4. RISK ASSESSMENT METHODOLOGY

In November 1995, Science Applications International Corporation (SAIC) conducted a baseline risk assessment as part of the Phase II Final Draft Resource Conservation and Recovery Act (RCRA) Facility Investigation (RFI) Report for the Group 3 suspected releases solid waste management units (SWMUs) at Deseret Chemical Depot (DCD) (SAIC 1995c). Baseline risks are the risks to human health and the environment in the absence of remediation or institutional controls at a site.

Since the Final Draft RFI was issued in 1995, additional groundwater and soil samples were collected during Phase IIA and IIB activities to fill data gaps. These data have been incorporated into the risk assessment presented in this Interim Final RFI Report. The general methodology used to conduct the risk assessment is described in Section 4. SWMU-specific details and results and conclusions of the risk assessment are provided in Sections 6 through 10.

4.1 BASELINE HUMAN HEALTH RISK ASSESSMENT

The baseline human health risk assessment was conducted in accordance with State of Utah, U.S. Environmental Protection Agency (EPA) Headquarters, and EPA Region VIII guidance, as specified in the following guidance documents:

- Utah Hazardous Waste Management Rules, Utah Administrative Code (R315-1 to R315-9, R315-12 to R315-14, R315-50, and R315-101) (UDEQ 1999)
- Risk Assessment Guidance for Superfund (RAGS): Human Health Evaluation Manual, Part A (EPA 1989a)
- EPA Region VIII Superfund Technical Guidance (EPA 1994c, d, and e).

The human health risk assessment is organized as follows:

- Data Collection and Evaluation (Section 4.1.1)
- Exposure Assessment (Section 4.1.2)
- Toxicity Assessment (Section 4.1.3)
- Risk Characterization (Section 4.1.4)
- Uncertainty Assessment (Section 4.1.5).

An additional section has been added that discusses the uncertainty associated with the risk assessment. As currently specified by EPA, risk assessments rely on conservative (i.e., health-protective) single-value point estimates. In this approach, point estimates are used as terms in equations that produce point estimates of risk when solved. Single point estimates by themselves, however, do not reflect the range of possible values that reasonably could be used. The resulting risk estimates appear to be more precise than they actually are, when in fact they often err to an uncertain degree on the side of health protection. The uncertainty assessment enhances the environmental decisionmaking process by providing additional information on the degree of conservatism within the risk estimates.

4.1.1 Data Collection and Evaluation

This section provides an overview of the approach used to interpret analytical data from samples collected for this RFI. This includes a brief discussion of the data used in the human health risk assessment and an examination of the steps used to select the human health chemicals of potential concern (COPCs). COPCs are defined as chemicals that are potentially site-related and present at concentrations that may impact human or ecological health.

4.1.1.1 Data Used in the Human Health Risk Assessment

Five SWMU are evaluated in the human health risk assessment. Data used to characterize the SWMU were collected from groundwater, soil, sediment, and surface water primarily during Phase II field activities. Phase II activities were initiated in 1992 with field work conducted in 1994 and 1995. Phase II data used in the risk assessment consist of soil samples collected at SWMU 11, 20, 33, and 37, and groundwater samples collected at SWMU 11 and 19. Since the 1995 Final Draft RFI (SAIC 1995c) was issued, additional data were collected to fill data gaps. During Phase IIA, groundwater samples were collected from existing wells at SWMU 11 and 19 and additional soil samples were collected at SWMU 33 from areas inside and immediately outside Building 536. During Phase IIB, soil samples were collected at SWMU 20 (around the septic tank), SWMU 33 (outside Building 536), SWMU 37, and 10 background locations.

A sediment and a sludge sample were collected from the septic tanks at SWMU 19 and 20, respectively, during the Phase II field effort. The tank at SWMU 20 since has been removed and there are no current or any realistic future exposures to contaminants within the septic tank at SWMU 19. Therefore, these data were excluded from the risk assessment. At SWMU 37, data were collected from slag piles present at the time of sampling. These piles have been removed from the SWMU. Therefore, data from the slag piles also were excluded from the risk assessment.

Historical data (i.e., not collected by SAIC) used in the risk assessment include both SWMU and background data. Pre-Phase I data include background and SWMU 11 groundwater samples collected in 1987 and 1988. Phase I data include groundwater samples collected at background and SWMU 11 and 19, and soil data collected at background and SWMU 19 in 1990, 1992, and 1993. Groundwater data for SWMU 11 and 19 collected in 1998 and 1999 also were used in the baseline risk assessment.

4.1.1.2 COPC Selection

COPCs were selected in accordance with RAGS (EPA 1989a) and EPA Region VIII guidance (EPA 1994d) using the following process:

- Data validation
- Data aggregation
- Background comparison.

These components are addressed in the following subsections.

Data Validation

Data used in the risk assessment have been validated in accordance with U.S. Army Environmental Center (USAEC) protocols (USAEC 1993 and 1995) or EPA Contract Laboratory Program (CLP) National Functional Guideline protocols (EPA 1994a and 1994b). During the validation process, data were qualified based on the results of the quality assurance/quality control (QA/QC) samples (e.g., laboratory and field blanks). The validation methods and results are described in Appendix J.

Data validated in accordance with USAEC protocols are used in the risk assessment (in both the background comparison and the calculation of the exposure point concentrations) as follows:

- Detected analytes that have no Installation Restoration Data Management Information System (IRDMIS) qualifiers or flagging codes are incorporated without changes.
- Detected analytes that have IRDMIS data qualifiers or flagging codes other than A, B, G, I, U, S, T, and + are incorporated without changes.
- Sample results with IRDMIS flag codes A, B, G, I, U, S, T, and + are treated as not detected and are included in the risk assessment as one-half the detected value; these flags are associated primarily with blank contamination and are explained in greater detail in the data quality assessment (Appendix J).

Data validated in accordance with EPA CLP National Functional Guideline protocols are used in the risk assessment (in both the background comparison and the calculation of the exposure point concentrations) as follows:

- Detected analytes that have no CLP qualifiers are incorporated without changes.
- Detected analytes that have CLP qualifiers other than U, UJ, or R are incorporated without changes.
- Sample results with CLP qualifiers U and UJ are treated as not detected and are included in the risk assessment as one-half the detected value.

For both USAEC and CLP data, not detected results were incorporated into the data set only if the analyte was detected at least once in a given data group. Data for which analytes were rejected during data validation were excluded from the risk assessment. Field duplicates were collected to assess variability in the sampling process. They are used in the data quality assessment, but were not included as part of the data set used to estimate risk.

Data Aggregation

Data aggregation refers to the manner in which sample data are combined for analysis and evaluation in the RFI. The data are aggregated into exposure units, which is a geographic area over which a receptor is likely to average exposure (both spatially and temporally) and is defined on the basis of observed or assumed patterns of receptor behavior, historic activities, and the nature and extent of contamination. Analytical data were aggregated by medium and by area.

Soil data also were aggregated according to the depth at which the samples were collected in order to distinguish between exposures involving only surface contact with the soil and exposures involving contact with both surface and subsurface soils.

Aggregation by Area—For the human health risk assessment, the exposure units for the Group 3 SWMU are as follows:

- SWMU 11
- SWMU 19
- SWMU 20
- SWMU 33A, area inside Building 536
- SWMU 33B, area immediately outside Building 536
- SWMU 33C, drainage swale
- SWMU 37, pit floor
- SWMU 37, slope.

Despite its large size, SWMU 11 was not divided into multiple exposure units. The data show that concentrations of contaminants across the SWMU are generally consistent and areas of elevated concentrations are not apparent.

SWMU 33 was divided into three exposure units due to differing patterns of receptor behavior. Workers periodically visit the SWMU to move drums in and out of the building. However, no work is conducted in or immediately around the drainage swale.

At SWMU 37, samples were collected both from the pit floor and the sloped walls of the pit. The associated data were aggregated into separate exposure units. Samples were collected from the pit floor to determine if contamination was introduced from the formerly removed slag piles. Samples were collected from the pit walls because the Phase II investigation revealed a disposal trench of thermate bombs that had been detonated in-place.

Aggregation by Depth—The risk assessment for soil focuses on two depths: surface soil and subsurface soil. For the Group 3 SWMUs, the following designations were made:

- Surface soil—collected from 0 to 0.5 feet below land surface (BLS)
- Subsurface soil—collected from >0.5 to 15 feet BLS.

Both surface and subsurface soil data are important for land use scenarios that include soil intrusion (e.g., construction of buildings or residences). Only surface soil data are needed for land use scenarios that do not include soil intrusion. Soil samples collected at depths greater than 15 feet BLS were excluded from the risk assessment because soils deeper than this are below the typical construction zone and human contact with soils at this depth is very unlikely.

At SWMU 19, historical soil and surface water data were collected within or near sumps located in the middle of the concrete foundation. Since these samples were collected, the sumps have been filled in with large rocks. The surface water data were eliminated from the risk assessment because the surface water no longer exists. The soil data were eliminated from the

surface soil data set because these soils are under rocks and are no longer exposed to the surface, but were included in the subsurface soil data set.

Appendix K presents the samples (site identification and sample depth) associated with each exposure unit.

Background Comparison

A background comparison for inorganic chemicals was conducted using two different methods: analysis of variance (ANOVA) (to support the baseline risk assessment) and upper tolerance limits (UTLs) (to evaluate the nature and extent of contamination). The ANOVA methods identify site-related chemicals, as specified in EPA Region VIII guidance (EPA 1994d). Inorganic chemicals determined to be site-related were selected as COPCs. For the nature and extent evaluation, UTLs were calculated using the background data set. The UTL comparison was not used to select COPCs. The UTLs help identify the extent of contamination by defining a threshold; concentrations above the threshold are considered site-related and those below are considered indistinguishable from background. In the ANOVA comparison, no threshold level is identified. Rather, an analyte is considered site-related or not based on the distribution of the entire data set. Therefore, for a given analyte, all detected values are considered either site-related or not site-related.

A background comparison was not conducted for organic compounds because background concentrations of organic compounds are assumed to be zero. The background comparison for groundwater data was conducted only at SWMU 11 because this is the only SWMU where the monitoring well samples were analyzed for inorganic analytes. The groundwater monitoring samples collected at SWMU 19 were not analyzed for inorganic analytes based on the fact that historical site usage indicated inorganics were not a concern (SAIC 1995a). Since the background comparison is applicable only to inorganic substances, no background comparison was conducted for groundwater at SWMU 19.

The background data set for soil is composed of samples that were collected from 1.5 to 3 feet BLS during previous investigations and additional samples from surface to 10 feet BLS that were collected as part of the Phase IIB activities. The background soil data have been aggregated into a single exposure unit with samples from all depths combined. Background surface and subsurface soils were combined because the soil types are essentially the same and the combination produces a more statistically robust data set. At each SWMU, the same background data set was compared to site surface and subsurface soils, as follows:

- Site surface soil (i.e., SWMU data from 0 to 0.5 feet BLS) was compared to background soil (0 to 10 feet BLS)
- Site subsurface soils (i.e., SWMU data from >0.5 to 15 feet BLS) was compared to background soil (0 to 10 feet BLS).

Therefore, two sets of soil COPCs were used in the risk assessment (one for the 0- to 0.5-foot soil interval and one for the >0.5- to 15-foot soil interval). Appendix K lists the samples associated with the background data set (site identification and sample depth).

For groundwater, results from multiple sampling rounds collected from a single monitoring well were averaged to create a single data point. Multiple rounds were collected to capture seasonal fluctuations in the groundwater concentrations.

"Not detected" results were treated as one-half the limit of detection and included in the background comparison. Field duplicates were collected to assess variability in the sampling process. Field duplicates are used in the data quality assessment, but were not included in the background comparison or risk assessment.

Background Comparison to Support the Baseline Risk Assessment—In the 1995 Final Draft RFI Report, the ANOVA background comparison was conducted using EPA Region VIII methods (EPA 1994d). The same methodology was used for this Interim Final RFI. Inorganic chemicals determined by the background comparison to be site-related were selected as COPCs. Because background concentrations of organic compounds are assumed to be zero, all detected organic compounds also were selected as COPCs.

The background comparison was conducted using different tests depending on the percentage of detected values in the site and background data sets. As shown in Table 4-1, ANOVA methods were used unless the frequency of detection in the background data set is low (i.e., less than 10 percent).

Table 4-1. Group 3 SWMU Background Comparison Methods
Deseret Chemical Depot, Tooele, Utah

% of Detects in Site Data Set	% of Detects in Background Data Set	Background Comparison Test/Method
0 – 100	0	No comparison
>0 – 100	<10	Poisson UTL
>0 – 50	>50	Mann-Whitney test
>0 – 100	10 – 50	Mann-Whitney test
>50 – 100	>50 – 100	Student's t-test* or Mann-Whitney test

* Student's t-test is used if the distributions in the site and background data sets are the same and the result of the F-test is equal; otherwise, the Mann-Whitney test is used.

The different background comparison methods are summarized as follows:

- ***Poisson UTL***—A UTL based on the Poisson distribution was calculated for the background data set and each detected site concentration was compared to the Poisson UTL. If any sample concentration exceeded the Poisson UTL, that analyte was identified as a COPC and included in the risk assessment.
- ***Mann-Whitney Test***—The Mann-Whitney test is a nonparametric test that compares the ranks of the site and background data sets to determine if they differ statistically. If the data sets differed statistically and the mean concentration in the site data was greater than the mean concentration in the background data, the analyte was identified as a COPC and included in the risk assessment.

- ***Student's t-test***—If the distributions of both data sets were determined to be normal or lognormal and the result of the F-test was equal, the Student's t-test was used to determine if the data sets differed statistically. If the data sets differed statistically and the mean concentration in the site data was greater than the mean concentration in the background data, the analyte was identified as a COPC and included in the risk assessment.

Background Comparison to Support the Nature and Extent Evaluation—UTLs were calculated from the background data set and used to help define the extent of contamination for inorganic chemicals identified as COPCs. The ANOVA tests establish the probability (for a given analyte) that the distributions of the site and background data sets differ statistically. However, in order to determine the nature and extent of contamination, a threshold concentration is needed below which levels are considered indistinguishable from background and above which levels are considered above background.

The UTL is calculated differently based on the distribution of the data. For normally or lognormally distributed data, the UTL is an upper confidence limit on a percentile (in this case, the 95 percent confidence on the 95th percentile) of the background data set. For data sets that are not properly represented using normal statistics, a nonparametric UTL is calculated. Similar to the normal and lognormal UTL, the nonparametric UTL represents a high-end value in the distribution of background data. For a given analyte, each site sample concentration is compared to the corresponding UTL. The UTL is considered a threshold concentration that defines for each chemical the concentration considered above background. Only sample results above the background UTLs were included in the discussion of the nature and extent of contamination for each SWMU.

A more detailed discussion, including pertinent equations and references, is presented in Appendix K. The results of the background comparison also are presented in Appendix K.

COPCs for DCD

The COPCs for each exposure unit under investigation are listed in Sections 6 through 10 and in the Appendix K tables entitled "Summary Statistics and Exposure Point Concentrations." COPCs for surface soil may differ from those in shallow subsurface soil because the background comparison was conducted separately for the two depth intervals. Because monitoring data were not collected for the food chain pathways, exposure point concentrations for produce and beef were derived from soil concentrations using uptake models. Therefore, COPCs for the food chain pathways are the same as those in soil.

4.1.2 Exposure Assessment

The objective of the exposure assessment is to estimate the type and magnitude of potential human exposures to COPCs. For a given receptor group, this results in an estimate of chronic daily intake or dose to COPCs in environmental media at DCD. The exposure assessment, in conjunction with the subsequent toxicity assessment (discussed in Section 4.1.3), supports the

characterization of potential risks to human health (discussed in Section 4.1.4). The exposure assessment consists of the following principal components:

- Land use assumptions and potentially exposed receptors
- Identification of potential exposure pathways
- Derivation of exposure point concentrations
- Development of chemical intakes or dose estimates.

Following the State of Utah and EPA guidance (UDEQ 1999, EPA 1989a and 1992a), exposure is quantified by developing a reasonable maximum exposure (RME) scenario, which is a conservative exposure case that is still within the range of possible exposures. In addition, a central tendency exposure (CTE) scenario also is evaluated and used to contrast average exposures with the RME estimates. The CTE estimate differs from the RME estimate in that the exposure assumptions (e.g., exposure frequency, exposure duration, and ingestion rate) are generally mid-range rather than high-end values.

Current EPA guidance requires that RME results be the primary focus of the risk assessment. The risk assessment conclusions and recommendations are based on the RME estimates. CTE estimates are included only as a point of comparison. Appendix L provides the RME and CTE estimates for all risk estimates. The inclusion of both central tendency and high-end estimates provides more information regarding the possible distribution of risks and enhances the decisionmaking process regarding the potential need for remediation.

4.1.2.1 Land Use Assumptions and Potentially Exposed Receptors

The following section describes the land uses of the SWMU under investigation and the receptor populations that potentially may be exposed to contaminants. The risk assessment evaluates exposures under both current and potential future land uses.

Current Land Use

Current land use at DCD is industrial. Three of the Group 3 SWMU (i.e., SWMU 11, 19, and 33) are active industrial sites or adjacent to active sites where Depot workers are present every day. The remaining SWMU (i.e., SWMU 20 and 37) are inactive sites that have been abandoned. Under current land use, a Depot worker scenario was evaluated for the Group 3 SWMUs. However, at SWMU 37, the frequency of exposure was reduced because there is little or no reason for a worker to visit this SWMU. At SWMU 20, soil contamination was anticipated and, therefore, samples were collected only from the subsurface layer. Because the Depot worker scenario assumes no exposure to subsurface soils, risks were not calculated for the Depot worker at SWMU 20.

Access to DCD is strictly controlled, precluding public exposure. The facility is surrounded by a fence and all personnel and visitors must enter through a guarded gate. Further security measures have been taken at SWMU 11. This SWMU is surrounded by a double fence and is under video surveillance. Armed guards patrol SWMU 11 and its perimeter on foot and

by vehicle. For these reasons, a trespasser scenario is unrealistic and was not evaluated in the risk assessment.

Future Land Use

Although a formal reuse plan is not available for DCD, the most likely future land use is the same as current land use (i.e., industrial). With the exception of SWMU 37, risks under a future industrial land use scenario would be the same as the risks under the current industrial scenario because the exposure pathways (e.g., soil ingestion and dermal contact with soil) and exposure assumptions (e.g., soil ingestion rate and exposure frequency) would be the same. Therefore, the discussion and risk tables for the industrial scenario (i.e., Depot worker) are labeled "Current/Future" to indicate that the risks are intended to represent both current and future exposure scenarios. At SWMU 37, separate current and future risk scenarios were evaluated varying the exposure frequency.

Two other scenarios were evaluated under future land use. In accordance with the Utah Hazardous Waste Management Rules (UDEQ 1999), a residential scenario was included at all SWMUs, although this is an unlikely scenario. This scenario assumes that residences will be constructed in the future on the SWMU under investigation. In addition, a construction worker scenario was evaluated in which a worker might construct buildings or build roads. The construction worker is different from the Depot worker in that some of the exposure assumptions are different (e.g., soil ingestion rate, exposure frequency, and exposure duration) and the construction worker is assumed to be exposed to subsurface soil in addition to surface soil. Under the residential scenario, subsurface soils inadvertently may be brought to the surface during excavation. In this manner, residential adults and children may be exposed to both surface and subsurface soils.

Considering the facility's current mission, development of DCD for residential land use in the future is unlikely. The population density in the area is low. The Depot is not currently zoned for residential development and some of the SWMU do not appear to be suitable for residential conversion because they are heavily industrialized. The U.S. Army has no plans to sell any of the DCD property and, considering its mission, it is unlikely that the Depot will close in the foreseeable future.

Potentially Exposed Receptors

Under the industrial scenario, the receptors at potential risk are Depot workers. This receptor is an adult and includes Depot personnel who regularly work at or visit the SWMU under investigation. This receptor group includes guards, Chemical Surety personnel, grounds keepers, and maintenance workers who may come into contact with contaminated media while working.

Under future land use scenarios, construction workers and residents (the latter including adults and children) are the receptors at potential risk of exposure. The resident child is defined as a 15-kilogram child between the ages of 1 and 6 years. The construction worker is an adult and is expected to work on jobs that involve shorter time periods than the Depot worker and to

be exposed to both surface and subsurface soil (the Depot worker is assumed to be exposed only to surface soil).

4.1.2.2 Exposure Pathways

Exposure pathways describe "the course a chemical or physical agent takes from the source to the exposed individual" (EPA 1989a). Four components comprise an exposure pathway:

- A source and mechanism of chemical release
- A retention or transport medium (or media)
- A point of potential human contact with the contaminated medium (the exposure point)
- An exposure route (e.g., ingestion).

Each element must be present for the pathway to be complete and considered further in the risk assessment. Some complete pathways may not be quantified if the contribution to hazard or risk is clearly minor relative to other major pathways.

Figure 4-1 presents the exposure pathways from source to receptor as a conceptual site model (CSM). The CSM is a simple diagram used to help define complete exposure pathways and understand the nature and extent as well as the fate and transport of contamination.

In a complete exposure pathway, exposure occurs at the point at which contact is made between contaminants and receptors. If there is no exposure point, there is no exposure, even if contaminants have been released into the environment (e.g., contaminants that are migrating in soils that are too deep to excavate).

If a complete exposure pathway is indicated, the exposure assessment estimates the contaminant concentration and potential for human uptake at the exposure point. Hazard and risk estimates then are calculated for exposures occurring to environmental media at the exposure point via the relevant exposure routes (e.g., ingestion, dermal contact, and inhalation). Table 4-2 shows the exposure pathways evaluated in the human health risk assessment.

Soil Pathways

Soil ingestion, dermal contact with soil, inhalation of suspended soil particulates, and inhalation of chemicals volatilized from soil were evaluated for each receptor in each SWMU.

For the dermal contact pathway, chemical-specific dermal absorption values recommended by EPA (EPA 1992b and 1997a) were used to calculate an absorbed dose from exposure to chemicals in soil. The dermal absorption values are listed in Appendix L. Dermal contact risks from soil were not calculated for chemicals without chemical-specific dermal absorption values.

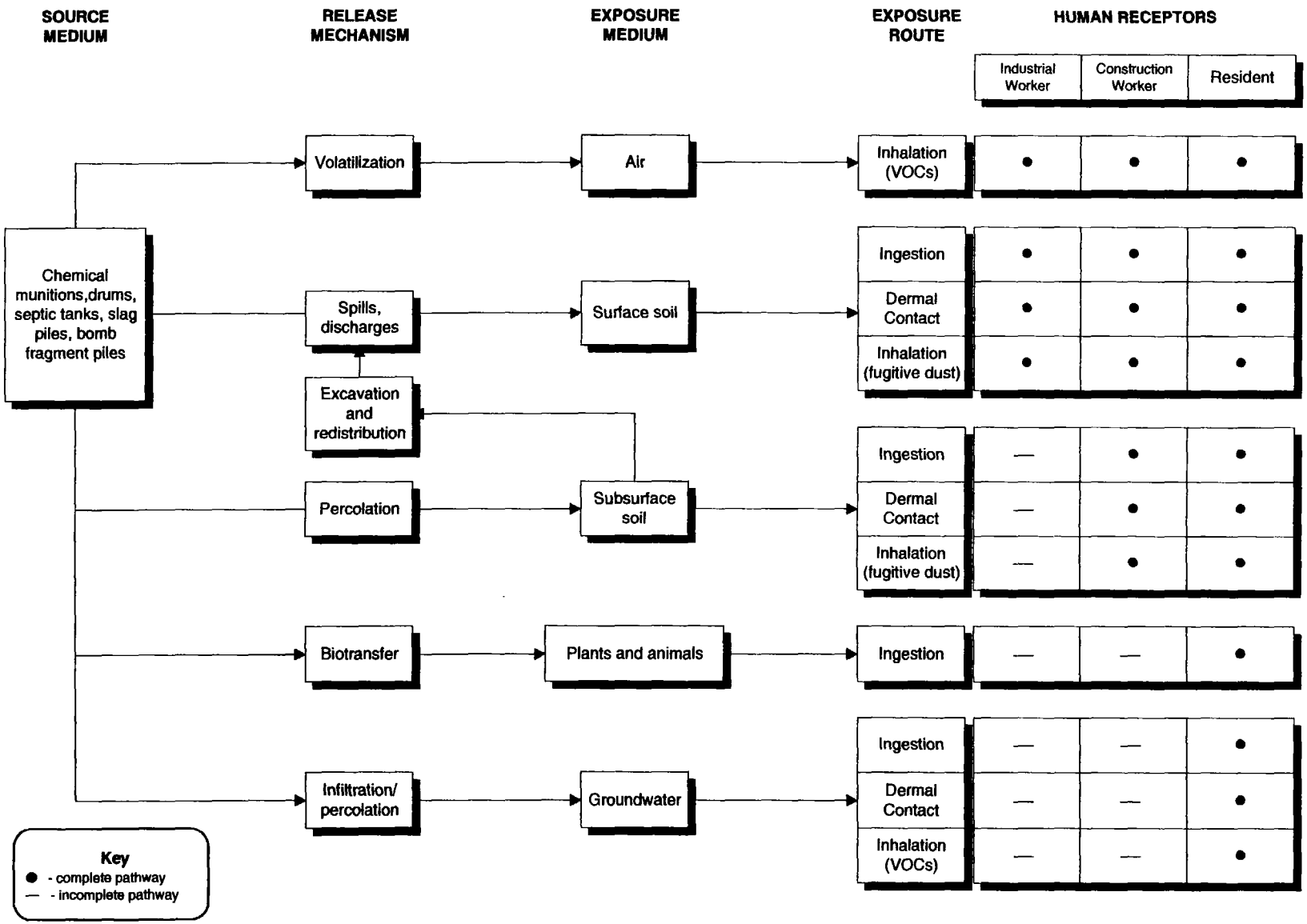

Figure 4-1. Conceptual Site Model for Human Health Risk Assessment
Deseret Chemical Depot, Tooele, Utah

Table 4-2. Exposure Pathways Evaluated in the Human Health Risk Assessment
Deseret Chemical Depot, Tooele, Utah

SWMU	Land Use	Receptor	Soil			Groundwater			Food Chain[b]
			Ingestion	Dermal Contact	Inhalation[a]	Ingestion	Dermal Contact	Inhalation	Ingestion
SWMU 11	Current/Future	Depot Worker	•	•	•				
Chemical Munitions	Future	Resident	•	•	•	•	•	•	•
Storage Area	Future	Construction Worker	•	•	•				
SWMU 19	Current/Future	Depot Worker	•	•	•				
Building 533 Foundation	Future	Resident	•	•	•	•	•	•	•
(Empty Drum Storage Area	Future	Construction Worker	•	•	•				
SWMU 20	Current/Future	Depot Worker	•	•	•				
Building 521	Future	Resident	•	•	•	Not evaluated[c]			•
(Crating Facility)	Future	Construction Worker	•	•	•				
SWMU 33	Current/Future	Depot Worker	•	•	•				
Building 536	Future	Resident	•	•	•	Not evaluated[c]			•
(CAMDS Salt Storage)	Future	Construction Worker	•	•	•				
SWMU 37	Current/Future	Depot Worker	•	•	•				
Slag Piles and	Future	Resident	•	•	•	Not evaluated[c]			•
Bomb Fragments	Future	Construction Worker	•	•	•				

[a] - Inhalation of particulates and volatile organic compounds (VOCs) were evaluated.

[b] - Food chain pathways include produce and beef ingestion.

[c] - The groundwater pathway was not evaluated at SWMUs 20, 33, and 37 because groundwater monitoring data were not collected at these SWMUs; a groundwate transport model was used to demonstrate that contaminant concentrations in soil are not a health threat to the groundwater.

Groundwater Pathways

Groundwater pathways were evaluated in accordance with the Utah Hazardous Waste Management Rules (UDEQ 1999) under the residential land use scenario. The risk assessment evaluates hypothetical exposure to groundwater from the shallow aquifer. Use of groundwater in the home allows residents to be exposed to contaminants through ingestion, dermal contact, and inhalation of volatiles while showering.

The groundwater monitored during this investigation is from the shallow aquifer, which is not a source of drinking water at DCD. DCD currently obtains its drinking water from two wells located upgradient of the facility in a deeper aquifer. The water from the deeper aquifer is tested regularly and has not been found to be contaminated. If future land use of DCD remains industrial, groundwater most likely would continue to be supplied by the two wells located in the deeper aquifer. For this reason, groundwater exposures are evaluated only for future residents requiring a new source of water and not for future construction workers.

Groundwater monitoring data were collected at SWMU 11 and 19. Risks from exposure to groundwater were evaluated quantitatively at these SWMUs. Quantitative risk assessment was not conducted at SWMU 20, 33, and 37 because monitoring data were not available. Instead, a groundwater transport model (PRIZM) has been run to address soil contaminants leaching to the groundwater.

The groundwater transport model uses the measured concentration of a highly mobile chemical in soil and several soil-related physical and chemical parameters. Of the key soil-related parameters, the organic carbon partition coefficient (K_{oc}—a measure of the chemical's mobility) is used to estimate the rate at which the chemical migrates through the unsaturated zone and how much attenuation occurs during the migration. The model was run using an indicator chemical (i.e., isopropyl methylphosphonic acid [IMPA]) with the lowest K_{oc} (the most mobile) and the highest solubility. Results of the modeling are discussed in the SWMU-specific results sections and in Appendix F.

Food Chain Pathways

The food chain pathways were evaluated in accordance with the Utah Hazardous Waste Management Rules (UDEQ 1999) only under the residential land use scenario. The pathways include ingestion of produce from a backyard garden (i.e., leafy and tuberous vegetables and fruits) and beef, and have been evaluated for each SWMU. Currently, there are no leases that allow cows to graze at DCD. Although the food chain pathways are included, exposure through these pathways is improbable because future land use most likely will be industrial, not residential. Because monitoring data were not collected for the food chain pathways, concentrations in vegetables, fruits, and beef were derived from soil concentrations using uptake models.

Pathways Not Evaluated

No surface water bodies are located at the SWMU under investigation. Therefore, surface water pathways are not included in the risk assessment. A sediment sample and a sludge sample were collected from septic tanks at SWMU 19 and 20, respectively, as part of the current and previous investigations. The septic tank at SWMU 20 since has been removed. Because there are no current exposures or any realistic future exposures to contaminants in these media, associated exposure pathways (e.g., ingestion of sludge) were not evaluated.

4.1.2.3 Exposure Point Concentrations

Exposure point concentrations are the chemicals in a given medium to which human receptors are exposed at the point of contact. Exposure point concentrations for the risk estimates were developed from monitoring data, which were aggregated as discussed in Section 4.1.1.2, or from the results of simple transport models (for produce and beef tissue). These concentrations have been calculated in accordance with EPA guidance (EPA 1992c and 1994c).

Exposure Point Concentrations from Monitoring Data

Methods used to derive exposure point concentrations are dependent upon the underlying shape of the distribution of the data set. The data used to estimate exposure point concentrations first were tested to determine if they were normally or lognormally distributed using probability plot correlation coefficients (EPA 1992d). If the data were found to be normally distributed, the exposure point concentration was calculated as the 95 percent upper confidence limit (UCL) on the arithmetic mean of the data using the Student's t-statistic. If the data were found to be lognormally distributed or a distribution could not be defined, the exposure point concentration was calculated as the 95 percent UCL using the H-statistic (EPA 1992c). One exception exists for the previous statements: if the 95 percent UCL exceeded the maximum value observed at the site, the maximum value was used as the exposure point concentration. This is not entirely consistent with the Utah Hazardous Waste Rules (UDEQ 1999), which discuss only the 95 percent UCL of the mean. When the sampling distribution is not well-characterized (e.g., as is more likely with small data sets), it is possible for the 95 percent UCL to exceed the maximum. In such cases, as written in EPA Federal guidance for risk assessment, the maximum detected concentration may be chosen as the exposure point concentration. The danger in this approach is that the true mean is actually higher than the maximum detected value, especially if the more contaminated area of the exposure unit was not sampled. However, as the sampling strategy was biased toward areas of contamination, SAIC believes that the maximum detected concentration is a more accurate estimate of the exposure point concentration and that the use of the 95 percent UCL would overestimate risks to humans and ecological receptors. Sections 6 through 10 present details concerning the biased nature of the sampling design.

"Not detected" results were treated as one-half the limit of detection and included in the calculations of the mean and UCL values. Field duplicates were collected to assess variability in the sampling process. Field duplicates are used in the data quality assessment, but were not included in the calculation of the exposure point concentrations.

For groundwater, results from multiple sampling rounds collected from a single monitoring well were averaged to create one data point for calculating the exposure point concentrations. Multiple rounds were collected to capture seasonal fluctuations in the groundwater concentrations. Monitoring data are available only for SWMU 11 and 19. Risk from exposure to groundwater at these SWMU has been evaluated using monitoring data. Groundwater monitoring data are not available for SWMU 20, 33, and 37. Instead, a groundwater transport model (PRIZM) has been run to address soil contaminants leaching to the groundwater. Therefore, quantitative risk assessment for groundwater at these three SWMU has not been conducted.

For soils, data from 0 to 0.5 feet BLS were used to calculate surface soil exposure point concentrations for the industrial worker, construction worker, and resident. In addition, data from >0.5 to 15 feet BLS were used to calculate shallow subsurface soil exposure point concentrations for the construction worker and resident. For receptors exposed to both surface and subsurface soil, risks are presented separately for each horizon.

The exposure point concentrations calculated from monitoring data and used in the human health risk assessment are included in the summary statistics and exposure point concentration tables in Appendix K. These tables also contain summary statistics for each chemical in each medium and exposure unit. The summary statistics include frequency of detection, minimum and maximum concentrations, the mean concentration, and the 95 percent UCL on the arithmetic mean. The exposure point concentrations also are included in the chemical-specific risk characterization tables in Appendix L.

Exposure Point Concentrations Derived Using Simple Models

The following paragraphs describe the models used to derive exposure point concentrations for the human health risk assessment. Because monitoring data are not available for produce and beef, these concentrations were derived from soil concentrations using biotransfer factors (BTFs).

Produce—The produce pathways evaluated in the human health risk assessment include ingestion of leafy and tuberous vegetables and fruits. The equation for uptake into produce is shown below (Baes et al. 1984):

$$C_{PRO} = C_{SO} \times BTF_{PRO} \quad (1)$$

where:

C_{PRO} = Chemical concentration in produce (mg/kg)

C_{SO} = Chemical concentration in soil (mg/kg)

BTF_{PRO} = Biotransfer factor from soil to plant for vegetation ([mg pollutant/kg plant] per [mg pollutant/kg soil]).

BTFs are used to estimate concentrations of substances in produce from measured concentrations in soil. BTFs for inorganic analytes were compiled from the Nuclear Regulatory Commission (NRC) report (1992). For organic substances, EPA (1991a) recommends methods

developed by Briggs et al. (1982). A computer model (Trapp et al. 1994 and Trapp and Matthies 1995) has been developed that is based on the method developed by Briggs. This computer model estimates tissue concentration of organic compounds in plant roots, stems, leaves, and fruits and was used to calculate risk from exposure to food chain pathways for the sites under investigation.

Beef Ingestion—The equation for uptake into beef is as follows (Belcher and Travis 1989):

$$C_{Beef} = [(Q_{FOR} \times C_{FOR}) + (Q_{SO} \times C_{SO})] \times BTF_{BEEF} \times FI \quad (2)$$

where:

C_{Beef} = Chemical concentration in beef (mg/kg)

Q_{FOR} = Quantity of ingested forage vegetation (kg dry weight/day)

C_{FOR} = Concentration of chemical in forage vegetation (mg/kg) (equal to the product of C_{SO} and $BTF_{Leafy\ Vegetables}$)

Q_{SO} = Quantity of soil ingested by cattle (kg/day)

C_{SO} = Chemical concentration in soil (mg/kg)

BTF_{Beef} = BTF for beef (day/kg)

FI = Fraction of ingested forage that is contaminated (unitless).

As with the produce pathway, BTFs for inorganic analytes were compiled from literature (NRC 1992). Concentrations of organic substances in beef were estimated from the K_{ow} using the equation presented below (Travis and Arms 1988):

$$BTF_{BEEF} = 10^{(-7.6 + \log K_{OW})} \quad (3)$$

In the risk assessment, the quantity of forage eaten by beef cattle (Q_{FOR}) was set at 3.6 kg dry weight (DW)/day and the quantity of soil ingested by beef cows (Q_{SO}) was identified as 0.39 kg/day (Belcher and Travis 1989).

4.1.2.4 Development of Chemical Intakes

This section provides information concerning the equations and exposure assumptions used to calculate chemical intakes. The risk assessment used intake equations that were developed and applied in accordance with methods presented by EPA in RAGS (EPA 1989a).

The oral and inhalation intake estimates are expressed as the administered dose of a chemical (i.e., the amount of chemical at an exchange boundary, such as the skin or the intestinal wall, that is available for absorption). However, dermal doses are estimates of absorbed dose (the amount of chemical actually absorbed into the blood stream). All chemicals are assumed

not to transform or degrade over the period of exposure (i.e., the concentration in the medium of concern remains the same).

Exposure Equations—Intake estimates (in mg/kg-day) were developed for each COPC using the corresponding exposure point concentration. Chemical intakes (or absorbed dose for dermal contact pathways) are estimated by means of the following general equation:

$$Intake\ (mg/kg - day) = \frac{C \times IR \times EF \times ED \times CF}{BW \times AT} \quad (4)$$

where:

C = Chemical concentration (exposure point concentration)
IR = Intake rate
EF = Exposure frequency
ED = Exposure duration
CF = Conversion factor (to attain units of mg/kg-day)
BW = Body weight
AT = Averaging time for noncancer or cancer effects.

Pathway-specific intake equations are presented in Appendix L.

Exposure Assumptions—Two sets of exposure factors or assumptions were developed, one representing CTE estimates and the other representing RME estimates. The CTE factors estimate average or mean exposures and may be compared with the high-end RME estimates. Following EPA direction, the CTE estimates are used only as a point of comparison and are not used in decisionmaking regarding the need for remediation. The exposure assumptions and corresponding guidance or rationale for their use are presented in Table 4-3.

4.1.3 Toxicity Assessment

The objectives of the toxicity assessment are to evaluate the inherent toxicity of the compounds under investigation and to identify and select toxicity values for use in risk characterization. The human health risk assessment for the Group 3 SWMU is concerned only with chronic (long-term) exposures because contaminant concentrations are generally low and contact rates for receptors are low and are averaged over long periods of time.

For the assessment of human health risks from exposure to chemicals, the following toxicity values are of principal importance:

- Reference doses (RfDs) for oral exposure—Acceptable intake values for chronic exposure (noncancer effects)
- Reference concentrations (RfCs) for inhalation exposure—Acceptable intake values for chronic exposure (noncancer effects); these have been converted to inhalation RfDs by multiplying by 20 m^3/day and dividing by 70 kg
- Cancer slope factors (CSFs) for oral exposure
- CSFs for the inhalation route.

Table 4-3. Exposure Factors for the Human Health Risk Assessment
Deseret Chemical Depot, Tooele, Utah

Pathway / Assumption	Units	Current/Future Land Use				Future Land Use											
		Depot Workers				Construction Workers				Resident Children				Resident Adults			
		RME		CTE		RME		CTE		RME		CTE		RME		CTE	
General																	
Body weight	kg	70	c	70	f	70	c	70	f	15	c	15	f	70	c	70	f
Exposure duration	years	25	c	5	f	5	h	2	h	6	c	2	f	24	c	7	f
Averaging time-noncancer	days	9125	c	1825	f	1825	c	730	f	2190	c	730	f	8760	c	2555	f
Averaging time - cancer	days	25550	c	25550	f	25550	c	25550	f	25550	c	25550	f	25550	c	25550	f
Soil Ingestion																	
Ingestion rate	mg/day	100	f	50	f	480	f	480	j	200	c	100	f	100	c	50	f
Bioavailability factor	none	1.0	m	1.0	m	1.0	m	1.0	m	1.0	m	1.0	m	1.0	m	1.0	m
Exposure frequency																	
SWMUs 11,19, and 33	days/year	250	c	219	f	50	h	30	h	350	c	234	f	350	c	234	f
SWMUs 20 and 37	days/year	5/250	h	5/219	h	50	h	30	h	350	c	234	f	350	c	234	f
Conversion factor	kg/mg	1.00E-06	-	1.00E-06	-	1.00E-06	-	1.00E-06	-	1.00E-06	-	1.00E-06	-	1.00E-06	-	1.00E-06	-
Soil Dermal Contact																	
Skin surface area available	cm^2/day	5800	e	5000	e	5800	e	5000	e	2010	e	1750	e	5800	e	5000	e
Soil-to-skin adherence factor	days/year	1	e	0.2	e	1	e	0.2	e	1	e	0.2	e	1	e	0.2	e
Dermal absorption factor	none	chemical-specific			i	chemical-specific			i	chemical-specific			i	chemical-specific			i
Exposure frequency																	
SWMUs 11,19, and 33	days/year	250	c	219	f	50	h	30	h	350	c	234	f	350	c	234	f
SWMUs 20 and 37	days/year	5/250	h	5/219	h	50	h	30	h	350	c	234	f	350	c	234	f
Conversion factor	kg/mg	1.00E-06	-	1.00E-06	-	1.00E-06	-	1.00E-06	-	1.00E-06	-	1.00E-06	-	1.00E-06	-	1.00E-06	-
Fugitive Dust Inhalation																	
Inhalation rate	m^3/day	20	c	20	j	24	b	24	j	10	b	10	j	20	c	20	f
Particulate emission factor	m^3/kg	8.62E+08	g	8.62E+08	g	8.62E+08	g	8.62E+08	g	8.62E+08	g	8.62E+08	g	8.62E+08	g	8.62E+08	g
Exposure frequency																	
SWMUs 11,19, and 33	days/year	250	c	219	f	50	h	30	h	350	c	234	f	350	c	234	f
SWMUs 20 and 37	days/year	5/250	h	5/219	h	50	h	30	h	350	c	234	f	350	c	234	f

Table 4-3. Exposure Factors for the Human Health Risk Assessment Deseret Chemical Depot, Tooele, Utah (Continued)

Pathway / Assumption	Units	Current/Future Land Use				Future Land Use							
		Depot Workers				Construction Workers				Resident Children			
		RME		CTE		RME		CTE		RME		CTE	
VOC Inhalation													
Inhalation rate	m^3/day	20	c	20	j	24	b	24	j	10	b	10	j
Volatilization factor	m^3/kg	chemical-specific			l	chemical-specific			l	chemical-specific			l
Exposure frequency													
SWMUs 11,19, and 33	days/year	250	c	219	f	50	h	30	h	350	c	234	f
SWMUs 20 and 37	days/year	5/250	h	5/219	h	50	h	30	h	350	c	234	f
Groundwater Ingestion													
Ingestion rate	L/day	not evaluated				not evaluated				1	b	1	b
Exposure frequency	days/year									350	c	234	f
Conversion factor	mg/µg									1.00E-03	-	1.00E-03	-
Groundwater Dermal Contact													
Skin surface area available	cm^2/day	not evaluated				not evaluated				8020	e	6980	e
Permeability coefficient	cm/hour									chemical-specific			l
Exposure time	hour/day									0.25	e	0.17	e
Exposure frequency	days/year									350	c	234	f
Conversion factor	L/cm^3 and mg/µg									1.00E-06	-	1.00E-06	-
Groundwater Inhalation													
Inhalation rate	m^3/day	not evaluated				not evaluated				15	c	15	c
Volatilization factor	L/m^3									0.5	d	0.5	d
Fraction Inhaled	none									0.13	n	0.06	n
Exposure frequency	days/year									350	c	234	f
Conversion factor	mg/µg									1.00E-03	-	1.00E-03	-
Leafy Vegetable Ingestion													
Ingestion rate	kg/day	not evaluated				not evaluated				0.045	k	0.045	k
Fraction ingested	none									0.4	b	0.25	b
Exposure frequency	days/year									350	c	234	f

Pathway / Assumption	Units	Future Land Use			
		Resident Adults			
		RME		CTE	
VOC Inhalation					
Inhalation rate	m^3/day	20	c	20	f
Volatilization factor	m^3/kg	chemical-specific			l
Exposure frequency					
SWMUs 11,19, and 33	days/year	350	c	234	f
SWMUs 20 and 37	days/year	350	c	234	f
Groundwater Ingestion					
Ingestion rate	L/day	2	c	1.4	f
Exposure frequency	days/year	350	c	234	f
Conversion factor	mg/µg	1.00E-03	-	1.00E-03	-
Groundwater Dermal Contact					
Skin surface area available	cm^2/day	23000	e	20000	e
Permeability coefficient	cm/hour	chemical-specific			l
Exposure time	hour/day	0.25	e	0.17	e
Exposure frequency	days/year	350	c	234	f
Conversion factor	L/cm^3 and mg/µg	1.00E-06	-	1.00E-06	-
Groundwater Inhalation					
Inhalation rate	m^3/day	15	c	15	c
Volatilization factor	L/m^3	0.5	d	0.5	d
Fraction Inhaled	none	0.13	n	0.06	n
Exposure frequency	days/year	350	c	234	f
Conversion factor	mg/µg	1.00E-03	-	1.00E-03	-
Leafy Vegetable Ingestion					
Ingestion rate	kg/day	0.067	b	0.067	b
Fraction ingested	none	0.4	b	0.25	b
Exposure frequency	days/year	350	c	234	f

Table 4-3. Exposure Factors for the Human Health Risk Assessment
Deseret Chemical Depot, Tooele, Utah (Continued)

Pathway / Assumption	Units	Current/Future Land Use	Future Land Use							
		Depot Workers		Construction Workers		Resident Children			Resident Adults			
		RME	CTE	RME	CTE	RME		CTE	RME		CTE	
Tuberous Vegetable Ingestion												
Ingestion rate	kg/day	not evaluated		not evaluated		0.073	k	0.073 k	0.11	b	0.11	b
Fraction ingested	none					0.4	b	0.25 b	0.4	b	0.25	b
Exposure frequency	days/year					350	c	234 f	350	c	234	f
Fruit Ingestion												
Ingestion rate	kg/day	not evaluated		not evaluated		0.093	k	0.093 k	0.14	b	0.14	b
Fraction ingested	none					0.3	b	0.2 b	0.3	b	0.2	b
Exposure frequency	days/year					350	c	234 f	350	c	234	f
Beef Ingestion												
Ingestion rate	kg/day	not evaluated		not evaluated		0.056	k	0.056 k	0.100	b	0.100	b
Fraction ingested	none					0.75	b	0.44 b	0.75	b	0.44	b
Exposure frequency	days/year					350	c	234 f	350	c	234	f

a - EPA 1989a, Risk Assessment Guidance for Superfund (RAGS), Volume I, Human Health Evaluation Manual, Part A.

b - EPA 1989b and 1996b, Exposure Factors Handbook (EFH) and Update; the inhalation rate for construction workers is based on the reasonable worst-case outdoor inhalation rate of 3 m^3/hour and assuming an 8-hour work day.

c - EPA 1991a, Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors.

d - EPA 1991b, Default values from Human Health Evaluation Manual, Part B.

e - EPA 1992b, Dermal Exposure Assessment; for skin surface area available during soil dermal contact, assumes 25 percent of total body surface area is exposed (pp. 8-10 and 8-12 of EPA 1992b).

f - EPA 1993a, Superfund's Standard Default Exposure Factors for the Central Tendency and Reasonable Maximum Exposure; the RME soil ingestion rate for the construction worker is for contact intensive activities.

g - From EBASCO 1994 RFI for Group 2 SWMUs (EBASCO used site-specific PM_{10} data from monitoring station located in Group 2 SWMUs).

h - The construction worker may be a contractor working full days for a limited time period (i.e., 10 weeks [RME] and 6 weeks [CTE]). This is repeated every year for a duration of 5 years. At SWMUs 11, 19, and 33, the exposure frequencies and durations of the Depot worker are the same as a standard full-time worker. At SWMUs 20 and 37, current and future land use have been evaluated separately. The exposure frequency is 5 days per year under current land use and 250 days per year under future land use.

i - Dermal absorption factors are presented in Appendix L.

j - If guidance is not available for the CTE but exists for the RME, the RME value will be adopted as the CTE.

k - For fruits and vegetables, the child ingestion rate was assumed to be two-thirds the adult ingestion rate using professional judgment; for beef, Yang and Nelson (1986) provide child ingestion rates (used data for children aged 1-9 years) for all meat and adult ingestion rates for beef and all meat; the adult ratio of beef-to-meat was used to convert the child meat ingestion rate to a child beef ingestion rate.

l - Dermal permeability constants and volatilization factors are presented in Appendix L.

m - A default value of 1.0 will be used, lacking EPA-verified or accepted bioavailability factors for the ingestion pathway.

n - Assumes that a fraction of the daily inhalation rate is spent inhaling vapors from the groundwater (e.g., while showering, washing, etc.).

Toxicity information preferably is obtained from the Integrated Risk Information System (IRIS) (EPA 2000). IRIS is a data base containing EPA risk assessment and risk management information for chemical substances. Data in the IRIS system are regularly reviewed and updated. If values are not available from IRIS, the Health Effects Assessment Summary Tables (HEAST) (EPA 1997b) were consulted. Where available, provisional toxicity values were used for chemicals that have no current EPA-approved RfD or CSF. These values are issued by EPA's Superfund Health Risk Technical Support Center-National Center for Environmental Assessment (SHRTSC-NCEA). Since the Final Draft RFI was issued, toxicity values have been developed for some agent and agent breakdown products (USACHPPM 1999a and 1999b). These values have been incorporated into the Interim Final RFI risk assessment.

EPA recommends two different approaches for evaluating noncancer and cancer health effects. The two approaches reflect a fundamental difference in the proposed mechanism of toxic action. In assessing the potential for noncancer health effects, EPA assumes that there is a toxicologic threshold below which no adverse health effects occur. These toxicologic thresholds are represented by RfDs for oral exposures and RfCs for inhalation exposures. The RfDs and RfCs are levels (with uncertainty spanning an order of magnitude or greater) of daily human exposures below which adverse health effects are not anticipated, even for the most sensitive members of a population (EPA 1989a). EPA derives RfDs and RfCs based on estimates of the no-observable-adverse-effect level (NOAEL) or lowest-observable-adverse-effect level (LOAEL) in humans or test animals. In this risk assessment, conversion from an RfC (concentration) to an inhalation RfD (dose) is employed.

For carcinogens, however, EPA believes that assumption of a threshold is inappropriate (EPA 1989a). An extremely low level of exposure to a carcinogen may result in chromosomal or enzyme changes leading to cancer. Therefore, EPA does not estimate a threshold for carcinogens. Instead, EPA uses a two-part evaluation in which: (1) a chemical is assigned a weight-of-evidence classification, and (2) a CSF is calculated for the chemical. In risk assessment, the CSF is used to estimate the probability of a cancer effect occurring in an exposed receptor over a lifetime.

The weight-of-evidence classification evaluates the evidence that a given chemical is a carcinogen to humans and animals. These ratings are as follows:

- A – Human carcinogen
- B1 – Probable human carcinogen—limited human data are available
- B2 – Probable human carcinogen—sufficient data in animals, and inadequate or no evidence in humans
- C – Possible human carcinogen
- D – Not classifiable as to human carcinogenicity.

EPA develops CSFs for carcinogens that have been classified as A, B1, and B2 and for many that have been classified as C. The CSFs are in units of inverse dose: $(mg/kg/day)^{-1}$.

Many of the toxic effects reported in the literature occur at much higher exposure levels than are likely for the substances that have been released into the environment at the Depot. This is an artifact resulting from the necessity of applying high doses to laboratory animals to elicit observable effects in a short period of time. Similarly, studies evaluating relatively high exposures to humans in occupational settings may have been used to develop some toxicity values that will be applied to residential exposures. In other cases, the route of exposure to a chemical influences the effects that are exhibited by a substance. The exposure route used in the experimental study (e.g., gastric gavage using a corn oil vehicle) may not relate to the exposure route being considered in the risk assessment.

The toxicity values used in the human health risk assessment are presented in Appendix L. Priority is given to the values obtained from the IRIS data base because they have been verified by the EPA RfD/RfC Work Group for noncarcinogens or the Carcinogen Risk Assessment Verification Endeavor (CRAVE) Work Group.

The toxicity assessment process is complicated by the fact that toxicity values are not readily available for all exposure routes or for all chemicals. However, EPA has provided guidance for the following: the dermal route, lead, polycyclic aromatic hydrocarbons (PAHs), and provisional toxicity values.

4.1.3.1 Dermal Route

Toxicity values are available only for the oral and inhalation routes. The intake equations for dermal contact exposures calculate absorbed dose (by incorporating a dermal absorption factor or a permeability coefficient). Thus, it is necessary to convert the administered dose toxicity value to an absorbed dose toxicity value in order to calculate risk. However, for many chemicals, a scientifically defensible data base does not exist for adjusting an oral slope factor or RfD to estimate a dermal toxicity value. Currently, EPA recommends this adjustment be made only for cadmium (EPA 1999a). For all other chemicals evaluated for dermal exposures, the oral slope factor or RfD was used to calculate risk for the dermal contact route.

For dermal contact exposures to soil, chemical-specific dermal absorption values recommended by EPA were used to calculate an absorbed dose. Dermal contact exposures to soil were not calculated for chemicals without chemical-specific dermal absorption values. Dermal absorption values used in the risk assessment are presented in Appendix L.

4.1.3.2 Lead

EPA does not provide a verified RfD or CSF for lead. The evaluation of lead was conducted by first comparing the maximum detected concentration at each exposure unit to the soil screening level of 400 parts per million (ppm) (EPA 1994f and 1994h) or to the action level in drinking water of 15 µg/L (EPA 1999b). If the maximum concentration is less than these screening levels or the background comparison demonstrated that lead concentrations detected at the SWMU are indistinguishable from background, no further evaluation was conducted.

If the maximum lead concentration is greater than the screening levels, exposures were evaluated using models to estimate blood lead levels in human receptors. Currently, EPA has

provided models for children in a residential setting (EPA 1994g) and for adult workers in an occupational setting. EPA's Technical Review Workgroup for Lead developed the model for adult workers (EPA 1996c). The model is designed to evaluate and protect the fetuses of pregnant working women and, therefore, incorporates a factor that converts the blood lead level in the mother to the blood lead level in the fetus. The adult worker model was not used in the 1995 Final Draft RFI Report because the model was developed after the Final Draft RFI Report was issued.

4.1.3.3 Polycyclic Aromatic Hydrocarbons

Most PAHs do not have published RfDs for noncancer effects, and only benzo(a)pyrene has a published slope factor for cancer effects. Because of this lack of data, EPA has provided interim guidance for evaluating some PAHs that are known to cause cancer (EPA 1993b and 1994e). In this interim guidance, EPA recommends using relative potency values (orders of magnitude) for seven PAHs to convert each carcinogen PAH concentration to an equivalent concentration of benzo(a)pyrene. These values are based on reliable studies in which PAHs caused cancer after repeated exposures to mouse skin. These relative potency values have been incorporated into the risk assessment and are listed in Appendix L.

Dermal exposures to carcinogenic PAHs have been evaluated using the relative potency values to adjust the concentration term and the oral toxicity value. Inhalation exposures to carcinogenic PAHs have been evaluated using the relative potency values and a provisional toxicity value that is based on an inhalation study on hamsters (EPA 1995a). For PAHs exhibiting noncancer effects without EPA-approved RfDs, the RfD for pyrene was used as a surrogate.

4.1.3.4 Provisional Toxicity Values

Provisional toxicity values are available for some chemicals that have no current EPA-verified RfD or CSF. These values are issued by EPA's SHRTSC-NCEA.

Risk managers should recognize that cases in which provisional toxicity values are used generally should not be regarded with the same level of review as for EPA-verified toxicity values. For example, using provisional toxicity values may cause trichloroethylene (TCE) to be identified as a chemical with human health effects exceeding EPA targets. The decision to remediate should be tempered with the understanding that the toxicity value is provisional, and may represent a low level of review relative to EPA-verified toxicity values. The provisional toxicity values used in the risk assessment are included in the toxicity tables in Appendix L.

4.1.3.5 Chromium

Chromium speciation (delineation of total chromium as chromium III versus chromium VI) will not be conducted for the Group 3 SWMUs. As in the 1995 Final Draft RFI Report (SAIC 1995c), the risk assessment has assumed a 6:1 ratio of hexavalent to trivalent chromium in sampled environmental media.

Hexavalent chromium is a carcinogen by inhalation (whereas the trivalent form is not) and has a more stringent (i.e., lower) RfD than trivalent chromium (chromium III). Based on the known history of past practices at the DCD Group 3 SWMUs, the 6:1 ratio of hexavalent to trivalent chromium is reasonable because there were no processes at the SWMUs in which hexavalent chromium was produced. In soil, trivalent chromium and its complexes are generally very stable, whereas hexavalent chromium is highly unstable. Hexavalent chromium is readily reduced to trivalent chromium in the presence of organic matter and by residual amounts of iron in weathering minerals (Eary and Rai 1989). The monitoring data for the Group 3 SWMUs indicated that iron is prevalent at DCD. In addition, soil that is dry for extended periods of time (such as that at DCD) prevents the oxidation of trivalent chromium to hexavalent chromium (Bartlett and James 1988).

4.1.4 Risk Characterization

Risk characterization combines the exposure and toxicity assessments by comparing estimates of intake or dose with appropriate toxicity values. The objective of the baseline risk characterization is to determine whether exposure to chemicals associated with the exposure units poses risks that exceed target levels for human health effects. The results of the risk assessment thus may support the determination of need for site remediation.

4.1.4.1 EPA Methods for Risk Characterization

The risk characterization presents a separate evaluation of noncancer and cancer effects. EPA methods distinguish cancer from noncancer effects because organisms typically respond differently following exposure to noncarcinogenic or carcinogenic agents.

The risk characterization requires that the potentially toxic effects associated with exposures to each COPC be combined across environmental media and exposure routes. As described in the exposure assessment, it is reasonable to assume that a receptor could be exposed to COPCs through multiple exposure routes in multiple media. Thus, it is reasonable to combine the hazards and risks to develop receptor total risk and hazard estimates.

The cancer risk is the probability of excess (incremental) lifetime cancer risk (ELCR) for an individual that can be attributed to long-term exposure to chemicals. To derive an estimate of risk, the CSF is multiplied by the estimated chronic daily dose (intake) experienced by the exposed individual. For multiple carcinogens, the risk for each compound has been summed to provide an overall estimate of risk for cancer effects (EPA 1989a).

For noncarcinogens, the chronic daily intake or dose experienced by the exposed individual is divided by the RfD. The resulting value is the hazard quotient (HQ) and is a measure of the possibility of adverse noncancer effects. To evaluate exposure from more than one noncarcinogen, the HQs are summed for all chemicals under evaluation to obtain the hazard index (HI).

4.1.4.2 Risk Characterization Methods for Lead

Health effects associated with low-level lead exposures include reproductive effects, nervous system effects, and learning disorders. At the present time, toxicological studies indicate that there may be no threshold of exposure to lead below which adverse effects do not occur. Given the uncertainty surrounding an acceptable exposure below which there would be no adverse effects for lead, EPA has withdrawn the RfD for lead from IRIS and HEAST. Lead also is classified as a B2 carcinogen (probable human carcinogen), but has no EPA-verified CSF. The noted noncancer effects related to lead exposure are likely to be more significant than the carcinogenic effects.

At exposure units where lead was not eliminated as a COPC in the background comparison and exceeded the screening level in soil (400 mg/kg) and action level in drinking water (15 µg/L), models were used to estimate blood levels of lead in human receptors. The risk characterization for lead is based on two uptake models: one for children and one for adult workers. Modeling is necessary because blood lead levels in exposed populations were not directly monitored. The lead models for children and adult workers was applied using measured soil and groundwater concentrations.

The analysis for children was conducted using a biokinetic model developed by EPA for this purpose. This model, LEAD 0.99d (EPA 1994g), was developed by EPA to estimate blood lead levels in children (from the ages of 0 to 6 years) based on uptake of lead originating from various sources in the environment. This model is not applicable to children older than 6 years or to adults. Therefore, it was used to calculate blood lead levels only for the resident child receptor. The model was designed to accept either site-specific or default inputs. SWMU-specific exposure point concentrations for soil and groundwater were used in the model and the concentration of indoor dust was assumed to be the same as soil. No adjustments were made to the default absorption methodology used in the model. The model does not distinguish between different forms of lead.

For adult workers, SAIC used a model developed by EPA's Technical Review Workgroup for Lead (EPA 1996c). The model assumes a baseline blood level and uses various exposure parameters along with a biokinetic slope factor to estimate blood lead levels. The model is designed to evaluate and protect the fetuses of pregnant working women and, therefore, incorporates a factor that converts the blood lead level in the mother to the blood lead level in the fetus. The basic equation uses the biokinetic slope factor to relate total intake of lead to blood lead. The equations used for evaluating lead exposures to adult workers are presented in Appendix L.

Both models provide default values for many of the parameters. In the case of the IEUBK model for children, these include dietary, maternal, and other sources of lead that are unrelated to site contamination, and the geometric standard deviation (GSD) for the blood lead uptake. The defaults were used, with the exception of the site-specific concentration of lead in soil. In the case of the model for adult workers, site-specific concentrations of lead in soil and exposure frequencies were used. Recommended default values were used for other parameters. However, a range of values is provided for the GSD and background or baseline level of lead. The proper value to select from each range is supposed to be based on site-specific demographics, such as

age, gender, and race. Specifically precluding certain demographic segments from the risk assessment is questionable. Given the hypothetical nature of the future residential scenario, the most conservative default values were used in the model for adults.

4.1.4.3 Interpretation of Risk Assessment Results

To determine the need for corrective action or management activities, target risk levels have been established in the Utah Administrative Rules (UDEQ 1999). According to these rules:

- If the noncancer HI is less than 1 and the cancer risk is less than 1×10^{-6} for the residential land use scenario, corrective action is not necessary and no further action may be recommended.
- If the cancer risk is less than 1×10^{-6} for the residential land use scenario and less than 1×10^{-4} for the actual or potential land use scenario, and the noncancer HI is less than 1 for the actual land use scenario, a site management plan is required that recommends management activities (e.g., monitoring and deed restrictions) and may or may not recommend corrective action.
- If the cancer risk is greater than 1×10^{-4} or the noncancer HI is greater than 1 for the actual or potential land use scenario, a site management plan is required, which recommends corrective action.

In the risk characterization, chemicals of concern (COCs) are identified to focus on the chemicals responsible for risks above targets. As opposed to COPCs, COCs are identified after the quantitative risk assessment has been completed. In order to be consistent with the guidelines set by the State of Utah for corrective action, COCs in the human health risk assessment are individual chemicals that contribute to pathway risks exceeding any of the following:

- HI of 1
- Cancer risk greater than 1×10^{-4} for the actual or potential land use scenario
- Cancer risk greater than 1×10^{-6} for the residential land use scenario.

Preliminary remediation goals (PRGs) have not been included in this Interim Final RFI. PRGs are designed to provide targets for the selection and analysis of remedial alternatives. Therefore, PRGs will be derived and presented as necessary in the Corrective Measures Study (CMS).

4.1.5 Uncertainty

The sources of uncertainty in the baseline human health risk assessment and the relative influence of these sources on the risk assessment results are discussed below. Uncertainty is inherent in the selection of key input parameters and in every step of the risk assessment process. Risk assessment of waste sites must not be viewed as yielding single-value, invariant results. Rather, the results of risk assessment are estimates that span a range of possible values and that may be understood only in light of the assumptions and methods used in the evaluation.

4.1.5.1 Analytical Data

Uncertainty will always surround estimates of environmental concentrations at waste sites. Uncertainty in the analytical data is linked to the adequacy of the sampling program (sample design), collection procedures in the field, and accuracy of the sample analyses.

In order to address the adequacy of the sampling program, a statistical analysis was conducted before the Phase II field program began to determine the adequacy of the number of samples available and proposed at each SWMU. Of primary concern was the minimum number of samples required to support risk assessment. The analysis is part of the Data Collection Quality Assurance Plan (DCQAP) (SAIC 1995a) and the results were used to support the field sampling program.

Procedures relating to sample collection were established to reduce uncertainty surrounding sample results. For example, multiple rounds of groundwater samples were collected to account for variability due to factors such as seasonal fluctuation. Standard QA/QC measures (e.g., proper decontamination of equipment and collection of trip blanks, field blanks, field duplicates, and matrix spike/matrix spike duplicates [MS/MSDs]) were followed to reduce uncertainty associated with the analytical data.

Uncertainty also may be introduced at the laboratory. The laboratory follows a complicated set of procedures to reduce this uncertainty. For example, these procedures include the use of surrogate spikes to monitor chemical recovery, internal standards to monitor instrument sensitivity, and laboratory blanks to determine if laboratory preparation has introduced contamination to the sample. These issues are explained in detail in the data quality assessment (Appendix J).

4.1.5.2 Exposure Assessment

Different types of uncertainty have been identified regarding the exposure assessment:

- ***Scenario Uncertainty***—Missing or incomplete information needed to define the exposure scenario or pathway
- ***Model Uncertainty***—Inability to quantify all assumptions in model variables
- ***Parameter Uncertainty***—Inadequate information to quantify an exposure variable or parameter.

Scenario uncertainty arises when pathways were not included in, or were eliminated from, the assessment. The pathways that have been included in and excluded from the human health risk assessment and corresponding rationale are presented in Section 4.1.2.2. In accordance with Utah guidance (UDEQ 1999), a future residential scenario has been evaluated in the risk assessment. Residential development of DCD has been considered unlikely. The U.S. Army has no plans to sell any of the DCD property and, considering the mission of DCD, it is unlikely that the Depot will close in the foreseeable future. In addition, DCD is not zoned for residential development and some of the area is heavily industrialized. However, these assumptions are not definitive, because it is impossible to know what will happen in the future.

Assumptions about the future land uses are speculative. In attempting to predict future exposures, assumptions must be made concerning contaminant fate and transport, future site activities, and receptor behavior. In particular, it was assumed that contaminant concentrations will be the same in the future as at present and that the contaminants themselves are immobile and will not decompose.

Models have been used to project concentrations in produce and beef. A considerable amount of uncertainty is associated with exposure estimates for these indirect food chain pathways. Limited guidance is available from EPA addressing the food chain pathways. In particular, literature concerning the use of the models for beef ingestion in risk assessment is not prevalent. A high degree of uncertainty also is associated with the food chain transfer coefficients due to limited studies and a high degree of variability among the existing studies. In addition, the surface soils at DCD, due to the predominance of sand and loam, do not readily support the growth of food crops without the addition of amendments. For this reason, the indirect pathway (i.e., food chain) risks have been reported and presented separately from the direct pathway (i.e., soil and groundwater) risks.

Parameter uncertainty results partly because many of the exposure parameters (i.e., exposure factors) used in the risk assessment are default values recommended by EPA. These default parameters, which are generally conservative, do not necessarily reflect actual behavior and have been used in the absence of site-specific information.

Each exposure parameter is commonly treated as a single point estimate. However, none of these parameters is truly a single value. Instead, a range or distribution of values would more accurately represent exposures. Defining a range of values for any given parameter is a measure of variability or uncertainty in the risk assessment. Quantitative uncertainty analysis may be used to propagate the uncertainty/variability in each input parameter. This analysis is difficult to perform because of the quantity and quality of data available, as well as the major commitment of time and resources required. Although a quantitative uncertainty analysis was not conducted, this risk assessment examines two point estimates (i.e., CTE and RME). The uncertainty associated with the exposure assumptions used in the risk assessment most likely overestimates the actual risks.

4.1.5.3 Toxicity Data

Although EPA provides toxicity values that are point estimates, a significant amount of uncertainty may surround these point estimates. Identification of the sources of this uncertainty enables the risk assessor to establish the degree of confidence associated with the toxicity measures.

Uncertainty is inherent within the toxicity assessment and is primarily due to differences in study design, species, sex, routes of exposure, or dose-response relationships. A major source of uncertainty involves the use of toxicity values based on experimental studies that substantially differ from typical human exposure scenarios. The derivation of the toxicity values must take into account such differences as using dose-response information from animal studies to predict effects in humans, using dose-response information from high-dose studies to predict adverse

health effects from low doses, using data from short-term studies to predict long-term (chronic) effects, and extrapolating from specific populations to general populations.

The CSFs in particular are based on studies that may differ greatly from realistic situations. Experimental cancer bioassays typically expose animals to very high levels of chemicals (i.e., the maximum tolerated dose) for their entire lifetime. After the appropriate studies have been identified, the slope factor is calculated as the 95 percent UCL of the slope of the dose-response curve. This introduces conservatism into the risk assessment.

The derivation of RfDs generally involves the use of animal studies. Uncertainty factors ranging from 1 to 10,000 are incorporated into the RfD to provide an extra level of public health protection. The factors used depend on the type of study from which the value has been derived (e.g., animal or human, long-term or short-term). The scientific basis for this practice is somewhat uncertain. In general, high uncertainty factors are meant to bias the results conservatively so that exposures at the RfD level will not result in adverse health effects.

In addition, no adjustments have been made for the medium of exposure (e.g., when the medium of exposure at the site differs from the medium of exposure assumed by the toxicity value). There are many chemicals for which no toxicity value exists and for which little information is available. Therefore, a quantitative risk estimate cannot be calculated for these chemicals. For example, many chemicals are not evaluated for the inhalation pathway because of limited inhalation-based toxicological information. The lack of toxicity information for some chemicals may contribute to the underestimation of risks.

Toxicity values are not available for most of the PAHs. Only one carcinogenic PAH (benzo[a]pyrene) has a toxicity value for use in risk assessment. Benzo(a)pyrene is one of several PAHs that were detected. When evaluating oral exposure to PAHs, the approach used in the risk assessment was to relate the toxicity of PAHs to that of benzo(a)pyrene. The factors used to relate the toxicity are called relative potency values. This approach, although currently under review by EPA, is based on scientific studies, and is thought to be more realistic than the alternative method of assuming that all carcinogenic PAHs have a potency factor equal to that of benzo(a)pyrene.

Arsenic is a class A human carcinogen, which is the most certain carcinogen classification. The oral unit risk (and resulting CSF) was based on studies of human *dermal* cancers occurring in populations *ingesting* drinking water with high levels of arsenic. EPA recommends that risk managers recognize the uncertainties associated with the CSF for arsenic:

> ...in reaching risk management decisions in a specific situation, risk managers must recognize and consider the quantities and uncertainties of risk estimates. The uncertainties associated with ingested inorganic arsenic are such that estimates could be modified *downwards* as much as an order of magnitude, relative to risk estimates associated with most other carcinogens (EPA 1995b).

4.1.5.4 Multiple Chemical Exposures

Uncertainties in any phase of the risk analysis are reflected in the risk estimates. Some uncertainty is associated with the summation of risks and HQs for multiple chemical contaminants. As stated in RAGS (EPA 1989a), "The assumption of dose additivity ignores possible synergisms or antagonisms among chemicals, and assumes similarity in mechanisms of action and metabolism."

Cancer and noncancer risks are summed in the risk characterization process (separately for carcinogens and noncarcinogens) to estimate potential risks associated with the simultaneous exposure to multiple chemicals. In the case of carcinogens, this approach gives carcinogens with a Class B or Class C weight-of-evidence the same weight as carcinogens with a Class A weight-of-evidence. It also equally weights CSFs derived from animal data with those derived from human data. Uncertainties in the combined risks also are compounded because RfDs and CSFs do not have equal accuracy or levels of confidence and are not based on the same severity of effect.

4.2 SCREENING-LEVEL ECOLOGICAL RISK ASSESSMENT

A screening-level ecological risk assessment (SERA) was conducted as part of the Phase II Final Draft RFI for the Group 3 Suspected Releases SWMU at DCD (SAIC 1995c). The SERA estimated the risks to plants and animals in the environment in the absence of remediation or institutional controls at each SWMU. Since the Final Draft RFI Report was issued in 1995, additional soil samples were collected during Phases IIA and IIB to fill data gaps. These additional data have been incorporated into the existing data base, maintaining continuity with the established exposure units and data evaluation procedures, and presented in this Interim Final RFI. New Federal (EPA 1997c and 1998) and State of Utah (UDEQ 1999) ecological risk guidance were promulgated since the Final Draft RFI and are incorporated into this document. In addition, regulatory comments received from the Utah Department of Environmental Quality (UDEQ) on the Final Draft RFI Report are incorporated into the SERA. This section describes the general methodology used to conduct the SERA. SWMU-specific details and the results and conclusions of the SERA are provided in Sections 6 through 10.

An ecological risk assessment (ERA) defines the likelihood of harmful effects to plants and animals and their habitats as a result of exposure to chemical contaminants. A SERA for the five SWMU at DCD is required by the RCRA hazardous waste permit, which governs the installation, to evaluate the risk to plants, animals, and the environment from current and future exposure to contamination at the site.

Regulatory guidance for ERAs is contained in *Risk Assessment Guidance for Superfund, Volume II, Environmental Evaluation Manual* (EPA 1989d) and subsequent documents (EPA 1991c and 1992e. Further discussion on the scientific basis for assessing ecological effects and risk is presented in *Ecological Assessments of Hazardous Waste Sites: A Field and Laboratory Reference Document* (EPA 1989e). Other ERA guidance is provided in the *Framework for Ecological Risk Assessment* (EPA 1992f). In addition, applicable portions of the State of Utah Hazardous Waste Management Rules (UDEQ 1999) were used.

A second generation of guidance consists of the *Procedural Guidance for Ecological Risk Assessments at U.S. Army Sites* (Wentsel et al. 1994), and its replacement, the *Tri-Service Procedural Guidelines for Ecological Risk Assessments* (Wentsel et al. 1996). The newly published *Ecological Risk Assessment Guidance* (EPA 1997c) supercedes *RAGS, Volume II* (EPA 1989d). This latter guidance makes the distinction between the interrelated roles of SERAs and baseline ecological risk assessments (BERAs). SERAs use conservative assumptions for exposures and effects, while BERAs use more realistic (and generally less conservative) exposures and effects. Lastly, *Guidelines for Ecological Assessment* (EPA 1998) supercedes the *Framework for Ecological Risk Assessment* (EPA 1992f).

These documents do not provide a detailed step-by-step approach to ERAs. Instead, they discuss an overall approach to considering ecological effects and identify sources of information necessary to perform ERAs. Thus, professional knowledge and experience are important in ERAs to compensate for the lack of specific guidance and established methods.

SERAs are simplified risk assessments that can be conducted with limited data by assuming values for parameters for which data are lacking. Assumptions for exposure and toxicity values are biased toward overestimating risk in a SERA. The objectives of a SERA are to identify any sites that pose negligible risk and require no further action and identify sites and chemicals having the potential to cause risks. Conservative assumptions are important to ensure that sites are not dismissed from further evaluation in a baseline or full ERA when an unacceptable risk actually exists at a site. If the results of the SERA indicate that additional evaluation is necessary, the ERA may continue with a BERA. A facility-wide ERA is planned for DCD prior to closure of the facility.

This section presents the scope and objectives, procedural framework, and steps to complete the SERA for the DCD Group 3 SWMUs.

4.2.1 SERA Scope and Objectives

The scope of the SERA is to characterize the risk to terrestrial plant and animal populations at SWMU 11, 19, 20, 33, and 37. Unlike the human health risk assessment, which focuses on individuals, the SERA focuses on populations or groups of interbreeding ecological receptors. In the ERA process, individual receptors may be addressed if they are protected under the Endangered Species Act (ESA).

The objective of the SERA for SWMU 11, 19, 20, 33, and 37 is to assess the risk of harmful effects on ecological receptors from exposure to chemical contamination during current and future exposures. These chemical contaminants are called ecological chemicals of potential concern (ecoCOPCs). At each SWMU, the SERA examines the direct effects of ecoCOPCs on the ecological receptors. When it has been demonstrated that these ecoCOPCs cause risk, they are called ecological chemicals of concern (ecoCOCs) and receive further scrutiny.

To assess the potential for a contaminant to pose a hazard at the SWMUs, the contaminants were subjected to quantitative estimates of exposure to ecological receptors. This was done for the most important pathways involving surface and subsurface soil. The evaluation of these pathways is based on chemical analysis of surface and subsurface soil samples collected during the RFI.

4.2.2 SERA Procedural Framework

This section presents an overview of the procedures for conducting SERAs and describes how this SERA is organized. The following sections describe the SERA procedures.

4.2.2.1 EPA Framework

The ERA process consists of three inter-related phases: problem formulation, analysis (composed of exposure assessment and ecological effects assessment), and risk characterization (EPA 1992f). In conducting the SERA for the five SWMU at DCD, these three phases were completed by performing four inter-related steps. These steps are discussed below.

Problem Formulation—Problem formulation establishes the goals, breadth, and focus of the SERA and provides a preliminary characterization of chemical stressors (chemicals that restrict growth and reproduction or otherwise disturb the balance of ecological populations and systems) present in the various habitats at the site. The problem formulation step also includes a preliminary characterization of the ecological resources, especially the receptor species, in the ecosystem likely to be at risk. The procedures for determining ecoCOPCs are presented and the ecoCOPCs are selected in this step. Lastly, the selection of assessment and measurement endpoints is made as a basis for developing a CSM of stressors, ecological resources, and effects.

Exposure Assessment—In this step, exposures of receptor species to chemical stressors are evaluated. The exposure point concentrations are determined, as well as the exposure doses for food chain receptors (Section 4.2.4).

Effects Assessment—Once exposure is characterized, the ecological response to chemical stressors in terms of the selected assessment and measurement endpoints is defined. The effects assessment results in a profile of the ecological response of populations of plants and animals and habitats to the chemical concentrations or doses. Data from both field observations and controlled laboratory studies are used to assess ecological effects (Section 4.2.5).

Risk Characterization—Risk characterization integrates exposures of ecoCOPCs with effects to receptor species using HQs (ratios of exposure to effect). The resulting data are used to define the risk from contamination at each SWMU, relative to background (naturally occurring) risk, and to assess the potential for population and ecosystem recovery (Section 4.2.6).

4.2.2.2 Organization of the Screening-level Ecological Risk Assessment

The discussion of the SERA presented in this report is organized by the four inter-related steps of the EPA framework. Section 4.2.7 evaluates the degree of reliability of these methodological steps and the data used. The major findings for the SERA for each SWMU also are summarized by medium and receptor in Sections 6 through 10.

4.2.3 Problem Formulation

The first step of EPA's approach to the SERA process, problem formulation, includes:

- Determination of the scope and objectives of the assessment (Section 4.2.1)
- Ecological surveys and descriptions of habitats and populations (Section 4.2.3.1)
- Selection of exposure units and receptor species (Section 4.2.3.2)
- Data collection, summarization, and identification of the hazards (i.e., ecoCOPCs) (Section 4.2.3.3)
- Selection of assessment and measurement endpoints for the SERA (Section 4.2.3.4)
- Formulation of a CSM based on existing information and reasonable assumptions, including habitats and populations, and any threatened and endangered (T&E) species (Section 4.2.3.5).

4.2.3.1 Ecological Surveys and Descriptions of Habitats and Populations

The methods for ecological surveys, which include vegetation mapping for DCD by EBASCO (1994a) at all Group 3 SWMU and field reconnaissance by SAIC at SWMU 11, 19, and 20 are described in Section 3.2.10. Habitat maps and species lists were developed from these efforts. Field survey and reconnaissance data then were used to develop food chain and contaminant flow models to illustrate how contaminants move from the abiotic environment through organisms over time. Originally, this was conducted within the framework developed by EBASCO for the site-wide ERA.

Given that habitat types and sizes are known at DCD, it was necessary to extrapolate those findings to each SWMU. At both the site-wide and SWMU-specific levels, it was necessary to determine the suitability of that habitat to support populations of a given species. In order to do this, the following must be known: the quality of the habitat relative to the habitat's capacity to supply the organism needs and the quantity of that habitat necessary for a given species to occupy it. The quality of habitat for the five SWMU was evaluated during the brief field reconnaissance by a biologist knowledgeable of the species that are known to be present in the area. The quantity of habitat is measured as areal extent.

The quantity of site-specific habitat necessary for a given species was determined by measuring the size of the area of each SWMU and comparing the area of the habitat to the area of the home ranges of one or more representative species documented through a scientific literature search. In some cases, the home range at a SWMU was entirely within the SWMU; in others, the home range extended beyond the SWMU boundaries. These site-specific differences were taken into account in determining the overall suitability of habitat for a given species at each SWMU and also eventually for the site-wide ERA.

Approximate habitat sizes derived for each investigation area are noted below:

- SWMU 11 (640 acres)
- SWMU 19 (2.5 acres)
- SWMU 20 (3 acres)
- SWMU 33 (1.15 acres)
- SWMU 37 (14 acres).

The total depot acreage is 19,364 acres. The SWMU occupy only a small portion of the entire DCD.

4.2.3.2 Selection of Exposure Units and Receptor Species

From the ERA viewpoint, an exposure unit is the area where ecological receptors are likely to gather food, seek shelter, reproduce, and move around, and as a result of these activities, be potentially exposed to SWMU contaminants. Thus, each exposure unit ideally would be defined on the basis of existing habitat and land use, observed and assumed patterns of behavior of the receptors, and the spatial area of SWMU habitats relative to the home range and foraging areas of the receptors. However, the spatial boundaries of the ecological exposure units were the same as the units defined for the human health risk assessment. These exposure units are the five specific SWMU (or subsets of each SWMU) included in the Group 3 Phase II investigation. Although selection of exposure units was biased toward SWMU boundaries rather than habitat boundaries, the evaluation of risks that are consistent with SWMU boundaries facilitates decisionmaking on a SWMU-by-SWMU basis. For the SERA, the exposure units for the Group 3 SWMU are as follows:

- SWMU 11
- SWMU 19
- SWMU 20
- SWMU 33B (area immediately outside Building 536)
- SWMU 33C, drainage swale
- SWMU 37, pit floor
- SWMU 37, slope.

Despite its large size, SWMU 11 was not divided into multiple exposure units. The data show that areas of elevated concentrations are not apparent. The area inside Building 536 at SWMU 33 (SWMU 33A) was not selected as an exposure unit because of the inherent physical boundaries of the building. At SWMU 37, additional samples were collected from a sloped area to determine the source of an area of stressed vegetation (i.e., the existence of disposed of and buried bomb fragments). The new data were incorporated into a separate exposure unit designated as the "slope" because the types of contaminants detected and their concentrations differed significantly from the existing SWMU 37 data.

The ecological receptors for the SERA were selected from plant and animal species found in terrestrial habitats. Five criteria were used to select the ecological receptors:

- Presence at DCD as reported by EBASCO (1994a) and others, including SAIC
- Representation of major biological pathways and trophic groups (species that share similar feeding habitats) in the dominant terrestrial habitats at DCD
- Potential sensitivity to contaminants
- Availability of toxicity data
- Rare, threatened, or endangered status.

For the terrestrial SWMU habitats at DCD, the ecological receptors are vegetation, jackrabbits, and eagles. Risks are quantitatively estimated for each receptor. The receptors were selected to be compatible with the site-wide ERA.

4.2.3.3 Data Collection, Summarization, and Selection of Ecological Chemicals of Potential Concern

This section provides an overview of the approach used to interpret analytical data from samples collected for this RFI. This includes a brief discussion of the data used in the SERA and an examination of the steps used to select ecoCOPCs.

Groundwater

Groundwater will not be considered an exposure medium because ecological receptors are unlikely to contact groundwater at its depth (approximately 15 to 125 BLS).

Sediment

A sediment sample and a sludge sample were collected from the septic tanks at SWMU 19 and 20, respectively, during the Phase II field effort. The tank at SWMU 20 since has been removed and there are no current or any realistic future exposures to contaminants within the septic tank at SWMU 19. Therefore, these data were excluded from the SERA. At SWMU 37, data were collected from slag piles present at the time of sampling. These piles have been removed from the SWMU. Therefore, data from the slag piles also were excluded from the SERA.

Surface Water

The surface water data were eliminated from the SERA because the surface water no longer exists.

Surface Soil

Historical data (i.e., not collected by SAIC) used in the risk assessment include both SWMU and background data. Phase I data include soil samples collected at background

locations and SWMU 19 in 1990, 1992, and 1993. Media samples collected during the DCD investigations and used in the SERA are listed in Table M-1 in Appendix M.

Most of the data used to characterize the SWMU were collected from soil during Phase II field activities. The initial Phase II activities began in 1993 with field work conducted in 1994 and 1995. Phase II data from 1994-95 used in the risk assessment consist of soil samples collected at SWMU 11, 20, 33, and 37. Since the 1995 Final Draft RFI Report was issued, additional data were collected to fill data gaps. During follow-on Phase IIA activities conducted in 1998-99, additional soil samples were collected at SWMU 33 from areas inside and immediately outside Building 536. During follow-on Phase IIB activities conducted in 2000, soil samples were collected at SWMU 20 (around the septic tank), SWMU 33 (outside Building 536), SWMU 37, and 10 background locations.

Data used in the SERA have been validated in accordance with USAEC protocols (USAEC 1993 and 1995) or EPA CLP National Functional Guideline protocols (EPA 1994a and 1994b). Sections 3.3, 3.5, and 4.1.4.3 of the human health risk assessment methodology provide additional details on data validation.

The data were aggregated by the exposure units defined in the previous sections; soil data also were aggregated according to depth. The revised SERA focused on two depths: surface soil and subsurface soil. For the Group 3 SWMUs, the following designations were made:

- ***Surface Soil***—Collected from 0 to 0.5 feet BLS
- ***Subsurface Soil***—Collected from >0.5 to 15 feet BLS.

Although the 0- to 3-foot BLS interval was evaluated during the 1995 Final Draft RFI, the collection of additional data suggested that these two intervals in the current SERA would be more appropriate. Ecological exposures are typically greater close to the surface. Thus, the evaluation of the 0.5- to 15-foot interval is associated with greater uncertainty.

Once the sampling data of the chemicals detected were grouped and summarized, ecoCOPCs were selected for further evaluation. EcoCOPCs are those substances detected at each SWMU that have the potential to pose a hazard to plants and animals. To streamline the SERA, chemical concentrations were screened against ecological screening values and background concentrations. This approach eliminates chemicals whose concentrations are either below ecotoxicity levels of concern or within background levels. The Interim Final RFI SERA did not use physical and chemical properties to select ecoCOPCs, as was done in the Final Draft RFI SERA (SAIC 1995c).

Chemicals were selected as ecoCOPCs if the maximum detected concentration in soil (surface or subsurface) was above the EPA Region V ecological data quality levels (EDQLs) for surface soil (EPA 1999c) and the site concentration was determined to be above background by ANOVA. In instances where only background concentrations were available, the chemical was selected as an ecoCOPC if the site concentration was determined to be above background by ANOVA. If neither a screening value nor background concentration was available, the chemical

was selected as an ecoCOPC by conservative default and evaluated further. Figure 4-2 presents a flowchart of the ecoCOPC selection process. These screening scenario outcomes are summarized below:

- Chemical Selected as an EcoCOPC
  - Chemical is above the EDQL and background (based on ANOVA)
  - Only one value (EDQL or background) is available and chemical is above that value
  - No EDQL or background concentration
- Chemical Eliminated as an EcoCOPC
  - Chemical is below EDQL and eliminated regardless of ANOVA results
  - Chemical is above EDQL, but is eliminated from further consideration because it is within background based on ANOVA
  - No EDQL is available, but chemical is within background based on ANOVA.

EDQLs were developed by EPA Region V for use in the RCRA Corrective Action Program and are initial screening levels against which site concentrations are compared (EPA 1999d). Comparison against these values serves to focus the SERA on those areas and chemicals most likely to pose unacceptable risks to the environment. The surface soil EDQLs are based on the most conservative NOAEL for plants, earthworms, meadow voles (*Microtus pennsylvanicus*), and masked shrews (*Sorex cinerus*). These soil EDQLs were developed by EPA Region V in the following manner. First, the following sources for soil quality criteria were identified and evaluated:

- Netherlands Ministry of Housing, Planning, and the Environment soil quality criteria
- Quebec Ministry of Environment and Wildlife soil quality criteria
- EPA ambient level multimedia goals for soil
- Great Britain Department of the Environment soil quality guidelines
- California Department of Health Services soil quality guidelines
- Oregon Department of Environmental Quality industrial soil cleanup levels.

According to EPA, this evaluation revealed significant discrepancies between the criteria related to methods of development, applicability to ecological receptors, toxicological endpoints, criteria values, and the availability of soil quality criteria documentation. Due to these inconsistencies, default soil EDQLs are based entirely on receptor-specific values that use adjusted toxicity reference values (TRVs). The adjusted TRV is the most relevant and available toxicological result modified with uncertainty factors (UFs) as appropriate to be equivalent to a chronic NOAEL for the selected receptor. TRVs were adjusted by EPA using UFs for scaling factor (test species to target species), endpoint (test endpoint to NOAEL), and duration (test duration to chronic exposure).

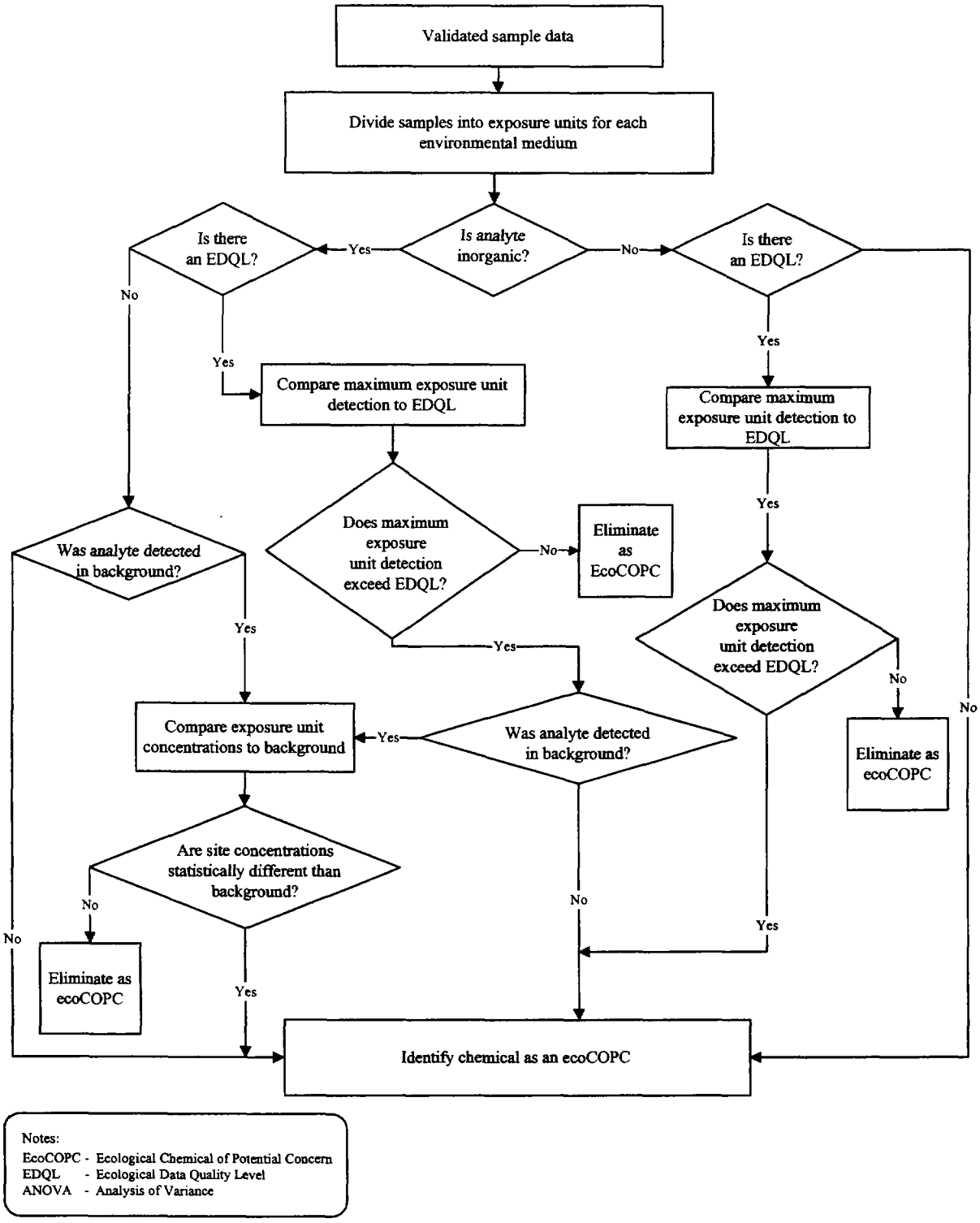

Figure 4-2. EcoCOPC Selection Process for the Screening-level Ecological Risk Assessment Deseret Chemical Depot, Tooele, Utah

EDQLs for plant and invertebrate receptors were developed based on a review of existing toxicological information. Plant-specific soil EDQLs also were calculated for some chemicals based on a review of toxicity threshold and background concentrations. Receptor-specific EDQLs for mammalian receptors were developed with a three-step ingestion and accumulation model. For example, the meadow vole represents mammalian herbivores and the masked shrew represents mammalian carnivores. The EDQLs for these two receptors were calculated by EPA based on the following equation:

$$C_{soil} = \frac{BW \times TRV_a}{[IR_{soil} + (IR_{plant/prey} \times BCF_{plant/prey})]} \quad (5)$$

where:

C_{soil} = Chemical- and receptor-specific EDQL in soil (mg/kg)

BW = Receptor-specific body weight (kg)

TRV_a = Adjusted toxicity reference value (mg/kg/day)

IR_{soil} = Receptor-specific soil ingestion rate (kg/day)

$IR_{plant/prey}$ = Receptor-specific plant/prey ingestion rate (kg/day)

BCF = Soil-to-plant or soil-to-prey bioconcentration factor (kg soil/kg plant or kg soil/kg invertebrate).

In addition to the Region V EDQLs, background concentrations for surface soil were used to select ecoCOPCs. Background surface and subsurface soil data were used to determine if detected concentrations of inorganic chemicals at the SWMU are representative of naturally occurring background levels. For the SERA, the background comparison consisted primarily of ANOVA methods to identify site-related inorganic contamination, as specified in EPA Region VIII guidance (EPA 1994c) and consistent with the methodology used in the 1995 Final Draft RFI. The background comparison was conducted using different methods depending on the percentage of detected values in the site and background data sets. Section 4.1.4.2 presents the different background comparison methods. Although many of the organic compounds detected at DCD are ubiquitous due to anthropogenic activities, all detected organic compounds were considered site-related and selected as ecoCOPCs if they were detected above the EDQL or did not have an EDQL; background comparisons were not conducted.

The background data set for soil is composed of samples that were collected from 1.5 to 3 feet BLS during previous investigations and additional samples from the surface to 10 feet BLS that were collected as part of the Phase IIB activities. The background surface and subsurface soil data set is composed of 20 sample locations with multiple sample depths as discussed in Section 4.1.4.3; the sample locations are presented in Section 5. The background data from several locations have been aggregated into a single exposure unit with samples from all depths combined. At each SWMU, the background comparison for soil was conducted separately for each depth horizon:

- Site surface soil (i.e., SWMU data from 0 to 0.5 feet BLS) was compared to background soil (>0 to 10 feet BLS)

- Site subsurface soil (i.e., SWMU data from >0.5 to 15 feet BLS) was compared to background soil (>0 to 10 feet BLS).

Therefore, two lists of soil ecoCOPCs were derived in the SERA (one for the 0- to 0.5-foot soil interval and one for the >0.5- to 15-foot soil interval).

Section 5 presents the samples included in the background data set. Acenaphthene, aluminum, antimony, barium, cadmium, chromium, cobalt, copper, dibenzofuran, iron, lead, magnesium, manganese, mercury, nickel, silver, thallium, zinc, and 2,4,6-trinitrotoluene (TNT) were selected as ecoCOPCs in surface soil. 4-Chloroaniline, antimony, arsenic, barium, benzo(a)anthracene, benzo(a)pyrene, beryllium, bis(2-ethylhexyl)phthalate (B2EHP), butyl benzyl phthalate, cadmium, calcium, chrysene, copper, di-n-butyl phthalate (DNBP), iron, lead, magnesium, manganese, mercury, naphthalene, nickel, silver, and zinc were selected as ecoCOPCs in subsurface soil.

All chemicals not eliminated by the above screening steps were identified as ecoCOPCs. The selection of a chemical as an ecoCOPC does not necessarily imply that a risk to ecological receptors exists; rather, the selection of a chemical indicates there is a need to evaluate that chemical further in the SERA to determine if exposures result in potential risk to receptors. The results of the ecoCOPC selection process are presented by SWMU in Sections 6 through 10.

4.2.3.4 Selection of Ecological Assessment and Measurement Endpoints

Protecting ecological resources, such as the species of plants and animals and habitats described in Sections 2.7 and 2.8, is the principal motivation for conducting ERAs. Key aspects of ecological protection are presented as policy goals (i.e., general goals established by legislation or agency policy) based on societal concern for protection of certain environmental resources. For example, environmental protection is mandated by a variety of legislation and Government agency policies (e.g., RCRA and the National Environmental Policy Act [NEPA]). To determine if this protection goal has been achieved, assessment and measurement endpoints were formulated to define the specific ecological values to be protected.

Assessment endpoints are statements specifying the desired ecological attribute to be protected in the environment (Suter 1993). Desirable attributes of the environment can be ecosystem functions, such as production and decomposition, or properties, such as biodiversity. Valued components of the environment can be organisms and trophic groups with symbolic, commercial, recreational, or ecological importance. Assessment endpoints often are not directly measurable. As a result, one or more measurement endpoints are chosen to determine site-specific impacts on the assessment endpoints.

Decision rules are specified for the assessment endpoints. Table 4-4 shows the decision rules that describe the logical basis for choosing from among alternative actions for the assessment endpoint. In some cases, toxicity tests or biological surveys provide direct evidence of the level of adverse effects. In cases where surveys were not conducted, receptor endpoints at DCD are assumed to exhibit the same level of adverse effects when a receptor is exposed to a

Table 4-4. Policy Goals, Ecological Assessment Endpoints, Measurement Endpoints, and Decision Rules Deseret Chemical Depot, Tooele, Utah

Policy Goals	Assessment Endpoint	Measurement Endpoint	Decision Rule
Policy Goal 1: The maintenance and protection of terrestrial populations and ecosystems.	**Assessment Endpoint 1:** Maintenance of plant community. **Endpoint Species:** Plants of various species.	**Measurement Endpoint 1:** Measured soil chemical concentrations.	**Decision Rule for Assessment Endpoint 2:** If the HQ is <1, the chemical alone is unlikely to cause adverse ecological effects, and therefore, the plant populations and communities are maintained. If the HQ >1, a weight-of-evidence evaluation will be conducted to determine the potential for ecological risk and the need for any additional measurements or calculations.
	Assessment Endpoint 2: Maintenance of populations of herbivorous animals. **Endpoint Species:** Black-tailed jackrabbits.	**Measurement Endpoint 2:** Modeled chemical concentrations in food chain based on measured soil chemical concentrations.	**Decision Rule for Assessment Endpoint 4:** If the HQ is <1, the chemical alone is unlikely to cause adverse ecological effects, and therefore, populations of the herbivores (e.g., black-tailed jackrabbits) are maintained. If the HQ >1, a weight-of-evidence evaluation will be conducted to determine the potential for ecological risk and the need for any additional measurements or calculations.
	Assessment Endpoint 3: Maintenance of terrestrial predators. **Endpoint Species:** Golden eagle.	**Measurement Endpoint 3:** Modeled chemical concentrations in prey (rabbits) based on measured soil chemical concentrations.	**Decision Rule for Assessment Endpoint 6:** If the HQ is <1, the chemical alone is unlikely to cause adverse ecological effects, and therefore, populations of terrestrial predators are maintained. If the HQ >1, a weight-of-evidence evaluation will be conducted to determine the potential for ecological risk and the need for any additional measurements or calculations.

given concentration of an ecoCOPC as that experienced by test species in published toxicity studies. Together, the assessment endpoint, measurement endpoint, and decision rule represent a mechanism for deciding whether the protection goal is being met.

The decision rules for the DCD SERA are stated quantitatively in terms of HQs. An HQ is the ratio of the measured or predicted concentration or dose of an ecoCOPC to which receptors are exposed in an environmental medium to the measured concentration or dose of an ecoCOPC that adversely affects an organism (benchmark or toxicity threshold). If the measured concentration or dose is less than the concentration producing an adverse effect (i.e., the ratio of the two [or the HQ] is less than 1), the risk is considered acceptable (protective of the ecological receptor). Any risk quotient at or above 1 indicates that the ecoCOPC should be investigated further. The policy goal and endpoints for the DCD SERA are for no harmful effects from soil contaminants to terrestrial plant and animal species.

4.2.3.5 Formulation of a Conceptual Site Model

The CSM of DCD was developed by EBASCO (1994a) for the SERA using the available site-specific information and professional judgment. The chemical sources, exposure media, exposure routes, and potential receptors are described below. A working version of the CSM, which illustrates the pathways by which ecological receptors are exposed to hazards at each SWMU, is described in this section and shown in each qualifying SWMU section.

Chemical Source—Chemical sources at the SWMU include surface soil at the terrestrial exposure units.

Exposure Media—Sufficient time (i.e., more than 10 years) has elapsed for the chemicals in original sources to have migrated to potential exposure media, resulting in possible exposure of plants and animals that come into contact with these media.

Surface water is limited to temporary puddles after occasional storms. There are no streams, and therefore, no sediment in or near the five SWMUs. Groundwater is not considered an exposure medium because ecological receptors are unlikely to contact groundwater at its depth (15 to 125 feet BLS). Air is not considered an exposure medium because potential VOCs are believed to have dissipated. Surface soil and biota were retained as the exposure media for this SERA. For this analysis, only soil data are available.

Exposure Routes—A principal exposure route is contact of biota with soils at the SWMUs. Animals also are exposed through ingestion of contaminated prey species and vegetation. Animals potentially may come into contact with soil at the SWMU by means of incidental ingestion, dermal contact, and inhalation of dust. Plants potentially are exposed by dermal contact (root uptake) from soil at the SWMUs.

Ingestion of soil and biota by animals are the only two potential exposure routes evaluated quantitatively for animals at the terrestrial SWMU in the SERA. The exposure of animals to contaminants in soil by dermal contact and inhalation is likely to be a small fraction of the direct exposure to contaminants in soil by incidental ingestion and the indirect exposure by ingestion of contaminated biota. Furthermore, the available toxicity data are almost exclusively for the ingestion pathway (e.g., Sample et al. 1996). External radiation is not a concern at DCD.

The exposure pathways that are the primary source of risk for ecological receptors are direct contact of plants with soil, incidental ingestion of soil by animals, and ingestion of terrestrial plant and animal matter by animals at the exposure units. The exposure pathways are evaluated quantitatively using site measurements, published exposure parameters, and toxicity data.

Receptors—Terrestrial receptors are recognized in the CSM, and are discussed in Section 4.2.4.2.

4.2.4 Exposure Assessment

Step 2 of EPA's four-step ERA process, as it applies to the SERA for DCD, is discussed in this section. The exposure assessment describes the receptor species and exposure media. The exposure point concentrations are determined as well as the exposure doses for food chain receptors.

Exposure assessment includes: (1) quantification of the release, migration, and fate of contaminants; (2) characterization of ecological receptors being exposed; and (3) quantification of concentrations at the point of contact with the exposed organisms. The release, migration, and fate of contaminants determine the concentrations of ecoCOPCs in the exposure media. The concentrations in exposure media are measured or estimated. For example, the concentrations in soil are measured. The concentrations in prey are estimated from these measured soil concentrations by using bioaccumulation factors.

Each receptor is characterized by different routes of exposure. These differences are captured in exposure parameters, which are used to adjust the measured concentration of ecoCOPCs in the exposure media as the chemical moves along its pathway to the receptor. Sufficient data exist to identify the source media leading to potential exposure of ecological receptors, quantify the concentration of ecoCOPCs in exposure media, and derive exposure parameters for a few of the ecological receptors.

4.2.4.1 Receptor Species and Their Exposure

Exposure is defined as contact between a receptor species and ecoCOPCs in an environmental medium. For exposure to occur, a chemical release must occur to an environmental medium with which a receptor species must have contact. The SERA evaluates the potential exposures of receptor species to ecoCOPCs in surface soil, and plants and animals ingested by other receptors. The primary receptor species' categories are subcategorized by exposure classes. Exposure classes group together species with similar feeding habits and physiologies. Each exposure class for DCD has one or more receptor species.

Terrestrial Exposure Classes and Receptor Species—The terrestrial exposure classes and associated receptor species for the DCD investigation areas are as follows:

- Vegetation (primary producers)
  - Grasses and forbs (broad-leaved plants)
  - Shrubs
- Mammalian herbivores (primary consumers)
  - Jackrabbits
- Bird predators (raptors)
  - Golden eagles.

These receptors or their ecological equivalents are likely to be present at DCD and were selected in accordance with EPA guidance (EPA 1992f, 1997c, and 1998). All are ecologically and socially relevant because they represent major receptor groups potentially exposed to

ecoCOPCs. These receptors have been defined in terms of their relationship to policy goals indicated in Section 4.2.4.4. The receptor species and major exposure routes are identified on the food web shown in Figure 4-3 and Table 4-5.

Each receptor species listed is linked directly to one of the assessment endpoints and provides an explicit expression of the environmental value to be protected. For example, vegetation is listed because the terrestrial plant community is ecologically important (vegetation forms the basis of the food web; herbivores eat plants and carnivores, in turn, eat herbivores), susceptible to ecoCOPCs in soil, and highly exposed to soil at the site. In addition, toxicity information for risk analyses is available.

Exposure pathways were chosen to provide the highest potential exposures to receptors under a variety of conditions. For example, eagles represent the top of the food web where exposures from bioaccumulated materials can be maximal.

Plants—Shrubs and grasses are exposed to soil chemicals primarily by direct contact with soil. These plants are assumed to be exposed to all measured soil concentrations (no exposure parameters need to be accounted for).

Mammalian Herbivores—Mid-sized herbivores (e.g., jackrabbits) are exposed primarily to soil chemicals that are in plant material. Exposure of jackrabbits to these chemicals by direct soil contact is assumed to be limited. For this class of receptor species, exposure is the sum of absorption from the soil and ingestion from plants. The estimated exposure for this class does not include exposure by direct contact or inhalation. Limited data are available on inhalation toxicity or toxicity by direct contact with contaminated soil (or the parameters required to model chemical absorption). Dermal exposure is expected to be minor and skin-associated soil that is ingested is included in the estimated daily soil ingestion rate. Exposure by direct contact with soil is assumed to be negligible for small mammals and birds at the DCD exposure units. The chemical contributions from these pathways are assumed to be less than from oral ingestion. As a result, exposure by direct contact and inhalation was not evaluated. Instead, conservative exposure values for soil ingestion and dietary composition are used for these herbivores.

Terrestrial Top Predators—Top predators are exposed to ecoCOPCs that have accumulated in their prey. Terrestrial top predators (e.g., golden eagles) feed on terrestrial prey. Some terrestrial predators also may consume soil incidentally; eagles do not. Eagles are assumed to forage over an area that is larger than the area of the DCD exposure units, so their realistic ecological exposure should include the fraction of their diet that comes from the investigation area, which is estimated as a function of the size of the home range relative to the sizes of the exposure units. However, in the SERA, each receptor species was assumed to forage only within each exposure unit. This assumption is overly conservative for wide-ranging species such as eagles, but is consistent with EPA (1997c) guidance.

The 95 percent UCL was used as the exposure point concentration, unless it was greater than the maximum detected concentration, in which case the maximum value was used. This is not entirely consistent with the Utah Hazardous Waste Rules (UDEQ 1999), which discuss only the 95 percent UCL of the mean. When the sampling distribution is not well-characterized (e.g., as is

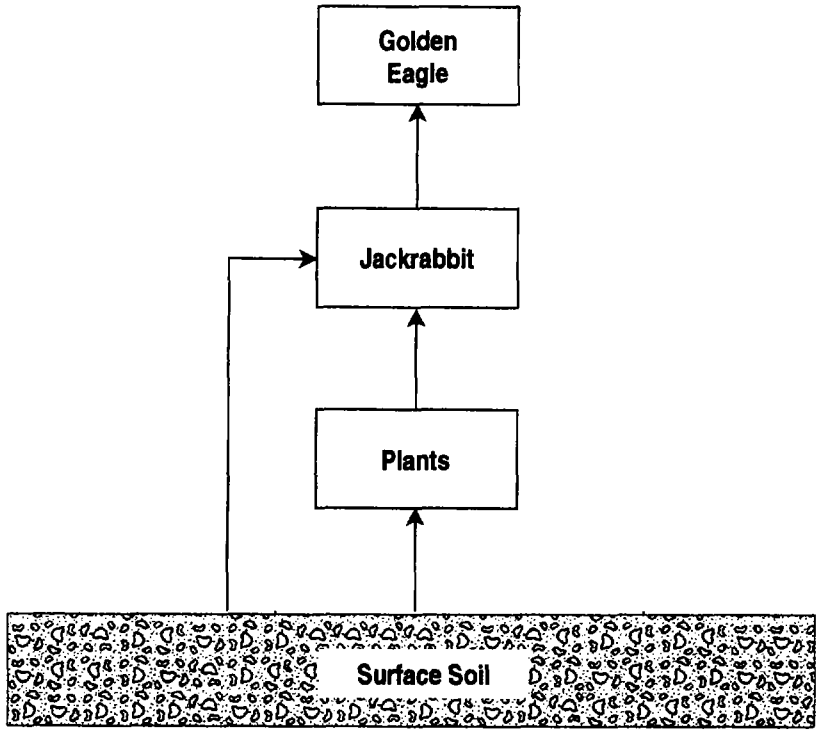

Figure 4-3. Food Web for DCD Screening-level Ecological Risk Assessment Deseret Chemical Depot, Tooele, Utah

Table 4-5. Exposure Pathways for the Screening-level Ecological Risk Assessment Deseret Chemical Depot, Tooele, Utah

SWMU	Receptor	Soil			
		Direct Contact	Ingestion	Ingestion of Plant	Ingestion of Animals
SWMU 11	Vegetation	•	—	—	—
Chemical Munitions	Jackrabbit	—	•	•	—
Storage Area	Golden Eagle	—	—	—	•
SWMU 19	Vegetation	•	—	—	—
Building 533 Foundation	Jackrabbit	—	•	•	—
(Empty Drum Storage Area)	Golden Eagle	—	—	—	•
SWMU 20	Vegetation	•	—	—	—
Building 521	Jackrabbit	—	•	•	—
(Crating Facility and Septic Tank Discharge Field)	Golden Eagle	—	—	—	•
SWMU 33	Vegetation	•	—	—	—
Building 536	Jackrabbit	—	•	•	—
(CAMDS Salt Storage)	Golden Eagle	—	—	—	•
SWMU 37	Vegetation	•	—	—	—
Slag Piles and	Jackrabbit	—	•	•	—
Bomb Fragments	Golden Eagle	—	—	—	•

• Pathway evaluated quantitatively.
— Pathway not evaluated.

more likely with small data sets), it is possible for the 95 percent UCL to exceed the maximum. In such cases, as written in EPA federal guidance for risk assessment, the maximum detected concentration may be chosen as the exposure point concentration. The danger in this approach is that the true mean is actually higher than the maximum detected value, especially if the more contaminated area of the exposure unit was not sampled. However, as the sampling strategy was biased toward areas of contamination, SAIC believes that the maximum detected concentration is a more accurate estimate of the exposure point concentration and that the use of the 95 percent UCL would overestimate risks to humans and ecological receptors. Sections 6 through 10 present details concerning the biased nature of the sampling design. The surface soil exposure point concentrations are presented in the SWMU-specific Sections 6 through 10. The exposures of terrestrial wildlife to ecoCOPCs were estimated from the measured concentrations in surface soil and adjusted by exposure parameters, as described below.

4.2.4.2 Quantification of Exposure for Terrestrial Wildlife

In support of exposure characterization, the estimation of exposure doses for terrestrial wildlife was completed. Consistent with EPA (1997c) guidance, conservative assumptions were used in the calculations of exposure doses in the absence of site-specific data. The assumptions tend to maximize exposure for each receptor. For instance, it was conservatively assumed that the temporal use factor (TUF) would be 1, indicating that the receptor species spends 12 months a year at an exposure unit and does not migrate. However, the area use factor (AUF) used in this assessment also is SWMU-specific, reflecting an estimate of time the receptor spends in an exposure unit. The AUFs for black-tailed jackrabbit were 1, 0.1, 0.1, 0.1, and 1, respectively, for SWMU 11, 19, 20, 33, and 37. The AUFs for golden eagle were 0.1, 0.01, 0.01, 0.01, and 0.1, respectively, for SWMU 11, 19, 20, 33, and 37. The absorption efficiencies of the chemical in ingested soil and tissue that is absorbed is assumed to be 10 and 100 percent, respectively. It is important to note, especially for exposures through the food chain, that all ecoCOPCs are assumed not to transform or degrade during the period of exposure (i.e., the concentration in the medium of concern remains the same). Receptor species-specific exposure parameters are presented in Tables 4-6 and 4-7 for black-tailed jackrabbit and golden eagle, respectively.

The dose equations are derived based on equations presented in the *Wildlife Exposure Factor Handbook* (EPA 1993c). The general exposure equations are presented below:

$$Dose_{total} = Dose_{food} + Dose_{soil} \quad (6)$$

where:

$Dose_{total}$ = Sum of all doses (mg/kg-day)

$Dose_{food}$ = Average daily dose ingested from food (mg/kg-day)

$Dose_{soil}$ = Average daily dose ingested from surface soil (mg/kg-day).

The component doses are derived using the following equations:

$$Dose_{food} = IR_{food} \times C_{food} \times AUF \times TUF \times AE \quad (7)$$

Table 4-6. Exposure Parameters for Black-tailed Jackrabbit Deseret Chemical Depot, Tooele, Utah

Parameter	Definition	Receptor: Black-tailed Jackrabbit (*Lepus californicus*)	
		Value	Reference/Notes
BW	Body weight (kg)	2.2	Average taken from 1.3 to 3.1 kg, grassland and open areas of the Western United States (EPA 1993c)
HR	Home range (ac)	40	Value for black-tailed jackrabbit (French et al. 1965)
TUF	Temporal use factor	1	Assumed to be 1
AUF	Area use factor	0.1, 1	Site-specific; see text for further details
IR_F	Food ingestion rate (g/g-d=kg/kgBW/d)*	0.205	Estimated by dividing free-living metabolic rate (203 kcal/kgBW/d) for Eastern cottontail by the product of the energy composition of young grasses (1.3 kcal/g wet wt.) and assimilation efficiency (0.76) per EPA (1993c)
PF	Plant fraction of diet	1	Based on data for Eastern cottontail (EPA 1993c); assumed to be vegetative parts
AF	Animal fraction of diet	0	Not reported in EPA (1993c) for Eastern cottontail; assumed to be negligible
SF	Soil fraction of diet	0.063	Estimated percent soil in diet (dry weight) for the jackrabbit (EPA 1993c)

* Food ingestion rate (g/g-d) re-expressed as kg/kgBW/d is assumed not to include ingested soil; therefore, PF+AF =1

Table 4-7. Exposure Parameters for Golden Eagle Deseret Chemical Depot, Tooele, Utah

Parameter	Definition	Receptor: Golden Eagle (*Aquila chrysaetos*)	
		Value	Reference/Notes
BW	Body weight (kg)	3.75	Arithmetic mean, adult, both sexes, Florida (EPA 1993c)
HR	Home range (ac)	8,600	Territory area for bald eagles, Arizona/desert, riparian river (EPA 1993c)
TUF	Temporal use factor	1	Assumed to be 1
AUF	Area use factor	0.01,0.1	Site-specific; see text for further details
IR_F	Food ingestion rate (g/g-d=kg/kgBW/d)*	0.12	Adult bald eagles, both sexes, Washington (free flying) (EPA 1993c)
PF	Plant fraction of diet	0	Not reported in EPA (1993c); assumed to be negligible
AF	Animal fraction of diet	1	Small mammals, snakes, birds, and carrion (EPA 1993c)
SF	Soil fraction of diet	0	Not reported in EPA (1993c); assumed to be negligible

$$Dose_{soil} = IR_{soil} \times C_{soil} \times AUF \times TUF \times AE \quad (8)$$

where:

Dose = Amount of chemical ingested per day (mg/kg-day)

IR = Ingestion rate (kg/kg-day)

C = Estimated concentration (mg/kg)

AUF = Area use factor (unitless)

TUF = Temporal use factor (unitless)

AE = Absorption efficiency of ecoCOPC in medium that is absorbed by the receptor (unitless).

Additional details on the derivation of terrestrial wildlife doses are presented in the following paragraphs.

Terrestrial Herbivores

Black-tailed jackrabbits are herbivores that feed primarily on vegetation and are considered to ingest negligible amounts of animal matter. The estimates of soil ingestion for black-tailed jackrabbits are based on reported fractions of incidental soil ingestion for those or related animals (EPA 1993c) and are presented in Table 4-6. The following equations were used to calculate the dose of the ecoCOPCs that the rabbit could be exposed to from ingestion of vegetation and surface soil:

$$Dose_{total} = Dose_{veg} + Dose_{soil} \quad (9)$$

The equation for soil ingestion is the same as equation 8. The equation for plant ingestion is presented below:

$$Dose_{veg} = IR_{veg} \times C_{veg} \times AUF \times TUF \times AE \quad (10)$$

where:

$Dose_{veg}$ = Amount of chemical ingested per day via the ingestion of vegetation (mg/kg-day)

IR_{veg} = Ingestion rate of plant matter (kg/kg-day)

C_{veg} = Chemical concentration in plant matter (mg/kg).

No data were available from the direct measurement of ecoCOPC concentrations in terrestrial plant tissue; therefore, soil-to-plant (SP) values were used to estimate these concentrations. The SPs are reported in Baes et al. (1984) and *Risk Assessment Methodology for Loring Airforce Base* (HAZWRAP 1994). These values are provided in Table M-2 of Appendix M. The default SP is 0.5 for all contaminants. The default SP is 0.5 for all

contaminants. The equation used to estimate the chemical concentration in vegetation is presented below:

$$C_{veg} = C_{soil} \times SP_v \quad (11)$$

where:

C_{veg} = Vegetation exposure point concentration (mg/kg)
C_{soil} = Soil exposure point concentration (mg/kg)
SP_v = Soil-to-plant transfer factor in vegetative part of the plant (unitless).

4.2.4.3 Terrestrial Top Predators

The diet for the golden eagle is composed primarily of small mammals (Table 4-7). For this assessment, several assumptions were made to cover receptors from each feeding strategy. The golden eagle diet assumes 100 percent ingestion of black-tailed jackrabbit, which provides the most conservative scenario for the top avian predator feeding on small herbivorous mammals. The fraction of soil ingested for these avian receptors is assumed to be negligible and not a major pathway of exposure.

Because no data were available from direct measurement of the concentration of ecoCOPCs in rabbit tissue, animal-to-animal bioaccumulation factors (BAF_v) for vertebrates were used to estimate these concentrations. The following equations were used to calculate the dose of the ecoCOPCs that the eagle could be exposed to from ingestion of small mammals. The general equation for the eagle is:

$$Dose_{total} = Dose_{animal} \quad (12)$$

The eagle animal dose is defined further as:

$$Dose_{animal} = Dose_{total\ rabbit} / (IR_{total\ rabbit}) \times IR_{eagle} \times BAF_v \times AUF \times TUF \times AE \quad (13)$$

where:

$IR_{total\ rabbit}$ = Ingestion rate of prey item (kg/kg-day)
IR = Ingestion rate of predator (kg/kg-day)
BAF_v = Bioaccumulation factors for vertebrates (kg_{soil}/kg_{tissue}).

The chemical-specific values for bioaccumulation for animal-to-animal (BAF_v) are provided in Table M-1 of Appendix M based on values compiled in *Risk Assessment Methodology for Loring Air Force Base* (HAZWRAP 1994). Both empirical and estimated values are documented. Default BAFs for ecoCOPCs without published BAF values are 0.1 for metals and 10 for organics, based on the range of values reported for these two types of contaminants (HAZWRAP 1994).

4.2.5 Effects Assessment

The purpose of the effects assessment is to evaluate the response to chemical stressors at the exposure units in terms of the selected assessment and measurement endpoints and select toxicity values for use in risk characterization. Depending on the exposure parameters, this effects assessment results in a profile of the response of receptor populations to stressors at concentrations or doses to which they are exposed.

4.2.5.1 Chemical Toxicity

Chemicals in the ecosystem may be directly toxic to plants and animals or indirectly harmful by reducing an organism's ability to survive and reproduce. These disparate effects are characterized by different dose response relationships and may result from different exposure pathways. The toxicity thresholds used for animals in the DCD SERA are based on toxic effects observed in laboratory studies.

Chronic (long-term) toxicity resulting from chemical exposure is the primary concern at the DCD SWMUs. Most organisms do not ingest large amounts of soil, and assuming that the soil is not acutely toxic, these organisms are unlikely to be affected in the short-term. Plants accumulate higher-than-background levels of some metals, resulting in chronic toxicity. Bioaccumulation is generally most significant in the roots of plants; however, several metals can be translocated to parts of the plants above the ground. Some metals (e.g., selenium) accumulate in animal tissues and can have subtle deleterious effects on animals over long exposure times. Many organic contaminants (e.g., polychlorinated biphenyls [PCBs] and pesticides) are extremely lipophilic and can biomagnify in organisms. No investigations into chronic effects on local plants and animals as a result of exposure to soils or plants and animals have been conducted at DCD.

Toxicity of soil contaminants varies, depending on the receptor species and the attending physical and chemical factors, the presence of complexing agents, or interaction with other chemicals at the site. Plants can be adversely affected by soil contaminants in numerous ways, including seed production, seed germination, growth rate, and plant biomass. Animals can be adversely effected in terms of behavioral and physiological changes, including reproductive impairment.

4.2.5.2 Toxicity Thresholds

Site-specific toxicological studies to determine whether the concentrations of ecoCOPCs at the site are toxic have not been completed for DCD populations. Therefore, this effects assessment uses toxicity data from the scientific literature. Published toxicity data on test concentrations, modes of exposure, and effects on similar species were used to establish toxicity thresholds. Toxicity values selected for the evaluation of the potential for adverse effects are referred to as TRVs and represent concentrations or doses of the ecoCOPCs that are protective of the receptor species being evaluated. The derivation of TRVs for receptor species is discussed below. TRVs then are compared to calculated exposure concentrations or doses in the risk characterization section to evaluate the potential for adverse effects for each ecoCOPC. Toxicity thresholds, including receptors, endpoints, and safety factors, are presented in Appendix M.

Surface Soil

Terrestrial Vegetation—TRVs reported by Efroymson et al. (1997) to be protective of terrestrial plants were used (Table M-3). TRVs were established at a level associated with a 20 percent reduction in plant growth or yield, which is consistent with other screening level benchmarks for ERA. Data for organic chemicals are not readily available for terrestrial plants. As a result, the plant TRV for acenaphthene was used for all PAHs. Efroymson et al. (1997) indicate that toxicological benchmarks that are exceeded by background levels may be a poor measure of the plant community. Inorganic chemicals detected in background locations at concentrations above benchmarks may be occurring at naturally elevated levels or are representative of widespread contamination. In the former instance, plants may have adapted to these higher levels.

Terrestrial Wildlife—TRVs for wildlife receptors represent doses of the ecoCOPCs that are protective of wildlife receptors. Chemicals identified as having the potential to adversely affect terrestrial species were evaluated by employing dose-based toxicological benchmarks. EPA has not developed toxicity criteria for terrestrial species. In addition, site-specific toxicological studies have not been conducted at DCD to determine the toxic potential of ecoCOPCs. Consequently, toxicity data in the scientific literature were reviewed to characterize the toxicity of the ecoCOPCs. Information on test concentrations, modes of exposure, and effects on similar species was used to establish TRVs for risk calculations.

Dose-based toxicological benchmarks (Sample, Opresko, and Suter 1996) were the preferred data source to evaluate the potential for adverse effects to the wildlife receptors of concern. The values presented in Sample et al. are chronic LOAELs and chronic NOAELs derived from bioassay studies of laboratory birds and mammals. The Oak Ridge National Laboratory (ORNL) benchmarks are derived from laboratory measurements of survival, growth, or reproduction. NOAELs are the highest dose of a chemical in a study that causes no observable adverse effect on a test species, while LOAELs are the lowest dose of a chemical in a study that causes an observable adverse effect on a test species. NOAEL-based dietary limits are the preferred toxicity threshold for the DCD SERA. In some instances, a LOAEL was used to derive a NOAEL if a NOAEL was not available.

A dose (d) was selected from Sample, Opresko, and Suter (1996) or available scientific literature for each ecoCOPC. The following criteria were used to select the dose values:

- Doses based on the receptor species selected for evaluation were used preferentially; however, if toxicity information was not available for these species, animal doses within the same class as the receptor species were used.
- Data for reproductive or developmental effects were used preferentially; otherwise, the lowest dose (i.e., most conservative) for which a LOAEL or NOAEL is available was used.
- Chronic data were used in preference to subchronic or acute data, and NOAELs were used in preference to LOAELs and LD_{50}s.

TRVs for chronic NOAELs then were derived according to the following equation:

$$TRV = \frac{d}{UF} \quad (14)$$

where:

TRV = Toxicity reference value (mg/kg bw-d)
d = Literature-based daily dose (mg/kg bw-d)
UF = Total uncertainty factor (unitless).

The uncertainty factors used in this equation are similar to those used by Sample, Opresko, and Suter (1996) and are generally consistent with Army guidance on uncertainty factors (Wentsel et al. 1996). Sample, Opresko, and Suter (1996) recommend an uncertainty factor of 10 for extrapolation from a LOAEL to a NOAEL whereas Wentsel et al. (1996) suggest 5 is more appropriate. As most of the TRVs used in this ERA were derived by Sample, Opresko, and Suter (1996) using a factor of 10 (if needed), the SERA is using the more conservative value of 10. The magnitude of the uncertainty factor is dependent upon both the duration (i.e., chronic, subchronic, or acute) and the endpoint measured (i.e., NOAEL, LOAEL, or LD_{50}). If the endpoint of the bioassay was not a chronic NOAEL, the following factors (Sample, Opresko, and Suter 1996; Wentsel et al. 1996) were used to extrapolate the available data to a chronic NOAEL:

	Divide by
A chronic LOAEL to a chronic NOAEL	10
A subchronic NOAEL to a chronic NOAEL	10
A subchronic LOAEL to a chronic NOAEL	20
An acute NOAEL to a chronic NOAEL	30
An acute LOAEL to a chronic NOAEL	50
An acute LD_{50} to a chronic NOAEL	100

Test species TRVs for mammalian and avian wildlife are presented in Tables M-4 and M-5, respectively, of Appendix M. In all instances, the available literature-based toxicological data were based on animals other than the selected receptor species (i.e., black-tailed jackrabbit and golden eagle). Study results have indicated that resistance to toxic chemicals usually varies among different species as a function of body size. This occurs because various physiological functions (e.g., metabolic rates) are related to body size such that smaller mammalian species, for instance, have higher metabolic rates and are more resistant to toxic chemicals, given their ability to more rapidly detoxify contaminants. The generic extrapolation equation, based on the relationship of body weight (Sample et al. 1996), is presented below:

$$d_a = d_b \times (bw_b/bw_a) \quad (15)$$

where:

d_a = Toxicity value (mg/kg bw-d) for species "a," species to be extrapolated to (e.g., jackrabbit)

d_b = Toxicity values for species "b," test species to extrapolate from (e.g., rat)

bw_a = Body weight of species a

bw_b = Body weight of species b.

This equation represents simple body weight scaling. Body weights of test species and receptor species are presented in Tables M-6 and M-7, respectively, of Appendix M.

4.2.6 Risk Characterization for Ecological Receptors

The procedures for the fourth step in the EPA SERA process are discussed below. Risk characterization integrates exposure and stressor response of receptor species used in the assessment and measurement endpoints, summarizes risk or the likelihood of harm to plants and animals, and interprets the ecological significance of these findings. The ecological assessment endpoints depend on this comparison by using HQs for ecoCOPCs at the qualifying SWMUs. The HQs form the quantitative basis of this risk characterization (EPA 1989d).

HQs compare the estimated exposure concentrations or doses to TRVs. This relationship is shown as:

$$HQ = \frac{\text{Environmental Concentration or Dose}}{\text{TRV}}$$

where an HQ could not be calculated because insufficient data were available to establish a TRV, ecoCOPCs were carried through the risk characterization as ecoCOPCs of uncertain risk to ecological receptors.

An HQ greater than or equal to unity (1) indicates that a potential exists for harmful ecological effects and that the ecoCOPC qualifies for further investigation. An HQ threshold of 1 assumes that the toxicity threshold and exposure concentrations are accurate. In reality, the range of values around 1 within which HQs may or may not indicate the existence of risk increases with the uncertainty of the estimated exposure and toxicity threshold concentrations. There is no clear consensus in regulatory guidance or the scientific literature concerning the significance of the level of departure of the HQ of 1. EPA Region III (1994i), for example, considers HQs greater than 100 to be indicative of extreme risk. However, Wentsel et al. (1994) indicate no statistical analyses exist to support this interpretation.

One further complicating issue is that an HQ greater than 1 by itself does not indicate the magnitude of effect nor provide a measure of potential population-level effects (Menzie et al. 1992). For example, a high soil HQ may be the result of an isolated "hot spot" rather than widespread contamination and may not indicate potential population/community-level effects because, no matter how high the HQ is above 1, the risk is likely limited to only receptors in the vicinity of the hot spot. Both current and future risk will be evaluated in the SWMU-specific SERA discussions. Future estimated risks to plants and animals are considered similar to current risks.

All ecoCOPCs with an HQ exceeding 1, and particularly those greater than 10, are potential ecoCOCs. One way to focus the ecoCOC evaluation and deal with risk characterization uncertainty is to use an order of magnitude adjustment following the safety factor of 10 sometimes used by regulatory agencies. As a first step in this evaluation, estimated or observed LOAELs are substituted for NOAELs as effects thresholds. Sample et al. (1996) estimate NOAELs to be 10 times lower than observed chronic LOAELs (100 times lower than observed subchronic LOAELs). The LOAELs are estimated to be 10 times greater than observed chronic NOAELs. EcoCOPCs with HQs less than 10 are, thus, judged unlikely to violate the assessment endpoints, and further attention is focused on those ecoCOPCs with HQs $\geq$ 10. Exposure assumptions and the conservatism, ecological relevance, and appropriateness of the LOAEL-based effects threshold concentrations derived from observed effects on test species during toxicity experiments are evaluated for those ecoCOPCs with HQs $\geq$ 10. If adjustments in one or more of these parameters is justified, an ecoCOPC with an HQ $\geq$ 10 may be judged not to be an ecoCOC. If there is no justification for adjusting the exposure or effects threshold concentrations, ecoCOPCs with NOAEL-based HQs $\geq$ 10 are identified as ecoCOCs. EcoCOPCs for which no HQs were calculated (because threshold concentrations are unavailable) are also potential ecoCOCs and will not be examined further.

Additionally, UDEQ indicated that the SERA should determine if there is an imminent threat (i.e., acute risks to ecological receptors) when evaluating recommendations. In the absence of an imminent threat, ecological decisions will be deferred to the installation-wide SERA to be conducted after the installation is closed and the future land use is determined (UDEQ 2000). The presence/absence of stressed plants and animals observed during the Group 3 RFI habitat surveys was used as the basis for determining an imminent threat. However, this evaluation method is associated with a large degree of uncertainty as each SWMU may have been visited only on one day and the presence/absence of receptors can be due to factors unrelated to environmental contaminants. In addition, the original purpose of the habitat surveys was to provide environmental setting information, not specifically to evaluate imminent threats.

4.2.7 Uncertainties

Uncertainties in the DCD SERA are discussed in this section according to EPA's procedural approach to ERA: problem formulation, exposure assessment, effects assessment, and risk characterization.

4.2.7.1 Problem Formulation

Concentrations of contaminants in the soil at the DCD SWMU are based on a limited number of samples. A degree of uncertainty exists about the actual spatial distribution of contaminants. Exposure concentrations could be overestimated or underestimated, depending on how the actual data distribution differs from the measured data distribution. However, the majority of samples were collected in areas most likely to have contamination. Although appropriate to achieving the goals of the RFI, such biased sampling is likely to overestimate the potential for exposure and adverse effects to receptor species. Use of data collected in the 0.5- to 15-foot sampling interval is also a source of uncertainty that could underestimate exposure

concentrations. Some receptors may contact soil only in the 0- to 0.5-foot interval rather than the >0.5- to 15-foot interval. Chemical distribution within the >0.5- to 15-foot interval also will differ. Because the estimated 95 percent UCL of the mean concentrations or maximum detected concentration was used as the exposure point concentration to calculate HQs, the estimates of risk from ecoCOPCs are conservative (i.e., protective). Using 95 percent UCL or maximum concentrations decreases the likelihood of underestimating the risk posed by each ecoCOPC and increases the likelihood of overestimating the risk.

The distribution and abundance of organisms comprising the ecological receptors at the SWMU have not been quantified by field studies. The lack of quantitative data introduces uncertainties concerning whether, and to what extent, the risk characterization based on the selected receptor species underestimates or overestimates the risk to organisms that were not used in the risk computations but that are found at DCD. Onsite reconnaissance established the nature and quality of habitat and confirmed the presence of vegetation types and active, visible animal species. Observations made during this reconnaissance justify assumptions concerning the presence of unobserved organisms that are essential to normal ecosystem functioning, such as soil-dwelling worms and arthropods, and herbivorous insects.

Another source of uncertainty relates to the receptor species. These species may or may not accurately reflect risks to observed or unknown species at a given SWMU. For example, some species not evaluated may be more sensitive than those receptors for which toxicity data were available. Conversely, exposures may be less for species not evaluated compared to the receptor species. Therefore, risks may either be overestimated or underestimated.

4.2.7.2 Exposure Assessment

The movement of contaminants from DCD contaminant source media to ecological receptors was not measured for this SERA. Therefore, uncertainties exist regarding the actual exposure modes and pathways and the ecoCOPC concentrations ingested by ecological receptors. Exposure concentrations can differ from the measured environmental concentrations as a result of physical and chemical processes during transport from source to receptor and as a result of biomagnification through the food web. These processes were not evaluated quantitatively in this SERA. For instance, risks to predators higher on the food chain may be underestimated if biomagnification processes are incompletely defined. Although bioaccumulation was estimated for those receptors ingesting food for which TRVs were available, it is possible that exposure to top predators is underestimated because the biomagnification of certain contaminants in their prey was overlooked. It is more likely that exposure to many ecoCOPCs, especially inorganics such as thallium, are overestimated by the default BAFs of 10 and 100.

The exposure modes and pathways are the most important for terrestrial receptors. Soil-dwelling terrestrial animals may be exposed to contaminants in soil by inhalation of volatiles; however, gaseous concentrations of contaminants in soil interstices, cavities, and burrows were not available for DCD. Therefore, the exposure of burrowing organisms to contaminated soil and soil interstitial water may be underestimated if gas concentrations are larger than soil concentrations, which is unlikely. The estimate of risk also will be underestimated if TRVs are lower for inhalation than they are for ingestion. Overestimating

exposure by using conservative exposure concentrations balances the underestimation of exposure that results from neglecting exposure modes and pathways of lesser importance.

Exposure concentrations may be overestimated because of conservative exposure factors. Use of published BAFs, irrespective of species and environmental conditions, may have resulted in the application of conservative values. For instance, it may be overly conservative to assume that jackrabbits obtain all of their diet from the exposure unit in which they are located.

Finally, plant and animal exposure to contaminants below detection limits is not considered in the SERA. In addition, the exposure of ecological receptors to tentatively identified compounds is not considered. There could be risks from these situations that are not evaluated; thus, there is uncertainty.

4.2.7.3 Effects Assessment

For some DCD organisms, the ecoCOPCs may have harmful effects at concentrations below the toxicity threshold concentrations. Toxicity thresholds were based on concentrations reported to have no or little effect on the test organism or were estimated conservatively from published toxicity data. Dietary limits used as TRVs for soils were derived from NOAELs or LOAELs using uncertainty. These TRVs would underestimate the risks only to organisms at DCD that are considerably more sensitive than the study organisms. The TRVs are more likely to overestimate the risk to organisms that are equally or less sensitive than the study organisms. The possibility remains that some TRVs were set at levels at or below which some harm would occur to organisms at DCD.

The calculated risks to the ecological receptors at DCD are the risks of individual contaminants. The risks from exposure to multiple contaminants depend on contaminant interactions; effects could be greater or less than those from a single chemical. This SERA provides findings for ecoCOPC-specific risk estimates. A true evaluation of risk from chemical mixtures cannot be conducted without additional data and evaluation of alternative models of contaminant interaction.

Additional uncertainty exists as to the pertinence of individual organism toxicity for characterizing the risk to populations and ecosystems. It is possible that populations may compensate for the loss of large numbers of juveniles or adults with increased survival or birth rates, and habitats or ecosystems may possess functionally redundant species that are less sensitive to contaminants. The uncertainty as to whether ecosystems at DCD possess these buffering mechanisms justifies a conservative approach to risk assessment based on organismal toxicity thresholds (i.e., NOAELs).

4.2.7.4 Risk Characterization

The uncertainties described above impacted the quantification of current and future ecological risks. Five additional areas of uncertainty in the risk characterization exist: acute risk, offsite risk, cumulative risk, future risk, and background risk.

Acute Risk—Acute risks were estimated based on the presence/absence of stressed plants and animals observed during the Group 3 RFI habitat surveys. This method is associated with a large degree of uncertainty as each SWMU may have been visited on only one day and the presence/absence of receptors can be due to factors unrelated to environmental contaminants. In addition, the original purpose of the habitat surveys was to provide environmental setting information, not specifically to evaluate imminent threats.

Offsite Risk—The risks to offsite receptors cannot be characterized without benefit of contaminant tracer studies and offsite plant and animal and habitat surveys. Offsite receptors can be exposed to contaminants via physical and food chain transport processes, but evaluating the magnitude of this exposure would require additional studies. In general, the risk to most offsite receptors is likely to be overestimated rather than underestimated by the risk estimate for onsite receptors.

Cumulative Risk—The SERA estimates the risk to populations of ecological receptors from individual contaminants. Yet, in nature, receptors are exposed simultaneously to mixtures of chemicals. Thus, cumulative risk is possible. Harmful effects in ecosystems (including effects on individual organisms) may cascade throughout the system and have indirect effects on the ability of a population to persist in the area even though individual organisms are not sensitive to the given contaminants in isolation. Therefore, the ecological risk characterization for DCD may underestimate actual risks to plants and animals from cumulative risks.

Future Risk—A fourth area of uncertainty in the ecological risk characterization is the future risk to the plants and animals from contamination at DCD. The SERA characterizes the current risk based on chronic exposure to measured concentrations of toxicants with the potential to persist in the environment for extended periods of time. Mechanisms exist that could significantly increase (e.g., erosion and leaching to surface water or groundwater) or decrease (e.g., enhanced microbial degradation) the risk to future plants and animals at the site.

Background Risk—Another source of uncertainty is ecological risk relative to background conditions. Some ecoCOPCs may be above background by a statistically significant amount, yet most of the ecological risk can be attributed to chemical amounts contributed by background.

Summary—The most important uncertainties in the DCD SERA are those surrounding the estimates of the contaminant concentrations to which ecological receptors are actually exposed (exposure concentrations) and the concentrations that present an acceptable level of risk of harmful effects (toxicity thresholds). These uncertainties arise from multiple sources, especially from the lack of site-specific data on contaminant transport and transformation processes, organismal toxicity, animal behavior and diet, population dynamics, and the response of plant and animal populations to stressors in their environments. Despite these uncertainties, the available site-concentration data and published exposure and effects information allow ecoCOCs to be identified as risks characterized for each exposure unit.

4.2.8 Summary of the Screening-level Baseline Ecological Risk Assessment

A SERA was performed in accordance with written and other guidance from EPA Headquarters and Utah UDEQ. This guidance recognizes step-by-step procedures. Despite

variances in guidance from different organizations, all guidance adheres to an ERA process that includes problem formulation, followed by exposure assessment and effects assessment, and culminating in risk characterization with attention to uncertainties and summarization.

DCD covers more than 19,000 acres of natural and man-made habitats. Sagebrush, grease wood, and rabbit brush, both native and disturbed, occupy a large portion of the total area. Disturbed grassland areas occupy most of the remainder. The appearance of the abundant vegetation and various animal life suggest no immediate endangerment.

Of the many observed plant and animal taxa, three terrestrial species (i.e., vegetation, jackrabbits, and eagles) were selected for terrestrial receptors. HQs were calculated for each qualifying SWMU or exposure unit.