hill, 1997). Though analytical results from both sampling protocols may be used to identify changing trends in surface water actinide activity, the two protocols generate results which generally serve different purposes and require unique interpretations.

Both continuous and storm-event samples are composites of multiple flow-paced discrete grabs collected by automated sampling equipment. However, the time span over which the discrete grab samples are collected differs between the two methods. The objective of storm event sampling is to collect a flow-paced composite for the time of increasing stage (rising limb) during a storm-event hydrograph. In contrast, continuous flow-paced composites are comprised of grab samples taken over the entire hydrograph for an estimated discharge volume. When a continuous flow-paced composite is removed from the field, a new composite is started immediately.

Analytical results of continuous flow paced samples are representative of what may be considered the average concentration/activity of a given analyte for the surface water discharge over the sampling period. This information can be used to estimate the total mass of constituent migration in the surface water during the sampling period.

In contrast to continuous sampling, storm event sampling is performed to assess storm event mobilization of constituents. The goal of sampling during the rising limb of the hydrograph is to collect composite samples containing only aliquots with the likelihood of having the greatest concentration of mobilized dissolved or particle bound constituents. Because this sample type contains exclusively storm event mobilized actinides, analytical results may provide insight as to the mechanisms of mobilization and transport. Additionally, these types of samples may be used to evaluate relationships between flow rates, radionuclides, and water quality parameters.

6.1.3.1 Detention Pond Effluent Sampling

Data presented in Section 6.4 for detention pond effluent sampling are from grab samples collected once per day, then composited, for discharges from terminal detention ponds A-4, B-5, and C-2 from 1991 through 1996 (dates vary for each pond and are noted Table 6.1). Pond locations are shown in Figure 6.1.

Average activities measured in discharges from Pond B-5 and C-2 are skewed somewhat higher than Pond A-4 average activities because of results from samples collected during the high precipitation period in Spring 1995 when Ponds B-5 and C-2 were emergency discharged without the normal settling time occurring.

6 1 4 Flow Monitoring

Past sampling and flow monitoring indicate that as the magnitude of flow increases the level of actinide activity in surface water tends to increase as well. This phenomena, further explained in Section 6 2 is caused by increased suspended solids in surface water at higher flow rates. Flow monitoring therefore represents an important component of the overall surface water actinide transport analysis.

6 1 4 1 Flow Monitoring for Industrial Area Runoff and Detention Pond Influent

Flow rates are automatically measured at the monitoring stations for Industrial Area runoff and detention pond influent. For each sample collected a corresponding flow rate is recorded at the time of sample collection. The existence of quantifiable relationships between flow and surface water actinide activity unique for each monitoring location can hence be examined.

Surface water flow rates at the six gaging stations monitoring runoff from the Industrial Area vary significantly in magnitude and occurrence. Baseflow is generally not observed at these stations. recorded flow at these locations is attributed exclusively to stormwater runoff. Among these stations maximum recorded flow rates during times of non zero flow varied from 0.8 cubic feet per second (cfs) at station GS24 to 11.3 cfs at station SW022². The flow rates and annual yields for these locations are directly dependent on sub-drainage characteristics such as area, gradient, and percent imperviousness.

² Maximum recorded flow values for Industrial Area runoff monitoring locations (GS21, GS22, GS24, GS25, GS27, and GS28) are based on 15 minute intervals recorded from October 1994 through March 1997 (where applicable). Data is preliminary and subject to review.

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In contrast to the Industrial Area runoff stations two of the detention pond influent stations SW093 (North Walnut Creek) and GS10 (South Walnut Creek) receive nearly continuous baseflow throughout the year. The other three detention pond influent stations discussed in this report (SW091, SW022, and SW027) receive intermittent flow.

For comparison between all Industrial Area runoff and detention pond influent stations average annual yields (acre feet per year) are presented in Table 6.1.

6.1.4.2 Flow Monitoring for Detention Pond Effluent

Correlations between flow and radionuclide activity are not presented for effluent from the terminal detention ponds. Water discharged from the ponds comes from the mid to upper portion of the water column (utilizing a standpipe for Pond A-4 outlet works discharges and floating inlets for pumping from Ponds B-5 and C-2). Consequently, flow rates of pond effluent water do not influence suspended solids concentration or radionuclide activity in water being discharged from the ponds.

6.2 RELATIONSHIP BETWEEN RADIONUCLIDE ACTIVITY, SUSPENDED SOLIDS, AND FLOW

A basic understanding of the behavior of radionuclides in surface soils suggests that a corollary relationship exists between radionuclide activity, suspended solids, and surface water flow rate. Radionuclides such as plutonium 239/240 (plutonium) and americium 241 (americium) have a tendency to form associations with particulate materials; therefore, it is reasonable to expect that the transport of these radionuclides may be defined by the transport of the material with which they are associated. Suspension of this particulate

material by raindrop impact and/or streamflow would facilitate transport of the associated radionuclides. Consequently, a dynamic relationship between Total Suspended Solids (TSS) concentration and the radionuclide activity of the surface water could be anticipated. The relationship would be dynamic because the state of drainages at the Site changes as contamination is removed, isolated, or exposed.

TSS can be related to flow rate if it is assumed that particulate matter is suspended by runoff to a varying degree depending on the magnitude of the flow rate. Generally, a more intense precipitation event would generate higher flow rates and would be expected to suspend more solids by various mechanisms, including:

- ditch and wetland scouring
- sheet flow on bare soils, and
- raindrop impact

Consequently, if the postulated relationship between radionuclide activity and TSS exists for a given location, then radionuclide activity may be correlated with flow rate as well. Although these relationships could be expected for most sampling locations, determination may be difficult to assess for several reasons:

- Since TSS is a measure of mass, a TSS (mg/L) to radionuclide (pCi/L) relationship assumes that the activity is proportional to the mass of solids independent of available surface area or particle composition. However, radionuclide association involves the physiochemical properties of the particles themselves and is more complex than a simple pCi/g relationship.
- If it is assumed that source areas in any particular drainage have unique physiochemical characteristics and that contamination levels vary within a drainage, the water quality characteristics of runoff from a sub drainage would be unique. In

other words since precipitation events do not occur uniformly over an area (especially for large drainages) it is expected that the water quality characteristics measured at a monitoring location will depend on the origination of that runoff. In fact there may be several or many concurrent relationships that could be established for a given location.

- Since a given drainage is continually changing either naturally through erosion or anthropogenically through D&D or construction activities there may not be time to collect sufficient information to determine one relationship before it is superseded by another.

6 3 WATERSHED CHANGES AND RADIONUCLIDE TRANSPORT

Erosion control measures have been implemented at RFETS during the past two fiscal years in an effort to stabilize and entrap soils and sediments likely to be transported from the watershed by storm water runoff. Installation of these measures is based on studies which indicate that radionuclides may associate with solids suspended in storm water (DOE 1996). Storm water data collected at the Site between 1991 and 1995 supports this conclusion (DOE 1996). Based on these characteristics of radionuclides and storm water removing particulate material from storm water runoff should reduce radionuclide loading to the water. Drainage areas targeted for control measures were those locations identified as most likely to contribute material that could provide a transport mechanism for radionuclides in Site runoff.

6 3 1 Selection of Watershed Improvement Locations

Several sources of information in conjunction with walkdowns of RFETS were used to determine locations where watershed improvements should be implemented. These information resources are listed below.

- Surface water monitoring data
Gamma spectroscopy data
- Industrial Area sediment quality data
- Industrial Area soils data and
- Historical Release Report information

Items of concern noted during site walkdowns included the following items

- Areas of concentrated fine sediments in drainage pathways
- Areas which contribute large quantities of runoff (e g steep dirt roads barren hillsides roof drains paved areas and slopes needing revegetation)
- Erosion on surface radionuclide-contaminated IHSSs (See Plate 1)
- Position of surface radionuclide-contaminated IHSSs in relation to storm water drainage pathways and
- Overall condition of storm drainage pathways

Results of the various investigative surveys were used in conjunction with findings from RFETS walkdowns to identify areas to target for watershed improvements Specific types of improvement measures implemented are discussed in the following section

6 3 2 Types of Watershed Improvements Implemented

Four types of watershed improvement measures have been implemented during the past two fiscal years Two different hydraulically applied erosion control products were utilized silt fences were installed to capture sediments suspended in runoff and overgrown vegetation

was removed from the SID to improve the flow capacity in the channel. Brief descriptions are provided below for the two hydraulically applied erosion control products.

SoilGuard® is a soil stabilizer and revegetation product hydraulically applied at locations of RFETS targeted for erosion control where revegetation was beneficial, such as exposed dirt areas. This material, a combination of wood fibers mixed with a guar gum tackifier and fertilizers, is sprayed on by a certified contractor using a hydroseeding truck. The product can be used strictly as a soil stabilizer without seed, or sprayed as a fixative on top of planted seeds. It dries within several hours to form a bonded fiber matrix that can withstand heavy rainfall while protecting the top layer of soil and does not impact water quality. New vegetative growth can protrude through the matrix without disrupting the surrounding sealed area.

TopSeal® was applied at areas of RFETS targeted for erosion control where revegetation was not practical, such as dirt roads. This acrylic copolymer emulsion product is mixed with water and sprayed on using a water truck. It dries within several hours to seal and bind the soil together and does not pose a threat to water quality.

6.4 ACTINIDE TRANSPORT ANALYSIS

Analysis of data involving actinide transport in surface water is divided into the following categories:

- Variation of radionuclides activity by location
- Relationships between radionuclides, suspended solids, and flow, and
- Impact of watershed changes on radionuclide transport

Based on relevant surface water actinide sampling results general conclusions are provided for each of these topics. Correlations of relationships between radionuclide activities, suspended solids, and flow are summarized in Section 6.4.2 with plots of these relationships located in Appendix B.

6.4.1 Variation of radionuclide activity by location and sampling methodology

A summary of radionuclide activities measured at surface water sampling stations is provided in Table 6.2. This provides an insight into the relative magnitude of activities categorized by drainage, measured at Industrial Area runoff stations versus detention pond influent and effluent locations.

Table 6 2 Radionuclide Activities by Surface Water Sample Location

Station	Designation	Sample Type	Number of Samples (n)	Activity (Bq/L)	Activity (pCi/L)	Activity (dpm/L)
Industrial Area Runoff Sample Locations						
GS27	SWC	Storm Only	13	24 98	9 06	1 47 (n = 12)
GS28	SWC	Storm Only	11	0 156	0 061	0 676
GS21	SID	Storm Only	10	0 033	0 023	0 801
GS22	SID	Storm Only	8	0 0128	0 0178	0 710
GS24	SID	Storm Only	11	0 0931	0 031	1 4811
GS25	SID	Storm Only	9	0 018	0 012	1 518
Detention Pond Influent Sample Locations						
SW093	NWC	Storm & Continuous	36	0 409	0 198	3 266
SW093	NWC	Storm Only	17	0 816	0 396	2 149
SW093	NWC	Continuous	19	0 044	0 020	4 265
SW091	NWC	Storm Only	8	0 498	0 515	4 771
GS10	SWC	Storm & Continuous	66	0 195	0 166 (n = 67)	2 336
GS10	SWC	Storm Only	48	0 228	0 200 (n = 49)	2 069
GS10	SWC	Continuous	17	0 112	0 075	2 975
SW022	SWC	Storm Only	16	0 187	0 075	0 958 (n = 15)
SW027	SID	Storm & Continuous	18	0 305	0 058 (n = 17)	2 859
SW027	SID	Storm Only	14	0 385	0 074 (n = 13)	3 247

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SW027	SID	Continuous	4	0 022	0 005	1 502
Detention Pond Effluent Sample						
Locations						
Pond A 4	NWC	Compos grabs	139	0 007	0 008	1 789
Pond B 5	SWC	Compos grabs	9	0 022	0 011	2 272
Pond C 2	SID	Compos grabs	9	0 100	0 017	2 797

Note: Average are calculated as arithmetic averages of individual grab samples

6 4 1 1 **Plutonium and Americium**

The following generalizations may be made from the actinide activities measured at Site surface water locations as summarized in Table 6 2

Plutonium and Americium General Characteristics

- At stations where both storm event and continuous flow samples have been collected (stations SW093 GS10 SW027 located at the influent to Ponds A 4 B 5 and C 2 respectively) storm event samples contain significantly higher plutonium and americium activities than continuous flow samples collected at the same location. This variation is expected if it is assumed that plutonium and americium are preferentially associated with particulate matter in the water column. Storm event samples consist of grabs taken during high flow rates on the rising limb of the hydrograph and therefore have higher TSS concentrations than continuous flow samples.

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- Radionuclide activities in detention pond influent are generally 1-2 orders of magnitude higher than the activities associated with the detention pond effluent. This indicates that the settling of particulate matter occurring in the ponds is accomplishing radionuclide removal from the water column³

Plutonium and Americium Location Specific Characteristics

- Stormwater plutonium and americium activities measured at Industrial Area runoff station GS27 are the highest measured at any automated monitoring location. A maximum value of 90 pCi/L was measured on 6/28/95. Initial response was that a significant source had been discovered and that immediate mitigating action was warranted. However, soil sampling did not indicate a hot spot or significant source. Further detail is included in Section 6.4.1.1.
- By simply comparing the arithmetic average plutonium activities for storm samples at Industrial Area runoff station GS27 and pond influent station SW022, coupled with each location's corresponding annual surface water yield, it is suspected that the GS27 sub-drainage may be contributing a significant portion of the plutonium load to SW022 (and hence to the South Walnut Creek drainage).
- SW022 has activities of similar magnitude to GS10 and represents approximately 34% of the surface water runoff entering South Walnut Creek from the Industrial Area. Although SW022 measures runoff from the portion of the Site associated with uranium, it still represents a significant portion of the total actinide load, inclusive of plutonium and americium, to South Walnut Creek.

³ Pond C-2 is the exception. Effluent activities from this pond are skewed higher by samples collected in spring 1995 as noted in Section 6.1.1.

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- Although station SW091 (Pond A 4 influent) has activities of similar magnitude to the other A 4 influent location (SW093) SW091 represents only 1% of the surface water entering North Walnut Creek from the Industrial Area, and therefore comprises only a small portion of the total actinide load to North Walnut Creek

- Comparing the arithmetic average plutonium activities for storm samples at Industrial Area runoff stations GS21 GS22 GS24 and GS25 (sub basin monitoring upstream from station SW027) with station SW027 (Pond C 2 influent) coupled with each location s corresponding annual surface water yield it is indicated that none of these Industrial Area locations is contributing a significant portion of the plutonium load to SW027 This supports the conclusion that the source for the activity measured at SW027 probably originates downstream in areas affected by contamination from the 903 Pad

- Although Pond C 2 generally has higher plutonium activities than Ponds B 5 and A 4 station SW027 (influent to Pond C 2) has lower average plutonium activities for continuous samples than do the stations that monitor the influent to Ponds A 4 and B 5 (SW093 and GS10 respectively)

The difference in the water quality for Pond C 2 compared to A-4 and B 5 may be attributable to several items

- ◇ A-4 and B 5 waters are diluted by WWTP effluent
- ◇ C 2 may recieve loading only during very large precipitation events which result in significant overland flow from contaminated areas near the 903 Pad and
- ◇ Other biological physio-chemical limnologic or management differences for Pond C 2 may result in resuspension of contaminants

6 4 1 1 Uranium

Uranium Location Specific Characteristics

- There is a significant increase in total uranium at SW093 (Pond A 4 influent) for continuous samples compared to storm events. This suggests that uranium may be preferentially associated with baseflow associated with groundwater seeps. This supports the concept that when storm events occur, overland runoff causes dilution of the uranium contained in the baseflow and hence the storm event samples contain lower total uranium activity than the continuous flow samples. The proximity of SW093 to the Solar Ponds could also be influencing the total uranium data.

- There is measurably less total uranium at SW027 (Pond C 2 influent) for continuous samples compared to storm events. This suggests that uranium may be preferentially associated with storm runoff. It may also indicate that the source of baseflow may be low in uranium or that baseflow from natural sources is minimal at this location. In fact, SW027 does not receive baseflow much of the year and when it does, it is suspected that this baseflow (or a significant portion) originates as a domestic leak flowing into the 400 Area stormwater collection system.

6 4 2 Relationships between radionuclides, suspended solids, and flow

In order to understand the relationships between the actinides of concern, suspended solids, and flow, plots were developed relating these variables to one another for each of the surface water sampling stations described in Section 6 1 2. These plots included

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- Plutonium activity versus flow
- Americium activity versus flow
- Total uranium activity versus flow
- Plutonium activity versus TSS and
- Americium activity versus TSS

Results from these plots are compiled in Table 6.3. Where trends from the regressions whether positive or negative were observed they are noted with (+) or () signs respectively. When the regression R^2 value is greater than 0.90 the trend is noted with (+) or () signs underlined and the R^2 value is listed in the Correlation Notes column of the table. Potential implications of these relationships are also noted in the last column. Discussion of these relationships follows.

Table 6 3 Summary of regressions between radionuclides flow and suspended solids

Sample Location	Flow Type	Flow Regime	n	Am vs Flow	Pu vs Flow	Am vs TSS	Pu vs TSS	Notes
Industrial Area Runoff Sample Locations								
GS27	SWC	Storm Only	13	(+)	(+)	none	(+)	Am vs TSS $R^2 = 0.89$ (Minimizing erosion could minimize rad transport)
GS28	SWC	Storm Only	11	none	none	none	(+)	Pu vs TSS $R^2 = 0.92$ (Minimizing erosion could minimize rad transport)
GS21	SID	Storm Only	10	(+)	(+)	none	none	low activity in samples
GS22	SID	Storm Only	8	(+)	(+)	none	none	low activity in samples
GS24	SID	Storm Only	11	(+)	(+)	none	none	low activity in samples
GS25	SID	Storm Only	9	(+)	(+)	none	none	low activity in samples
Detention Pond Influent Sample Location								
SW093	NWC	Storm & Continuous	36	none	none	()	(+)	(Tot U contained in baseflow diluted by stormwater)
SW091	NWC	Storm Only	8	(+)	(+)	none	(+)	Pu vs flow $R^2 = 0.96$ Pu vs TSS $R^2 = 0.92$ (Minimizing erosion could minimize rad transport)
GS10	SWC	Storm Only	48	none	none	none	(+)	Pu vs TSS $R^2 = 0.89$
SW022	SWC	Storm Only	16	none	none	none	(+)	(Large basin variability possible cause for weak correlations)
SW027	SID	Continuous	4	(+)	(+)	()	none	(Low TSS at SW027 makes TSS trending difficult)

Note: Actinide activity as pCi/L. Flow measured as cfs. TSS measured as mg/L.

Where the signs from the regressions show their positive or negative we observed they are noted with () or () signs respectively. When the regression coefficient is greater than 0.90 then the data is considered with () or () symbols. In addition, the regression coefficient is noted in the notes column. Correlation is not shown if the value is less than 0.90.

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6 4 2 1 Relationship between radionuclides and suspended solids

As discussed in Section 6 2 2 a relationship between radionuclide activity and suspended solids concentration is anticipated based on the tendency for plutonium and americium to adsorb to particulate matter TSS and plutonium/americium relationships for surface water monitoring stations as summarized in Table 6 3 are discussed here by location Hypotheses are presented regarding surface water radionuclide transport mechanisms

Stations SW093 SW022 and SW027 Plutonium and americium activities from storm event samples collected from gaging stations SW093 SW022 and SW027 (influent to Ponds A 4 B 5 and C 2 respectively) show a general positive trend for activity versus TSS although statistical correlations for these relationships are weak The absence of a statistically significant relationship may be attributed to the relatively larger size of these drainage basins A larger basin is likely to exhibit greater variability in contaminant distribution as well as rainfall intensity distribution among events In fact the variability of sediment contamination across the SW027 drainage is documented (Eford et al) These factors combine to yield variable exposure of runoff waters to contaminated soils For instance a large storm event may produce rainfall on those areas of the drainage basin with minimal radionuclide content in the soil while another similarly large storm event may produce rainfall primarily on areas in the basin with significant contamination Both of these events of similar intensity might suspend similar concentrations of solids however the actinide levels in the samples would be quite different Consequently storm event samples from large basins may not be as readily comparable as those from smaller drainages

Station GS10 Gaging station GS10 (influent to pond B 5) also monitors surface water from a fairly large drainage basin however dramatically better correlations for plutonium and

americium versus TSS (Figure 1 Appendix B) are observed from storm event samples as compared to SW093 SW022 and GS10 This result seems incompatible with the argument that the quality of the radionuclide TSS correlation deteriorates as the size of the drainage basin increases This apparent inconsistency may be due to a factor which minimizes the variability of contaminant mobilization within the basin A possible explanation is that a significant actinide source exists near the GS10 sampling point Such a source downstream from the confluence of drainage sub basins would lead to regular contaminant mobilization with each event regardless of where the precipitation occurred in the basin because all water flowing to station GS10 would pass through the same contaminated source area This theory is further supported by the Site *Historical Release Report* which indicates potential actinide sources may exist in the South Walnut Creek drainage upstream from station GS10 and downstream from the Protected Area fenceline (DOE 1992) Causes of these actinide sources are attributed to discharges of untreated laundry wastewater below Building 995 from 1953 to 1965 and resuspension of contaminated sediments resulting from drainage reconstruction activities from 1971 to 1973

Station SW091 Significant positive correlations for plutonium and americium activities with TSS do exist for storm event samples collected from gaging station SW091 (Figure 2 Appendix B) Gaging station SW091 monitors surface water influent to the A series ponds in North Walnut Creek but from a smaller basin than monitored by SW093 The smaller size of the basin may account for the improved correlation due to reduced variability of precipitation and runoff intensity It should be noted that these correlations are based on relatively small data sets consequently the relationship is strongly influenced by the points of high activity

Stations GS27 and GS28 Results of plutonium americium and TSS analyses of storm water from GS27 and GS28 are similar exhibiting good correlations between actinide activity and suspended solids Data from GS27 and GS28 are presented in Figures 3 and

Figure 4 (Appendix B) respectively. The strong correlations may be attributed to the small drainage areas (Industrial Area runoff sub basins in the South Walnut Creek drainage comprising less than 1 acre each) monitored by these stations. However, it should be noted that only limited data sets are available for these locations (GS27 n = 13, GS28 n = 11) and the relationships are influenced by samples with high activity.

Analyses of sediment samples from near GS27 may indicate that plutonium and americium are associated with a specific, more mobile fraction of the soil. Samples of sediment materials were taken from a drainage gutter on the south side of Building 884 and analyzed for actinide activities. These results are presented in Table 6.4.

Table 6.4 Analytical Results from Sediment Samples near Building 884

Sample Name	Am-241 (pCi/g)	Pu-239 (pCi/g)
SD01001JE	6 602 ± 0 263	1 269 ± 0 133
SD01002JE	5 717 ± 0 248	1 833 ± 0 175
SD01003JE	5 672 ± 0 232	1 453 ± 0 150
SD01003JE DUPLICATE	5 523 ± 0 231	1 850 ± 0 200

These results do not show sufficient actinide content to account for the plutonium and americium levels observed in storm samples given the measured TSS concentrations. This apparent discrepancy, it is theorized, is caused because radionuclides selectively associate with a specific fraction of the sediment which is essentially concentrated in the more easily mobilized TSS. This fraction may simply be the smaller particles which possess greater surface area for radionuclide adsorption.

Stations GS21, GS22, GS24, and GS25 Data from storm event samples from GS21, GS22, GS24, and GS25 (Industrial Area runoff in the SID basin) exhibit no significant correlation between radionuclides and TSS. Only results from GS24 show even an apparent positive

trend for plutonium versus TSS. Although these gaging stations all monitor fairly small basins and a better relationship with TSS might be expected, the lack of good correlations may be attributed to the overall low radionuclide activity in all the samples. Even the samples with the highest activities yielded actinide levels of similar magnitude to the reported analytical error. As a result, comparison of variation among the samples is not meaningful.

6.4.2.2 Relationship between radionuclides and flow

SW093 Monitoring results from station SW093, which include storm event and continuous flow paced samples, show a negative relationship between flow and total uranium activity as shown in Figure 5 (Appendix B). This station receives near continuous baseflow. These results indicate that runoff from storm events dilutes the total uranium activity in the baseflow and causes lowered activities at higher flow rates. This phenomenon may be caused by contaminated groundwater that supplies the baseflow and constitutes the primary source for total uranium observed in the surface water.

No relationship was observed for plutonium and americium activities as a function of flow at station SW093. This is to be expected as SW093 monitors a large, complex basin and no significant relationship was observed at this station between plutonium and americium activities with TSS.

SW091 Results from station SW091, in contrast to station SW093, showed a weak relationship between total uranium and flow. These results are to be expected. Station SW091 monitors a relatively small sub-basin with ephemeral flow. Because no baseflow exists at this site, the dilution of baseflow containing total uranium (as seen at station SW093) does not occur.

Plutonium and americium activities exhibit positive correlations with flow at station SW091 as shown in Figure 6 (Appendix B)

GS10 No strong correlations were observed between total uranium and flow at station GS10 (Pond B-4 influent) At low flow rates (less than 1 cfs) the average total uranium activity measured at GS10 is approximately 2 pCi/L This contrasts with station SW093 where a strong negative total uranium to flow relationship exists and total uranium activity at low flow rates is approximately 4 pCi/L or twice the activity of GS10

Plutonium and americium activities do not exhibit good correlations with flow at Station GS10

SW022 Total uranium and flow exhibit no relationship at SW022 (Pond B 5 influent) Similar to station SW091 this is expected as flow at this site is ephemeral Sources of groundwater with measurable levels of total uranium are not available for dilution by runoff therefore a negative correlation between total uranium and flow (as exists at station SW093) does not exist

Plutonium and americium activities do not exhibit a good correlation with flow at station SW022

GS27 This small Industrial Area sub basin has no baseflow Total uranium exhibits a weak correlation between activity and flow Plutonium and americium also exhibit weak correlations but positive trends between activity and flow Figure 7 (Appendix B)

SW027 When analyzing data from storm events none of the radionuclides exhibit a good relationship with flow at SW027 However when limited data points (n = 4) of

continuous flow paced samples are examined trends in radionuclide activity are observed Total uranium trends to a negative relationship between activity and flow for continuous samples Plutonium and americium trend to a positive relationship between activity and flow

GS21, GS22, GS24 and GS25 These Industrial Area sub basins (within the SID basin) are similar to GS27 with respect to size lack of baseflow and similar radionuclide activity to flow relationships Total uranium exhibits a weak correlation between activity and flow Plutonium and americium also exhibit weak correlations but with positive trends between activity and flow as shown in the plot of GS24 in Figure 8 (Appendix B)

6 4 4 Impact of watershed changes on radionuclide transport

Two separate monitoring locations GS27 (Industrial Area runoff) and SW027 (influent to Pond C 2) are well suited to demonstrate the impact on water quality of changes on the watershed In addition monitoring results from these sites reflect the challenge in collecting an adequate number of samples at variable flow rates after watershed improvements have been implemented in order to properly assess whether or not the improvements are reducing activity in runoff for both high and low flows

6 4 4 1 Impact of watershed improvements on GS27

Surface water monitoring at Industrial Area sites in smaller localized drainage basins can be used to monitor the effectiveness of improvements implemented within the specific basins Station GS27 is of specific interest because it is a small basin draining less than 1 acre and samples have been collected before during and after changes have occurred

within the basin Station GS27 was installed to monitor for possible impacts of the decontamination and decommissioning (D&D) at Building 889 (completed in July 1996) and of watershed improvements implemented in the basin These watershed improvements included

- Removing and drumming sediments measured to contain 5 to 6 pCi/g of plutonium accumulated on the asphalt south of Building 884 (completed July 1996) and
- Applying TopSeal® on the exposed dirt areas south of Building 884 (completed October 1996)

Plutonium and americium activities in samples collected at GS27 before and after the initial improvements were implemented are shown in Figure 9 (Appendix B) The majority of post improvement samples were collected at low flow rates Although these data points indicate actinide activities as low or lower for a given flow than prior to the improvements it is impossible to extrapolate this data to define the impact on water quality of the control measures when higher flows occur Station GS27 remains on line to fill this data gap by collecting samples when a higher flow rate does occur

6 4 4 2 Impact of watershed improvements on SW027

Sample results indicate a trend toward reduced plutonium and americium activity to flow ratios at Station SW027 since the time when improvements were first implemented in this basin A dirt road running south down the hill from the 903 Pad toward Pond C 1 was revegetated during the summer of 1996 and treated with SoilGuard® In addition dirt roads encircling the 903 Pad were treated with TopSeal® during the past year However it should be noted that the sampling protocol at station SW027 changed from storm-event sampling to continuous flow paced sampling as a result of RFCA requirements in the

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midst of the improvements being implemented. Results of storm event samples only however, also indicate reduced radionuclide loading following the implementation of improvement measures. Plots of plutonium and americium activity versus flow, before and after watershed improvements were implemented, are contained in Figure 9 (Appendix B).

6 5 CONCLUSIONS FOR ACTINIDE TRANSPORT IN SURFACE WATER

6 5 1 Uranium Transport in Surface Water

6 5 1 1 General Uranium Transport Mechanisms

Surface water data from monitoring locations with baseflow near the Solar Ponds suggest a negative correlation between total uranium activity and flow rate. As surface water flow rate increases due to storm events, uranium activity decreases with dilution of baseflow. This trend is evident in the results from gaging station SW093 downgradient from the Solar Ponds (Figure 5, Appendix B). This phenomenon may be due to groundwater, which supplies the baseflow and is the primary source for uranium observed in the surface water. This conclusion is further supported by data from monitoring locations with ephemeral flow, which exhibit no significant relationship between total uranium and flow rate. At stations with baseflow, the primary mechanism for uranium contamination of surface water may not be mobilization of particle-bound nuclides by storm events, but transport via groundwater supply of baseflow.

6 5 1 2 Location Specific Uranium Transport Observations

Uranium in surface water drainages is associated with baseflow in areas that drain the solar ponds (SW093 GS 10) but appears to be associated with storm runoff in the area south of the 881 building and south of the old landfill (SW027) where there is surface uranium contamination (see detailed discussion of the data in Section 6 4) Total uranium activity in the discharge ponds is similar for all three ponds In North Walnut Creek Pond A 4 (1 79 pCi/L) contains about 1/4 the total uranium activity than the continuous flow for SW093 (4 265 pCi/L) which is below the Solar Ponds In South Walnut Creek Pond B 5 (2 272 pCi/L total uranium) is slightly lower than the continuous flow sampled at GS10 (2 975 pCi/L) In the SID Pond C 2 (2 797 pCi/L total uranium) contains slightly lower total uranium than is recorded for SW027 (3 247 pCi/L) Since the flow paths are not direct from the surface water sampling locations to A and B series ponds correlation of actinide content in the ponds to the influent data may not be meaningful

6 5 2 Plutonium and Americium Transport in Surface Water

6 5 2 1 General Plutonium and Americium Transport Mechanisms

Sampling results indicate that plutonium and americium transport is related to both total suspended solids concentration and flow rate Figure 3 and Figure 4 (Appendix B) depict positive correlations between plutonium/americium activity and TSS for Industrial Area runoff stations GS27 and GS28 that monitor smaller drainage sub basins In general sampling sites in small drainages provide more distinct positive correlations compared to sites located in larger basins The correlation is also stronger in drainages that collect runoff from areas with widespread surface soil radionuclide contamination Results from station GS10 a large basin capturing runoff from roughly two thirds of the Industrial Area represent the exception to this observation by exhibiting a strong correlation between plutonium/ americium activity and TSS as shown in Figure 1 (Appendix B)

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A positive correlation is observed between plutonium and americium activity and flow rate across the Site. This trend is depicted in Figure 6 and Figure 7 (Appendix B) for gaging stations SW091 and GS27 respectively. As with the plutonium/americium relationship to TSS, the plutonium/americium correlation with flow is markedly better for smaller drainages. Again, this may reflect diminished contaminant level and rainfall distribution variability inherent in smaller basin areas.

The positive correlations of activity with both TSS and flow strongly suggests that transport of plutonium and americium is facilitated by dislocation and transport of suspended particles whose suspension is a function of flow rate and precipitation intensity.

Despite mobilization of plutonium and americium detected during storm events as shown by storm event samples, pond effluent sampling indicates the pond system performs well to attenuate and settle contaminants in storm water prior to discharge off site. Contaminant levels measured in storm water flowing from the ponds are typically decreased at least 10 fold from that of influent levels.

6.5.2.2 Location Specific Plutonium and Americium Transport Observations

North Walnut Creek at GS 10 This station has continuous baseflow with relatively high levels of plutonium and americium activities. The plutonium and americium activities for continuous flow samples at GS 10 (0.112 pCi/L Pu) are about an order of magnitude above those for SW093 (0.044 pCi/L Pu) and SW027 (0.022 pCi/L Pu). A 72 CMP culvert feeds a continuous baseflow to GS10. This water flows over IHSS 141 just prior to being sampled at GS 10. IHSS 141 is shown in Figures 4.1 and 4.2 which show elevated plutonium and americium activities in surface soils.

South Walnut Creek at SW 093 This station has continuous baseflow but does not have high levels of plutonium and americium in the continuous flow samples. However, SW093 (0.816 pCi/L Pu) storm event samples have higher plutonium and americium activities than either GS 10 (0.228 pCi/L Pu) or SW027 (0.385 pCi/L Pu). SW093 collects water draining from IHSS 150.8, an area of surface contamination containing slightly elevated plutonium and americium contamination as shown in Figures 4.1 and 4.2. Stormwater flushes this area and drains into a drop box. This storm flushing across the IHSS and through the sediments accumulated in the drop box is suspected of causing an increase in plutonium and americium loading for storm events over baseflow. SW093 also collects drainage from east of Building 771. This water flows along IHSS 172, a plutonium related IHSS. However, no elevated plutonium or americium is shown in this area on figures 4.1 and 4.2.

SW027 and Pond C 2 SW027 collects water draining the south Industrial Area, which is collected in the SID and discharged into Pond C 2. Pond C 2 also collects water from areas downwind of the 903 Pad with surficial plutonium and americium soil contamination as shown in Figures 4.1 and 4.2. The average plutonium activity for continuous flow sampling at SW027 (0.022 pCi/L) is approximately an order of magnitude less than the average plutonium activity within the pond discharge (0.100 pCi/L).

Average plutonium concentrations at Pond C 2 are skewed by high levels during the spring and summer of 1995. Both C 2 and SW027 had peak plutonium activities in the spring/summer of 1995. The loading in C 2 is from both the input from SW 27 and from water flowing into C 2 from the surrounding area. The current data cannot segregate contaminant loading into Pond C 2 from the drainage via SW027 from the loading due to the low level contaminated surface soils draining into Pond C 2 which are not captured by SW027.

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7 0 SOIL EROSION AND TRANSPORTATION OF ACTINIDES TO SURFACE WATER/SEDIMENTS

7 1 ESTIMATING SOIL MOVEMENT DUE TO RAINFALL EROSION AT RFETS

Data presented in Section 6 show that actinides are moving in surface waters and exceed surface water standards during storm events above the A and B Series Ponds on North and South Walnut Creeks and in drainages to the SID. The transport of actinide contaminated soils by overland flow to surface water in the Walnut and Woman Creek Drainages has the potential to cause exceedances of the surface water standards and cause the movement of actinides off site.

The surface soil action levels (DOE 1996a) for actinides did not consider transport pathways to surface water, surface water ingestion, or the effects of the action levels on compliance with surface water standards. Therefore, it is important that when cleanup levels are established for areas with surface soil exceedances of Tier I actinide action levels that the long term effects on surface water quality be considered.

During remediation activities in the IA and Buffer Zone, soil transport due to rainfall erosion may increase temporarily and affect surface water quality. The magnitude of these problems must be understood so that surface soil cleanup levels are set that are protective of surface water and steps are taken to control the effects of soil erosion during and after remedial actions, if necessary. The following presents past erosion modeling efforts at RFETS and discusses plans for further work.

7 1 1 Past Erosion Modeling for the 903 Pad and East Trenches Area

Surface soil in the 903 Pad Area is one of the major contaminant sources for the surface water system at RFETS. Precipitation events induce soil movement and surface soils are carried by overland flow to surface water drainages. Soils and associated actinides move into the creeks and increase the concentrations of actinides in sediments and surface waters. Two reports have been published that use the Universal Soil Loss Equation (USLE) model developed by the USDA (Renard et al. 1997) to estimate soil movement in the 903 Pad Area. *Estimated Soil Erosion and Associated Actinide Transport for the South Interceptor Ditch Drainage (DOE 1992)* and *Phase II RFI/RI Report 903 Pad Mound and East Trenches Area Operable Unit No. 2 (DOE 1995)*

The 1992 report (DOE 1992) modeled soil erosion and associated transport of soil bound actinides within the SID drainage basin. Its purpose was to assess the impact of 881 Hillside construction and SID maintenance activities on soil erosion and the transport of actinides to Pond C2 via the SID. The hillside on the south side of the Industrial Area from west of the old landfill to Pond C 2 was divided into seven sub basins for the study. The following is a summary of the analysis conducted in 1992.

The Soil Conservation Service Curve Number Method (SCSCN) was used to estimate runoff in the basin. This method generated runoff hydrographs for storms of various intensities and durations on a variety of soil types. The rainfall hydrographs were then used to estimate soil erosion within the basin using the Modified and Revised USLE (MUSCLE/RUSLE) model developed by the Agricultural Research Service (ARS) of the United States Department of Agriculture (USDA).

The basic USLE presented by Wischmeier and Smith (1978) and Mills et al (1985) is

$$A = R * K * L * S * C * P * S_a$$

where

A is the soil loss tons/acre

R is rainfall erosivity the product of the total storm energy and its maximum 30 minute intensity foot tons/acre

K is a measure of the erodibility of a given soil type foot¹

L is the slope length factor a dimensionless ratio of soil loss from a field of a given length to the soil loss from an experimental unit plot 72.6 feet in length with all other conditions equal

S is the slope steepness factor a dimensionless ratio of the soil loss from a field of a given steepness to the soil loss from an experimental unit plot with a 9 percent slope with all other conditions equal

C is the dimensionless cover and management factor reflecting the type of vegetation percent basal cover and percentage foliar cover on a sub basin

P is the dimensionless support practice factor reflecting any erosion prevention practices in use and

S_a is the dimensionless sediment delivery ratio which accounts for the attenuation of sediment through deposition and filtering during movement from source area to watershed outlet

The total annual quantity of soil transported into the SID was estimated. The EPA Storm Water Management Model (SWMM) was used to model channel dynamics and the EPA recommended Sediment Removal Model (SRM) was then used to estimate sediment movement and the amount of sediments that would be deposited in both the SID and Pond C2 during pre and post maintenance conditions. The models were calibrated using site specific data and information from technical journals and other public sources.

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The models were verified using SID flow and TSS data from a sampling station located at the downstream end of the SID

Average annual actinide loading was calculated using the annual soil loss computed by the RUSLE model and average actinide activity/concentrations in surficial soil samples from the drainage basin. The 1992 report estimated that with the basins vegetated 70 tons of soil per year would erode into the SID of which 7.2 percent (5.04 tons) would be transported into Pond C2 resulting in an annual deposition in Pond-C2 of about 6.9×10^6 pCi of plutonium 239/240 and 1.1×10^6 pCi americium 241. Under post renovation conditions it was estimated that the same amount of soil per year would enter the SID with 11.4 percent (7.98 tons) reaching Pond C2 resulting in an annual deposition of 10.9×10^6 pCi of plutonium 239/240 and 1.8×10^6 pCi of americium 241. The results of this study are similar to actinide loads calculated for Pond C2 from 1996-1997 data presented in Section 8 (Figure 8.2)

The above estimates were verified using data from a storm that occurred on March 28, 1992. Storm data were used to generate runoff hydrographs for each basin using the SCSCN. The hydrographs were then input into MUSLE and estimated a total soil loss from the basin of 10 tons for the storm. Total suspended solids and flow data from the SW027 sampling station were used to estimate a total sediment load of 0.85 tons to Pond C2 which is 7.3 percent of the sediment load to the SID estimated by the MUSLE. This is in very close agreement with the 7.2 percent transport efficiency estimated by the SWWM model.

The 1995 study (DOE 1995) performed as part of the OU2 RFI/RI used the MUSLE estimated soil erosion from the 903 Pad Mound and East Trenches Area and estimated loading to the South Walnut and Woman Creeks. The purpose of the study was to estimate concentrations of chemicals of concern in the Walnut and Woman Creek Drainages at Indiana Avenue over a thirty year period. The study assumed that no engineered

structures such as Pond C 2 were present For modeling 65 small sub basins were delineated to produce sub basins of roughly uniform gradient and meet model criteria The sub basins were subsets of the six major drainage basins identified in the former OU2 Two of the six major basins drain directly into Walnut and Woman Creeks

Precipitation was simulated over a 30 year period to calculate the erosivity factor (R) The erodibility (K) of the eleven soil types within OU2 soils was determined by the Soil Conservation Service (Price and Amen 1983) The percentages of each soil type within each sub basin were determined using a geographical information system (GIS) and areally weighted averages calculated to estimate the erodibility factor for each sub basin The slopes and lengths of the sub basins were estimated from topographic contours The cover factor (C) was estimated using observations of vegetation and ground cover in OU2

Activities of americium and plutonium were estimated over the entire area using data from 1991 soil sampling events The estimation method assumed that the concentration at an unsampled location was equal to that at the nearest sampled location Areal percentages of each sub basin associated with different activity/concentrations were used to calculate weighted averages of americium and plutonium activity/concentration for each sub basin The weighted plutonium activity/concentrations for the 65 sub basins ranged from 0.67 to 1206 pCi/g These values were multiplied by the estimated soil loss from each sub basin to estimate the loads of americium and plutonium from each of the six drainages

Runoff volume was estimated using the CUHPE PC software developed by the Denver Urban Drainage and Flood Control District The model subtracts infiltration and retention from precipitation to estimate overland flow (runoff) Overland flow was estimated at 1 percent of the total precipitation for the Woman Creek basin and at 13 percent for the Walnut Creek basin The larger value for the Walnut Creek basin was attributed to a larger area of impervious land Overland flow at RFETS has previously been estimated at 1.4 percent by Hurr (1976)

Observed TSS concentrations in the creeks were considered an indicator of soil loss and were used for calibration of soil loss from the watershed and estimation of S_d . The geometric mean TSS concentration at sampling station SW 027 was selected 31 g/L. It was assumed that soil loss contributed only to the TSS and not to the bed load during storm events. This assumes that all eroded soil remained suspended in the surface water flow. This may have underestimated soil loss however as the model did not estimate the contribution of bank erosion to TSS.

The results of one of 100 30 year simulations were presented (DOE 1995). During the 30 simulations total annual precipitation was varied from a minimum of 4 inches to a maximum of 30 inches with an average of 17.1 inches. Estimated total annual soil losses for all sub basins draining into Woman Creek varied from 0.2 tons of soil per year to 19.3 tons per year with an average of 3.8 tons per year. This is considerably less than the 70 tons per year estimated in the SID study even though the area modeled was larger. The estimated total annual loading into Walnut Creek ranged from 0.4 to 33.7 tons of soil per year with an average of 6.6 tons per year.

Based upon this simulation it was calculated that the average annual plutonium loading from sub basins to Woman Creek was $8.75E+08$ pCi per year and to Walnut Creek was $2.1E+07$ pCi per year. The Woman creek value appears to be high. In the report (DOE 1995) these values were used as input to estimate surface water concentrations in the creeks at Indiana Avenue. The 30 year value for plutonium activity/concentrations for Woman Creek was calculated to be 1.8 pCi/L which is an order of magnitude or more greater than average observed values. This indicates that the model was significantly overestimating loading.

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7 1 2 Other Modeling Efforts for the Woman Creek and Walnut Creek Drainages

Two modeling studies were completed in support of the former OU5 and OU6 RFI/RI reports (DOE 1996b and DOE 1996c) For both reports modeling was done to support the Human Health Risk Assessment Both efforts used Version 10 of the Hydrological Simulation Program FORTRAN (HSPF) which is a comprehensive package for simulation of watershed hydrology and water quality that allows the integrated simulation of land and soil runoff with in stream hydraulic and sediment/chemical interactions (Bicknell et al 1993)

The purpose of both of these modeling efforts was to predict chemical of concern (COC) concentrations in sediments and surface waters up to 30 years in the future They do not specifically address soil erosion but rather estimate long term concentrations of COCs in the water column and stream and pond sediments In the OU5 report (DOE 1996b) only estimated 30 year sediment and water column concentrations are reported

The OU6 report (DOE 1996c) estimated the transport of COCs in the Walnut Creek drainage system A seven year (1986 1992) calibration period was used for sediment deposits to the A and B Series ponds Prediction errors were less than 7 percent for the pond systems although prediction error was as high as 37 percent for individual ponds Predicted sediment loads to the ponds over a 30 year simulation varied from a low of 8.9 tons to Pond B2 to a maximum of 2215 tons in Pond B4 Plutonium 239/240 loading to the ponds during the 30 year simulation varied from a minimum of 5.7×10^{-6} pCi in Pond B1 to a maximum of 2.7×10^{-8} pCi in Pond A3

7 1 3 Future Erosion Modeling

The purpose of modeling soil erosion by water is to provide input for the conceptual model for actinide migration at RFETS to quantify the transfer of sediments and soil bound actinides from soil to surface water under a variety of plausible future scenarios and to provide information to facilitate the choice of cleanup levels for actinides in surface soils that will be protective of surface water under long term conditions. Erosion models are currently being evaluated for selection. The modeling effort will begin in fiscal year 1997 and be completed in 1998 with results linked to the Conceptual Model and Mass Loading Analysis for the Walnut and Woman Creek drainages.

The RUSLE model is currently being evaluated. A basic description was given in the previous section. The documentation, databases used for calculations, and the computer model have been recently updated (Renard et al. 1997). The RUSLE is an empirically based erosion model predicting long time average annual soil loss resulting from rainfall and runoff for a variety of field conditions including rangeland. Its development and widespread use over a period of 40 years has confirmed the RUSLE's usefulness and validity in quantitatively predicting soil loss and evaluating conservation practices. The soil loss computed by RUSLE is a guide, not a precise estimator of soil loss. It has limitations in scope. It is best used on a field scale rather than on a watershed scale. RUSLE has been used to estimate watershed sediment yields, however care must be taken to account for depositional areas on the landscape. The long term average soil loss computed with RUSLE is not sediment yield from the field, but an estimate of total sediment production by sheet and rill erosion. Sediment yield may be much less or if ephemeral gully erosion occurs, sediment yield may be much greater than estimated by RUSLE.

The Water Erosion Prediction Project (WEPP) model developed over the last 10 years by the ARS in cooperation with other federal agencies and universities is also being evaluated. It is a process based distributed parameter continuous simulation erosion prediction model that simulates hydrologic and erosion processes on a watershed scale.

It can be used to estimate soil loss, soil deposition, sediment yield and how these are distributed in space and time (ARS 1997). This model is considered to be a great advance over all previous erosion models.

7.2 ACTINIDE MIGRATION THROUGH WIND DRIVEN PARTICULATE RESUSPENSION

The Site has been a source of airborne actinides throughout its history. The three largest contributors are building ventilation systems, leakage from containers of actinide laden waste stored outdoors, and resuspension of contaminated soils. Building ventilation systems and outside storage are not responsible for movement of actinides that are already in the environment. They are instead sources of actinide deposition to the environment. Resuspension, however, is a mechanism for movement of actinides that are already in the environment. Airborne migration of actinides in the environment around the Site is primarily due to soil resuspension (see *Radionuclide Air Emission Annual Report* DOE 1995a). Movement can be caused by natural processes alone (resuspension in a natural ecosystem caused by high winds, or by a combination of rain splash and wind movement of vegetation) or by a combination of artificial and natural processes (transport by wind of actinide-contaminated soil that is disturbed by earth moving activities) (see Appendix C *Resuspension of Soil Particles from Rocky Flats Containing Plutonium Particles* DOE 1995a).

Through these transport mechanisms, actinide and non actinide particles are lifted from the ground and plants and carried through the air. Resuspension, dispersion, and deposition of actinides in the environment is highly dependent on the physical and chemical form of the

actinides For example the size of the particles affects their potential to be resuspended and the chemical form of the actinide affects its potential to attach to other particles

Resuspension has not always been the major source of plutonium in the ambient air around the Site When the Site was producing triggers plutonium from the building ventilation systems contributed a significant fraction of airborne plutonium and masked the resuspension component Cleanup activities which can potentially contribute significantly to measured emissions from the Site also mask the contribution from the natural resuspension processes

Measurements of plutonium in the ambient air have been routinely conducted at the Site for over twenty years using medium volume ambient air particulate samplers These samplers have been operated continuously on Site at the Site perimeter and in several nearby communities Not surprisingly trends within the ambient air data show higher concentrations of plutonium near the contamination sources and decreasing concentrations with increasing distance from the sources (see Figure 7 1 based on data in CY92 Annual Report) This trend applies whether the source is a building a remediation activity or an area of contaminated soil It is interesting to note however that simple modeling to account for airborne concentrations of plutonium activity at a sampling location near the Site perimeter suggests that as much as 30% of that activity is contributed by resuspended material near the sampler and not from the more highly contaminated but distant source area near the 903 Pad

7 2 1 Source Summary

The migration of actinides already in the environment through the air pathway is contingent on the actinides being available for entrainment and dispersion in the ambient air The major source of resuspendable actinide material at the Site is the area around the 903 Pad (see Figure 7 2) The 903 Pad itself was contaminated with plutonium laden cutting oil stored in metal drums which over time leaked onto the soil beneath the drums Removal

of the drums and associated cleanup activities in the early 1970 s allowed the contaminated soil to disperse to the east and south of the pad area (Rocky Flats Plant Radioecology and Airborne Pathway Summary Report 1986) The pad itself is now covered with asphalt and is no longer a source of resuspendable plutonium however the initial spread of plutonium-contaminated soil during cleanup resulted in a plume of plutonium in the surface soils extending from the pad to the east and east southeast The plume decreases in concentration from approximately 1 000 pCi/g just east of the 903 Pad to well under 1 pCi/g at the eastern boundary of the Site (DOE 1994)

Because the Site measures plutonium in the ambient air all sources will contribute indistinguishably to the concentration measured in the sampling network Through 1989 the Site was in production and the plutonium from the building effluent systems was the major contributor to the plutonium in the ambient air Resuspension from contaminated soil areas was a source of plutonium during this time but the contribution was not directly quantifiable due to the effluent source contribution

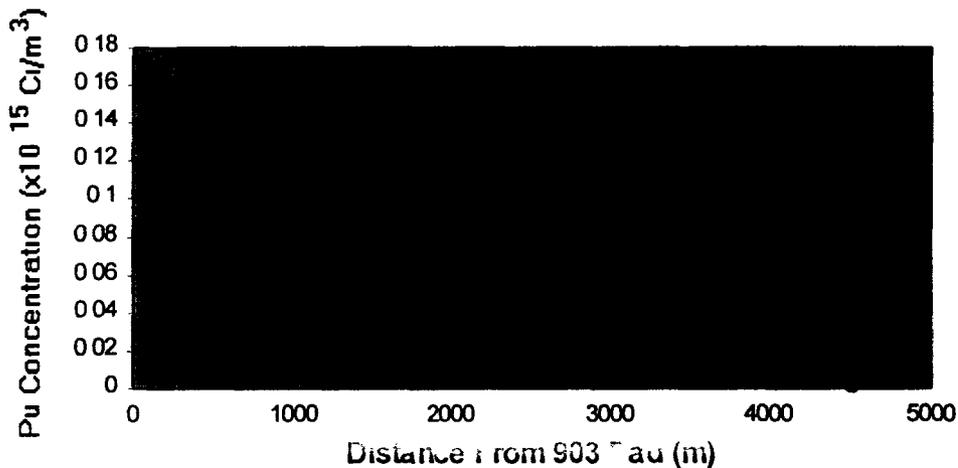


Figure 7 1 Plutonium Air Concentration vs Distance from 903 Pad (92 Data)

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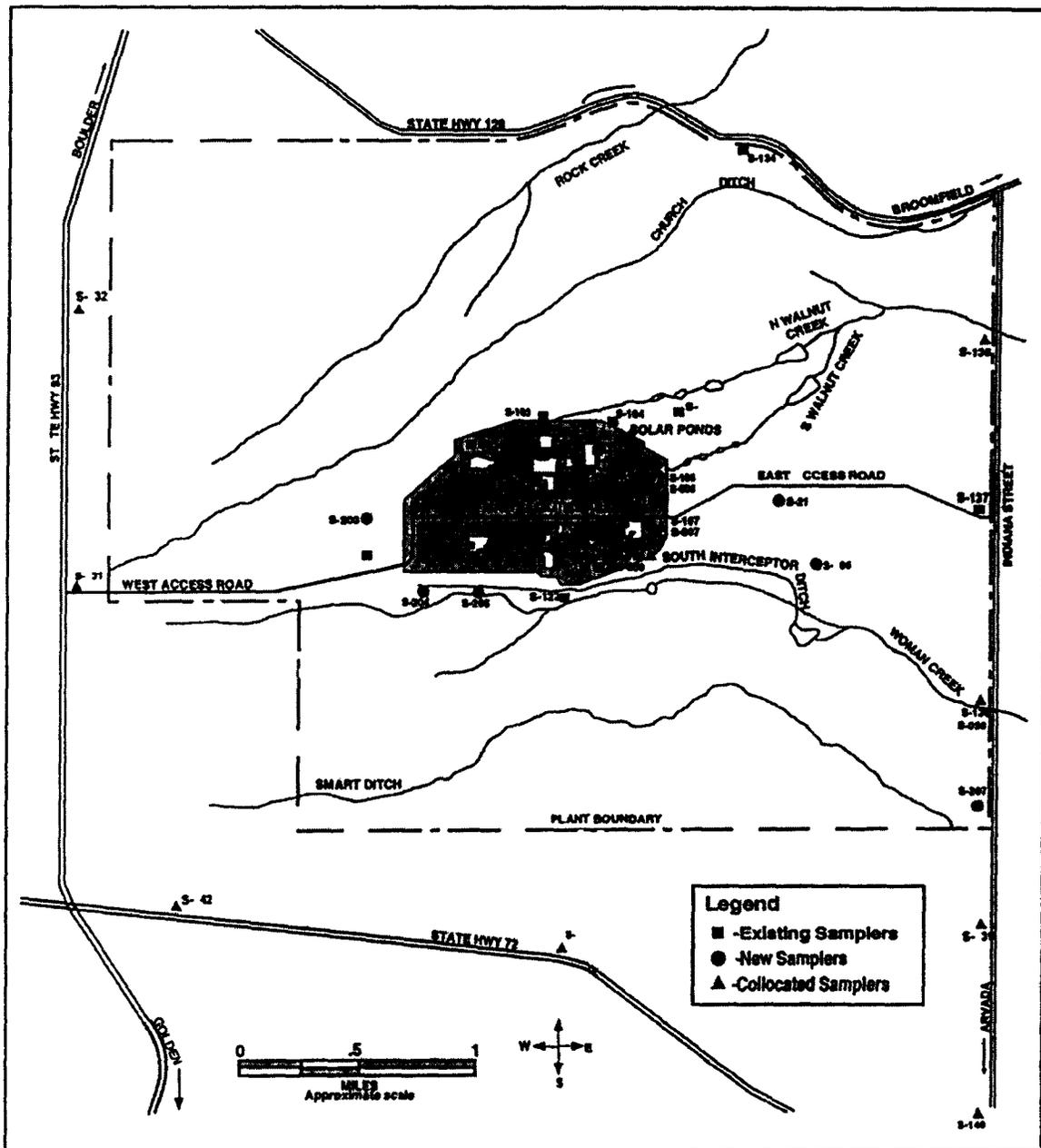


Figure 7 2 RAAMP Sampler Locations

From 1989 through 1995 the Site was essentially dormant and activities shifted to planning for cleanup and decontamination and demolition of Site structures. During this period overall Site airborne emissions decreased and resuspension from the contaminated soils contributed the majority of the plutonium captured in the air samplers. In 1996 cleanup of

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contaminated soils waste disposal areas and buildings was initiated and these project specific emissions became measurable contributors of plutonium to the air once again masking the contribution from 903 plume area emissions (see 1996 *Radionuclide Air Emissions Annual Report*) If a study were to better quantify the natural transport processes the best data available for the quantification of the movement of actinides appears to be from 1990 to 1995

Figure 7 3 shows the historical trend in plutonium concentration at a sampler (S-007) located just southeast of the 903 Pad Note that this sampler was not significantly impacted by emissions (mostly uranium) generated from the remediation activities conducted in 1996 because a large fraction of the cleanup emissions occurred during several relatively short (two week) periods when the prevailing winds transported primarily project emissions from the project site toward the northwest away from the 903 Pad Area and the S 007 sampler The short periods of emission are documented in a CDPHE report (Feb 97 Quarterly Data Exchange Meeting) from a sampling platform located at the project site

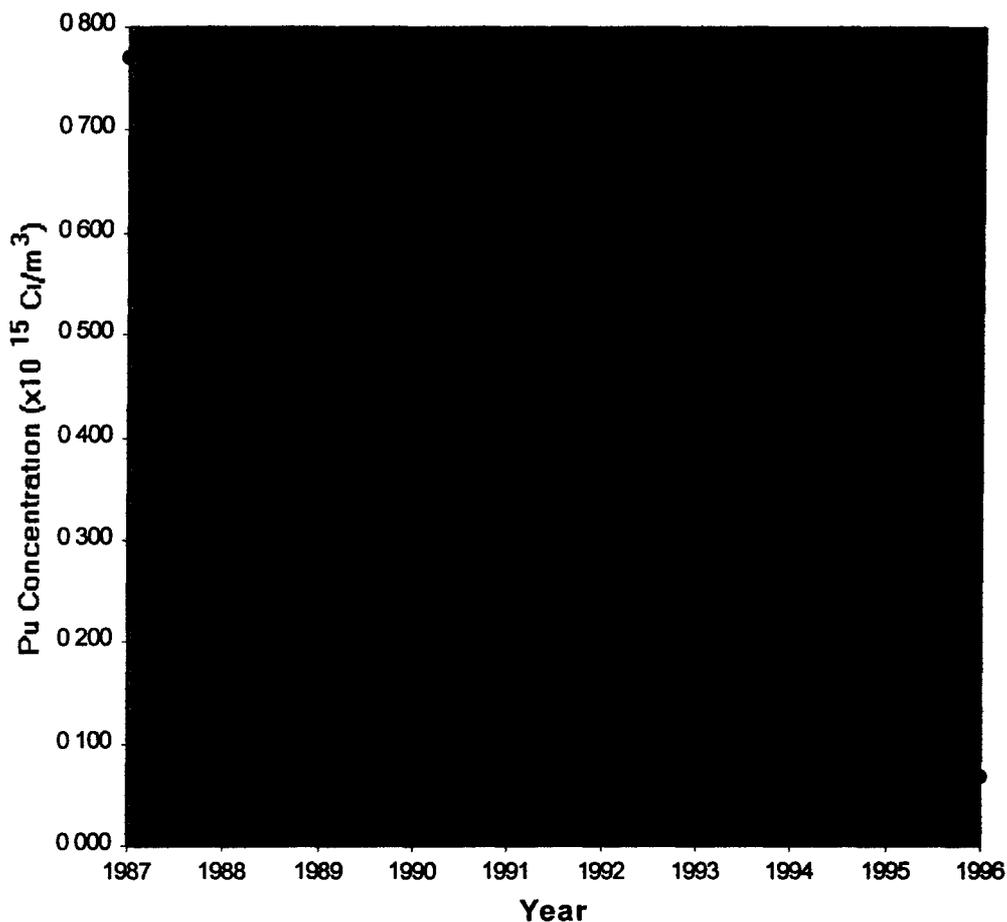


Figure 7 3 Annual Mean Plutonium Air Concentrations at S 007

7 2 2 Resuspension and Dispersion Mechanisms

Research on particulate resuspension took place for over 10 years at the Site beginning in the late 1970s. The Site offers a unique opportunity to study resuspension. Most dust resuspension studies have focused on resuspension from earth moving activities including piles of dirt exposed to the wind. Because the area contaminated by the 903 emissions is covered with vegetation, the resuspension rate from the soil is orders of magnitude less than that observed from earth moving activities on exposed soil.

Several studies (Citation) have suggested that resuspension at the Site does not occur solely from surface soils at high wind speeds through saltation as in a typical resuspension

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scenario but much of resuspension may occur instead from vegetated surfaces and may occur at much lower wind speeds than are typically associated with resuspension (DOE 1995a) This study also suggests that the vegetated cover receives the plutonium through rain splash The splash deposits fine particulates on the surface of plants where they are available for resuspension at very low wind speeds The rain splash resuspension mechanism may explain the "chronic year round low level ambient air plutonium concentrations observed at the Site from 1990 to 1995

Plutonium is a very heavy element giving it an aerodynamic characteristics the same as those of a much larger soil particle This means plutonium particles that are actually able to be resuspended and held aloft by the ambient air behave aerodynamically like much larger soil particles but have the activity of the small particle Fine plutonium particles may also be attached physically or chemically to soil particles resulting in radioactive particles that are transported in the same manner as soil The resuspended material from the 903 Pad Area is likely to be made up of three types of particles soil particles which are of typical soil density and that therefore disperse typically and soil particles with attached plutonium which have the same dispersion characteristics as pure soil and possibly some fine dispersable plutonium particles

Data have been collected near the 903 Pad that indicate the typical sizes of ambient airborne radionuclides The data show that the amount of plutonium collected is generally proportional to the total mass collected and is distributed rather uniformly across size ranges from sub micron to greater than ten microns The study was completed under conditions when no major cleanup activities were in progress and the Site was not in production Consequently the size fractionation data available are characteristic of resuspended material from the 903 Pad Area (DOE 1995a)

Other data collected specifically for cleanup activities by new Radioactive Ambient Air Monitoring Program (RAAMP) samplers could be used in a limited capacity to determine

size fractionation of resuspended material from earth moving activities

Data have been taken from samplers at a range of distances from cleanup activities and have been fractionated at approximately 10 micrometers aerodynamic diameter

7 2 3 Sampling Summary

The RAAMP samplers in use during most of the 1990 to 1995 time frame were Site designed medium volume air samplers. All samplers were located within the typical adult breathing zone (about 5 feet from the ground) operated continuously at a flow of 25 cubic feet per minute (cfm) and collected total suspended particulates (TSP). Samples were collected on an 8 x 10 inch fiberglass filter which is rated to be 99.97% efficient for conditions routinely encountered in ambient air sampling. Filters were collected biweekly from all samplers and composited for monthly analysis. Isotopic analysis was then performed for plutonium 239. These samplers are known as the old RAAMP samplers. New size fractionating RAAMP samplers that operate at 40 cfm and collect material on two substrates were installed in 1995. Particles larger than approximately 10 micrometers are collected on an oiled substrate while particles smaller than 10 micrometers are collected on a standard 8 x 10 inch fiberglass filter.

Forty three samplers were used in the RAAMP network. Twenty five were located inside the Site boundary, 14 were at the Site perimeter and 14 were in the communities adjacent to the Site. Typically many of the samplers collected such small amounts of radioactive material that the activity was below quantifiable detection limits.

Several of these samplers were located near the 903 Pad Area. Three samplers were located at the east end of the 903 Pad itself. Three others were located east of the 903 Pad area along a several mile stretch of the Site boundary. Several other samplers were located well to the east of the Site boundary. Those samplers outside the Site boundary typically do not yield data above quantifiable detection limits.

The samplers at the 903 Pad historically showed results above the analytical uncertainty while the samplers at the Site boundary only occasionally yield data above quantifiable detection limits

7 2 4 Suggested Uses of the Previously Collected Data

Several possible uses of existing data and samples might be beneficial to the Actinide Migration study

- Quantify the different resuspension mechanisms at the Site using data from various resuspension studies and using known dispersion mechanisms develop a long term model of actinide migration then validate the model using monitoring data
- Characterize the composition of actinide contaminated particles and compare the information with the observed size activity distribution
- Review data to estimate settling rates of resuspended actinide-contaminated particles for use in long term migration scenarios

7 3 REFERENCES

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DOE 1996b *Final Phase I RFI/RI Report Woman Creek Drainage Operable Unit 5* Rocky Flats Environmental Technology Site Golden CO RF/ER 96-0012 UN April 1996

DOE 1996c *Final Phase I RFI/RI Report Walnut Creek Drainage Operable Unit 6* Rocky Flats Environmental Technology Site Golden CO RF/ER 95 0119 UN February 1996

Hurr R T 1976 *Hydrology of a Nuclear Processing Plant Site* Rocky Flats Jefferson County Colorado U S Geological Survey Open File Report 76 268

Renard K G Foster G R Weesies G A McCool D K and Yoder D C coordinators 1997 *Predicting Soil Erosion by Water A Guide to Conservation Planning With the Revised Universal Soil Loss Equation (RUSLE)* U S Department of Agriculture Agricultural Handbook No 703 404 pp

8 0 CONCEPTUAL MODEL OF ACTINIDE TRANSPORT AT RFETS

Conceptual models are developed to understand how a site "works" both chemically and physically. In environmental work, conceptual models are used to tie together the relationships between surface and subsurface geology, soil types, contaminant sources, contaminant types, and distribution by air, groundwater, and surface water transport of contaminants. Development of a conceptual model based on existing data and an understanding of the problem forces the identification of data deficiencies and where the problem is not known or not understood completely. Completion of the conceptual model will include filling in the data deficiencies and integrating the new data into the model.

The conceptual model of actinide distribution and transport at the Site explains the relationship between soil types, actinide distribution, hydrogeology, and the potential effects of the actinides on surface water quality and air quality. The conceptual model will be used to assist in evaluating and designing remedial actions. This model of actinide transport at RFETS is a living document that will be revised and updated as new data are obtained. The conceptual model in this section updates the conceptual model distributed to Stakeholders on June 6, 1997. New data will be generated over the next two years (fiscal years 1998 and 1999) that will refine the conceptual model of actinide distribution and transport at RFETS. In addition, data from sitewide and project specific monitoring will be reviewed and additional analysis of existing data will be performed to confirm, enhance, or change the conceptual model.

Transport of actinides at RFETS includes both chemical and physical mechanisms. Physical transport is the movement of the media that contains the actinides (i.e., soil or water). Chemical transport is the transfer of actinides in solution or from one physical form to another, for example, the chemical transfer of actinides from soil particles to solution in

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groundwater Once the site specific transport mechanisms are understood they can be quantified and remedial design decisions can be made based on this understanding

Potential transport pathways data and data evaluation needs for better understanding actinide migration from RFETS have been identified These pathways are shown in Figure 8 1 and are generally broken down into discussion by environmental media

- Soil The transport of surface soil particles by overland flow to surface water in the Walnut and Woman Creek Drainages The factors affecting transport of actinides in soil are discussed in terms of the data for actinides in RFETS soils Location of soils on slopes or flat areas grain size and sorting or armoring of soil all affect the amount of soil available for erosion The soil type moisture conditions prior to a storm event and the length and intensity of a storm also affect soil erosion rates
- Surface water and sediments Potential remobilization of actinides in the ponds and sediments by both physical erosion (mobilization of pond sediments to surface water) and chemical partitioning (from actinide-contaminated sediments to surface water in a dissolved phase) will impact surface water quality
- Air The air monitoring performed at RFETS collects the fraction of actinides that is transported by wind The long term transport of actinides by wind erosion is discussed based on the factors that affect air transport
- Groundwater Chemical and physical mechanisms are discussed that contribute to actinide migration from groundwater to surface water under the environmental conditions encountered at RFETS The discussion also addresses groundwater transport and the soil action levels in relation to long term protectiveness of surface water

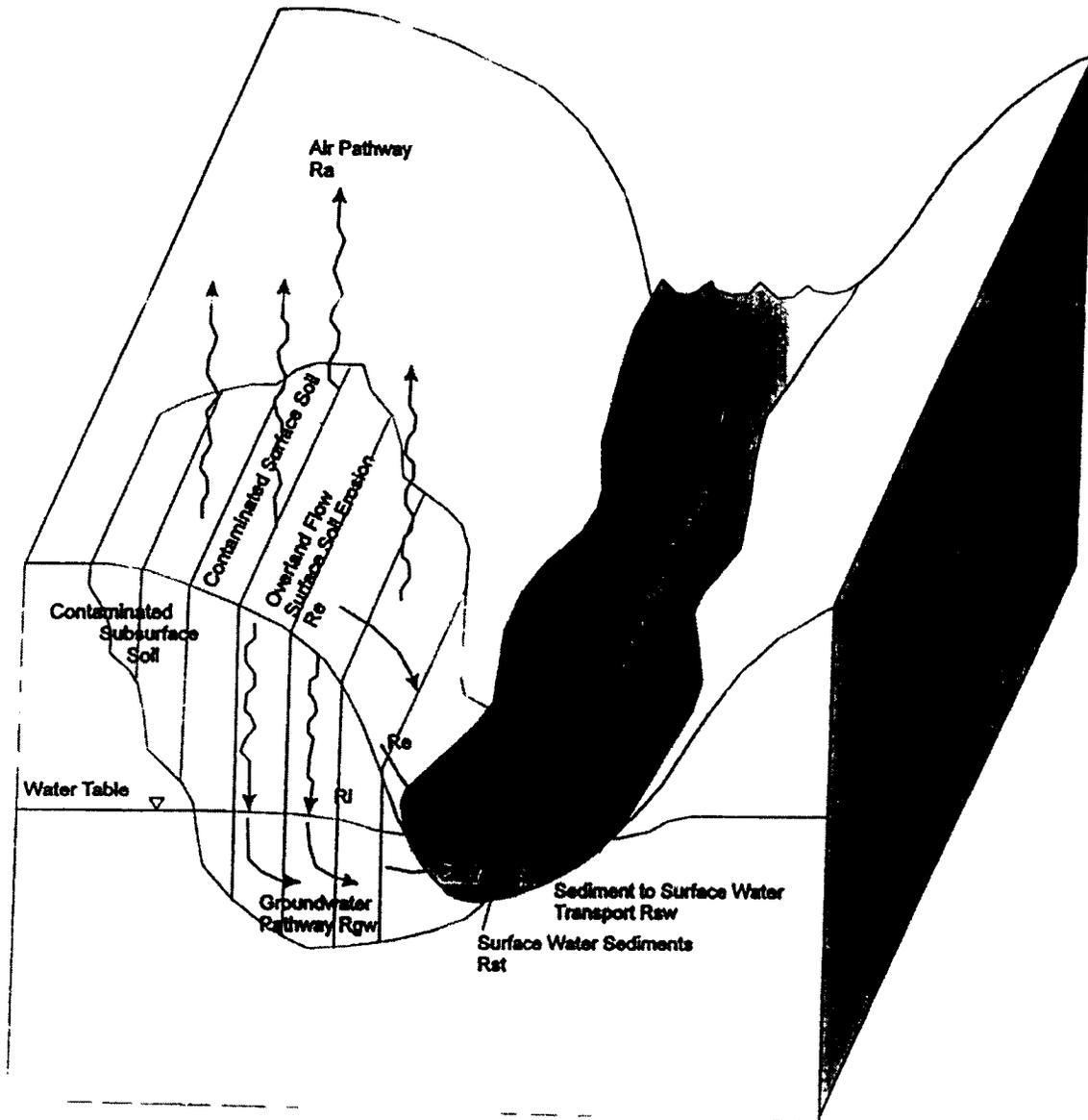


Figure 8 1 Preliminary Conceptual Model of Potential Actinide Transport Pathways

R = rate of actinide transport (subscripts indicate transport mechanism)

Ra = air

Re = soil erosion

Rgw = groundwater

R_i = infiltration

Rsw = sediments to surface water

Rst = sediment transport

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Actinides in these media can be broken down into two phases a solid (particulate) phase and a dissolved phase The particulate phase consists of the solid parts of soil or sediments and solids suspended in surface water and groundwater Figure 8 1 illustrates the environmental media that contain actinides at RFETS and the contaminant transport pathways

The rates of transport from the soils are represented by R in Figure 8 1 For example R_a is the rate of transfer of actinides from soil to air by wind erosion The subscripts designate the transfer process Once the ranges of transport rates are known for the various mechanisms the relative proportions of total actinide transport by each mechanism can be determined

8 1 SUMMARY AND SIGNIFICANCE OF ACTINIDE DISTRIBUTION AT RFETS

This section summarizes the current actinide distribution at RFETS and relates this distribution to specific sources and events Plutonium and americium will be discussed separately from uranium because they generally have different sources and types of contamination as well as different chemical associations and transport mechanisms

8 1 1 Actinide Distribution In Soils

The general conceptual model of plutonium and americium in soils at RFETS is that the majority of soil contamination was deposited by drum leakage at the 903 Drum Storage Pad and subsequent air borne redistribution during repairs of the Pad area in the late 1960s/early 1970s as discussed in Section 2 5 Minor redistribution of plutonium and americium in RFETS soils has occurred since this initial set of events (Meyers et al 1996)

As noted in Section 6 and discussed below and in the surface water/sediment section plutonium loading to North and South Walnut Creek surface waters appears to be due to low level plutonium soil contamination in IHSSs 101 141 and 165 in the northeastern IA. The history of the IHSSs is discussed in Section 2.5.

Plutonium and americium surface soil contamination northeast of the Solar Ponds is shown in Figures 4.1 and 4.2. This contamination corresponds in part with IHSSs 101, 165 and 141. This surficial contamination may be affecting sediment quality and water quality in the North Walnut Creek (IHSS 165) and South Walnut Creek (IHSS 101 and 141) and to a lesser extent 165) drainages. Comparing the surface distribution of these actinides as shown in Figures 4.1 and 4.2 there appears to be some minor wind blown surface soil contamination related to these IHSSs.

Plutonium and Americium in Subsurface soils

Vertical profiles of plutonium and americium activities with depth reflect transport by all applicable mechanisms over the past 30 years. As presented in Section 4, plutonium and americium are concentrated in the upper 15 to 20 cm (6 to 7 in) in RFETS soils. This is true for soils with widely varying concentration activities and at varying distances east and southeast of the 903 Pad (Figures 4.7 and 4.8) implying that the plutonium and americium in these soils was deposited from the same source in the same manner and at the same time. These data also imply that the same downward transport mechanisms apply to these soils and that downward transport is extremely slow. Aside from particle transport along macropores, the vertical migration of plutonium and americium is quite limited (Litaor et al. 1995, DOE 1995). Krey et al (1976) suggested that for depths below 6 cm, diffusion transport better describes plutonium distributions than particle transport via infiltration flow.

Downward transport in the top 6 inches of soil is dominated by macro pore flow and by bioturbation caused by both vegetation (roots) and animals (worms, insects, burrowing

animals) Root channels can extend to a depth of two to three feet Downward transport below this zone of bioturbation and macro pores is dominated by solution chemistry The data consistently indicate that downward movement of plutonium and americium via solution and colloid transport is very limited

As expected the vertical profile of plutonium and americium in soil beneath the 903 Pad in areas where previous surface soil removals have occurred and in areas where there has been surface disturbance since 1970 is different than in undisturbed soils

Uranium in Surface and Subsurface Soils

Elevated concentration activities of uranium in surface soils are found in the area of the Old Landfill (IHSS 115) and the Solar Ponds (IHSS 101) (Figures 4 9 4 10 and 4 11) In general wind transport does not appear to be a significant transport mechanism for uranium in surface soils in the 903 Pad Area (Litaor 1995) High activities of uranium isotopes are present in some RFETS soils (see Tables 3 2a and 3 2b) Uranium contaminated soils have been transported into some site drainages (Efurd et al 1993) but appear to be effectively controlled by the onsite detention pond system

The spotty occurrence of high levels of uranium in RFETS soils is also characteristic of naturally occurring uranium in Jefferson County including the RFETS area (DOE 1993 DOE 1995 Morse and Moody 1992 Bolivar et al 1978) Through the use of TIMS analysis to individually measure uranium 234 uranium 235 uranium 236 and uranium 238 one can approximate the relative proportions of natural and anthropogenic uranium at RFETS

If total uranium activities lie within the range of activities for background uranium even though the isotopic mixture indicates an anthropomorphic origin remediation may not be justified as it would constitute cleanup below background levels

Low levels of uranium in subsurface soils below the Solar Ponds may contribute to a uranium and nitrate-contaminated groundwater plume that is partially intercepted by the ITS. Figure 5.5 shows that there is no actinide contamination above the Tier II actions in the subsurface below the Solar Ponds. This is discussed further in Sections 8.1.4 and 8.2.2. The surface soil data for uranium do not show a pattern that would be indicative of the Old Landfill or the Ash Pits as sources of wind-blown uranium contamination.

8.1.2 Actinide Distribution In Surface Waters and Sediments

As discussed in Section 6, plutonium and americium activities in pond influent waters are higher for storm events than for continuous flow sampling, confirming that these actinides are associated with fine particulates and total suspended solids. In a study by Rokop et al (1993) to determine the source and amount of radioactivity in pond water and sediments at RFETS, the following conclusions were drawn: (1) the largest amount of anthropogenic activity detected was in the pond sediments; (2) one gram of pond sediment contained about 50 times more plutonium than 1 liter of pond water, and both depleted uranium and plutonium are mobile through the drainage system; (3) the largest source of alpha activity is naturally occurring uranium and its decay product radium, which provides 70-459 times more alpha activity than that produced by the plutonium in the terminal ponds; and (4) the largest source of anthropogenic activity was depleted uranium, which comprised 20-50% of the total alpha activity in the samples.

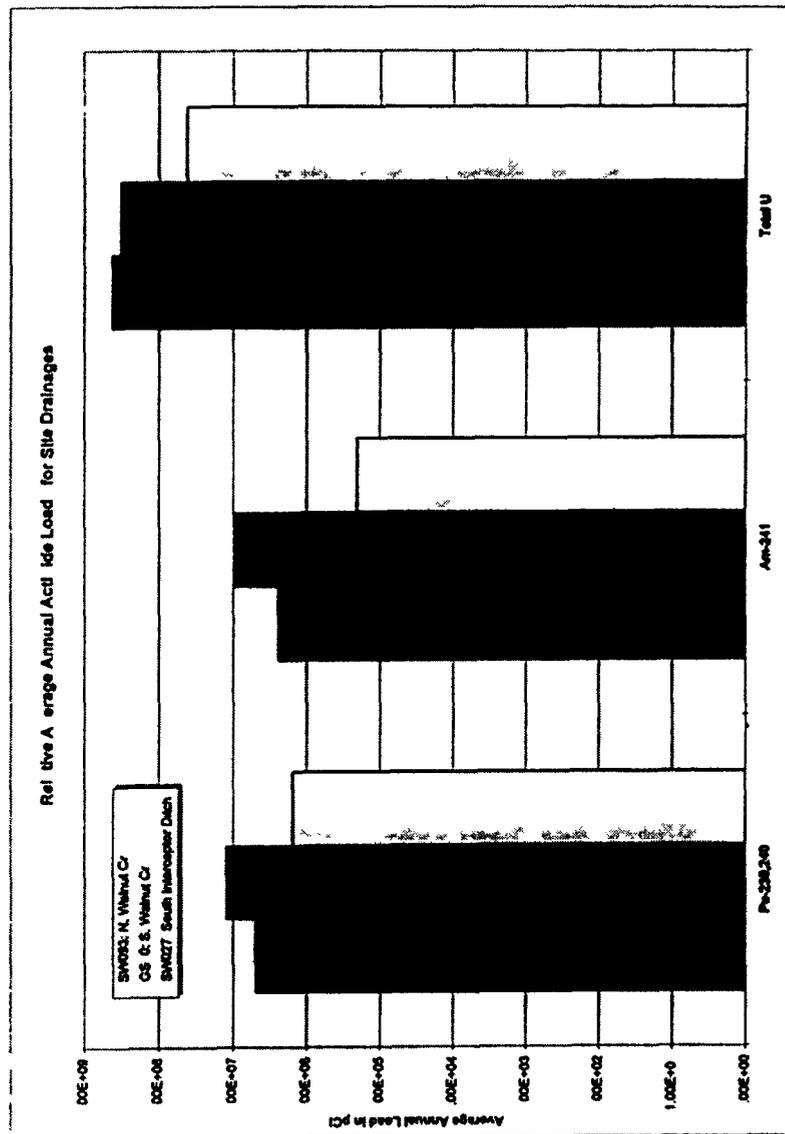
The surface water data presented and summarized in Section 6 indicate that plutonium and americium are entering surface water at locations north and east of the protected area in the center of the IA (GS27 and GS 28) and in Woman Creek (SW 027).

(6)

Uranium in Surface Water

Uranium in surface water drainages is associated with base flow in areas that drain the area around the Solar Ponds (SW093 and GS10) but appears to be associated with storm runoff in the area south of the 881 building and south of the Old Landfill (SW027) where there is surface uranium contamination (see detailed discussion of the data in Sections 4 2 and 6 2) Total uranium activity (the sum of the three isotopes) in the discharge ponds is similar for all three ponds (A-4 B 5 and C 2) In North Walnut Creek Pond A-4 (1 79 pCi/L) contains about 1/4 the total uranium activity of the continuous flow for SW093 (4 265 pCi/L) which is below the Solar Ponds At South Walnut Creek Pond B 5 (2 272 pCi/L total uranium) is slightly lower than the continuous flow sampled at GS 10 (2 975 pCi/L) Pond C 2 (2 797 pCi/L total uranium) contains slightly lower total uranium than is recorded for SW027 (3 247 pCi/L) in the SID Since the flow paths are not direct from the surface water sampling locations to A and B Series Ponds correlation of actinide content in the ponds to the influent data may not be meaningful

Efurd et al (1994a) examined pond water data comparing Uranium 235 and Uranium 238 ratios in the RFETS ponds to determine the amount of depleted uranium from RFETS operations that was contributed to each pond The calculation assumed a simple two component system and that the depleted uranium released contained 0 5% U 235 This conservative estimate determined the maximum amount of U 235 in the pond water that is attributable to the depleted uranium from RFETS The upper ponds A 1 A 2 B 1 B 2 B 3 and B-4 contained measurable quantities of depleted uranium This study estimated that essentially 100% of the uranium in Ponds A 1 and A 2 originated as depleted uranium All other ponds except Pond C 1 were estimated to contain mixtures of naturally occurring and depleted uranium Approximately 50% of the uranium present in Ponds A-4 and C 2 and 20% of the uranium in Pond B 5 was estimated to have originated as depleted uranium Efurd et al (1994b) noted that no uranium enriched in U 235 was found in the pond waters sampled at RFETS



Gaging Station SW003
 Volume-Weighted Average Actinide Activity for 10/1/96-5/24/97 (pCi/L)

Analyte	Pu-239,240	Am-241	Total U
	0.034	0.017	2.911

Average Annual Discharge 505271 CF

Average Annual Load (pCi)

Analyte	Pu-239,240	Am-241	Total U
	4865155	2432577	416643106

Gaging Station GS10
 Volume-Weighted Average Actinide Activity for 10/1/96-5/24/97 (pCi/L)

Analyte	Pu-239,240	Am-241	Total U
	0.116	0.081	3.068

Average Annual Discharge 3712117 CF

Average Annual Load (pCi)

Analyte	Pu-239,240	Am-241	Total U
	12194749	9566570	32230077

Gaging Station SW027
 Volume-Weighted Average Actinide Activity for 10/1/96-4/30/97 (pCi/L)

Analyte	Pu-239,240	Am-241	Total U
	0.045	0.006	1.221

Average Annual Discharge 1212400 CF

Average Annual Load (pCi)

Analyte	Pu-239,240	Am-241	Total U
	1545063	206011	41923257

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Actinides in Pond and Stream Sediments

Woman Creek Within the SID plutonium appears to be entering the ditch at locations below SW038 in the vicinity of sampling locations SW035 and SW036 (Efrud et al 1993)

Walnut Creek Efrud et al (1993) noted that the highest plutonium concentrations were in sediments collected from Ponds B 1 B 2 and B 3 More plutonium was detected in the sediment from the A 1 Bypass than from sediments in the A series ponds The area above the B series ponds was disturbed in 1972 (Johnson et al 1974) and pond sediment profiles from a 1993 1994 study show a marked increase in plutonium concentrations in pond sediments during this disturbance

Off Site Sediments Core profiles of Pu 239/240 in Great Western Reservoir and Stanley Lake both show a peak concentration corresponding to the 1969 1970 903 Pad releases Concentrations in other sections of the core samples are generally greater than background levels before and after the peak concentration until more recent deposits corresponding to the early 1980s This indicates that liquid as well as airborne releases contributed to the contamination in the Great Western Reservoir sediments (Meyer et al 1996)

The maximum concentrations measured in Stanley Lake sediments were much lower than those from the Great Western Reservoir sediments Concentrationa prior to and after the 1969 1970 peak were generally within the range of concentrations measured at background locations indicating sediment contamination was air borne

8 1 3 Actinide Movement in Air

As discussed in Sections 2 5 4 1 7 2 and 8 1 above the major plutonium and americium contamination of surface soil at RFETS is the result of airborne soil contamination from the 903 Pad and to a lesser extent from the Solar Ponds Through 1989 the Site was in production and the plutonium from the building emissions systems was the major contributor to the plutonium in the ambient air Resuspension from the area contaminated by the 903 Pad was a source of plutonium during this time but the contribution was masked by the effluent source contribution

Several studies (DOE 1995b) have suggested that resuspension at RFETS does not occur solely from surface soils at high wind speeds as in a typical resuspension model Significant resuspension also occurs from plant surfaces and occurs at much lower wind speeds than are typically associated with resuspension These studies also suggest that the plant surfaces receive the plutonium through rain splash The splash deposits fine particulates on the surface of the plants where they are available for resuspension at low wind speeds The rain splash resuspension model may explain the "chronic" year round very low level ambient air plutonium concentrations observed at the Site from 1990 to 1995

8 1 4 Actinide Distribution in Groundwater

Plutonium and Americium in Groundwater

Analytical data for plutonium and americium in samples of filtered (0 45 mm) groundwater suggest that there are no plutonium/americium contaminant plumes in Site groundwater As discussed earlier (Section 5) a few wells contained activities of total (unfiltered) plutonium or americium greater than the 0 15 pCi/L plutonium or 0 145 americium Tier II

action levels Most of these wells were drilled through the 903 Pad or the surficial plutonium contamination associated with the 903 Pad Other wells where plutonium has been detected do not show a pattern indicative of a groundwater plume The data suggest that the plutonium detected in groundwater wells is the result of down hole cross contamination of the well from plutonium-contaminated surface soils As discussed in Section 3 plutonium and americium tend to sorb onto soils with a wide range of cation exchange capacity and the transport of colloidal plutonium long distances through groundwater has not been proven

Uranium in Groundwater

Analytical data for uranium isotopes in samples of filtered (0.45 mm) and unfiltered groundwater show relatively high (>40.0 pCi/L for U-233+234 and U-238) maximum activities for both background and non background areas

As discussed in Section 5.2 groundwater in some nearby background areas (Coal Creek Canyon east of RFETS) contains significantly higher amounts of naturally occurring uranium (Moody and Morse 1992) than does the sitewide groundwater at RFETS Concentrations of naturally-occurring uranium increase from west to east across RFETS and dissolved (filtered) uranium concentrations are generally higher than total (unfiltered) uranium concentrations This supports the discussion in Section 3 (geochemistry) that the naturally occurring alkaline and oxidized groundwater at RFETS favors dissolution of uranium and therefore transport of uranium via groundwater

8 2 TRANSPORT

The previous discussion of actinide sources and concentrations in various media indicates that the following major sources and transport mechanisms should be incorporated into the conceptual model

Actinide loading to surface waters has been calculated based on continuous flow monitoring (Section 6). The continuous flow samplers monitor storm events as well as baseflow so these calculations are a reasonable estimate of the amount of contaminants entering the drainages at the measurement points. Tables 6 5 6-6 6 8 6 9 and 6 14 show the calculated actinides to North Walnut Creek South Walnut Creek and Pond C 2. The calculated loading to these drainages based on surface water monitoring data is shown in Figure 9 2. The current data does not distinguish contaminant loading into Pond C 2 from the drainage via SW027 versus the loading due to flow over the low level contaminated surface soils surrounding Pond C 2 which are not captured by SW027. The amounts entering the drainages are less than the amount leaving the ponds based on the pond discharge data. Actinides are accumulating in the sediments of the ponds. The main sources of plutonium and americium actinide loading to surface water appear to be the 903 Pad and related surface soil contamination areas IHSS 141 IHSS 165 and as yet unidentified low level sources.

8 2 1 Surface Soil Transport

The three main transport pathways for actinides in surface soils are surface soil erosion to surface water surface soil leaching to subsurface soil and groundwater and wind transport to other surface soils and to surface water. Surface soil leaching to subsurface soil and

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groundwater is not considered to be a major transport pathway at RFETS as supported by the data presented in Sections 3 4 and 5

8 2 2 Groundwater Transport to Surface Water

Plutonium and Americium

Transport of plutonium and americium via groundwater to surface water is not a significant mechanism at RFETS The presence of plutonium and americium in groundwater at RFETS is related to surface contamination and well construction There is no evidence of a plutonium or americium "plume" of groundwater contamination There is no contaminated groundwater plume to intercept surface water and no effect on surface water quality

Uranium

Groundwater downgradient of the Solar Ponds contains levels of uranium that are elevated relative to the groundwater underlying most of RFETS although not generally above background levels The isotopic ratio and activity ratios of uranium isotopes in groundwater downgradient of the Solar Ponds do not indicate an anthropomorphic origin Transport of uranium via groundwater to surface water is occurring in the area of the Solar Ponds Surface water data indicate that at SW93 uranium is associated with baseflow which is fed by groundwater SW93 is north of the Solar Ponds at the toe of the slope and is where groundwater not intercepted by the ITS will surface

Two groundwater wells on the south side of the IA contain isotopic ratios of uranium that may indicate anthropomorphic uranium Well 61093 is located just south of the Old

Landfill between the landfill and the SID and Well 07391 is located several hundred feet south of the 903 Pad

8 2 3 Surface Water and Sediment Transport

Plutonium and Americium Migration in Surface Water and Sediment

Although transport of plutonium and americium-contaminated particles from surface soils to the creek beds by erosional processes occurs in the Woman Creek and Walnut Creek drainages the pond systems established and maintained by DOE effectively inhibits sediment and actinide transport within the drainage systems Plutonium and americium mobility can be controlled by careful management of the ponds and by controlling the timing of pond discharges

Uranium Migration in Surface Water and Sediment

Dissolved uranium is transported by both surface water and groundwater (Sections 3 and 5) Uranium-contaminated surface soils are also transported overland to drainages where the particulates ultimately accumulate as sediment in the series of onsite detention ponds Isotopic studies of the pond water and pond sediment indicate that depleted uranium from RFETS has accumulated in the pond system (Efurd et al 1993) The summary statistics (Section 6) for uranium isotopes in sitewide surface water show activities significantly higher than those for background surface water

Overall the pond systems perform well in containing uranium such that surface water leaving the site generally meets the site specific standards for uranium

8 3 REFINEMENT OF CONCEPTUAL MODEL

Distribution and transport mechanisms for the actinides of concern at RFETS are different for various areas of RFETS due to contamination history and the physical setting of the contaminants. Based on the data summarized in this report, the conceptual model of the site has been further subdivided into three main areas of actinide contamination, each with a different combination of dominant actinides and transport mechanisms. These areas are

- Pad Area and related soil contamination
- Northeast Protected Area including the Solar Ponds and IHSS 141 and 165 and
- Old Landfill (IHSS 115)

The 903 Pad Area and related plutonium surface soil contamination is shown in Figure 8 3 (low levels of plutonium extend to the east of this figure boundary). This figure demonstrates the extensive spread of surface contamination contrasted to the limited spread of actinide contamination in groundwater. The location of some of this contamination on relatively steep slopes can be seen by examining the topographic contours on Figure 8 3. Soils on slopes are more subject to erosion by water than soils at the top of slopes, such as along Central Avenue. In the 903 Pad Area, the dominant actinide contaminants are plutonium and americium, and the dominant transport mechanism to surface water is erosion by overland flow. If large expanses of this area were de-vegetated, wind transport and wind loading to the surface water would increase.

The Northeast PA and Solar Ponds area is shown in Figure 8 4 with surface plutonium contamination. Although the surface plutonium concentrations in this area are much lower than in the 903 Pad area, plutonium and americium are entering surface water at or above the RFCA surface water action level immediately downgradient of this area. In this area, the main contaminant in surface soil is plutonium (and americium) and the main actinide contaminant in groundwater is uranium. The uranium in groundwater is possibly being

mobilized by the high nitrate concentrations in groundwater originating in the solar ponds. The existing data do not indicate the presence of anthropomorphic uranium downgradient of the Solar Ponds, possibly indicating that the increased uranium downgradient of the Solar Ponds is due to the mobilization of naturally occurring uranium by the high nitrate groundwater plume in this area.

The Old Landfill, IHSS 115 is shown in Figure 8.5. The landfill is of interest because wells downgradient from it indicate the potential presence of anthropomorphic uranium in groundwater. Uranium-238 activities up to 1000 pCi/g have been noted in the Old Landfill.

It is recognized that additional areas of actinide contamination exist; however, based on the objective of protection of surface water, these are the three main areas of interest. Additional areas may be added to the conceptual model as more surface water monitoring is conducted in the IA related to site decontamination and decommissioning of buildings and potential exposure of under building contamination to erosion and transport mechanisms.

Actinide loading to surface water via groundwater appears to be an issue for uranium in the Solar Pond Area and potentially downgradient of the Old Landfill. Since there is no plutonium or americium plume of contamination, loading of these actinides to surface water cannot be calculated and is considered minimal compared to the other transport mechanisms for plutonium and americium.

Subsurface uranium contamination contributions to groundwater are addressed in the groundwater discussions and in the discussion of further data needs.

8 4 IMPLICATIONS FOR REMEDIAL ACTIONS

Erosion modeling will be conducted in fiscal year 1998 assuming 903 Pad Area soils above the action level are either removed or covered. The modeling results will be compared to the results discussed in Section 7.1 and the stream loading summarized in Figure 8.2.

It is expected that the planned soil remediation activities for the 903 Pad Lip and surrounding actinide contaminated soil areas together with watershed improvement projects currently in progress (e.g., application of road sealant, revegetation of buffer zone roads, installation of silt fences, SID improvements) and demolition and decommissioning activities will serve to reduce actinide levels in pond influent and effluent waters.

8 4 1 903 Pad and Lip Area Surface Soil Cleanup

Because the 903 Pad and Lip area contains activities of plutonium and americium above the radionuclide RFCA action levels, a remedial action must be performed to reduce potential exposure of the public and workers to the soils contaminated above action levels. The action levels are dose based and were not determined in conjunction with determination of the surface water quality protection levels. Erosion calculations and surface water mass loading calculations indicate that surface water continues to be affected by the 903 Pad and Lip Area especially after significant storm events. The Conceptual Model needs further refinement specifically in the erosion estimates of the 903 Pad and related soil contamination to determine the benefits and extent of remediation recommended.

Remedial action may include all or some of the following:

- Reduction of source term concentration by soil removal

- Steps to reduce rate of transport to surface water

 - redistributing soils away from slopes

 - armoring soils with cobbles and gravel

 - other anti-erosion measures and

 - Capping

The extent of cleanup will be determined not only by the extent of soils contaminated above action levels but on the amount of soil needed to be removed or otherwise prevented from eroding to protect surface water quality

Volume reduction potential will be addressed based on treatability studies. Indications are that because the actinides are concentrated in the fine grained fraction, significant volume reduction can be achieved.

Air quality impacts during cleanup are a significant concern to the public, especially since the major surface soil contamination is due to wind borne contamination. Previous actions and studies indicate that air emissions can be controlled by use of dust suppressants, limiting the size of the excavation, and limiting the amount of time the excavation is open.

The previous air releases from the 903 Pad occurred when heavy equipment operations exposed large areas of soil contamination without mitigation over several months to several year periods (Meyers et al 1996).

Surface water impacts during cleanup can be controlled by some of the same engineering controls that reduce airborne spread of contaminants, as well as common engineering practices such as the use of silt fences, isolation of contamination areas with berms or other surface water flow control measures, and covering contaminated soil stockpiles.

8 4 2 Northeast Protected Area And Solar Ponds

This area has a two fold contamination and migration problem. The nitrate/uranium groundwater plume is affecting the surface water quality downgradient/downstream of the Solar Ponds. Remediation of the groundwater may involve use of a passive barrier to intercept and precipitate the contaminants from the groundwater or otherwise remove the contaminants from the groundwater. Gathering additional data and refining the conceptual model of groundwater transport in this area is a focus of the Fiscal Year 1998 activities discussed in Section 9.

The occurrence of plutonium above the RFCA surface water standard is driving additional surface water sampling in this area. The current conceptual assumes that this contamination comes from low level surface soil contamination however this has not been confirmed and is based on the spatial association of the surface contamination with the surface monitoring stations showing exceedences. Since the levels of surface contamination are low investigation of other upstream sources of the exceedences is necessary. Further evaluation of existing data for these IHSSs and related surface water monitoring locations may be needed.

8 4 3 Watershed Improvements

The data summarized herein indicate that actinide loading is occurring in areas that are not affected by the 904 Pad or related soils. The detailed subbasin surface water monitoring data will be examined more closely in an effort to determine specific sources of actinide loading. Once the sources have been located watershed improvements may be used as temporary measures to reduce the actinide loading to surface water.

Changing transport rates and efficiency in surface water might be accomplished by allowing controlled areas for sediments to drop out of the water column

8 4 4 Pond Operations/Pond Flow Through System

At or before the time of RFETS closure the surface water system will be converted to a flow through system that does not require active pumping for surface water management. The design of this flow through system must take into account that the pond sediments are the reservoir for actinide accumulation on site.

Although contaminated pond sediments are currently diluted by sediments from uncontaminated areas they contain significant zones of contamination from past activities.

8 5 REFINING/CONFIRMING THE CONCEPTUAL MODEL

Key areas where the conceptual model needs data (including additional interpretation of existing data) to be complete are

- Further analysis of groundwater data from the Solar Ponds area
 - Correlation of uranium with other analytes including nitrate, nitrate carbonate, iron, sodium, pH, and TSS. These correlations should be performed on uranium versus each of the other major ions, and
 - TIMS analysis of groundwater to determine uranium isotope and activity ratios. At this time the conceptual model is incomplete because it is not known whether the uranium increase downgradient is due to mobilization of natural uranium or the movement of anthropogenic uranium that was introduced into the Solar Ponds.

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- analysis of existing nitrate/nitrate groundwater data downgradient of the solar ponds to determine whether there has been a change since the ponds were cleaned out
- temporal analysis of existing uranium isotope data downgradient of the old landfill and in the well downgradient of the 903 pad that showed skewed isotope and activity ratios to determine whether anthropomorphic uranium is migrating in these areas
- Additional analysis of the groundwater data downgradient of the Old Landfill to determine if other analytes are showing the influence of contaminants migrating
- Further evaluation of the drainages in the IA that have high plutonium and americium loading to determine the sources of these actinides
- Temporal analysis of data at all surface water stations to determine if actinide loading is related to on site activities as well as to storm events
- Surface sediments and pond configuration to determine what would happen if RFETS went directly to a flow through system
- What kinds and amounts of loading to offsite surface water would be expected following a 100 year 24 hour storm

8 5 1 Incorporation of 1997 Soil Analysis

The main transport mechanism for plutonium and americium at RFETS is from surface soil to surface water via overland erosion. The 1997 evaluation of speciation and K_d of plutonium and americium in soils will be used as input to modeling to quantify the contribution of actinide migration to groundwater via chemical transport and hence the relative contribution to surface waters. This is similar to the modeling approach that was used to set the RFCAs soil action levels for volatile organic compounds that are protective of surface water via the groundwater pathway. The pond sediment K_d values will be used to quantify the chemical transport of the actinides from sediments to surface water.

Both of these chemical transport mechanisms are expected to be relatively small compared to overland transport however they are being addressed to assure completeness and so the basic transport factors are not overlooked.

8 5 2 Incorporation of 1997 Pond Sediment Analysis

Surface sediments and pond configuration answers to the questions what would happen if RFETS went directly to a flow through system and what kinds and amounts of loading to offsite surface water would be expected following a 100 year 24 hour storm will depend in part on the 1997 data from pond sediment analysis

Sediment inventory and depth profiles collected during 1997 will be used to calculate the amount and rate of actinides that would be released for each major drainage for various scenarios of conversion of the ponds to a flow through system. Converting the ponds to wetlands may mitigate both actinide transport and may deplete the nitrates from the Walnut Creek Drainage thereby reducing the amount of uranium that can be transported in solution

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9 0 RECOMMENDATIONS

The near term projects to evaluate and control actinide migration include finalization of the conceptual model for actinide migration for the Woman Creek and Walnut Creek Drainages remediation of the 903 Pad Lip Area and containment of the Solar Ponds plume Recommendations for finalization of each of these tasks are summarized below

9 1 WOMAN CREEK DRAINAGE

The 903 Pad Lip Area is a major source of windblown contamination Initiation of remediation activities is scheduled for the 2000 2001 timeframe In order to attain this schedule the Conceptual Model will have to be finalized in order to determine site specific cleanup levels for protection of surface water The goals of the proposed actinide migration evaluation work for fiscal year 1998 is to complete a draft Conceptual Model for actinide migration at RFETS to quantify the rates of actinide transfer among media and to model and test transport rates on proposed remediation scenarios for the 903 Pad In order to accomplish this the following activities will occur

- Based on existing data 1997 column studies to determine actinide speciation and partition coefficient/investigation results on soils and sediments 1998 erosion modeling and air dispersion modeling rates will be established for each transport mechanism (air erosion and leaching to groundwater) for plutonium and americium
- Additional laboratory studies will be conducted to determine mobility under various environmental conditions
- A mass loading analyses will be performed in order to allocate the maximum contribution of plutonium and americium to each basin that will not exceed the Segment 5 stream action levels

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- The above data will be input into the draft Conceptual Model for actinide migration and tested on the watershed below the 903 Pad Lip Area to determine site specific cleanup levels and
- Using the rates established in the draft Conceptual Model the feasibility of remediating the 903 area soils by reducing erosion rates to surface water will be evaluated

The draft Conceptual Model for the Woman Creek Drainage will be peer reviewed in 1998 and comments incorporated into the final Conceptual Model in 1999 This information will be used to finalize the IM/IRA for the 903 Pad and Lip Area

9 2 WALNUT CREEK DRAINAGE

The sources of actinide contamination in the Walnut Creek Drainage are uranium from the Solar Ponds and plutonium americium and uranium from the IA

9 2 1 SOLAR PONDS PLUME

An action is required to protect surface water in South Walnut Creek from contaminated groundwater originating from the Solar Ponds There is a proposed RFCA milestone to construct a plume containment system in fiscal year 1999 In order to ensure attainment of this milestone additional plume characterization will be conducted in 1998 to refine knowledge of the extent of the uranium and nitrate plumes originating from the Solar Ponds area A groundwater model will be developed specifically for the uranium and nitrate plumes to evaluate alternatives for managing the plume The model will incorporate 1997 laboratory study data that determined site specific uranium K_d s and speciation which affect the mobility of actinides The groundwater model will be used to help evaluate select and design a preferred alternative for remediation of the Solar Ponds plume

 August 1997

9 2 2 WALNUT CREEK DRAINAGE

The North and South Walnut Creek Drainages will be further examined to define the source of plutonium loading and a Conceptual Model for actinide migration will be developed. The following tasks will be performed:

- Complete inventory of plutonium, americium, and uranium in each pond based on the fiscal year 1997 data and all previous historical data.
- Review of existing data from the IA and B2 to determine source areas.
- Evaluate pond sediment grain size data in relationship to contamination to determine what stream energy is required to erode contaminated sediments from the ponds and under what circumstances these energies are likely to occur and
- Establish rates for each transport mechanism.
- Perform a mass loading analysis in order to allocate the maximum contribution of plutonium and americium to a basin that would not exceed the Segment 5 stream action levels and
- The above data would be input into the draft Conceptual Model for actinide migration and tested on the Walnut Creek Drainage to determine site specific cleanup levels.

APPENDIX A

A 0 TREATABILITY STUDIES

Radioactive contamination affects large volumes of soil at RFETS and this contaminated soil will have to be remediated before the site can be released for other uses. At the 903 Pad and Lip Area, for example, up to 31,300 cubic meters of soil may require remediation. Offsite disposal costs for this may be as high as \$96 million at current prices (if it must be handled as LDR-compliant low level and low level mixed waste). The radioactive contaminants cannot be destroyed by any type of treatment, but the high costs of disposal make treatment to concentrate contamination and reduce the volume of soil requiring disposal extremely attractive.

A number of treatability studies have been conducted at RFETS focusing on reducing the volume of contaminated soil which must be remediated. These studies have focused either on concentrating contaminants into a fine grained, high radioactivity fraction through physical sizing/classification (Section 8.1) or partitioning contaminants into a liquid through chemical extraction (Section 8.2). Both approaches have shown some promise in laboratory scale tests. Calcination, heavy liquid separation, and magnetic separation also have been examined, but do not appear promising and will not be discussed further here (Arnold et al. 1980, Navratil et al. 1979, Navratil and Kochen 1982, Olsen et al. 1980).

A 1 SIZE CLASSIFICATION FOR VOLUME REDUCTION

The surface soil of RFETS is rocky and plutonium contamination is concentrated in the fine grained portion of the soil (Kochen and Navratil, Little 1980, Olsen et al. 1980, Garnett et al. 1980, Stevens and Rutherford 1982, Pettis and Kallas 1988). Undisturbed soil from Rocky Flats has a highly variable grain size and the soil has been mixed and commingled with fill during construction, maintenance, and Site cleanup. In general, RFETS soil has bimodal grain size distributions with a natural break in the 5 to 35 mesh (4.5 and 0.7 mm) range (Table A.1).

Table A 1 Size Distribution of a Plutonium Contaminated Soil Sample from the 903 Pad Area

Size Fraction	Fraction of soil
> 1 inch	47/
1 inch to 5 mesh	31/
5 to 8 mesh	2/
8 to 35 mesh	5/
<35 mesh	15/

From Stevens and Rutherford (1982)

Separation of coarse and fine soil fractions generally involves three processes liberation of fine clasts from coarse clasts separation of the size classes and dewatering of the coarse and fine fractions (Eagle et al 1993) These processes have been extensively studied at RFETS because of the potential usefulness of size separation in reducing the volume of radioactive soil which must be disposed (Olsen et al 1980 Stevens and Rutherford 1982) Techniques which have been investigated for liberation include scrubbing attrition scrubbing ultrasonic cleaning and use of surfactants/dispersants Techniques which have been investigated to separate fine-grained clasts from coarse grained clasts include dry screening wet screening hydraulic classification desliming flotation and mineral jigs (Garnett et al 1980 Kochen 1991)

Pettis and Kallas (1988) conducted bench scale testing and reported that simple room temperature screening of 903 Pad soils was effective at decontamination greater than 4 mm size fraction (approximately 60 wt % of the total) to less than 5 dpm/g (2.3 pCi/g) plutonium and americium The greater than 2.4 mm size fraction (approximately 65 wt % of the total) was decontaminated to less than 12 dpm/g (5.5 pCi/g) Plutonium and 6 dpm/g (2.7 pCi/g) americium by wet attrition scrubbing The remaining soil fraction of less than 2.4 mm was treated by attrition scrubbing ultrasonic scrubbing oxidation calcination desliming flotation and heavy liquid density separation

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Although somewhat guarded because of results for selected size fractions they concluded that attrition or vibratory scrubbing and either mineral μg or acid leaching of this fraction would be effective for a decontamination goal of 14 pCi/g

Kochen and Navratil (1982) examined dry sieving and wet sieving techniques to separate soil into fine grained contaminated and coarse grained uncontaminated fractions (the results are shown in Table A 2) Clasts coarser than 4 mm removed by dry sieving had average plutonium/americium activities of 100/67 (3% and 13% of untreated soil values respectively) Wet screening even without overt attempts to liberate contamination from particle surfaces reduced plutonium/americium activities to 2/2 pCi/g (0.07% and 0.4% of untreated soil values respectively) Attrition scrubbing and wet screening were tested on the underflow for liberation and separation respectively These techniques reduced the volume of contaminated fine grained soil by a further 30% (Kochen and Navratil 1982)

A conceptual three step process for concentrating soil contamination in soils from the 903 Pad was developed by Rockwell (1978) Primary treatment would separate soil clasts coarser than 4 mm for return as clean fill The underflow of contaminated fine grained material from primary treatment would enter secondary treatment for sizing at 0.4 mm The primary and secondary treatment were projected to reduce the volume of contaminated soil approximately 70% "

Olson et al (1980) attempted to separate 903 Pad soil into contaminated and uncontaminated fractions using the following process a pH 11 NaOH solution and wet screening attrition scrubbing with high pH Calgon solution and decantation attrition scrubbing with low pH solution and decantation and scrubbing followed by cationic flotation (Table A 3)

The high pH and low pH attrition scrubbing were both more effective than simple wet sieving in liberating plutonium contamination but both techniques produced secondary aqueous waste streams with admixed cations and anions. Wet sieving by itself reduced Pu 239 in the coarse fraction to approximately 100 pCi/g (the results are shown in Table A-4). The plutonium activity in the greater than 5 mesh fraction can be reduced further to approximately 11 pCi/g using vibratory grinding.

Table A 2 Actinide Activity in Size Fractions of Soil from the 903 Pad

	Weight%	Pu		Am	
		pCi/g	/	pCi/g	/
Untreated		3370		518	
Dry screening					
>4 0 mm	60	100	3/	67	7.76/
4 0 to 2 4 mm	4	630	19/	122	0.94/
2 4 to 0 42 mm	12	1395	41/	252	5.83/
< 0 42 mm	24	13 100	390/	1845	85.5/
Wet screening					
>4 0 mm	61	2.25	0.07/	2.25	0.26/
4 0 to 2 4 mm	4	54	1.6/	10.8	0.08/
2 4 to 0 42 mm	11	302	9/	72	0.14/
< 0 42 mm	24	12 150	360/	2 025	95.1/
From Kochen and Navratil, Navratil and Kochen (1982), Hicks and Blakeslee (1981)					

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Table A 3 Effectiveness of Sizing Treatments

Technique	Solution composition	Clean fraction	Waste fraction
Untreated			20 300 pCi/g
Wet screening	pH 11 with NaOH 1g to 1 ml	35 mesh+ 13 5 pCi/g 60 70 wt%	<35 mesh 50 700 pCi/g 30-40 wt%
Attrition scrubbing high pH solutions	Calgon soln at high pH	fast settling 2 pCi/g 80 wt /	slow settling 101 000 pCi/g 20 wt%
Attrition scrubbing low pH solutions	2/ HNO ₃ 0 2/ HF 2/ pine oil 5/ Calgon	fast settling 2 2 pCi/g 84 wt /	slow settling 101 000 pCi/g 16 wt%

Compiled from Olsen et al (1980) these results also are summarized in Pettis and Kallas (1988)

Table A 4 Pu 239 Activity by Size-Fraction in Soil from the 903 Pad Area Treated by Wet Sieving

Size Fraction	²³⁹ Pu Activity
> 1 inch	Not determined due to analytical difficulties
1 inch to 5 mesh	100 pCi/g
<5 mesh	approximately 15 100 pCi/g

After Stevens and Rutherford (1982) Initial activity of soil approximately 3400 pCi/g

Stevens and Rutherford (1982) tried a variety of solutions in conjunction with vibratory grinding to remove plutonium from the coarse fraction. Use of more aggressive acid based solutions increased the removal of plutonium. Caustic solutions increased the rate at which plutonium was removed from the coarse fraction but did not improve the final decontamination effectiveness. Navratil and Kochen (1982) tested similar solutions in combination with attrition scrubbing to remove actinides from sand sized clasts (2.4 to 0.42 mm). Large uncertainties in the determination of actinide activity preclude any conclusion concerning the relative effectiveness of extracting solutions (the results are shown in Table A 5).

Table A 5 Effectiveness of Attrition Scrubbing in Removing Actinide Activity from Sand Sized Soil Fraction

Solution	Pu		Am	
	pCi/g	/ remaining	pCi/g	% remaining
Untreated	333±128		54±32	
Distilled water	56±40	17/	13±2.8	23/
80°C distilled water	81±0	24/	16±1.6	29/
Calgon 5 wt%	63±8	19/	8±7	15/
Calgon 10 wt%	30±7	9/	7±0.8	13/
Citric Acid 0.1M	72±16	22/	14±9	27/
Oxalic Acid 0.1M	31±1.2	9/	9±4	18/
Sodium Silicate 0.1 wt%	67.5±8	20/	12±1.6	23/
Basic H Detergent 1 vol /	207±64	60/	27±9.5	51/
OT B (Anionic Surf) 0.1 wt%	99±48	30/	21±8	39%
C 61 (Cationic Surf) 0.1 wt%	77±8	23/	16±8	3/

From Navratil and Kochen (1982)

Kochen et al (1991) reinvestigated attrition scrubbing and rotary scrubbing to liberate contaminated particles and dry sieving and wet sieving to separate contaminated and uncontaminated soil fractions. They also experimented with a gravimetric separator (mineral jig) to separate contaminated and uncontaminated fractions. The soil used in these experiments was skewed fine by the exclusion of large rocks but even with this exclusion more than 50% of all soil samples were caught on a 6 mesh (4 mm) screen. Attrition scrubbing helped liberate actinide contamination from the 4 to 0.42 mm size fraction and mineral jigs removed some contamination from the 0.42 to 0.25 mm size fraction. The low efficiency of plutonium removal from the 0.42 to 0.25 mm size fraction (69%) the high actinide activity in this fraction and the low abundance of this size material in RFETS soil (4% by weight) makes the technique of questionable value (as shown in Table A 6)

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Table A 6 Liberation and Separation Techniques for Various Size Fractions of Windblown Soil from the 903 Pad

Size fraction	/ of Soil	Liberation Technique	Separation Technique	Pu/Am removal Efficiency
50 to 4 0 mm	>50 /	NaOH pH 12.5	wet sieving	98%
4 to 2 4 mm	1 /	NaOH pH 12.5	wet sieving	83 /
4 to 2 4 mm	1 /	NaOH pH 12.5	wet sieving	95 /
4 to 2 4 mm	1 /	attrition scrubbing NaOH pH 12.5	wet sieving	78 /
2 4 to 0 42 mm	6 /	rotary scrubbing NaOH pH 12.5	wet sieving	66 /
2 4 to 0 42 mm	6 /	NaOH pH 12.5	wet sieving	90 /
2 4 to 0 42 mm	6 /	attrition scrubbing NaOH pH 12.5	wet sieving	53 /
2 4 to 0 42 mm	6 /	rotary scrubbing NaOH pH 12.5	wet sieving	53 /
<0 42 to 0 250 mm	4 /	NaOH pH 12.5	mineral jig	69%

Compiled from Kochen et al (1991)

Garnett et al (1980) attempted a pilot scale test of size based decontamination of actinide contaminated soil from the 903 Pad. A trommel removed material coarser than 5 mesh (50.5% of soil by weight) and a screener removed clasts between 5 and 35 mesh (20% of soil by weight). The fraction coarser than 35 mesh was "clean" and was returned to the area of contamination and the fraction passing through 35 mesh was dewatered for disposal using a clarifuge, flocculant, and centrifuge. A number of problems were encountered in the pilot scale study including:

- Recontamination of coarse clasts by recycled washwater
- Recontamination of "clean" material from the build up of contamination on the trommel screen and other equipment
- Breakdown of the flocculated waste by shear in the centrifuge
- Plugging of hydrocyclones studied for use to separate 35 mesh material
- Plugging of drum filters tested to replace centrifugation

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Similar problems have been encountered by other workers attempting to treat radioactively contaminated soils (Eagle et al 1993)

A 2 CHEMICAL EXTRACTION

Chemical extraction has been investigated at the laboratory scale to concentrate plutonium contamination in solution and allow clean soil to be returned to the environment (Stevens et al 1982 Triay and Loge 1994 DOE 1995a DOE 1995b) None of the techniques have been scaled up to pilot scale and there has been no full scale application of any of the chemical extraction techniques either at this site or at other sites

Techniques attempted on RFETS soils have included adding surfactants to disburse fine grained soil particles adding acids to depress the solution pH and increase actinide solubility adding complexing agents such as carbonate or EDTA to produce soluble complexes and adding redox agents to raise or lower the redox potential and favor soluble actinide species [such as Pu(III) or Pu(VI)] over insoluble actinide species [such as Pu(IV)] Encouraging results have been obtained with several combinations of reagents but practical chemical extraction of bulk soils may be difficult expensive and fraught with unexpected complications as demonstrated by the Pit 9 cleanup effort at the Idaho National Engineering and Environmental Laboratory

It also may be possible to use bacterially mediated reduction in place of reducing agents to facilitate removal of multivalent actinides from soils Iron reducing bacteria produce reducing conditions which presumably function in a manner similar to reducing agents to mobilize actinides Use of this technique as a practical remediation approach however is likely to be extremely difficult and expensive

Stevens et al (1982) investigated the effects of a variety of potential lixivants on finely ground soil from Rocky Flats and other DOE sites 1 kg aliquots of soil were extracted in a 8L ball mill The initial extraction was with 1L of solution and samples were subjected

to 3 subsequent extractions with 500 mls of fresh scrub solution. A final rinse consisted of 500 ml of water. Extraction efficiency was highest for the acidic solutions and lowest for the basic solution (Table A 7)

Table A 7 Extraction of Actinides from Finely Ground RFETS Soil by Scrub Solutions

	NaOH Solution pH 12.5	2% HNO ₃ 0.2% Pine	15% Calgon
Pu 239 removed	2.15±5%	57.5±7.8%	60.2±5.7%
Am 241 removed	15±17%	90.3±13%	84.3±8.4%

From Stevens et al (1982) error values are twice the reported standard deviation for duplicate measurements

Hicks and Blakeslee (1981) summarized a decade of soil characterization and bench scale attrition scrubbing studies on RFETS 903 Pad soils. Plutonium was removed with varying effectiveness by attrition scrubbing with hot (80°C) distilled water, hot solutions of reducing/oxidizing agents (H₂O₂, NaClO, and Na₂SO₃), hot solutions of complexing agents (Na₂CO₃, oxalic acid, and citric acid), hot detergent and surfactant solutions (Oakite NST and sodium dioctyl sulfosuccinate), Oxalic acid (0.1 M), sodium hexametaphosphate (10%), and 10% detergent solutions were among the more effective decontaminating systems for the 2-4-45 0 mm soil fraction and removed 95-98% of the plutonium and americium.

Table A 8 shows the results of experiments conducted by Triay and Loge (1994) to test simple complexants, standard leaching techniques, and redox agents to remove actinides from 903 Pad soils. The tests were designed on the basis of earlier experiments and knowledge of plutonium chemistry and included a range of reducing agents, oxidizing agents, complexing agents, and pH conditions. There was no attempt to optimize extraction conditions and results are for 2.5 g of RFETS soil equilibrated with 20 ml of extracting solution for 24 hours. The highest extraction efficiencies for plutonium were observed with citric acid and sodium dithionite (51%), with sodium citrate and ascorbic acid (51%), and with sodium citrate and sodium dithionite (61%).

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The effects of elevated temperature on the extraction efficiency of various combinations of reducing/oxidizing agents complexing agents and pH was investigated subsequently (DOE 1995b Table A 9) Americium removal efficiencies were similar in all complexant systems studied with the exception of carbonate Heating to 80°C improved americium extraction efficiency by 10 15% for most complexants High pH had little effect on americium removal again with the exception of the carbonate system

Plutonium extraction was less efficient than americium extraction across the board but converged with americium extraction efficiency when reducing agents were employed This could either be due to either valence difference between the two actinides (Am^{3+} should be relatively more mobile than Pu^{4+}) or to some difference in the location of americium and plutonium in soil (with americium concentrating in a more accessible location)

Plutonium extraction efficiency decreased with increasing pH for all complexants except carbonate The increased efficiency at high pHs in the carbonate system would be driven by transformation of bicarbonate to carbonate with a concomitant increase in the stability of anion actinide complexes The overall negative slopes of the other complexant systems represents precipitation/sorption of actinide hydroxide complexes on solid phases which is partially offset by increased actinide-complexant complexation as the complexants dissociation increases

The EDTA plutonium system and some other systems show more effective plutonium extraction at high temperatures than at low temperatures (see Figure A 1) This suggests that plutonium removal may be limited by dissolution rate and that it may be possible to increase plutonium extraction by increasing extraction time or taking some step to increase the dissolution rate

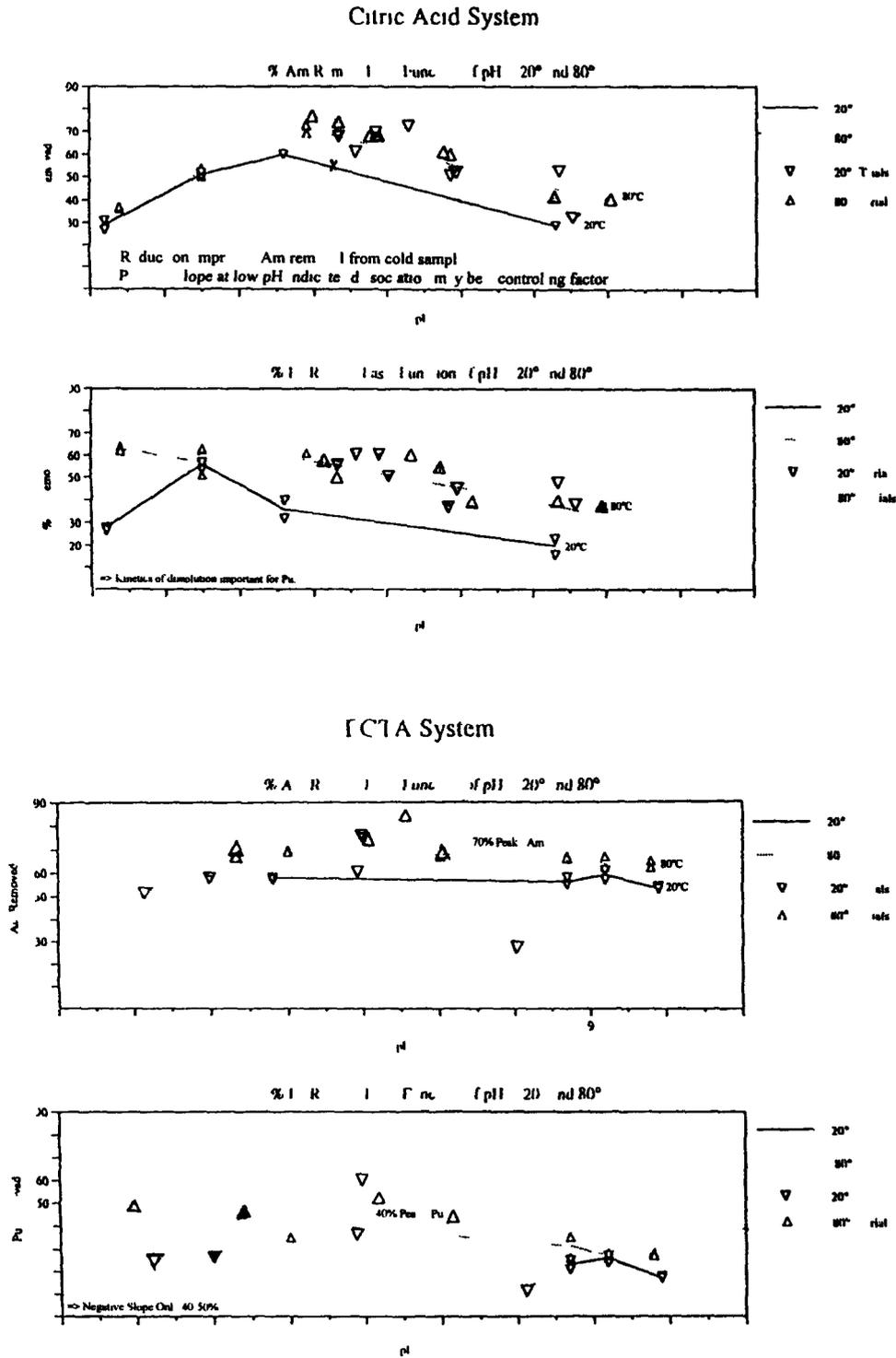


Figure A 1 Plutonium Extraction

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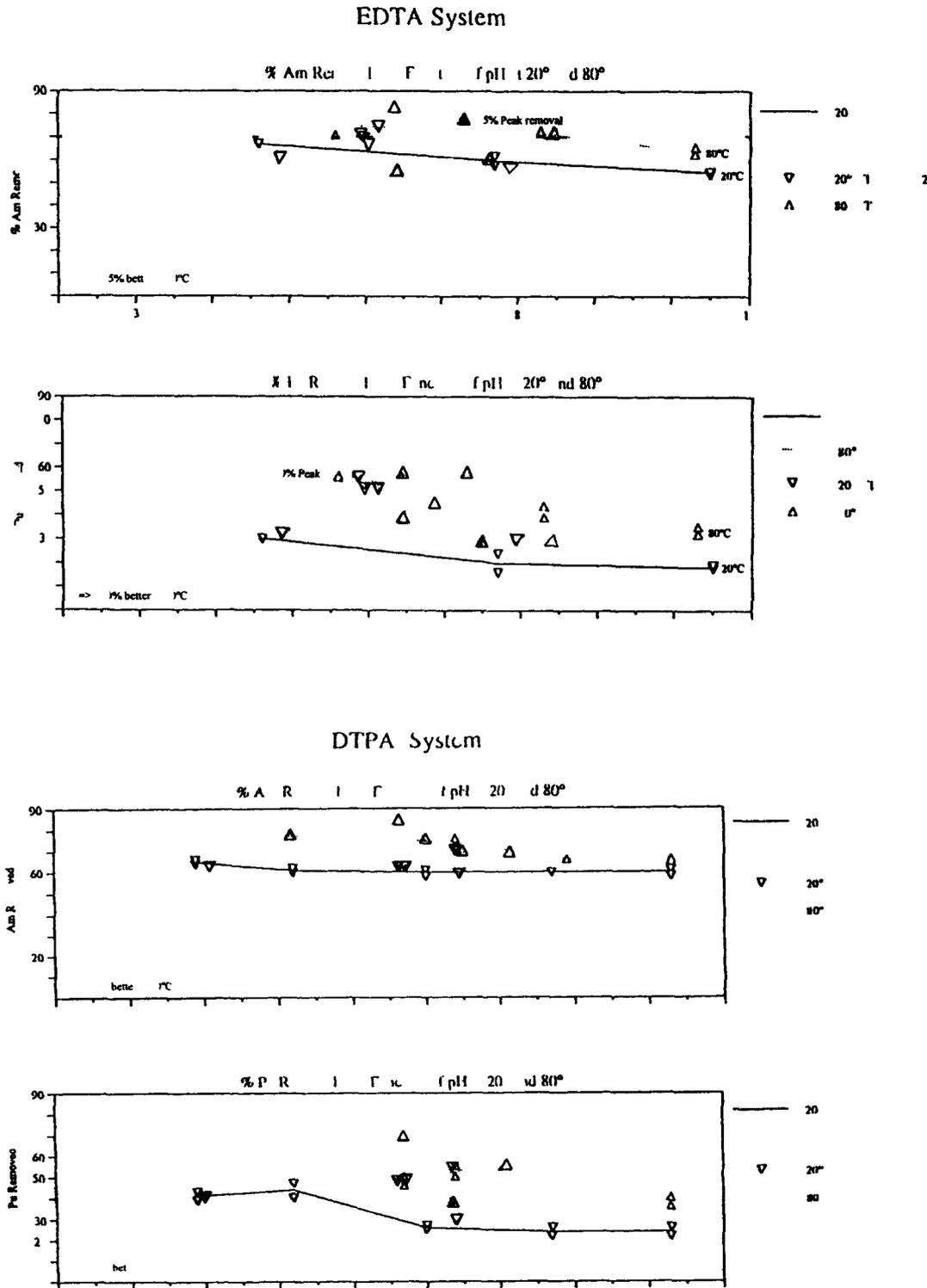


Figure A 1 cont Plutonium Extraction

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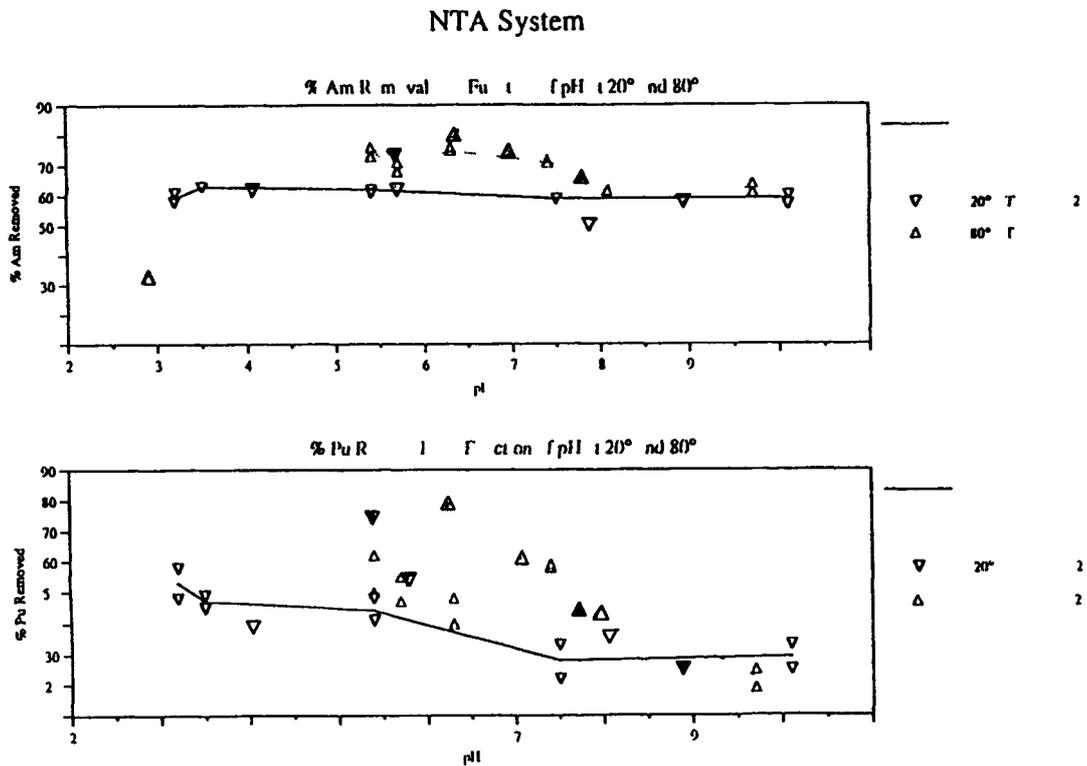


Figure A 1 cont Plutonium Extraction

Reducing agents generally increased removal of both americium and plutonium from RFETS soils when combined with most complexants (Table A 10 Figure A 1) The gain was largest for plutonium and room temperature solutions and plutonium extraction efficiencies approached the efficiencies for americium under non reducing conditions Hydrogen peroxide and oxidizing agents had little effect on the removal of actinides from soils and at high temperatures actually decreased removal in some systems possibly as a result of the destruction of the complexing agents

A 3 DATA GAPS

Remaining data gaps primarily relate to the application of particle sizing and chemical extraction techniques tested at the laboratory scale to practical cleanup of Site soils Of the two categories of techniques physical sizing appears to be the more promising for volume reduction because of the relatively uncomplicated nature of secondary waste One attempt

has been made to scale up physical separation techniques and a variety of issues were encountered during this scaleup. These issues will have to be addressed before the approach can be applied during cleanup.

Scaleup from laboratory scale processes to pilot scale processes has been systematically studied by a variety of authors (Bisio and Kabel 1985). Generally scale up ratios on the order of 50 to 100 are advisable for processes involving solids such as physical sizing to concentrate plutonium. Laboratory tests were limited to fractions of a cubic yard/hour and a pilot plant with a capacity of not more than 10 cubic yards/hour should be tested before physical sizing is applied in actual cleanup.

Chemical extraction schemes have larger practical problems than do physical techniques. None of the chemical extraction technologies have been demonstrated at pilot scale and many of the questions which will need to be answered before scaleup have not been addressed. The nature of the final waste stream following treatment, for example, is unclear. Many of the promising techniques use chelating or complexing agents and these agents are banned from radioactive disposal sites because they mobilize contaminants. Transition metals and other ions, moreover, will be present along with the actinides in the secondary waste and could require sophisticated and expensive treatment prior to disposal.

Table A 8 Plutonium and Americium Removal from Soil at Room Temperature for Various Combinations of Extractants

Extractant				Reactions	/ Am	/ Pu
Reducing agent	Oxidizing agent	Complexing Agent	pH			
		0.1 M Citric Acid		Complex formation	13 /	24 /
		0.1 M Citric Acid	2 eq NaOH	Complex formation Dissolution of silicates Destruction of organic matter	39 /	18 /

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3/ H ₂ O ₂	same	0 1 M Citric Acid	Destruction of organic matter reduction/oxidation of plutonium complex formation	14/ 42/	
	5/ NaOCl	0 1 M Citric Acid	Oxidation of organic matter oxidation of plutonium complex formation	3/ 32/	
	0 1 M Na ₂ S ₂ O ₈	0 1 M Citric Acid	Oxidation of organic matter oxidation of plutonium complex formation	7/ 30/	
0 1 M Na ₂ S ₂ O ₄		0 1 M Citric Acid	Reduction of iron sesquioxides reduction of plutonium complex formation	15/ 51/	0 1 M Na ₂ S ₂ O ₄ is the salt of an acid and should be weakly to strong basic
0 1 M NH ₂ OH HCl		0 1 M Citric Acid	Reduction of iron sesquioxides reduction of plutonium complex formation	2/ 25/	pH of 0 2 M aq soln 3 2
0 1 M Ascorbic Acid		0 1 M Citric Acid	Reduction of iron sesquioxides reduction of plutonium complex formation	16/ 18/	
3/ H ₂ O ₂	same	0 1 M Sodium Citrate	Destruction of organic matter reduction/ oxidation of plutonium complex formation	42/ 19/	
3/ H ₂ O ₂	same+0 01 M Fe ³⁺	0 1 M Sodium Citrate	Destruction of organic matter reduction/ oxidation of plutonium complex formation	43/ 43/	
	5/ NaOCl	0 1 M Sodium Citrate	Oxidation of organic matter oxidation of plutonium complex formation	12/ 0/	

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0.1 M Na ₂ S ₂ O ₈	0.1 M Sodium Citrate		Oxidation of organic matter oxidation of plutonium complex formation	45 / 29 /	
0.1 M Na ₂ S ₂ O ₄	0.1 M Sodium Citrate		Reduction of iron sesquioxides reduction of plutonium complex formation	57 / 61 /	Sodium citrate solution should be less acidic than a citric acid solution
0.1 M NH ₂ OH HCl	0.1 M Sodium Citrate		Reduction of iron sesquioxides reduction of plutonium complex formation	41 / 36 /	
0.1 M Ascorbic Acid	0.1 M Sodium Citrate		Reduction of iron sesquioxides reduction of plutonium complex formation	58 / 51 /	
	0.1 M EDTA	2 eq NaOH	Dissolution of silicates destruction of organic matter complex formation	39 / 8 /	
	0.1 M EDTA	3 eq NaOH	Dissolution of silicates destruction of organic matter complex formation	45 / 8 /	
3 / H ₂ O ₂	same	0.1 M EDTA	Destruction of organic matter reduction/oxidation of plutonium complex formation	47 / 26 /	
3 / H ₂ O ₂	same + 0.01 M Fe ³⁺	0.1 M EDTA	Destruction of organic matter reduction/oxidation of plutonium complex formation	0% / 0%	
	0.1 M EGTA	2 eq NaOH	Dissolution of silicates destruction of organic matter complex formation	30 / 7 /	
	0.1 M DTPA	3 eq NaOH	Dissolution of silicates destruction of organic matter complex formation	46 / 25 /	
	0.1 M (NH ₄) ₂ CO ₃		Weak complexant	0 / 6 /	

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3 / H ₂ O ₂	same	0 1M Na ₂ CO ₃	Reducing/oxid agent, weak complexant	0 /	0 /
3 / H ₂ O ₂	same+0 01 M Fe ³⁺	0 1M Na ₂ CO ₃	Reducing/oxid agent weak complexant	0 /	14 /
	5 / NaOCl	0 1M Na ₂ CO ₃	Oxidizing agent weak complexant	11 /	28 /
	0 1 M Na ₂ S ₂ O ₈	0 1M Na ₂ CO ₃	Oxidizing agent weak complexant	0 /	10 /
0 1 M Na ₂ S ₂ O ₄		0 1M Na ₂ CO ₃	Reducing agent, weak complexing agent	0 /	3 /
0 1 M NH ₂ OH HCl		0 1M Na ₂ CO ₃	Reducing agent weak complexing agent	0 /	0 /
0 1 M Ascorbic Acid		0 1M Na ₂ CO ₃	Reducing agent weak complexing agent	27 /	20 / (Ascorbate must complex in its own right)
		0 1 M NaHCO ₃	weak complexing agent	2 /	12 /
3 / H ₂ O ₂		0 1 M NaHCO ₃	Reducing/oxid agent, weak complexant	0 /	0 /
3 / H ₂ O ₂	same + 0 01 M Fe ³⁺	0 1 M NaHCO ₃	Reducing/oxid agent weak complexant	0 /	5 /
	5 / NaOCl	0 1 M NaHCO ₃	Oxidizing agent weak complexant	13 /	14 /
	0 1 M Na ₂ S ₂ O ₈	0 1 M NaHCO ₃	Oxidizing agent, weak complexant	0 /	10 /
0 1 M Na ₂ S ₂ O ₄		0 1 M NaHCO ₃	Reducing agent weak complexant	0 /	4 /
0 1 M NH ₂ OH HCl		0 1 M NaHCO ₃	Reducing agent, weak complexant	0 /	2 /
0 1 M Ascorbic Acid		0 1 M NaHCO ₃	Reducing agent weak complexant	22 /	11 / (Ascorbate must complex in its own right)
		0 1 M Nitrilotriace tic Acid	2 eq NaOH Complexing agent	48 /	2 /

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	0 1 M Nitrilotriacetic Acid	3 eq NaOH	Complexing agent	41 / 23 /	(Must be the effect of further dissociation of the NTA)
3 / H ₂ O ₂	0 1 M TETA		Reducing/oxid agent, complexant	0 / 9 /	
3 / H ₂ O ₂	0 1 M TETA	0 1 M NaHCO ₃		0 / 0 /	
	5 / NaOCl	0 1 M TETA	0 1 M NaHCO ₃	9 / 13 /	
	0 1 M Na ₂ S ₂ O ₈	0 1 M TETA	0 1 M NaHCO ₃	5 / 18 /	
3 / H ₂ O ₂	0 1 M TETA	0 1 M NaHCO ₃		0 / 0 /	
0 1 M NH ₂ OH HCl	0 1 M TETA	0 1 M NaHCO ₃		0 / 12 /	
0 1 M Ascorbic Acid	0 1 M TETA	0 1 M NaHCO ₃		3 / 11 /	
0 1 M Tartaric Acid		1 eq NaOH		0 / 5 /	
	0 1 MTEG	0 1 M NaHCO ₃		0 / 0 /	
3 / H ₂ O ₂	0 1 MTEG			0 / 0 /	
3 / H ₂ O ₂	Same +0 01 M Fe ³⁺	0 1 MTEG		4 / 3 /	
	0 1 MTEG	0 1 M DME	0 1 M NaHCO ₃	0 / 0 /	
	5 / NaOCl	0 1 MTEG	DME	0 / 8 /	
	0 1 M Na ₂ S ₂ O ₈	0 1 M TEG	DME	0 / 0 /	
0 1 M Na ₂ S ₂ O ₄	0 1 M TEG	DME		0 / 0 /	
0 1 M NH ₂ OH HCl	0 1 M TEG	DME		0 / 3 /	
0 1 M Ascorbic Acid	0 1 M TEG	DME		0 / 0 /	
	0 1 M Tiron			0 / 24 /	
	0 1 M Nitric Acid			7 / 0 /	
	5 / NaOCl			15 / 8 /	
0 1 M NH ₂ OH HCl				0 / 0 /	
0 1 M Ascorbic Acid				0 / 0%	Should try a range of pHs

From Triay and Loge 1994

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Table A 9 Plutonium and Americium Removal from Soil at Elevated Temperature for Various Combinations of Extractants

Reducing Agent	Oxidizing Agent	Complexing Agent	Ph		Am Removed		Pu Removed	
			20°C	80°C	20°C	80°C	20°C	80°C
none	none	Citric Acid 0.1 M	2.2	2.4	29%	36%	28%	63%
none	none	Citric Acid 0.1 M	3.5	3.5	51%	52%	56%	57%
none	none	Citric Acid 0.1 M	4.6	4.9	60%	71%	36%	58%
none	none	Citric Acid 0.1 M	8.3	8.7	29%	42%	20%	35%
Ascorbic Acid		Na ₃ Citrate 0.1 M	5.4	5.4	69%	73%	56%	51%
Ascorbic Acid		Na ₃ Citrate 0.1 M	6.8	7.2	52%	61%	38%	40%
Na ₂ S ₂ O ₄		Na ₃ Citrate 0.1 M	5.9	6.4	71%	67%	62%	57%
Na ₂ S ₂ O ₄		Na ₃ Citrate 0.1 M	5.6	6.7	63%	61%	63%	53%
Na ₂ S ₂ O ₄ (0.5 M)		Na ₃ Citrate 0.1 M/NaHCO ₃ 0.01 M	6.1	5.8	77%	70%	52%	
H ₂ O ₂ 3%	same	Na ₃ Citrate 0.1 M	8.4	8.3	54%	42%	48%	40%
	Na ₂ S ₂ O ₈	Na ₃ Citrate 0.1 M	6.9	5.1	54%	77%	45%	57%
	NaOCl 1%	Na ₃ Citrate 0.1 M	8.6	9.2	33%	41%	39%	37%
none		EGTA 0.1 M	4.8	5.5	58%	69%		42%
none		EGTA 0.1 M	8.7	8.7	56%	67%	23%	31%
none		EGTA 0.1 M	9.2	9.2	59%	64%	26%	27%
none		EGTA 0.1 M	9.9	9.8	53%	64%	17%	28%
ascorbic acid		EGTA 0.1 M	5.9	6.2	61%	76%	37%	53%
Na ₂ S ₂ O ₄		EGTA 0.1 M	6	6.4	75%	81%	62%	70%
H ₂ O ₂ 3%	same	EGTA 0.1 M	4	4.3	57%	68%	27%	46%
none	Na ₂ S ₂ O ₈	EGTA 0.1 M	3.2	2.9	53%	72%	26%	48%
none	NaOCl 1%	EGTA 0.1 M	8.1	7.2	27%	67%	12%	43%
none		Na ₂ EDTA 0.1 M	4.6	5.6	67%	75%	30%	57%
none		Na ₂ EDTA 0.1 M	7.7	8.3	60%	72%	20%	42%
none		Na ₂ EDTA 0.1 M	10.5	10.3	55%	64%	18%	33%
ascorbic acid		Na ₂ EDTA 0.1 M	6	7.4	67%	77%	60%	59%

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Na ₂ S ₂ O ₄		Na ₂ EDTA 0.1 M	6.2	6.5	73%	83/	55%	60/
H ₂ O ₂ 3/	same	Na ₂ EDTA 0.1 M	5.9	7.5	69/	62/	54/	28/
	Na ₂ S ₂ O ₈	Na ₂ EDTA 0.1 M	4.8	6.3	61/	56/	32/	39/
NaOCl 1/		Na ₂ EDTA 0.1 M	7.9	8.4	57/	68/	30/	29/
none		DTPA 0.1 M	3.9	3.8	65/	79/	41/	
none		DTPA 0.1 M	5.2	6.7	61/	75/	44/	48/
none		DTPA 0.1 M	7	7.4	60%	74/	26/	52/
none		DTPA 0.1 M	8.7	8.9	60/	66/	24/	64/
none		DTPA 0.1 M	10.3	10.3	60/	66/	24/	38/
ascorbic acid		DTPA 0.1 M	6.7	6.9	66/	76/	50/	
Na ₂ S ₂ O ₄		DTPA 0.1 M	7.4	6.5	71/	86/	56/	68/
H ₂ O ₂ 3/	same	DTPA 0.1 M	6.6	7.4	65/	70/	50/	38/
	Na ₂ S ₂ O ₈	DTPA 0.1 M	4	5.1	64/	77/	42%	
	NaOCl	DTPA 0.1 M	7.4	8.1	60/	70/	31/	54/
none		NTA 0.05 M	3.2	5.7	59/	69/	53/	51/
none		NTA 0.08 M	3.5	5.4	63/	75/	47/	56/
none		NTA 0.1 M	5.4	6.3	62/	75/	44/	44/
none		NTA 0.1 M	7.5	7.4	59/	71/	28/	58/
none		NTA 0.1 M	10.1	9.7	59/	62/	29/	22/
ascorbic acid		NTA 0.1 M	5.7	7.6	63/	75%	55/	61/
Na ₂ S ₂ O ₄		NTA 0.1 M	5.4	6.3	75/	80/	75/	78/
H ₂ O ₂ 3/	same	NTA 0.1 M	8.8	7.7	58/	65/	26%	43%
none	Na ₂ S ₂ O ₈	NTA 0.1 M	4.1	2.9	60/	32/	40/	
none	NaOCl 1/	NTA 0.1 M	7.9	8.1	50/	62/	42/	36%
none		2 M Na ₂ CO ₃	11	10.8	26/	20/	17/	18/
none		0.1 M NaCO	10.4	10.1	1/	3/	2/	2%
none		0.1 M NaHCO ₃	8.7	8.6	0/	1/	0/	1%
ascorbic acid		0.1 M NaHCO ₃	7.2	7.1	25/	42%	13/	28/
Na ₂ S ₂ O ₄		0.1 M NaHCO	6.6	7.1	0/	1/	1/	2/
H ₂ O ₂ 3/	same	0.1 M NaHCO ₃	8.9	9.1	1/	0/	1/	1/
none	Na ₂ S ₂ O ₈	0.1 M NaHCO	8.3	2.2	0/	54/	2/	
none	NaOCl 1/	0.1 M NaHCO ₃	8.3	8.5	2/	3/	4/	8/

From DOE (1995b)

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Table A 10 Effects of Reducing Agents Hydrogen Peroxide, and Oxidizing Agents on Removal of Actinides from RFETS Soil at 20°C and 80°C

	Americium	Plutonium	Comparison
CITRIC ACID			
Reduction	Increases removal at 20°C to approximate values seen at 80°C without reduction May slightly increase removal at 80°C	Increases removal at 20°C to approximate values seen at 80°C without reduction No apparent effect on removal at 80°C	Reduced plutonium at 20°C plots along the extraction line for non reduced americium and below the extraction efficiency for americium under reducing conditions Reduced plutonium at 80°C falls in the same vicinity as reduced plutonium at 20°C
Hydrogen Peroxide	Increases removal at 20°C to approximate values seen at 80°C without reduction May slightly increase removal at 80°C	Increases removal at 20°C to approximate values seen at 80°C without reduction No apparent effect on removal at 80°C	
Oxidation	No obvious changes	Increases removal at 20°C to approximate values seen at 80°C without reduction No apparent effect on removal at 80°C	
EGTA			
Reduction	Increases removal at 20°C to approximately the levels seen at 80°C May also increase removal at 80°C by 10 /	Reduction at 20°C comparable to non reduced results at 80°C Reduction at 80°C increases removal approximately 25 / over non reduced levels	Reduction increases plutonium removal efficiency to approximately values seen for americium under non reducing conditions but removal remains less than for americium under reducing conditions
Hydrogen Peroxide	No clear effects	No clear effects	
Oxidation	No clear effects	No clear effects	
EDTA			

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Reduction	May slightly increase extraction at both 20°C and 80°C	Increases extraction efficiency at 20°C to values seen at 80°C in absence of reducing agents Reduction at 80°C did not have a major effect	Reduction increases plutonium removal efficiency to approximately values seen for americium under non reducing conditions but removal remains less than for americium under reducing conditions
Hydrogen Peroxide	No obvious effect	At 20°C increased extraction efficiency to approach levels seen at 80°C without redox agents	
Oxidation	No effect at 200°C May decrease extraction efficiency at 800°C	At 20°C increased extraction efficiency to approach levels seen at 80°C without redox agents At 80°C decreases extraction efficiency	
DTPA			
Reduction	May increase extraction efficiency slightly both for 20°C and 80°C solutions	Increases extraction efficiency at both 20°C and 80°C	Reduction increases plutonium removal efficiency to approximately values seen for americium under non reducing conditions but removal remains less than for americium under reducing conditions
Hydrogen Peroxide	Not much effect	Doesn't have much effect	
Oxidation	Not much effect	Doesn't have much effect	
NTA			

205

RF/ER 96 0048 UN
Draft Summary of Existing Data on Actinide Migration
at the Rocky Flats Environmental Technology Site

Reduction	Little effect	Significantly increases plutonium removal	Reduction increase plutonium removal at 20°C to near values seen for americium removal at 20°C under non reducing conditions Reduction increase plutonium removal at 80°C to near values seen for americium removal at 80°C under non reducing conditions
Hydrogen Peroxide	Little effect	Little effect	
Oxidation	Little effect	Little effect	

206

A 4 IMPLICATIONS FOR THE MOBILITY OF ACTINIDES IN GROUNDWATER AND SURFACE WATER

Actinide chemistry actinide behavior at sites other than RFETS and the distribution of actinides at RFETS all suggest that trivalent and tetravalent actinides associate strongly with soil particles and have low concentrations in groundwater. Treatability studies of chemical treatments discussed in this appendix however show that the high affinity of trivalent and tetravalent actinides (as exemplified by americium and plutonium respectively) for soil can be overcome under some conditions and that the actinides can be moved into solution. High water to soil ratios, high concentrations of strong synthetic complexing agents, pH values favoring deprotonation of the complexing agent, and strong reducing agents all increase the dissolution of available trivalent and tetravalent actinides.

High concentrations of strong synthetic complexing agents such as NTA or EDTA are indispensable for the dissolution of the majority of actinides (Table A 9). Relatively weak complexing agents such as carbonate also can solubilize an appreciable fraction of trivalent and tetravalent actinides when applied in high concentrations at high pHs. When applied at dilute concentrations and intermediate pH conditions which are more similar to conditions found in the environment however, the fraction of the actinides which dissolve is small. The fraction solubilized by weak complexing agents from RFETS soils remains small even when strong reducing agents such as sodium dithionite are applied (Table A 9).

The low solubility of actinides under these near natural conditions show that while low concentration of actinides may dissolve in groundwater and form complexes, most of the actinides will remain in the solid phase. This will be true even under strongly reducing conditions and with high concentrations of natural complexing agents. The strong affinity of soils for actinides also ensures that actinides which enter solution in the source area will partition onto aquifer materials as they migrate and this will retard movement.

The treatability studies clearly show that trivalent actinides such as americium are more soluble than tetravalent actinides such as plutonium. Reducing conditions which can convert some species of tetravalent actinides into trivalent actinides therefore will increase the solubility and movement of the actinide. This will have little practical significance over short time periods because actinides are insoluble and strongly retarded even in the trivalent form.

A 5 REFERENCES

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Triay I R and Loge G W 1994 Batch experiments for desorption of plutonium and americium in contaminated soil from the Rocky Flats Los Alamos Report LA UR 94 1165 16 pp

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APPENDIX B SURFACE WATER FIGURES

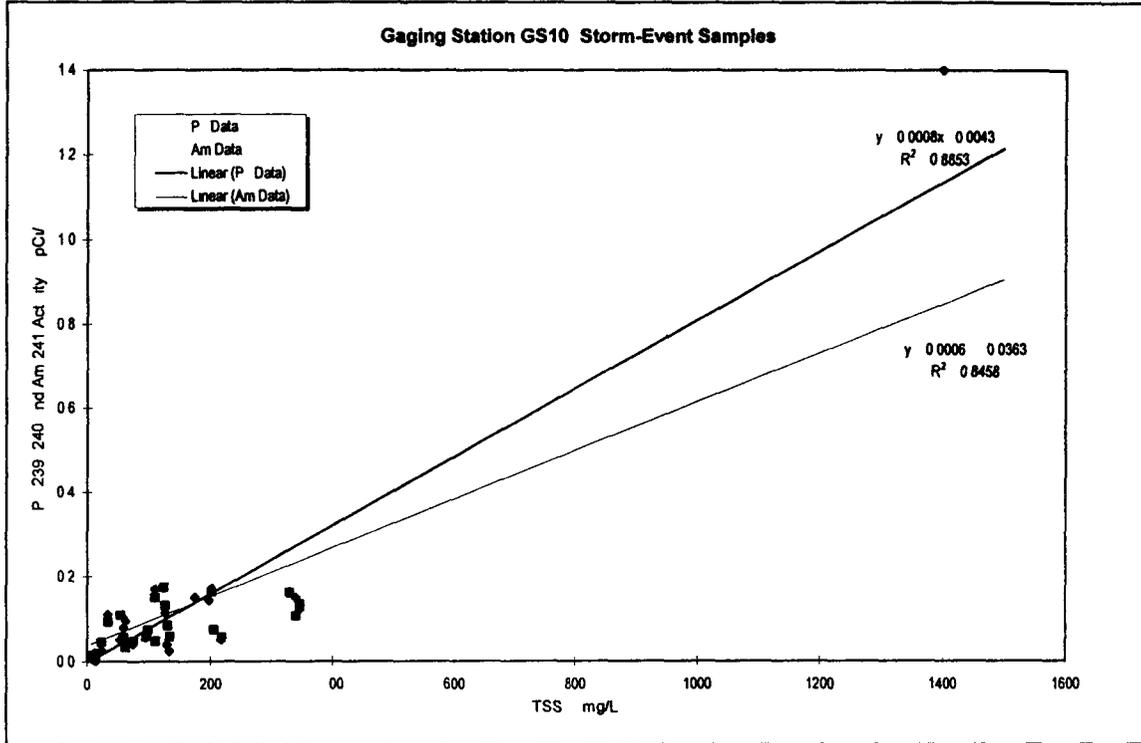


Figure 1 Station GS10 Variation of Pu 239 240 and Am 241 Activities with TSS

210

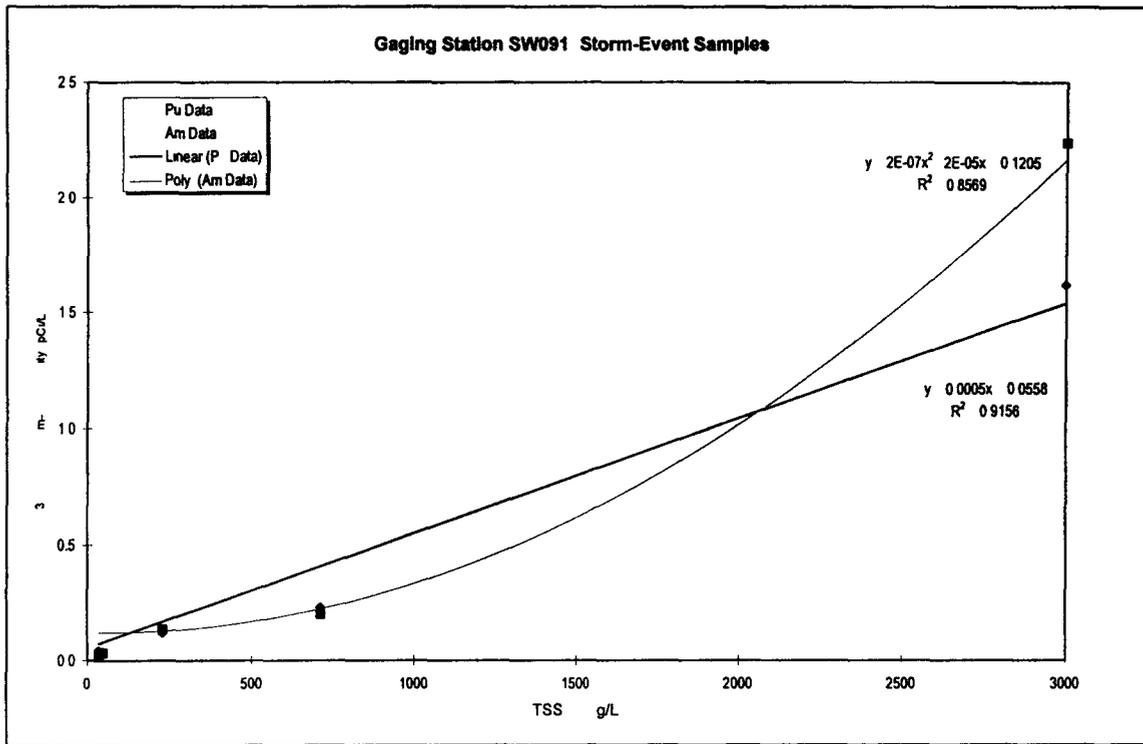


Figure 2 Station SW091 Variation of Pu 239 240 and Am 241 Activities with TSS

211

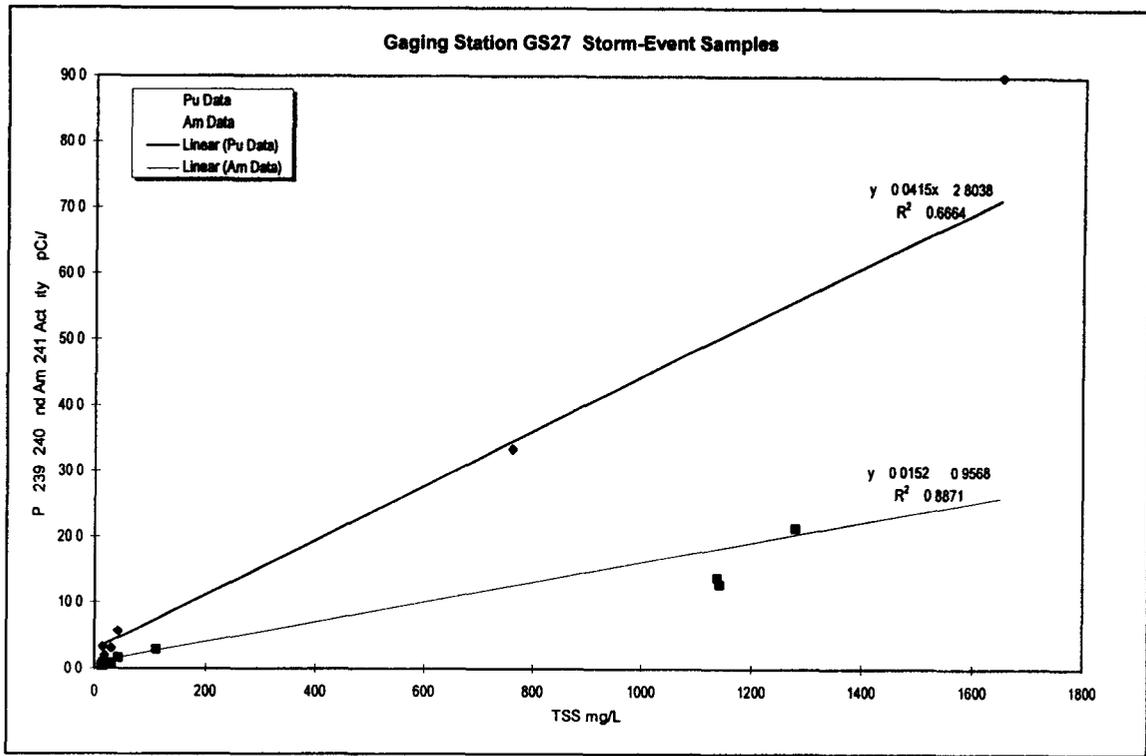


Figure 3 Station GS27 Variation of Pu 239 240 and Am 241 Activities with TSS

212

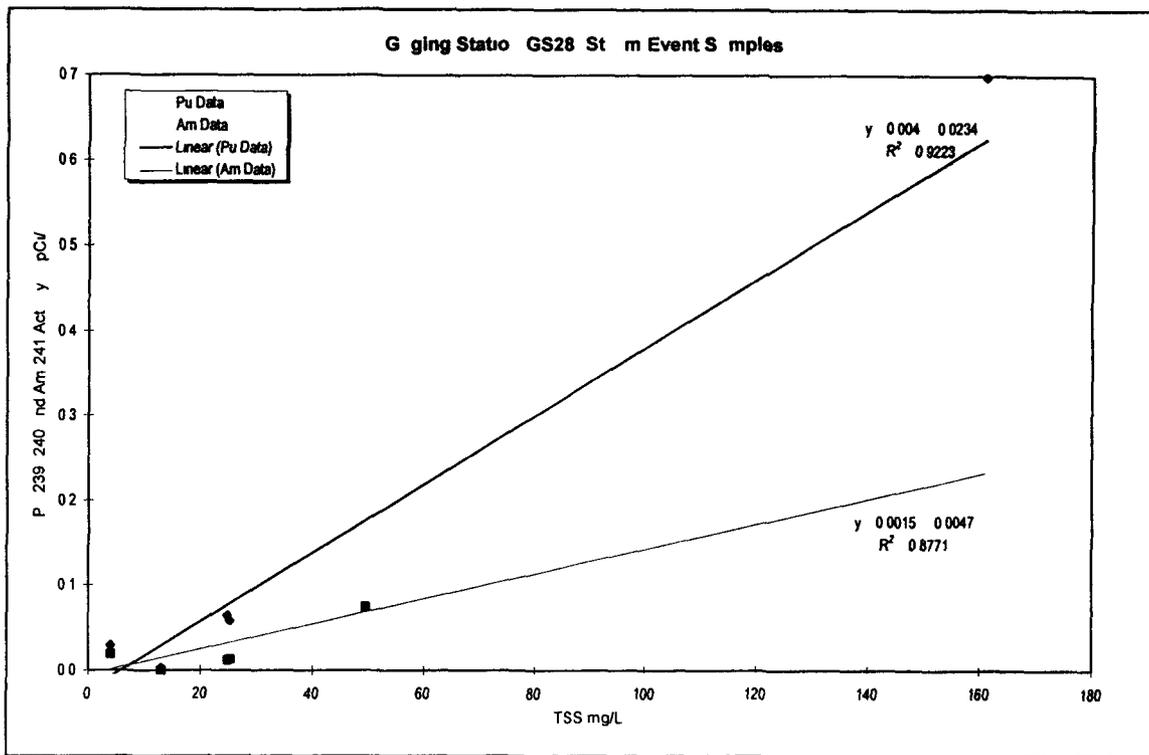


Figure 4 Station GS28 Variation of Pu 239 240 and Am 241 Activities with TSS

2/3

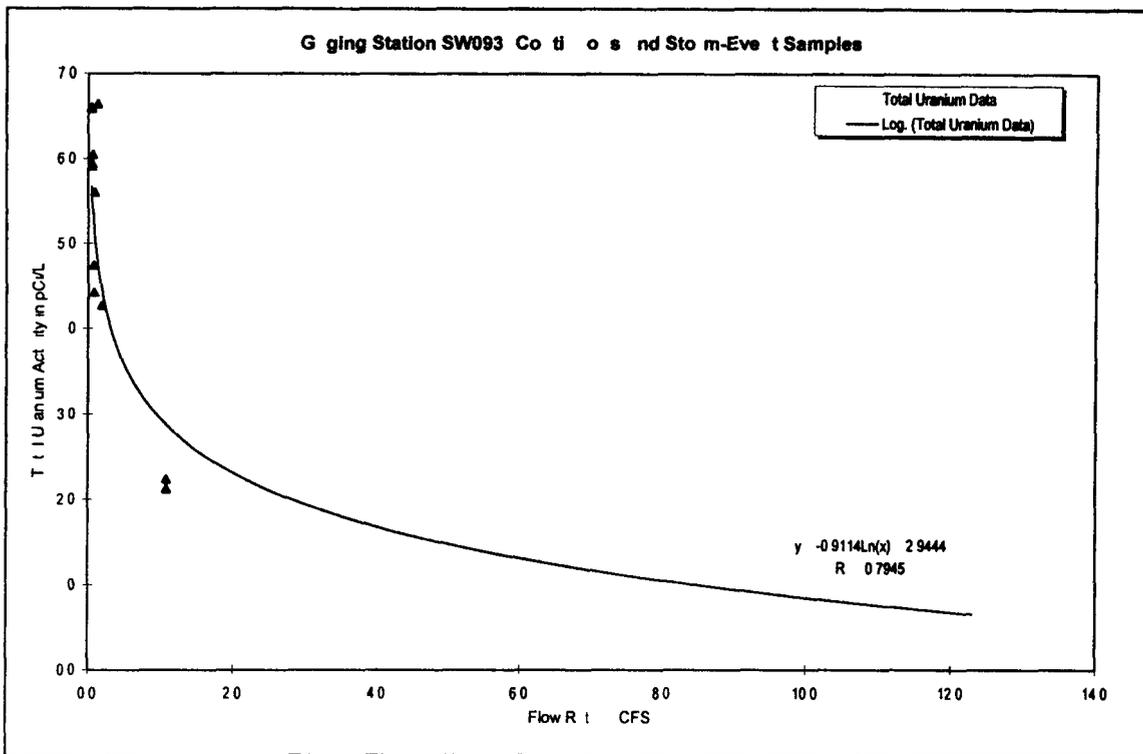


Figure 5 Station SW093 Variation of Total Uranium Activity with Flow

214

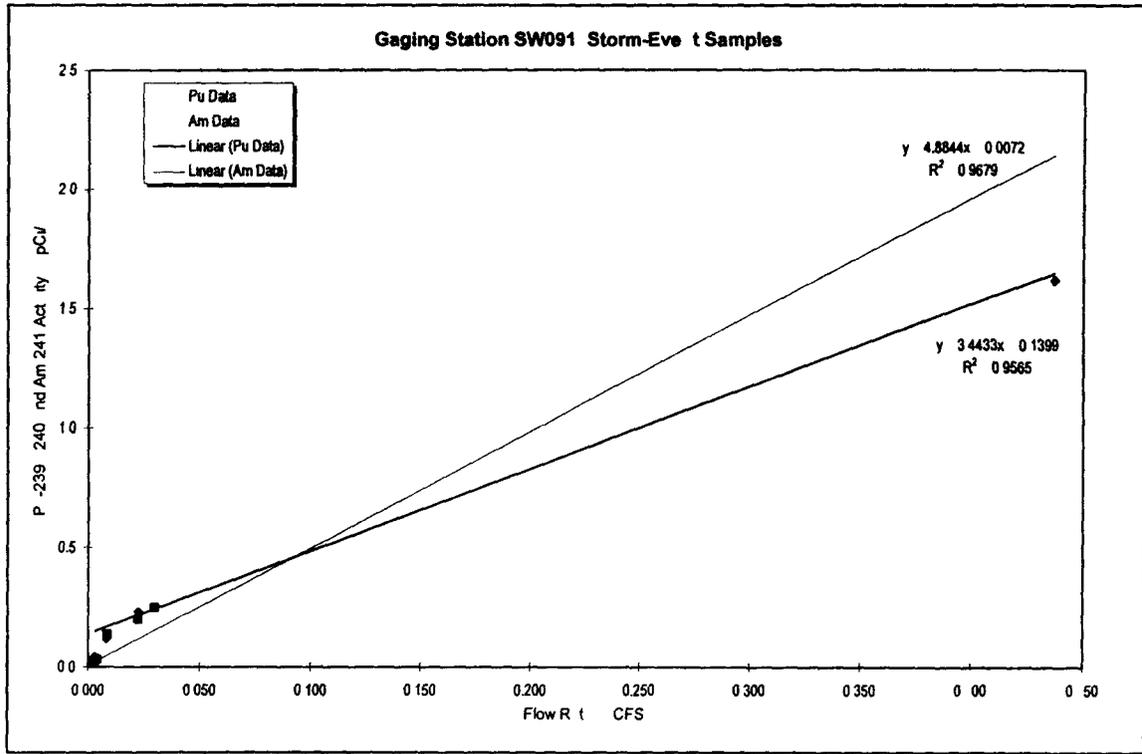


Figure 6 Station SW091 Variation of Pu 239 240 and Am 241 Activities with Flow

215

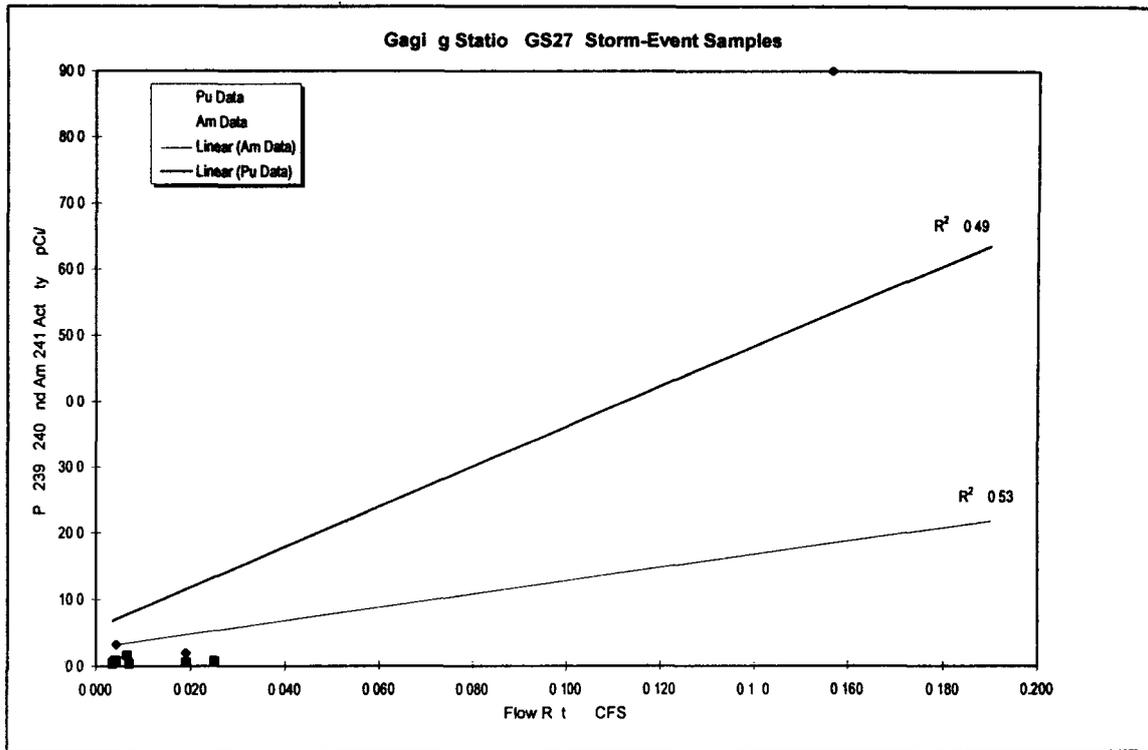


Figure 7 Station GS27 Variation of Pu 239 240 and Am 241 Activities with Flow

216

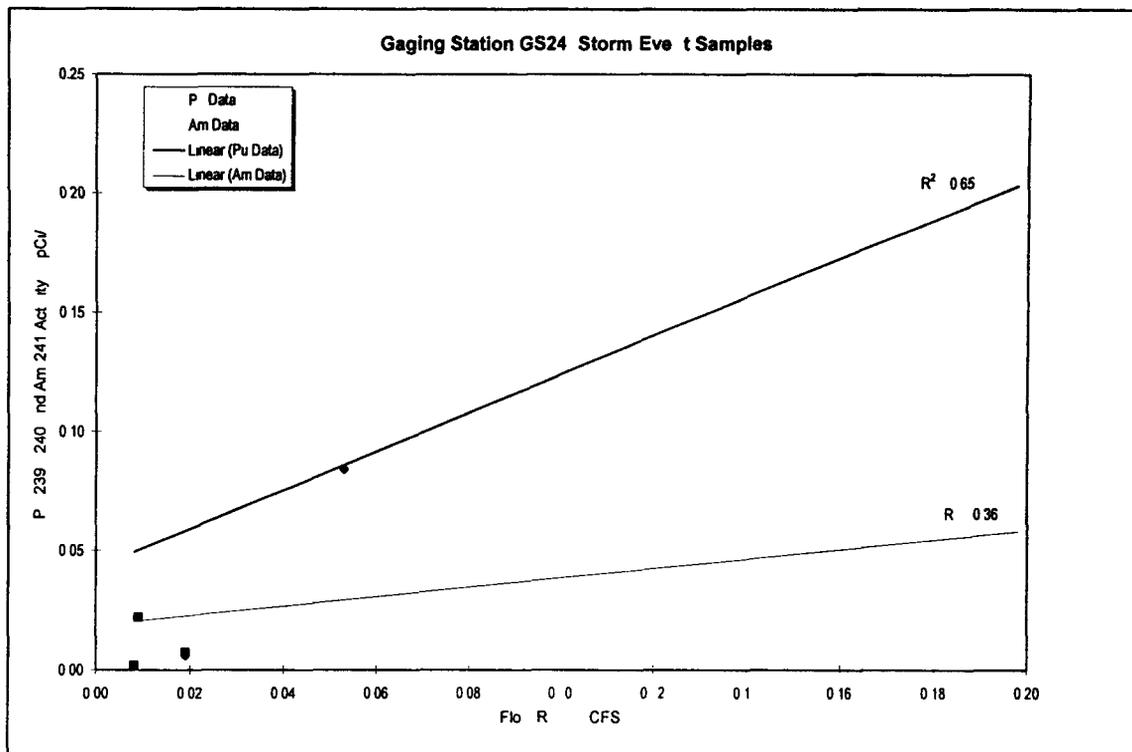


Figure 8 Station GS24 Variation of Pu 239 240 and Am 241 Activities with Flow

217

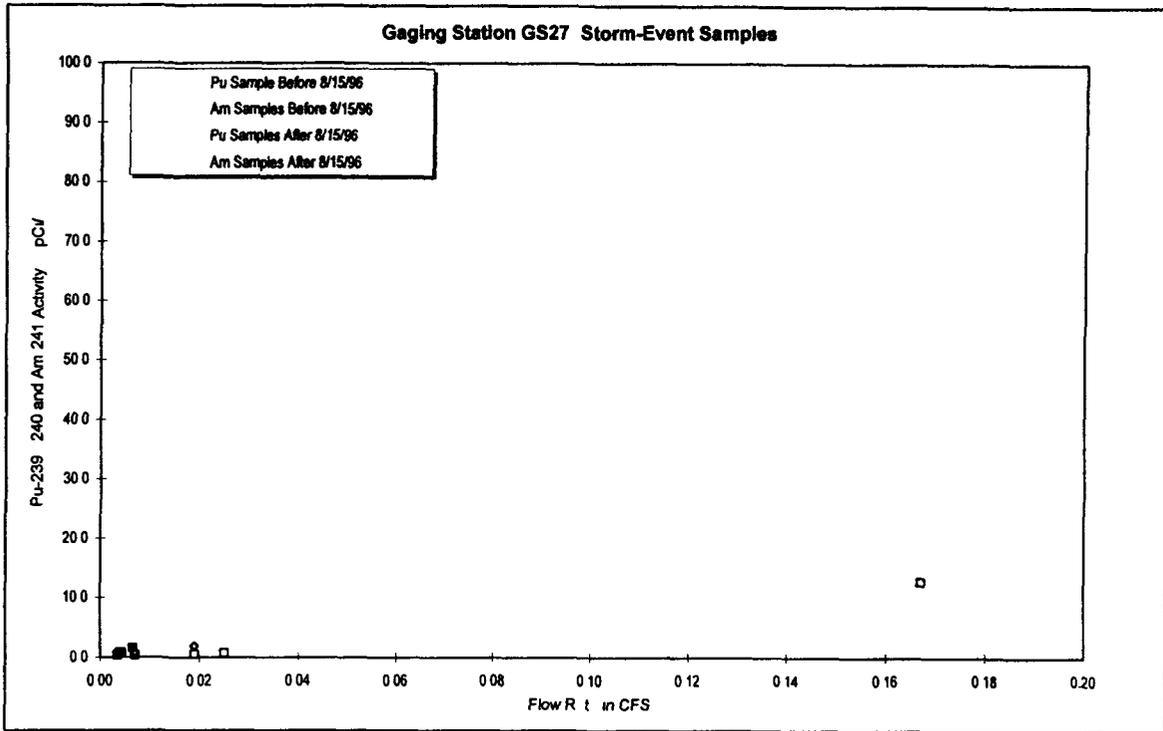


Figure 9 Station GS27 Pu 239 240 and Am 241 Activities Before and After Implementation of Watershed Improvements

218

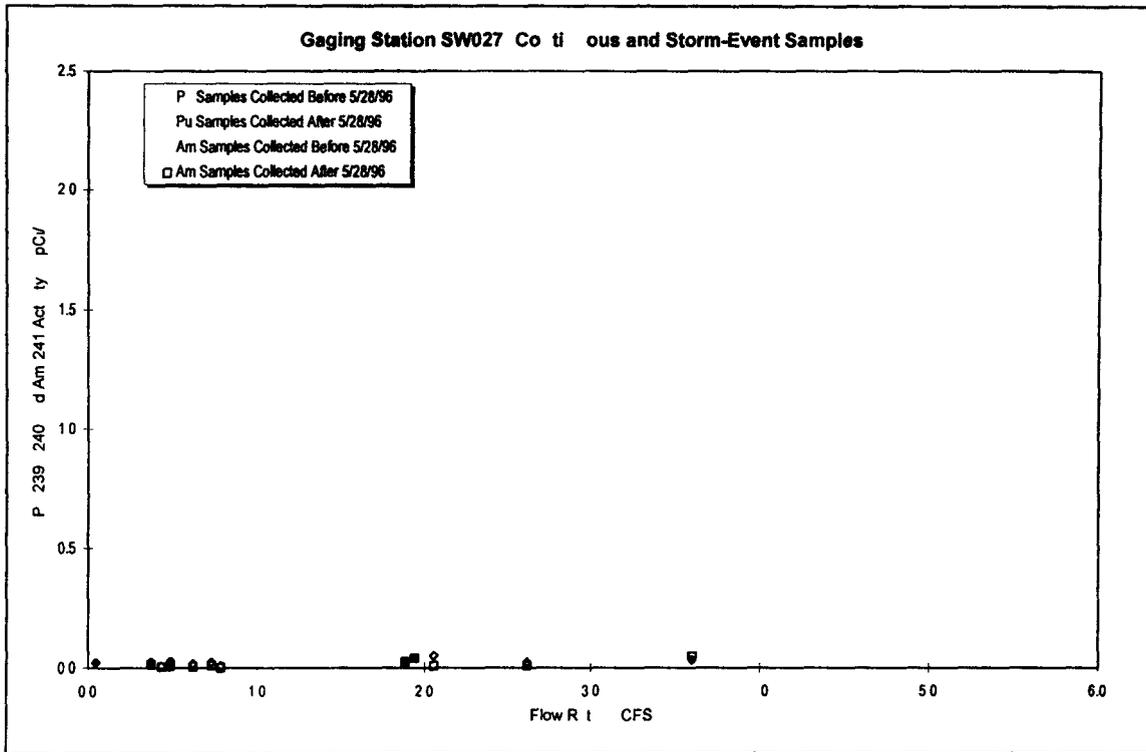
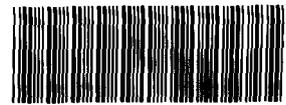


Figure 10 Station SW027 Pu 239 240 and Am 241 Activities Before and After Implementation of Watershed Improvements

48100



000067221

ER/WM&I DDT

Source/Driver (Name & Number from
ISP IAG milestone Mgmt Action Corres
Control, etc)

Closure # (Outgoing Correspondence
Control # if applicable)

Due Date

J. K. Hopkins
Originator Name

G. D. DiGregorio
QA Approval

for A.M. Tyson
Contractor Manager(s)

Chris Dayton
Kaiser Hill Program Manager(s)

T. G. Hedahl
Kaiser Hill Director

Document Subject

8/20/97

TRANSMITTAL OF THE DRAFT SUMMARY OF EXISTING DATA ON ACTINIDE MIGRATION AT ROCKY FLATS ENVIRONMENTAL TECHNOLOGY SITE REV 0 AMT-096 97

Discussion/Issues

Seven copies of the report Draft Summary of Existing Data on Actinide Migration at Rocky Flats Environmental Technology Site Rev 0 are enclosed for transmittal to the Department of Energy (DOE) for review This document incorporates Kaiser Hill comments Comments received on August 20 from Bruce Honeyman and David Janecky will be addressed along with DOE comments

Two copies are for the Kaiser Hill Project Manager and five copies are for DOE RFFO

If you have any questions regarding this document please contact John Hopkins at extension 4974

JKH cb

Attachments
As Stated

cc

J K Hopkins
A M Tyson (w/o attachment)
Correspondence Control



ADMIN RECCRD

1/240

ERWWM & I DDT 7/95

SW-A-004332

EXPLANATION

- V g l l d Area As oca l d W th Gro d wat
- Se pag (V g l to on ls f ll d ba llic
- nushe nd ome woody bu h) Se pag Foc
- Al g Uph ll Bou d ry (V g l l d Ar



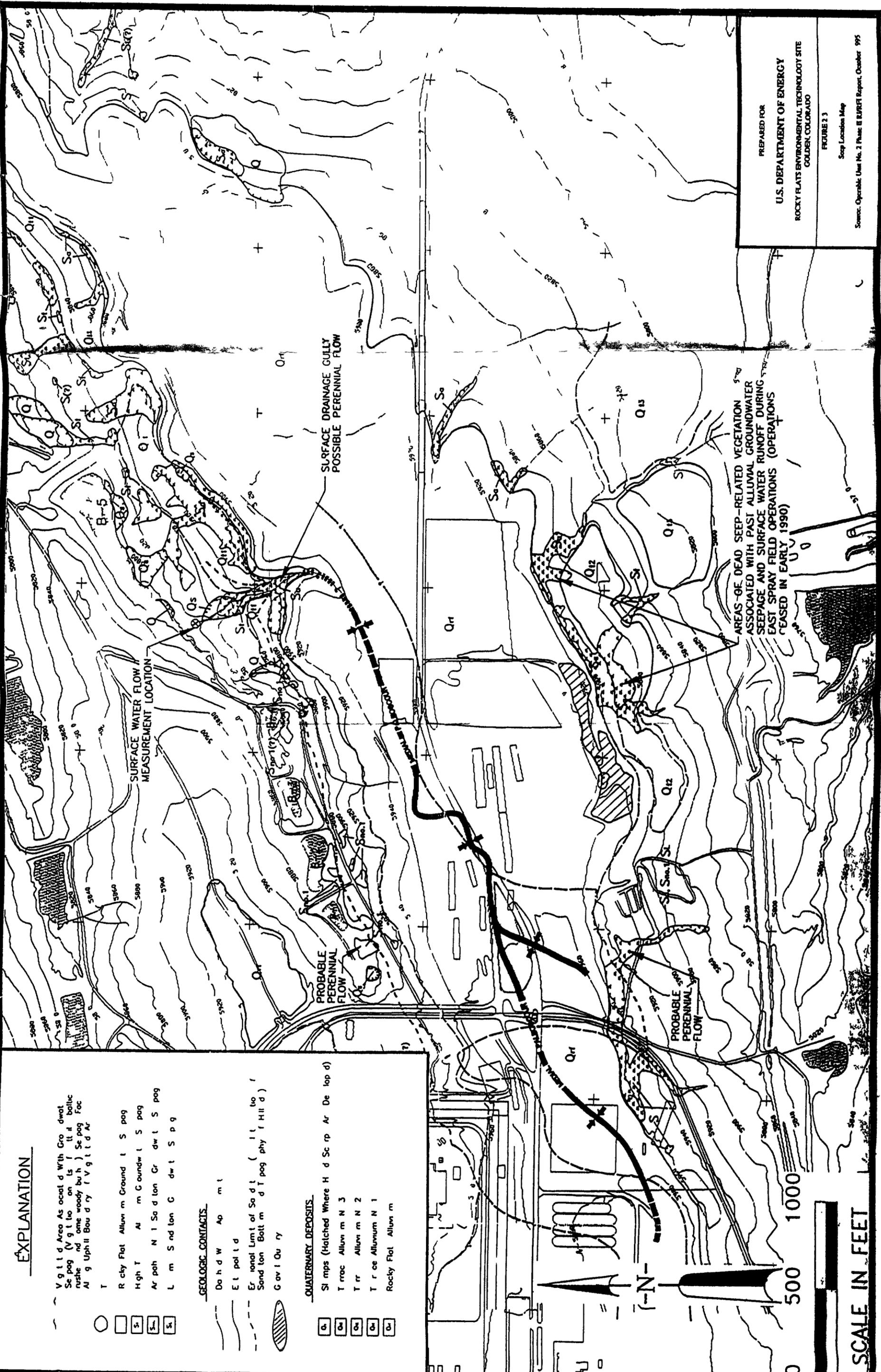
- T R cky Flat Alluv m Ground l S pag
- H gh T Al m Groundw l S pag
- Ar path N l Sa d lon Gr dw l S pag
- L m S nd lon G dw l S p g

GEOLOGIC CONTACTS

- Da h d w Ap m l
- El pol d
- Er nonal Lmt of Sa d t (l l to f
- Sand ton Bolt m d T pag phy (H l l d)
- Gav l Ou ry

QUATERNARY DEPOSITS

- Sl mps (hatched Where H d Sc rp Ar De lop d)
- T rrac Alluv m N 3
- T r r Alluv m N 2
- T r ee Alluvum N 1
- Rocky Flat Alluv m



AREAS OF DEAD SEEP-RELATED VEGETATION ASSOCIATED WITH PAST ALLUVIAL GROUNDWATER SEEPAGE AND SURFACE WATER RUNOFF DURING EAST SPRAY FIELD OPERATIONS (OPERATIONS CEASED IN EARLY 1990)

PREPARED FOR
 U.S. DEPARTMENT OF ENERGY
 ROCKY FLATS ENVIRONMENTAL TECHNOLOGY SITE
 GOLDEN, COLORADO

FIGURE 2.3
 Seep Location Map
 Source: Operable Unit No. 2 Phase II RI/FSI Report, October 1995

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Figure 4-1
Distribution of Americium 241
in Surface Soil

- EXPLANATION**
- Less than 1 pCi/g
 - 1 pCi/g
 - 10 pCi/g
 - 100 pCi/g
 - 1000 pCi/g
 - Solid SWM

Standard Map Feature

- Building
- ▨ Lake
- Stream
- Fence
- Rocky
- Paved
- Road

DATA SOURCE
 Aerial photography, ground truth, and other data were used to develop this map. The data were collected by the U.S. Environmental Protection Agency, Office of Research and Development, in 1995.

NOTE:
 This map was developed using data from surface soil sampling locations shown in Figures 4-2 and 4-3. Contours were added using professional judgment from data obtained by sampling.

Scale 1:21330
 1 inch = 1778 feet



State Plane, Colorado North
 Zone 10N, NAD83

U.S. Department of Energy
 Rocky Flats Environmental Technology Site

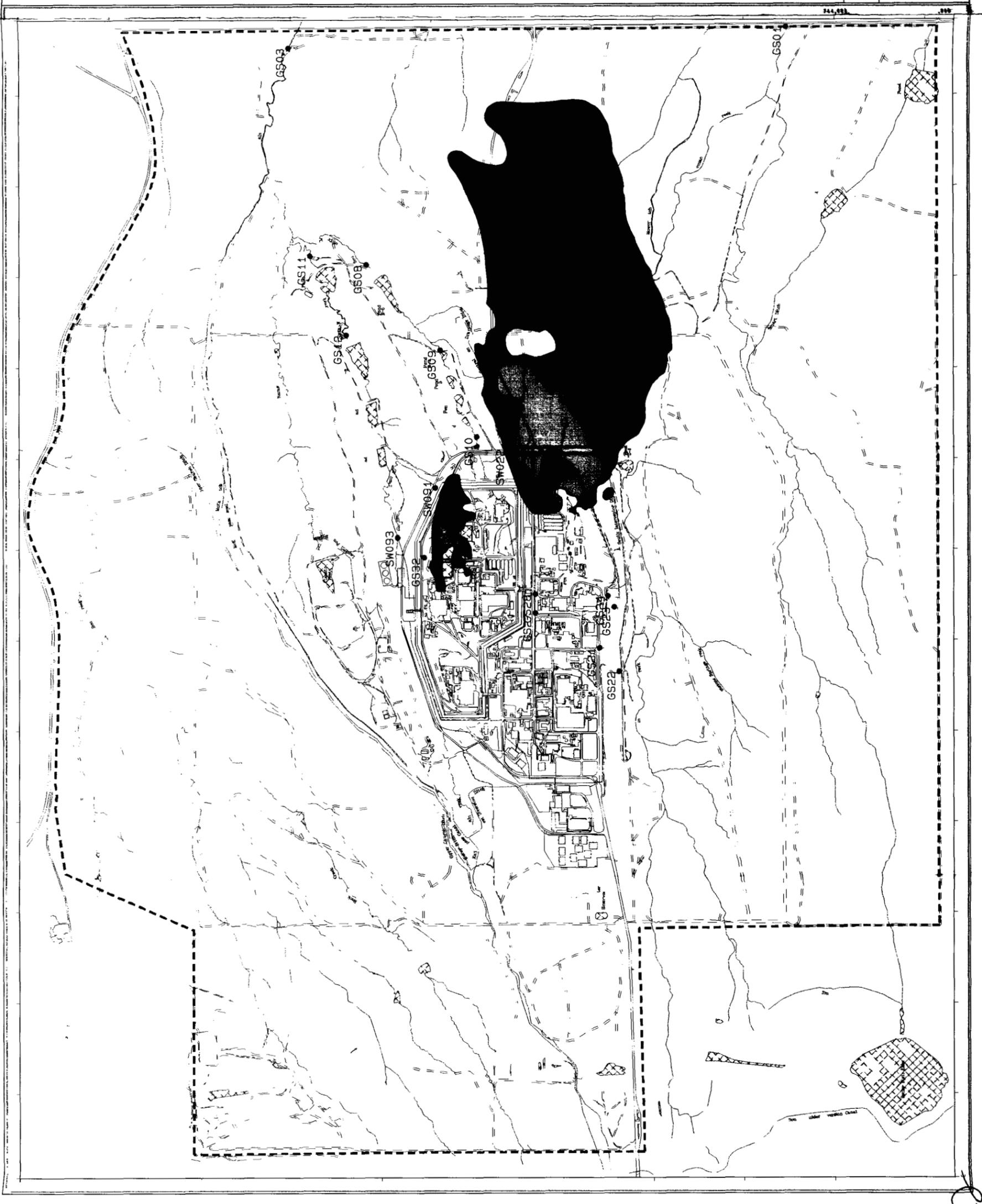
Prepared by



Rocky Mountain Remediation Services, LLC
 Geospatial Information Systems Group
 10000 North Central Expressway, Suite 100
 Denver, CO 80231

MAP ID: 97-0130

August 15, 1997



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Figure 4-2
Distribution of Plutonium 239/240
in Surface Soil

- EXPLANATION**
- Less than 1 pCi/g
 - 1 pCi/g to but less than 10 pCi/g
 - ▨ 10 pCi/g to but less than 100 pCi/g
 - 100 pCi/g to but less than 1000 pCi/g
 - 1000 pCi/g to
 - Solid SWM infiltration

Standard Map Features

- Building structure
- ▨ Lake pond
- Street highway
- Railway
- Dry
- Paved
- Dotted

DATE SOURCE: Aerial photography, 1984; topographic maps, 1984; aerial photography, 1984; aerial photography, 1984; aerial photography, 1984.

NOTE: Contours were developed using data from the 1984 aerial photography. Contours were adjusted using professional judgment from grids overlaid by lighting.



Scale 1:21,330
 1 inch represents approximately 1778 feet



State Plan Coordinator
 Colorado Department of Natural Resources
 Denver, Colorado

U.S. Department of Energy
 Rocky Flats Environmental Technology Site

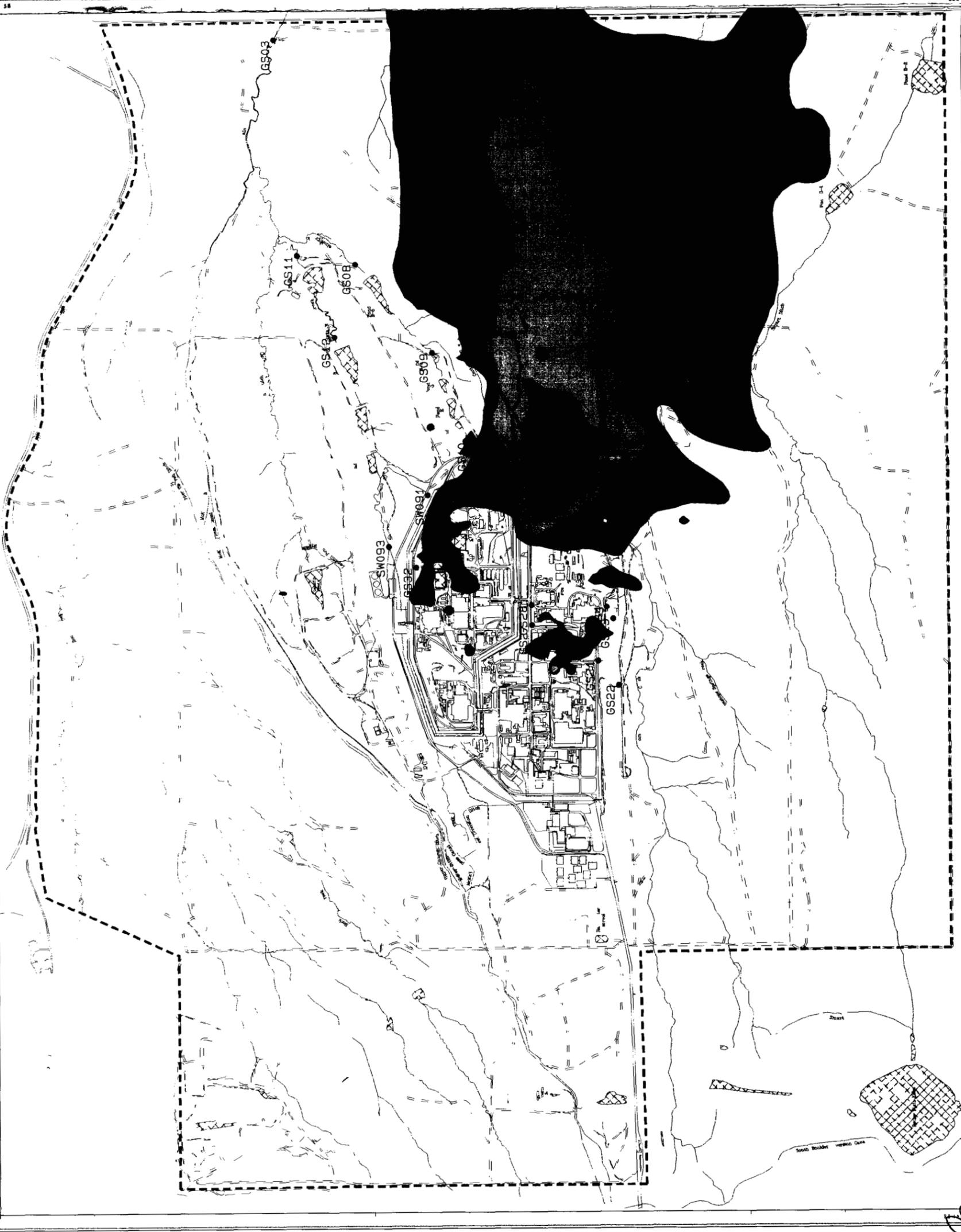
Prepared by



Rocky Mountain Remediation Services, L.L.C.
 Environmental Remediation Services Group
 10000 North 10th Street, Suite 100
 Denver, Colorado 80231

MAP ID: 97-0130

August 16, 1997



22

Figure 4-3
Surface Soil and Sediment
Sampling Locations
 Protactinium, Uranium & Americium Activities
 by Action Level

Legend

Action Level Sum of Ratio

- Below the Action Level
- Above the Action Level
- Above the Action Level
- Surface soil to which has been remediated

Sampling Location

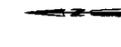
- Surface Soil Sample
- △ Sediment Sample

Standard Map Features

- Building structure
- ▨ Sole evaporative pond
- Lake and pond
- Stream ditches and linear features
- Fence and other barriers
- Rocky Flats boundary
- Paved road

NOTE:
 All values are in picograms per gram (pg/g).
 The values for each feature were determined by the Rocky Flats Environmental Technology Site (RFETS) Environmental Monitoring System (EMS) and are based on the most recent data available. The values for each feature were determined by the Rocky Flats Environmental Technology Site (RFETS) Environmental Monitoring System (EMS) and are based on the most recent data available. The values for each feature were determined by the Rocky Flats Environmental Technology Site (RFETS) Environmental Monitoring System (EMS) and are based on the most recent data available.

SCALE:
 Scale: 2540 feet represents approximately 875 feet

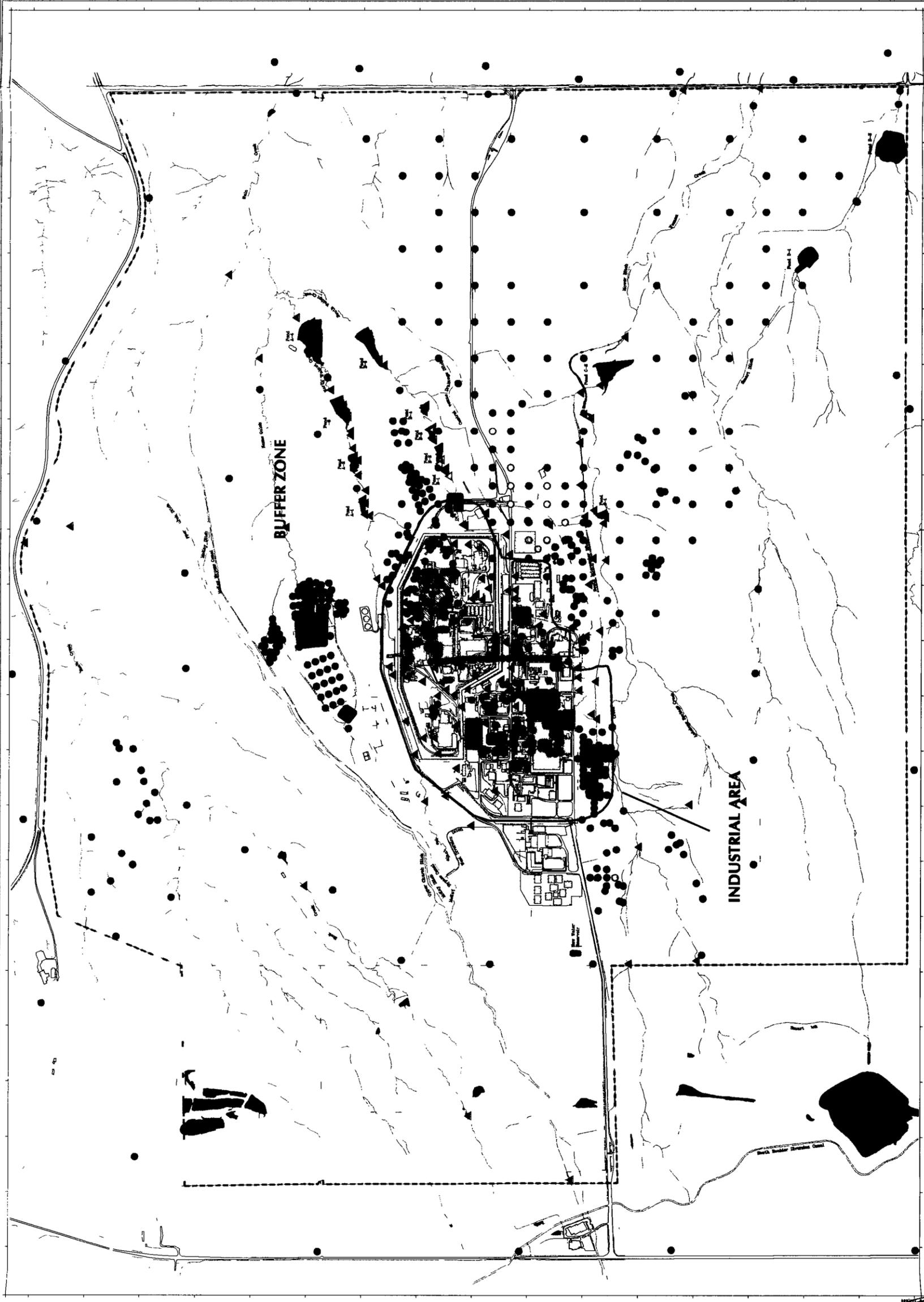


PROJECTION:
 State Plane Coordinate System
 Colorado North Zone
 Datum: NAD83

U.S. Department of Energy
 Rocky Flats Environmental Technology Site



Rocky Mountain Remediation Services, LLC
 10000 North Federal Boulevard, Suite 100
 Denver, Colorado 80231
 Phone: 303.752.3444



file

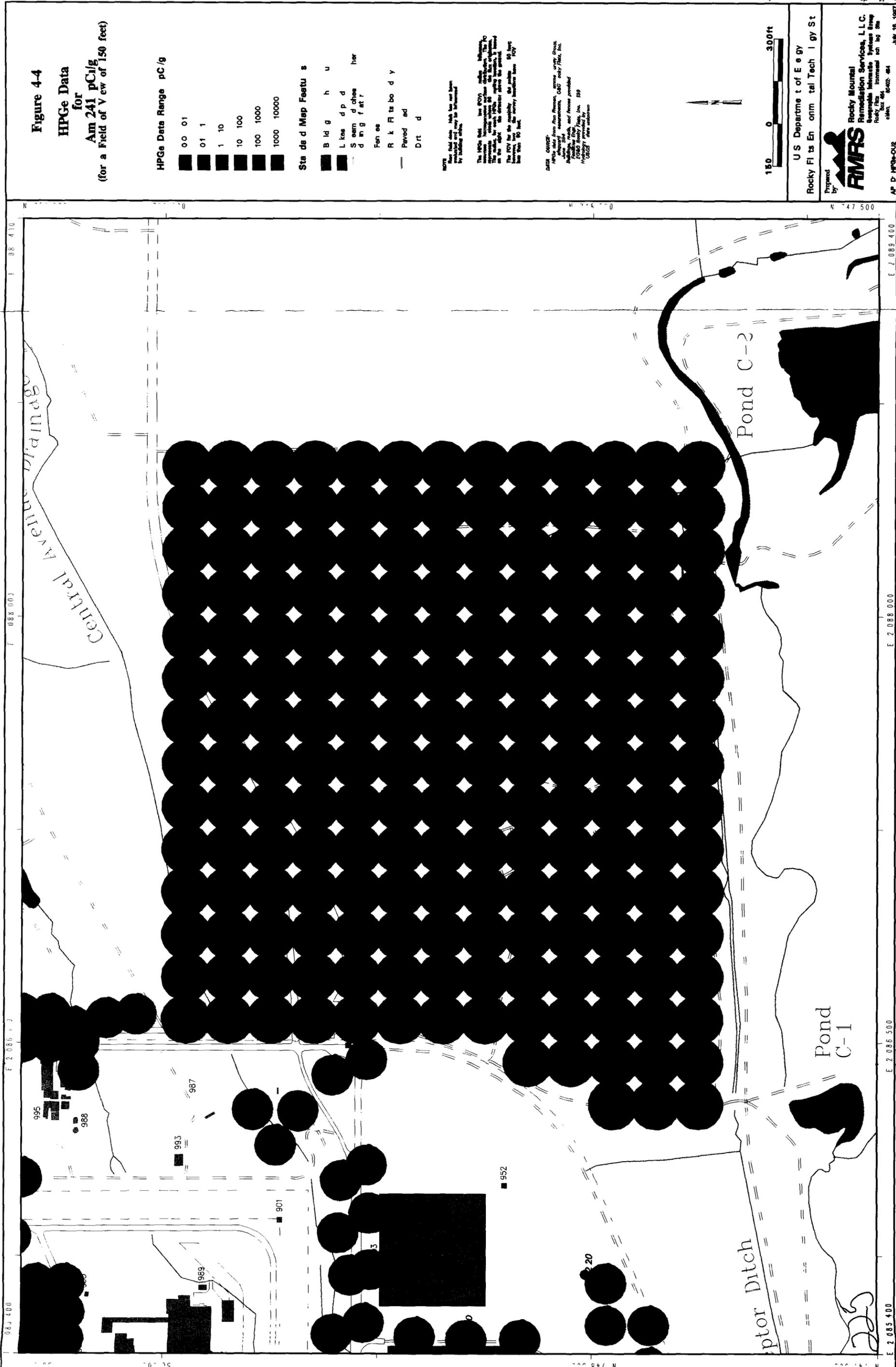


Figure 4-4
HPGe Data
 for
Am 241 pCi/g
 (for a Field of View of 150 feet)

HPGe Data Range pCi/g

0.01
0.1
1
10
100
1000
10000

- Site Data Map Features**
- Buildings
 - Lakes
 - Fences
 - Roads
 - Paved
 - Dirt

Notes:
 1. This field data was not taken and may be influenced by background noise.

The HPGe field was 200V, which is the maximum recommended voltage for the detector. The detector was calibrated for the field. The detector was calibrated for the field. The detector was calibrated for the field.

DATA SOURCE:
 Rocky Mountain Remediation Services, LLC
 Rocky Mountain Remediation Services, LLC



U.S. Department of Energy
 Rocky Mountain Remediation Services, LLC
 Rocky Mountain Remediation Services, LLC
 Rocky Mountain Remediation Services, LLC
 Rocky Mountain Remediation Services, LLC

Prepared by
RMRS
 Rocky Mountain Remediation Services, LLC
 Rocky Mountain Remediation Services, LLC
 Rocky Mountain Remediation Services, LLC
 Rocky Mountain Remediation Services, LLC

AP D HPGe-018
 01811 00402-004
 July 19, 1997

Figure 4-5
Subsurface Soil
Sampling Locations
 (Borehole & Groundwater Well)
 Plutonium, Uranium & Americium Activities
 by Action Level

- Legend**
- Action Level Sum of Ratio**
- Below the II Action Level
 - Above the II but below the I Action Level
 - Above the I Action Level
 - Borehole Well location that has been removed (remediated)

- Standard Map Feature**
- Building structure
 - ▨ Sole evaporative pond
 - Lake and pond
 - Streams, ditches, their draining features
 - Fence and other barriers
 - Rocky Flats boundary
 - Paved road

DISCLAIMER
 Aerial photography was collected by the Rocky Flats Environmental Technology Site (RFETS) in 1994. The data was processed and digitized by the Rocky Flats Environmental Technology Site (RFETS) in 1994. The data was processed and digitized by the Rocky Flats Environmental Technology Site (RFETS) in 1994. The data was processed and digitized by the Rocky Flats Environmental Technology Site (RFETS) in 1994.

NOTE:
 All coordinates and elevations were used as they are. The data was processed and digitized by the Rocky Flats Environmental Technology Site (RFETS) in 1994.

Scale: 25840
 Inch represents approximately 675 feet



These Plans Conform with the
 Construction Specifications
 Manual, 1995

Prepared by
FMRS
 Rocky Mountain Remediation Services, LLC
 Environmental Remediation Services
 10000 North 10th Street, Suite 100
 Denver, CO 80231-4444

U.S. Department of Energy
 Rocky Flats Environmental Technology Site



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Figure 4-6
Soil Pit Locations

- EXPLANATION**
- S I P L o
 - S t a d a d M a p F e a t u r e s
 - B l d g & h r u r e
 - L e k d p d
 - S t r e m d t h
 - d r a g f t u r e
 - F e
 - R k y F l b n d r y
 - P a e d d
 - D r t r o d

DATA SOURCE
Buildings, roads, and features provided by Rocky Mountain Remediation Services, LLC. Hydrology provided by USGS. Data unknown.

Scale = 1:21330
1 inch approximately 1778 feet



State Plane Coordinate System
North American Datum 1983

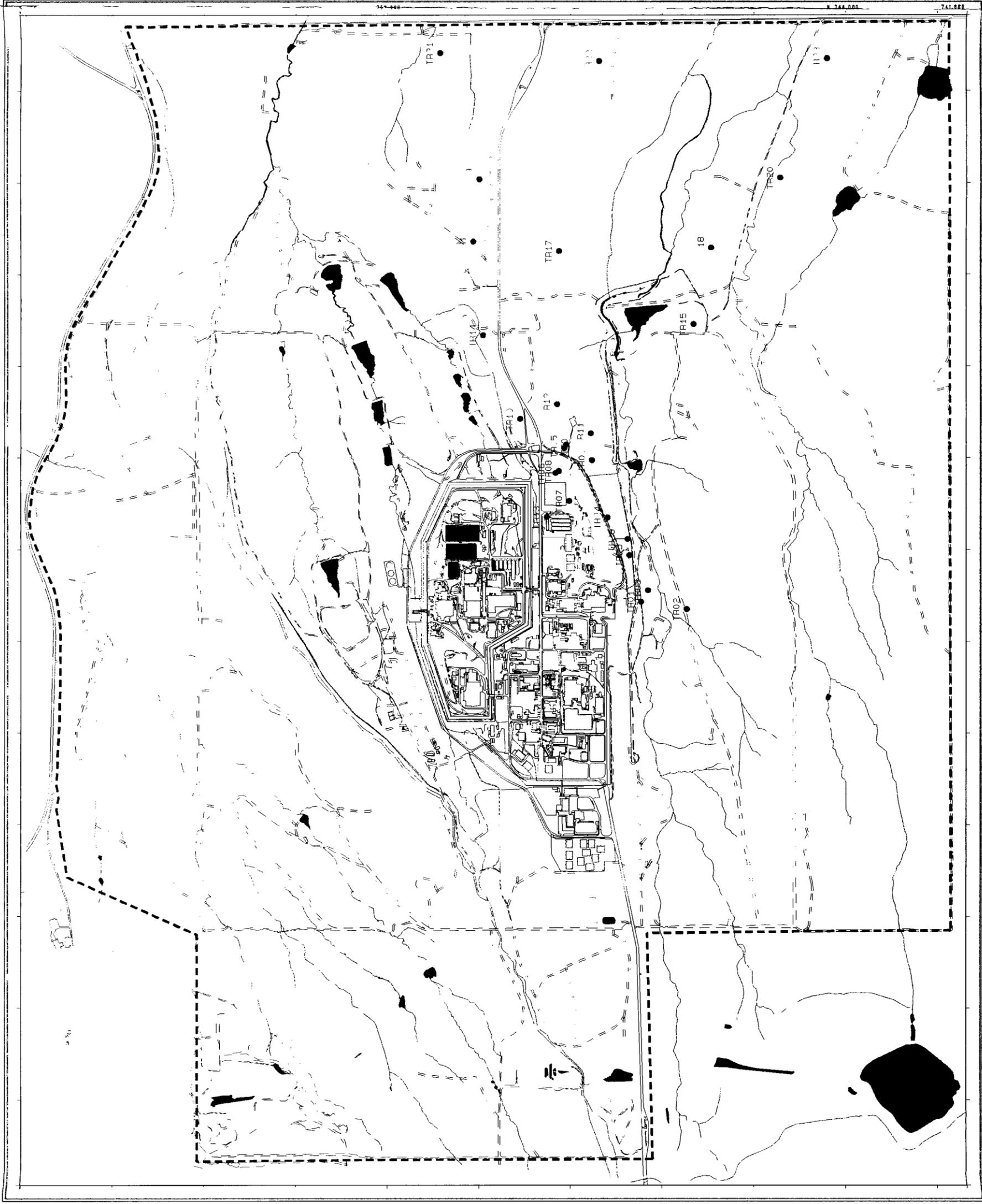
U.S. Department of Energy
Rocky Flats Environmental Technology Site



Rocky Mountain Remediation Services, LLC
3000 14th Street, Suite 100
Boulder, CO 80501
Phone: 303-442-4444

MAP ID: 97-0130

July 08, 1997



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Figure 4-9
Distribution of Uranium 233/234
in Surface Soil

- EXPLANATION**
- Less than 10 pCi/g
 - 10 pCi/g to but less than 100 pCi/g
 - 100 pCi/g to but less than 1000 pCi/g
 - 1000 pCi/g to greater than 1000 pCi/g
 - Selected SWM treatment

- Standard Map Features**
- Building & structure
 - Lake pond
 - Stream ditch
 - Fe
 - Rocky Flats dry
 - Paved
 - D road

DATA SOURCE
 Surface soil data from 1000 grid cells and other locations were collected by ESI, Inc. in August 1997. Digitized from aerial photograph, 1995.

NOTE
 Contours were developed using data from surface soil sampling locations shown in Figures 4-1 through 4-3. Contours were not developed for areas where data were not available. Judgment from grids prepared by ESI, Inc.

Scale: 1 inch represents approximately 1778 feet



State Plane Coordinates, Project
 Colorado North Zone
 Datum: NAD83

U.S. Department of Energy
 Rocky Flats Environmental Technology Site



Rocky Mountain
 Remediation Services, LLC
 Geospatial Information Systems Group
 10000 North
 100th Avenue
 Denver, CO 80231

MAP ID: 07-0130

August 15, 1997

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Figure 4-10
Distribution of Uranium 235
in Surface Soil

- EXPLANATION**
- Less than 10 pCi/g
 - 10 pCi/g through 100 pCi/g
 - 100 pCi/g through 1000 pCi/g
 - 1000 pCi/g or greater
 - Selected SWM Pit

Standard Map Features

- Boundary
- Lakeland
- Storm drain
- Fe
- Rocky Flats
- Paved
- Ditch

DATA SOURCE
 Public domain hydrography, roads and other features were digitized from the 1:50,000 scale map of the Rocky Flats area prepared by ESRI, Inc. in 1986. Digitized from orthorectified aerial photography, 1986.

NOTE
 Contours were developed using data from the field sampling locations shown in Figure 4-3. Contours were edited using professional judgment. Contour values are not guaranteed.

Scale: 1 inch = 21330 feet



State Plan Code: P
 Colorado: 2
 District: NAD27

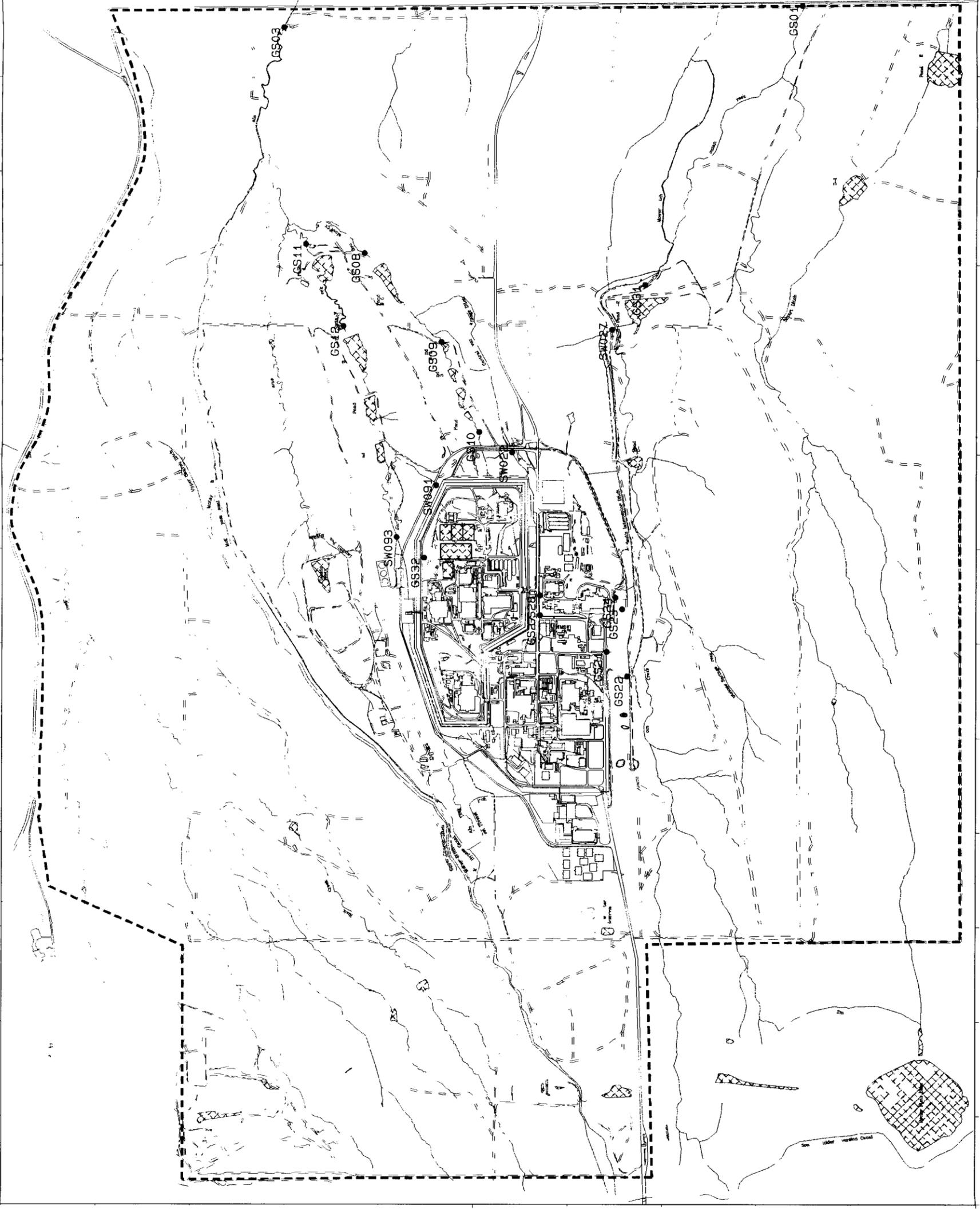
U.S. Department of Energy
 Rocky Flats Environmental Technology Site



Rocky Mountain Remediation Services, LLC
 10000 Interstate 25
 Suite 404
 Denver, CO 80231

MAP ID: 87-0130

August 16, 1987



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Figure 4-11
Distribution of Uranium 238
in Surface Soil

EXPLANATION	
	Le th 10 pC/g
	10 pC/g g but l th 100 pC/g
	100 pC/g g b l th 1000 pCi/g
	1000 pC/g gre te
	Sele t d SWM rt g St

Standard Map Feature

	B l d g & th ru re
	Lak dp d
	St m dlt h h d g f t
	Fs
	Ro ky Fl b d ry
	Paved d
	D ro d

DATA SOURCE:
 Aerial photos, maps, utility, roads and other features were digitized from a 1:25,000 scale map prepared by ESRI, Inc., Las Vegas, NV. Digitized from aerial photography, 1985.

NOTE:
 Contours were developed using data from surface soil sampling locations shown in Figures 4-2 through 4-5. Contour values were estimated where necessary. Judgment on grid created by NCHS.

1 h p ee t app ximet ly 1778 feet
 S al 1 21330

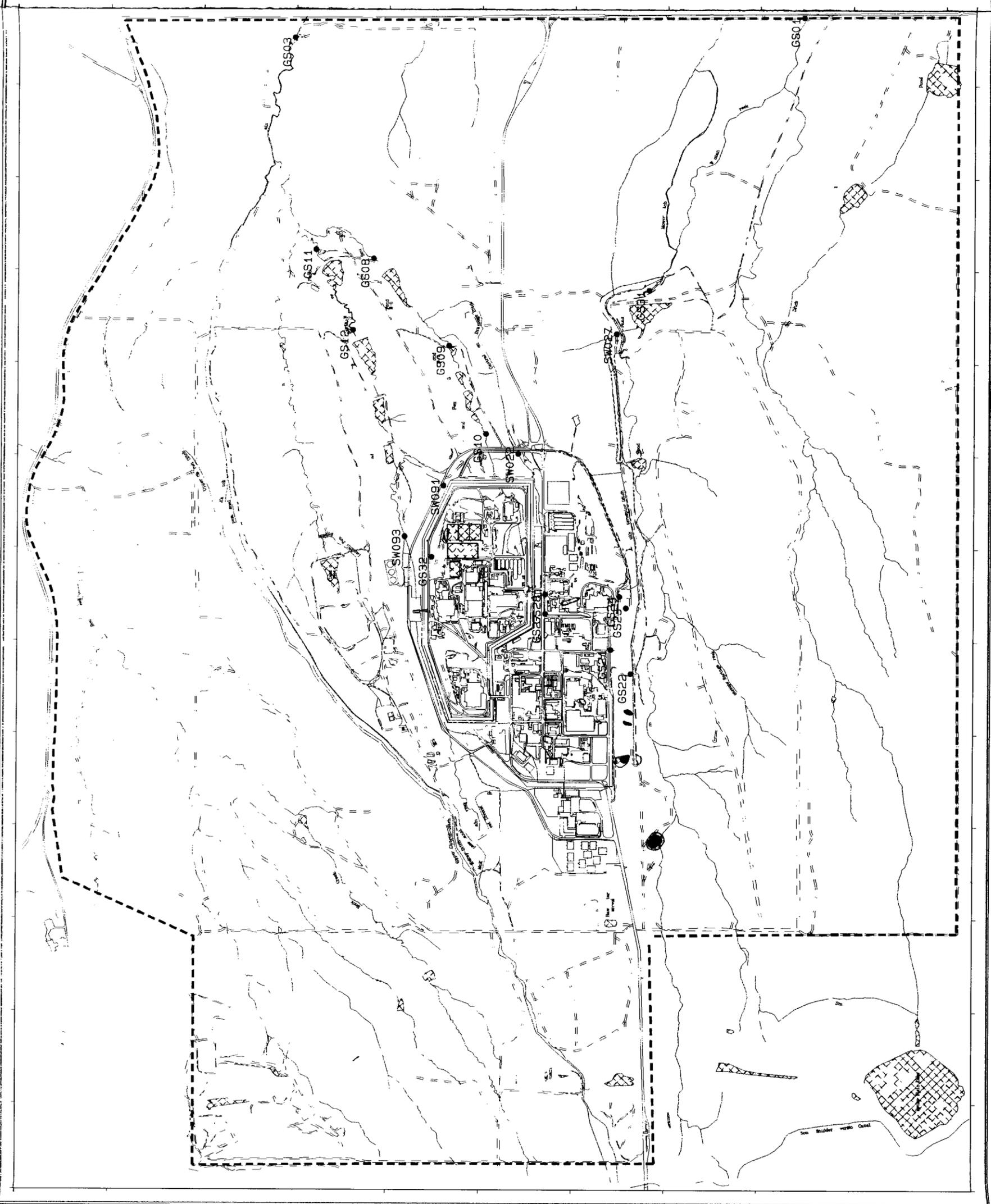


Stet Plan Coo d at P feet
 Col d C t al Z
 Det m NAD27

U S Department of Energy
 Rocky Flats Environmental Technology Site

Prepared by:
 AMRS Rocky Mountain Remediation Services, LLC
 6000 North 10th Street
 Fort Collins, Colorado 80501

MAP ID: 87-0130 August 16 1997



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Figure 5 1

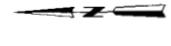
BackGround Groundwater Monitoring Wells and Flow Paths

EXPLANATION
 ● B kg o d Gro dw M g W ll

Stand d Map Featu es

- B u l d g & h t r u
- L k e
- S e m d t h d r a g f t u r e
- Fe
- R k y F l a t e s
- - - P a d
- D i r t r o a d

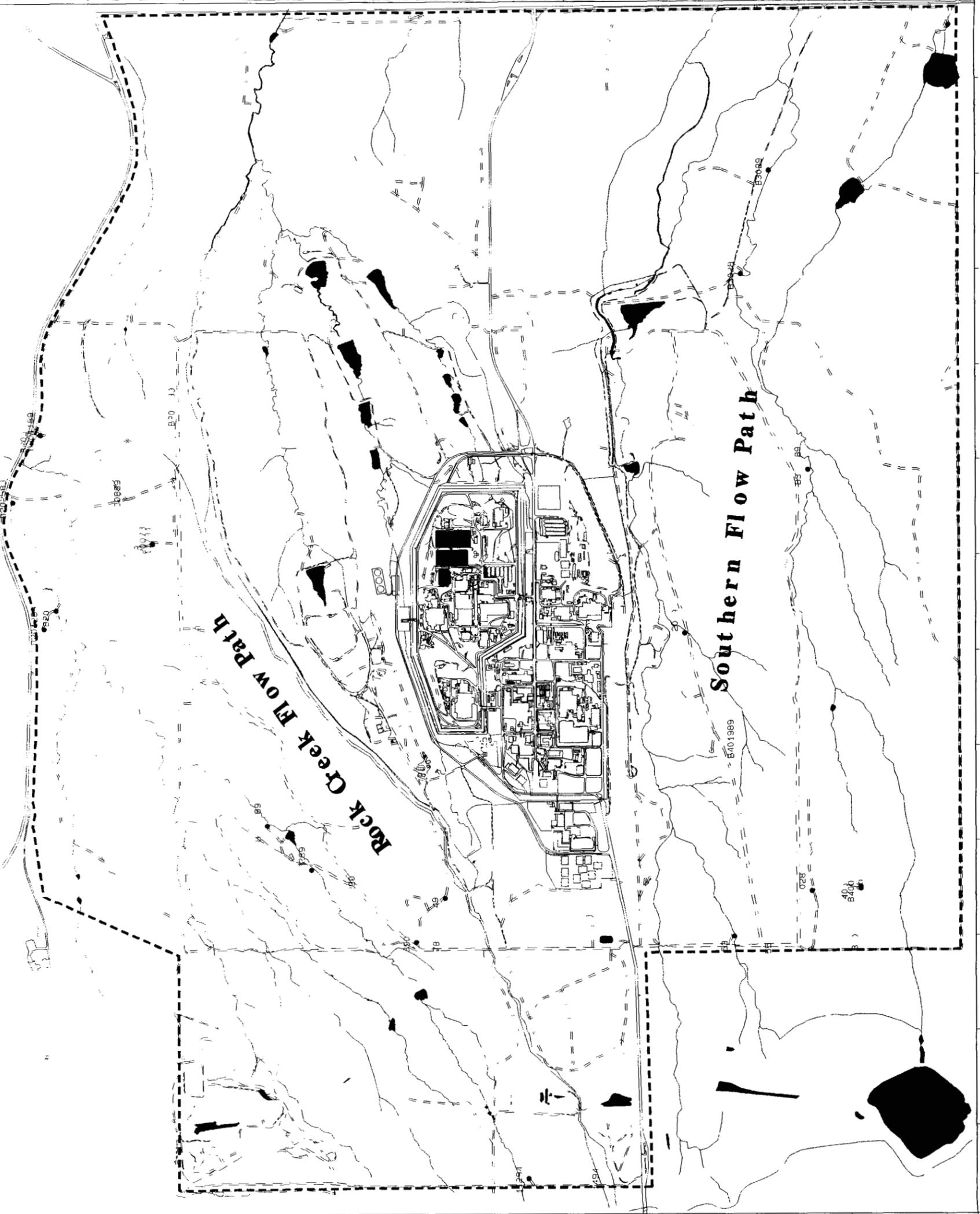
DATA SOURCE
 Background, topographic, and elevation provided by Rocky Mountain Remediation Services, Inc. All Rocky Flats, Inc. Hydrology provided by USGS data unknown



1 i n c o r r e c t i o n 2 1 3 3 0 m e t e r s 1 7 7 8 f e e t



Stat Plan Co d et P J t
 C o r d C o t L Z
 D t m NAD27



U S Department of Energy
 Rocky Flats Environmental Technology Site



Rocky Mountain Remediation Services, LLC
 Being paid information systems by
 Rocky Flats Environmental Technology Site
 Contract # 44-02-044

MAP ID 97-0130

July 23 1997

Figure 5-2
Unfiltered Plutonium 239/240
in Alluvial Groundwater
1991 1995

Plutonium 239/240 Activity-Concentration (pCi/l)

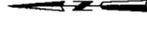
- 0.0 0.05
- 0.05 0.15
- 0.15 0.5
- 0.5 1.0
- 1.0

BACKGROUND BENCHMARK 0.047 pCi/L

- Standard Map Feature
- Buildings the structures
 - ▨ Soils vaporatio ponds
 - ▧ Lakes and pond
 - Streams, ditches, other drainage features
 - Fences
 - - - Rocky Flats boundary
 - == Paved roads
 - DI roads

DA SOURCE:
 Analysis, Remediation, and Safety work and other
 information from the Rocky Flats site
 inventory, 1991. RFL Site Inventory data
 collected from the site inventory, 1991.

NOTE:
 This map represents the data collected from
 January 1991 through December 1995.



Scale 1:21330
 1 inch represents approximately 1778 feet



State Plane Colorado North
 Colorado Central Zone
 Datum NAD27

U.S. Department of Energy
 Rocky Flats Environmental Technology Site



Rocky Mountain
 Remediation Services, LLC
 Geographic Information Systems Group
 10000 W. Environmental Technology Site
 Suite 100, Littleton, CO 80120-4004

MAP ID: 07-0188

August 15, 1997



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Figure 5-3
Unfiltered Americium 241
in Alluvial Groundwater
1991 1995

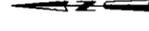
- Americium 241 Activity-Concentration (pCi/l)
- 0.0 0.04
 - 0.04 0.15
 - 0.15 1.0
 - 1.0 5.0
 - > 5.0

BACKGROUND BENCHMARK 0.038 pCi/l

- Site Land Use Feature
- Buildings and structures
 - ▨ Solar evaporative ponds
 - Lakes and ponds
 - Stream and ditches drainage features
 - Fences
 - Rock Flats boundary
 - Paved roads
 - Dirt roads

Map prepared by Rocky Mountain Remediation Services, LLC
 Project: Rocky Mountain Remediation Services, LLC
 Project: Rocky Mountain Remediation Services, LLC
 Project: Rocky Mountain Remediation Services, LLC

Map prepared by Rocky Mountain Remediation Services, LLC
 Project: Rocky Mountain Remediation Services, LLC
 Project: Rocky Mountain Remediation Services, LLC



Scale 1:21,330
 1 inch represents approximately 1778 feet



State Plane Colorado Projection
 Central Zone
 Datum NAD27

U.S. Department of Energy
 Rocky Flats Environmental Technology Site



Rocky Mountain Remediation Services, LLC
 Remediation Services, LLC
 Remediation Services, LLC

MAP ID: 97-0133

August 15, 1997



233

Figure 5-5
Filtered Uranium 233/234
in Alluvial Groundwater
1991 1995

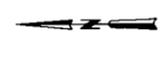
Uranium 233/234 Activity-Co concentration (pCi/l)

- 0 1
- 1 20
- 20 40
- 40 60
- 60 100
- > 100

BACKGROUND BENCHMARK 60.7 pCi/l

- Site and Map Feature**
- Buildings the structures
 - ▨ Solar evaporative ponds
 - ▤ Lakes and ponds
 - Streams ditches other drainage features
 - Fences
 - Rocky Flats boundary
 - Paved roads
 - Dirt road

NOTES:
 1. Alluvial groundwater is shown in the map area. The map shows the location of the alluvial groundwater and the location of the Rocky Flats boundary. The map is based on data collected from the 1991-1995 period.



Scale 1:21330
 1 inch represents approximately 1778 feet



State Plane Colorado North
 Central Zone
 Datum NAD27

U.S. Department of Energy
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Rocky Mountain Remediation Services, LLC
 Remediation Information Services Group
 2010 West 10th Avenue, Suite 100
 Golden, CO 80401-4404

MAP ID: 87-0183

August 15, 1997



h2

Figure 5-6
Filtered Uranium 235
in Alluvial Groundwater
1991 1995

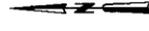
- Uranium 235 Activity Concentrations (pCi/l)
- 0.01
 - 0.1 0.5
 - 0.5 1.0
 - 1.0 1.8
 - 1.8 5
 - > 5

BACKGROUND BENCHMARK 158 pCi/l

- Standard Map Feature
- Buildings and structures
 - ▨ Solar vaporatio ponds
 - Lake and pond
 - Stream and ditches
 - drainage features
 - Fences
 - Rocky Flats boundary
 - Paved roads
 - Dirt roads

Source: Data derived from the 1991 and 1995 filtered uranium 235 activity concentration data collected from the Rocky Flats Environmental Technology Site. Data derived from the 1991 and 1995 filtered uranium 235 activity concentration data collected from the Rocky Flats Environmental Technology Site.

Note: Data are representative of average data collected from the site from January 1991 through December 1995.



Scale 1:21,330
 1 inch represents approximately 1778 feet



State Plane Colorado Projection
 Contour Interval 2 feet
 Datum NAD27

U.S. Department of Energy
 Rocky Flats Environmental Technology Site



Rocky Flats Environmental Remediation Services, LLC
 4800 West 10th Avenue, Suite 100
 Denver, CO 80202-1404

MAP ID: 07-0183

August 15, 1997

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Figure 5-7
Filtered Uranium 238
in Alluvial Groundwater
1991 1995

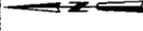
- Uranium 238 Activity Concentration (pCi/l)
- 0.077
 - 0.775
 - 5.10
 - 10.42
 - 42.100
 - > 100

BACKGROUND BENCHMARK 418 pCi/L

- Standard Map Feature
- Buildings and structures
 - ▨ Solar evaporator ponds
 - Lakes and ponds
 - Streams, ditches, drainage features
 - Fences
 - Rocky Flats boundary
 - Paved roads
 - Dirt roads

Source: Aerial photography, maps and other data. Digitized by RMRS, Inc. in cooperation with the U.S. Environmental Protection Agency. Digitized from the orthophotograph, 1995.

NOTE: Each dot represents the average of all samples collected from January 1995 through December 1995.



Scale 1:21,330
 1 inch represents approximately 1778 feet



State Plane Colorado Projection
 Central Zone
 Datum NAD27

Prepared by
U.S. Department of Energy
Rocky Flats Environmental Technology Site



Rocky Mountain
 Remediation Services, LLC
 Remediation Information Systems Group
 10000 W. Environmental Parkway, Suite 100
 Denver, CO 80244-4604

MAP ID: 97-0183

August 15, 1997

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Figure 6 1
Surface Water
Monitoring Stations

- EXPLANATION**
- ▲ N w s D t
 - P o f C m p l
 - A c L e v l F r a m w r k
 - P e r f m
 - ▲ O p r a / M

Standard Map Features

- B u i l d i n g
- L a k e / P o n d
- S r e m d i t h d r a g f t r e
- F R o a d
- R o a d
- P a v e d r o a d
- D i r t r o a d

DATA SOURCE:
 Aerial photography and maps provided by
 Rocky Flats, Inc. 89
 Contour data provided by
 Rocky Flats, Inc. 89
 Hydrology data unknown



Scale = 1 : 15640
 1 inch represents approximately 1303 feet



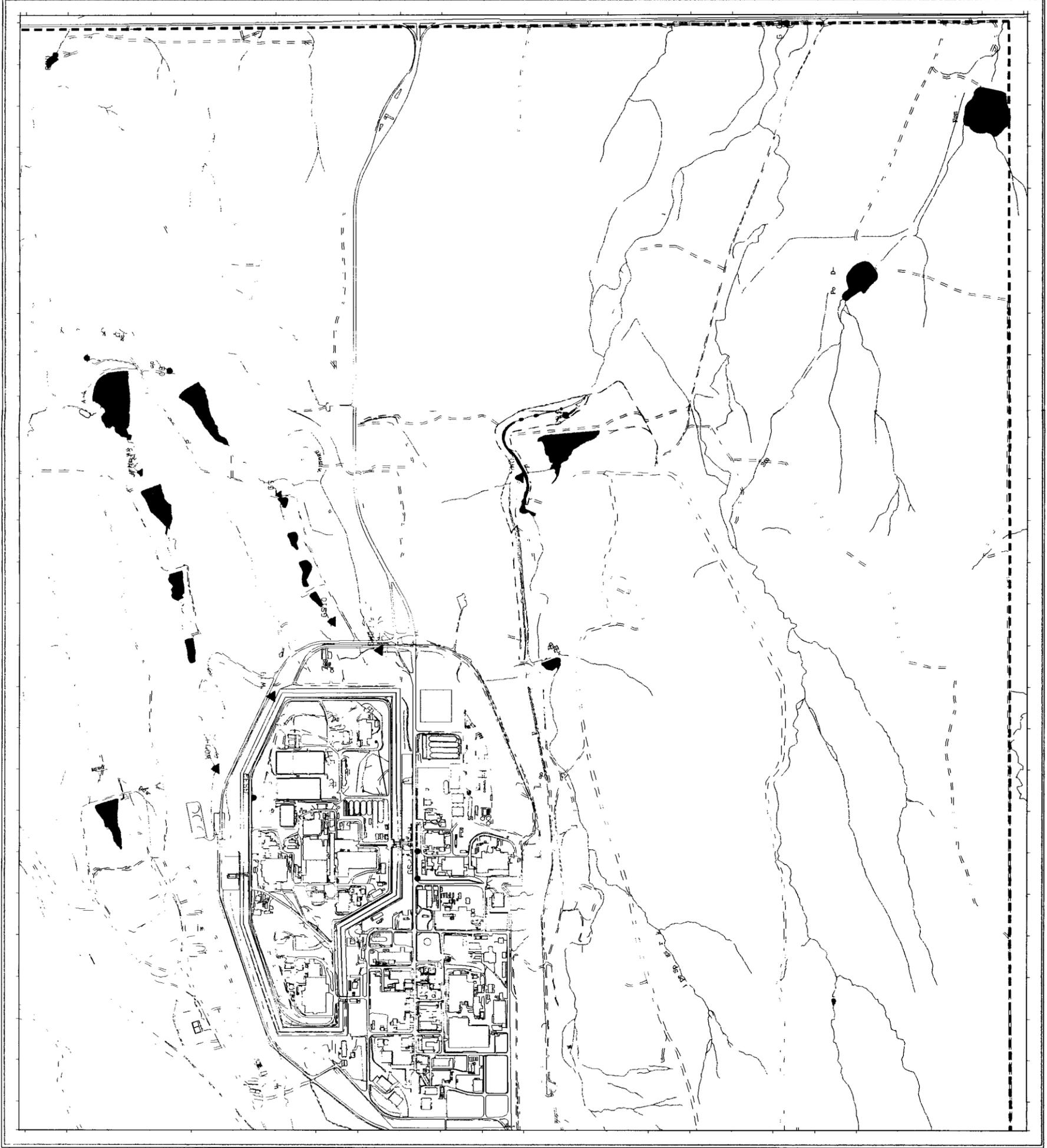
State Plane Coordinate System
 Zone 12
 Datum NAD27

Prepared by
U S Department of Energy
Rocky Flats Environmental Technology Site

RMRS Rocky Mountain Remediation Services LLC.
 6665 Park Lane, Suite 100
 Golden, CO 80401
 Phone: 303.440.4400
 Fax: 303.440.4402

MAP ID: RF-0085

July 24, 1997



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Figure 8-4
Distribution of Uranium 233/234
in Surface Soil and Groundwater

EXPLANATION

- U-233/234 Surface Soil
- Less than 10 pCi/g
- 10 pCi/g greater than 100 pCi/g
- 100 pCi/g greater than 1000 pCi/g
- 1000 pCi/g or more
- U-233/234 Groundwater pCi/L
- 0.1
- 1.20
- 20.40
- 40.80
- 80.100
- > 100
- BACKGROUND BENCHMARK = 60.7 pCi/L
- Select SWM坑 Starting

Standard Map Features

- Building & other structure
- Lake or pond
- Stream, ditch, or other linear feature
- Fence
- Contour (20-Foot)
- Rocky flat boundary
- Paved
- Dirt road



State Plane Coordinate Projection
 Colorado Central Zone
 Datum: NAD27

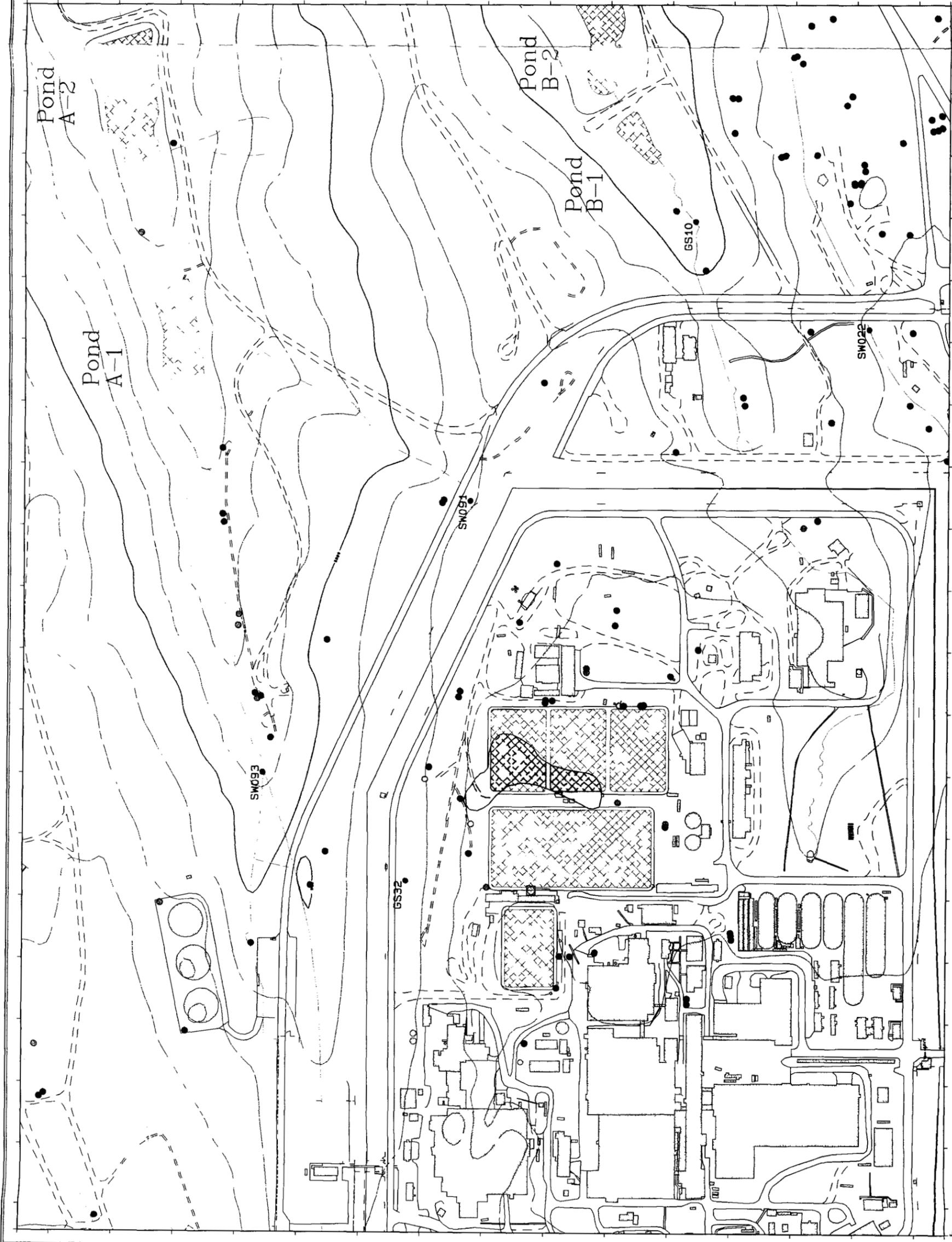
U.S. Department of Energy
 Rocky Flats Environmental Technology Site



Rocky Mountain
 Remediation Services, L.L.C.
 Remediation Services Group
 Environmental Technology Site
 Golden, CO 80642-4404

MAP ID: 97-0190

August 20, 1997



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Figure 8 5
Distribution of Uranium 238
in Surface Soil and Groundwater

- EXPLANATION**
- U 233/234 Surf Soil
 - La h 10 pC/g
 - 10 pC/g get btl th 100 pC/g
 - 100 pC/g g b l h 1000 pC/g
 - 1000 pC/g g t
 - U 238 G dw pC/L
 - 0 0.77
 - 0.77 5
 - 5 10
 - 10 42
 - 42 100
 - 100
 - BACKGROUND BENCHMARK 418 pC/L
 - S lec d SWM g S t

- St d d M p Fe tu
- Build g & th at uct
 - Lak d p de
 - St m d i h d g f t h
 - Fe
 - Cont (20 Fe t)
 - R ky FI b d y
 - Pa d d
 - D rt d

Stat Plan Co di et Ph lect
 Call d C rit al Z
 Det m NAD27

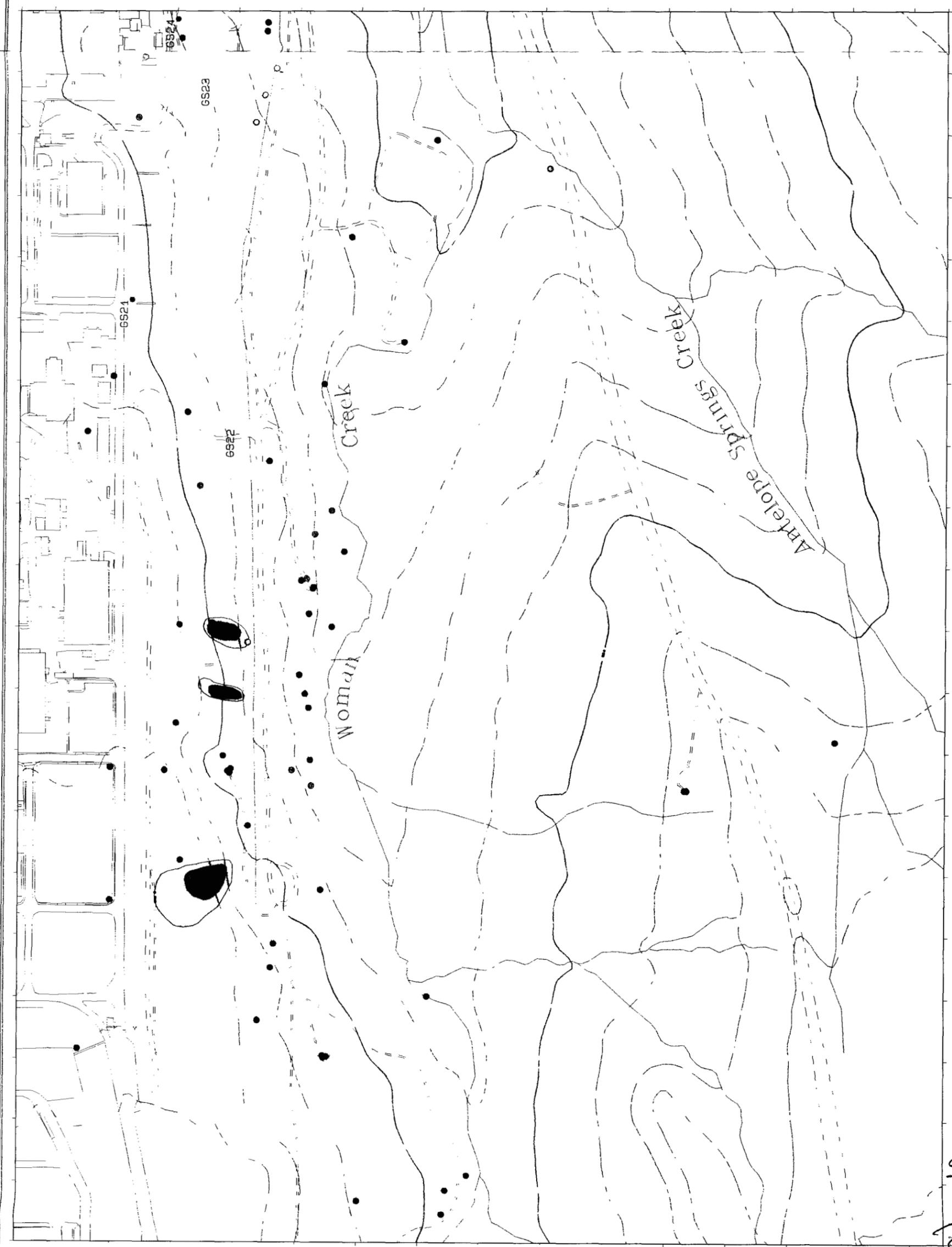
U S Department of Energy
 Rocky Flats Environmental Technology Site



Rocky Mountain Remedial Services LLC
 Environmental Remediation Services
 17700 E. 4th Ave.
 Suite 404
 Aurora, CO 80014

MAP ID: 97 0130

August 20, 1997



08/21/97

Figure 8 3
Distribution of Plutonium 239/240
in Surface Soil and Groundwater

EXPLANATION

- P 239/240 Surf Soil
- Le th 1pC/g
- 1pC/g g t b l i th 10pC/g
- 10pC/g g b t l i th 100pC/g
- 100pC/g g b l i h 1000pC/g
- 1000pC/g gre

- P 239/240 G dw
- 0.0 0.05
- 0.05 0.15
- 0.15 0.5
- 0.5 1.0
- 1.0

- BACKGROUND BENCHMARK 0.047pC/L
- S l d SWM g s

Standard Map F t e

- B l d g & h e t
- L a k d p d
- S m d t h h
- d g f t
- F e
- C o t (20 F o)
- R o k y F l a t s
- P a d r o d
- D d
- 1 h p e e t p p x m a t l y 378 f t



Stat Pl C d at Pt J et
 Col do Co t J Z
 Det m NAD27



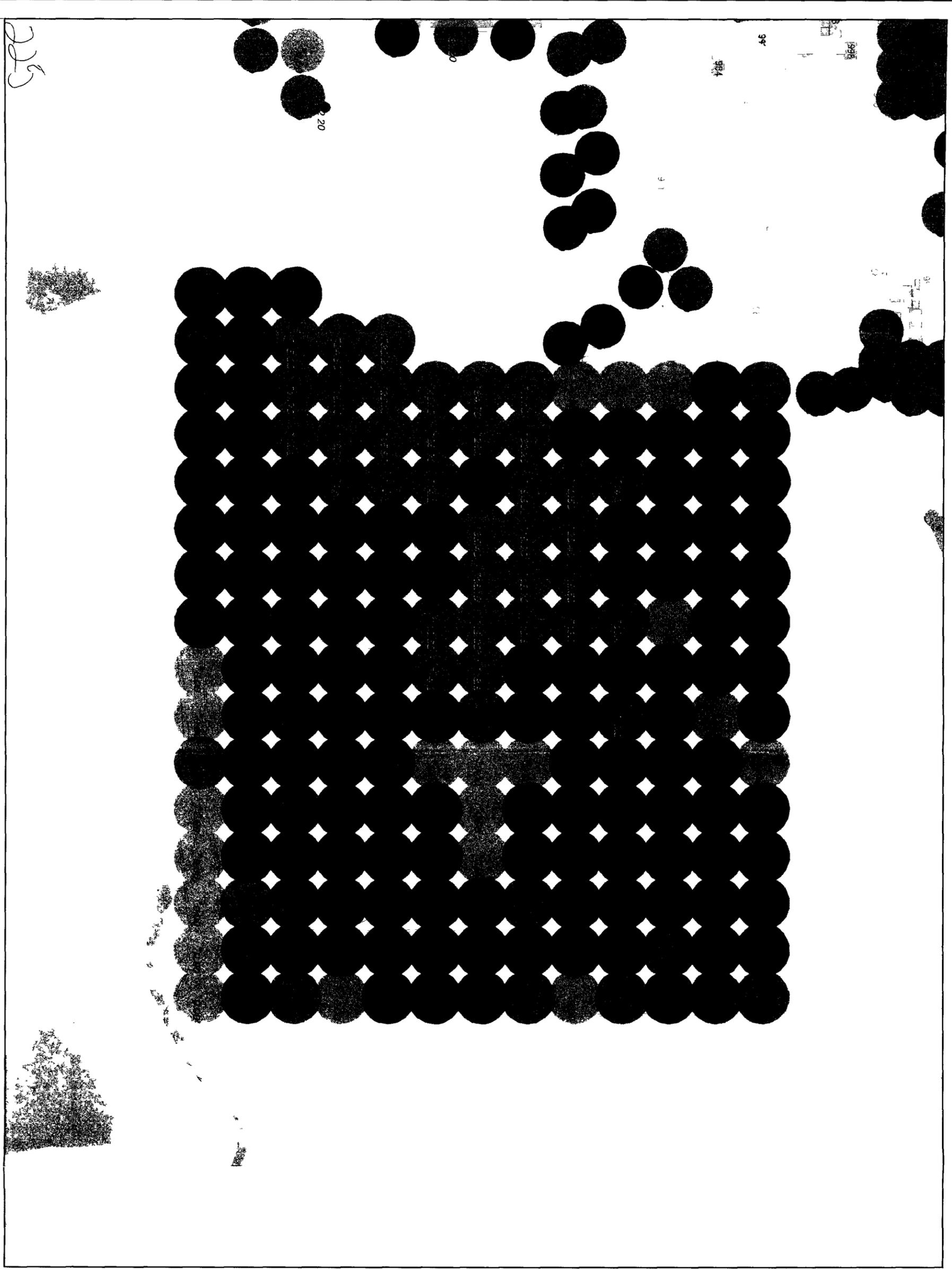
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 Rocky Flats Environmental Technology Site



RMRS R ky Mo tal
 R m d i e t i S r v i e s L L C
 a n n u a l a l l o c a t i o n p l a n
 o n l y m i 401

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Figure 4-4
 IPGe Data
 for
 Am 241 pCi/g
 (to a Field of View of 150 feet)



HPG D t R g pCi/g



St d d M P F t

B I g
 L K d p
 m d h h
 F R K E b d
 P d d
 D d

UM OUT CE
 1 3rd 4th 5th 6th
 7th 8th 9th 10th 11th 12th



US D P t m t f E g y
 R k y F i t E m t i t h l i g y S t

FMRS R k y M m t a
 R m e d i S r v L L C

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The Administrative Record Staff