

## 2.0 METHODOLOGY OF THE BASELINE RISK ASSESSMENT

This section presents the methodology used to derive exposure and risk estimates for the sites assessed. Sections 3.0 through 7.0 present the details of the BRAs for each site. Any deviations from the methodology presented in this section are identified in the detailed discussion for each site. The methods presented here follow EPA and DTSC guidance.

The steps used to perform the data evaluation and selection of chemicals of potential concern (COPCs) for each site are discussed in Section 2.1. The methods for the exposure assessment for each of the RI sites are presented in Section 2.2, which includes a description of the exposure setting, of the receptors, and of potential exposure pathways for each site. The methods used to derive the exposure point concentrations (EPCs) for each COPC in the relevant media are also presented in Section 2.2.

Section 2.3 presents a summary of the toxicity information for all of the COPCs evaluated at the four RI sites. Section 2.4 presents the methods for the risk characterization for each of the BRAs, including the methods used to evaluate possible noncancer health effects, possible cancer risks, and blood lead levels. The uncertainties of the methods used are summarized in Section 2.5.

### 2.1 Data Evaluation and Selection of Chemicals of Potential Concern (COPCs)

This section presents the methods used to evaluate the sample data and the methods used to select the chemicals of potential concern (COPCs) to be included in the quantitative risk evaluation for each RI site. Summaries of the sample data for each site and the site-specific COPCs are presented in the detailed discussions for each BRA in Sections 3.0 through 7.0.

The data considered for the human health BRAs include validated data from the RI, select data from surface water outfall points, and select data from the basewide investigations that are reviewed in the RIs for Sites 2 and 12, Sites 16

and 17, and Site 39 in Volume II of this basewide RI/FS. These data are presented in summary form in the Appendixes for each RI site in Volume II. Diskettes containing all of the raw data for each site were submitted to the reviewing agencies under separate cover. This risk assessment considers chemicals that were detected in each site area. Current onsite source areas of chemicals detected at each site are identified in the RI and are summarized in the detailed site-specific BRAs in Sections 3.0 through 7.0; potential releases from onsite and offsite sources are considered.

Section 2.1.1 describes the parameters used to evaluate the data used in the BRAs. This discussion includes a review of the analytical methods, of the data validation procedures, and of the procedures used to evaluate tentatively identified compounds (TICs) in each data set. Section 2.1.2 presents the steps used to select COPCs for each RI site.

#### 2.1.1 Identification of Usable Data

Much data have been collected at the four RI sites, but only a subset of these data were used in the BRA evaluations. The screening steps recommended by EPA guidance were used to select the dataset for the quantitative BRAs (EPA, 1989b). The dataset selected for each BRA is defined in Section 2.1.1.5.

##### 2.1.1.1 Analytical Methods

The analytical methods used to evaluate sample data from Fort Ord were presented in the Sampling and Analysis Plan and the RI/FS Work Plan for Fort Ord (HLA, 1991b, 1991c). Additional information about the analytical methods is presented in Volume II of this Basewide RI/FS. EPA-approved analytical test methods were used to analyze samples from various media, including soil and groundwater. Screening test results, such as soil gas and total petroleum hydrocarbon (TPH) analyses, are not considered appropriate for use in risk assessment and therefore, were not included in the data

considered for the BRAs (EPA, 1989b). The methods used for collection and analysis of soil gas samples were not designed to support risk assessment needs. Soil gas data were collected to identify areas of potential contamination for additional soil investigation. No patterns indicative of source areas were identified.

### 2.1.1.2 Data Validation

To verify that consistent QA/QC methods were used when evaluating RI/FS data for the RI sites, all data considered for use in the BRAs underwent independent validation. Analytical results from the RI sites were validated according to procedures specified in the Fort Ord QAPP (Part 2 of HLA, 1991b). The validation included an evaluation of the quality of the data with respect to quality control (QC) criteria including precision, accuracy, and completeness. The QC samples used to assess data quality consisted of laboratory duplicate samples, matrix spike/matrix spike duplicates (MS/MSD), blank spike/blank spike duplicate (BS/BSD, also known as laboratory control samples [LCSs]), method blanks, source water blanks, trip blanks, equipment rinsate blanks, and field duplicate samples. Holding times and laboratory surrogate spike recoveries were also evaluated. In addition, 10 percent of all sample delivery groups were subjected to detailed data validation, including review of initial and continuing calibrations, and sample results calculations. The details of the data validation are presented in the Appendixes to the RI (Volume II).

### 2.1.1.3 Evaluation of Detection Limits, Quantitation Limits, and Data Qualifiers

The detection limits, quantitation limits, and data qualifiers for all of the chemicals analyzed for at the RI sites were reviewed. In general, detection limits indicate the concentration at which a small amount of chemical in a sample can be detected, whereas quantitation limits indicate the concentration at which measurements can be trusted. The quantitation limit of interest in the evaluation of RI data for the BRAs is the reporting limit, or sample quantitation limit (SQL). Compounds reported by the laboratory as "below detection limit" or "not detected" (ND)

were analyzed for but were not detected above the reporting limit. These compounds are reported in the laboratory data as the reporting limit value followed by ND or by a U qualifier.

Data qualifiers are coded information about a particular piece of sample data. Data qualifiers can be added to a data set either in the laboratory or during validation. Laboratory qualifiers used in the Fort Ord data set are presented and defined in the site characterization reports and in the appendixes to the RI (Volume II). Some common laboratory qualifiers are: "U," "B," and "J." For inorganic chemical data, "B" qualifier indicates that the reported concentration is below the level of accurate quantitation, whereas for organic chemical data, a "B" qualifier indicates that the analyte was found in the associated blank as well as in the sample. A J qualifier indicates that the compound was detected in the sample but that the value reported is estimated. These and other laboratory qualifiers are reviewed as part of the data validation process. Additional qualifiers are added to the dataset during data validation. These qualifiers are presented in the Appendix to the RI (Volume II). An example of a qualifier that could be added during data validation is an "R." An R means that this piece of data is "rejected," or not considered to merit further evaluation.

For the evaluation of RI site data, all compounds reported with U, B, or J qualifiers after validation were retained in the dataset; and all compounds reported with R qualifiers were omitted from the data set, as recommended by EPA guidance (EPA, 1989b). Because of the uncertainty of the concentration of a compound in samples reported as ND, as "below detection limit" (BDL), or as U qualified samples, EPA guidance recommends that one half the reporting limit be used as a proxy concentration when calculating chemical concentration terms for the BRA. This was done in the calculation of exposure point concentrations (EPCs) for the BRAs.

### 2.1.1.4 Tentatively Identified Compounds (TICs)

Each laboratory analysis is limited to a subset of chemicals that can be reported accurately. This

subset of chemicals may not represent all of the chemicals actually present at a site. Although the identity and reported concentration of TICs are questionable, the laboratory may prepare a list of TICs to accompany a particular dataset.

Any TIC data available for the Fort Ord RI were reviewed according to EPA guidelines as part of the data evaluation for the BRAs (EPA, 1989b). These reviews are presented in the data evaluation section for each site.

### 2.1.1.5 Data Used in the Baseline Risk Assessment Methodology

The data considered for the BRAs are summarized in Table 2.1. For each RI site, the following information is summarized in Table 2.1: the area of the site from which the samples were collected, the sampling medium, the number of samples collected, and the analyses run on those samples. The raw data for each RI site are summarized in the Appendixes of the RI (Volume II). Summaries of concentrations of all compounds detected in each area are presented in the RI text. The data from each site were segregated into several different groups by depth for the BRAs. Sample analyses for screening tests, such as TPH and soil gas samples, were not used in the BRAs (EPA, 1989b). Summary tables presenting concentrations of all detected compounds in the area-depth groupings for consideration in the BRAs are presented in the data evaluation section for each site (Sections 3.2, 4.2, 5.2, and 6.2).

In general, soil data were separated into three separate depth groupings for each area: samples from 0 to 2 feet below ground surface (bgs); samples from 2 to 10 feet bgs, and samples from below 10 feet bgs. Soil data were separated in this way for evaluation of the different potential for human exposures at different depths. Groundwater data were segregated into separate aquifers, where appropriate. Groundwater data from 1993 to May 1994 were used in this evaluation. Groundwater data collected before 1993 were not included due to the potential for migration and degradation of chemicals in groundwater.

Because some samples were analyzed by two test methods for the same compounds, two data points were sometimes available in the data set for the same compounds at one sampling location. This was true for samples analyzed for benzene, toluene, ethylbenzene, and xylene (BTEX) by EPA Test Method 8020 and for volatile organic compounds (VOCs) by EPA Test Method 8240. For this evaluation, when two data points were available for one compound at one sampling location, both data points were used to derive summary statistics for the BRAs.

### 2.1.2 Selection of Chemicals of Potential Concern

The COPCs were selected so that the most prevalent, persistent, and potentially toxic compounds detected at each site were quantitatively evaluated in the BRAs. Criteria for establishing COPCs included consideration of the toxicity, physical properties, and concentration of each of the detected chemicals. Only chemicals reported at concentrations above the laboratory reporting limit (i.e., detected compounds) were considered for evaluation in each discrete study area.

The EPA recommends the use of alternate exposure and toxicity methods to estimate the potential risk from exposure to lead. Therefore, the COPC screening steps reviewed in this section were not applied to the evaluation of lead as a COPC. Lead was retained as a COPC in soil if it was detected at concentrations above a health-based screening level (HBSL). The HBSL used in this assessment is the preliminary remediation goal (PRG) for soil estimated for a child (240 mg/kg [HLA, 1993e]). Lead was not detected in groundwater samples considered for the BRAs.

The following sections describe the methodology used to select COPCs for each of the BRAs for the Fort Ord RI sites. The COPCs selected for each site are presented in Sections 3.3, 4.3, 5.3, and 6.3.

#### 2.1.2.1 Background Chemical Concentrations

As recommended in EPA guidance, chemicals associated with background soil conditions need not be included in the quantitative risk assessment (EPA, 1989b). To evaluate the potential contribution of background chemicals in soil, site-specific background soil data were collected and reported in the Draft Final Basewide Background Soil Investigation report for Fort Ord (HLA, 1993e). Background soil concentrations for organochlorine pesticides and 13 priority pollutant metals were investigated in this report. The infrequent detection of pesticides in onbase soil samples and the significantly higher frequency of detection of pesticides in offbase samples as compared with onbase samples precluded estimating background thresholds or maximum values for pesticides in Fort Ord soil. Site-specific background soil concentrations were determined for 13 priority pollutant metals in the background soil report: antimony, arsenic, beryllium, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, thallium, and zinc (HLA, 1993e).

Background metal concentrations were identified for four geochemically significant conditions in Fort Ord soil: (1) shallow QTP (derived from the Paso Robles Formation), (2) deep QTP, (3) shallow NQTP (non-QTP soil, i.e., derived from the alluvium, older and recent dune sand, Aromas Sand, and Santa Margarita Formation), and (4) deep NQTP. Shallow soil was defined as soil less than 2 feet bgs; deep soil was defined as soil deeper than 2 feet bgs. Background concentrations of metals in the NQTP subsets adjusted for data outliers are shown in Table 2.2. The background dataset for all soil types is presented in Appendix G.

For the BRAs, priority pollutant metals detected at concentrations below maximum site-specific background concentrations were not considered as COPCs. Background concentrations selected for this evaluation were those for the soil type at the site considered; the soil type for the five RI sites evaluated here is NQTP.

As discussed in the background soil report, arsenic, beryllium, and chromium were present at background concentrations that exceeded the lowest, most conservative, preliminary remediation goals (PRGs) estimated for those

metals (HLA, 1993e). This indicates that adverse health effects may occur as a result of exposure to background concentrations of these metals. EPA guidance recommends calculating the potential risks of background concentrations at a site separately from potentially site-related risks if there is reason to believe that the background risks for the site are of concern (EPA, 1989b). Because some metals have been detected at elevated background concentrations in soil, a detailed analysis of potential background risks from metals in soil is provided in Appendix A.

### 2.1.2.2 Further Limitations on the Number of Chemicals

Before the final selection of COPCs for each BRA, several additional points were considered as recommended in EPA guidance (EPA, 1989b):

- Chemicals known to be of high toxicity and known from historical data to be associated with past site activities are to be retained as COPCs
- Chemicals known either to be highly mobile or persistent or known to have a high bioaccumulation potential are to be retained as COPCs
- Chemicals known to be essential human nutrients, present at low concentrations, and known to be toxic only at high doses are not to be considered as COPCs. The details of the essential nutrient evaluation are presented in Appendix B.
- Chemicals that can be identified as laboratory contaminants or artifacts of laboratory analysis are to be eliminated as COPCs by EPA recommendations (1989b). As stated in the Quality Assurance Project Plan (Part 2 of HLA, 1991b), EPA recognizes acetone, methylene chloride, toluene, and phthalate esters as common laboratory contaminants. In areas where these chemicals were detected at low concentrations (i.e., less than 10 times the method blank concentration), they were eliminated as COPCs.

- Compounds detected in groundwater are to be eliminated if historical data shows decreasing concentrations in wells over time and if the current groundwater concentrations for the compounds do not exceed Maximum Contaminant Levels (MCLs)
- Chemicals most likely to contribute significantly to risk are to be retained as COPCs. These chemicals are identified through the use of a toxicity screen. This screening technique involves the calculation of a screening risk value to evaluate potential carcinogenic risks and a screening hazard index (HI) value to evaluate potential noncarcinogenic health effects. Potentially carcinogenic chemicals with carcinogenic screening risks of less than one in one hundred million ( $1 \times 10^{-8}$ ) are eliminated as COPCs. Chemicals not assumed to be carcinogenic with screening HI less than 0.01 are eliminated as COPCs. A summary of the results of the toxicity screen for each BRA is presented in Sections 3.3, 4.3, 5.3, 6.3, and 7.3. The details of the toxicity screens for all BRAs are presented in Appendix C.

### 2.2 Exposure Assessment

The exposure assessment section of each BRA identifies the populations assumed to be exposed to COPCs at each site. The exposure scenarios developed describe the potentially exposed populations, the potential pathways of human exposure to the COPCs at each site, and reasonable estimates of the frequency and duration of contact with COPCs in each of the site areas. The methods used to define these factors are presented in Sections 2.2.1, 2.2.2, and 2.2.3. The exposure scenarios for each site are described in detail in Sections 3.4.3, 4.4.3, 5.4.3, 6.4.3, and 7.4.4.

Section 2.2.4 defines the general approach for estimating potential human exposure doses for each scenario and presents the equations used to estimate pathway-specific doses for all chemicals except lead. Section 2.2.5 presents the exposure assumptions used to estimate dose via each pathway; both receptor-specific and pathway-specific assumptions are presented. Most of the exposure assumptions used in the

BRAs are taken directly from current EPA risk assessment guidance; other assumptions are taken from the available scientific literature. As recommended by EPA, two separate exposure conditions for each scenario were evaluated: (1) a reasonable maximum exposure (RME), and (2) an average exposure. As suggested by current California Environmental Protection Agency (Cal/EPA) and EPA guidance, an appropriate mix of 50th and 95th percentile exposure assumptions were used to estimate both RME and average potential risks.

Section 2.2.6 presents the chemical-specific absorption factors used to estimate potential exposure dose. Section 2.2.7 presents the methods used to estimate exposure point concentrations (EPCs) for each chemical in each environmental medium selected for quantitative evaluation. Section 2.2.8 presents the methods used to perform fate and transport modeling for certain chemicals in select media. Fate and transport modeling is required when exposure is anticipated to occur at a point for which no measured data are available. The specific scenarios and site areas for which fate and transport modeling was conducted are identified in Section 2.2.8. The details of these evaluations are presented in the site-specific discussions. The methodology used to evaluate potential exposures to lead is presented in Section 2.2.9.

#### 2.2.1 Exposure Setting

The Fort Ord facility has been used as a military training facility since 1917, and was undeveloped prior to that time. Previous uses of the sites addressed in the RI/FS include:

- Site 2 - sewage treatment plant with sludge drying beds and unlined pond areas
- Site 12 - automotive storage, maintenance, repair, and dismantling; fuel and solvent storage; refuse disposal; and railroad right of way
- Site 16 - corporation yard, stormwater runoff percolation area, and open space
- Site 17 - motor vehicle storage and maintenance; storage of petroleum products,

solvents, and other chemicals; incinerator site; refuse disposal, including incinerated and unincinerated medical waste and other materials; and baseball field

- Site 31 - obstacle course used for training, incinerator building, disposal of refuse which included ashes apparently from an incinerator at the site, and open space
- Site 3 - small arms fire training ranges and open space.
- Site 39 - Ordnance training ranges, including those for naval gunfire from offshore; antitank rocket (bazooka) range; and open space.

The decision-making process to identify the reuse of these and other areas of Fort Ord is described in Volume 1 of this RI/FS. The exposure assessment developed land use scenarios based on the projected future land uses identified in the planning documents available at the time of preparation: the Fort Ord Reuse Group *Summary of Base Reuse Plan (FORG, 1994)*, the *Installation-Wide Multispecies Habitat Management Plan for Fort Ord, California (COE, 1994)*, and the *Final Environmental Impact Statement Fort Ord Disposal and Reuse (COE, 1993)*. The general land uses upon which the exposure scenarios were based are:

- Sites 2 and 12 - Aquaculture and oceanographic research facilities, commercial and industrial development, a transit center, medium- to high-density residential development, and a school
- Sites 16 and 17 - Part of a university campus, and a corporation yard for public agencies
- Site 31 - Open space for wildlife habitat and an agricultural center with production, processing, distribution facilities, and worker housing
- Site 3 - A limited-access state park.
- Site 39 - Habitat reserve: a limited-access natural resource management area (NRMA)

managed by the Bureau of Land Management (BLM).

The scenarios used to evaluate exposure for individual sites considered the projected land use at individual areas in which chemicals have been detected in soil or groundwater. Additional specific assumptions about land uses are presented in the site-specific discussions.

### 2.2.2 Potential Exposure Pathways

For this assessment, the exposure scenarios evaluated in the BRAs for the five RI sites represent complete exposure pathways that meet the following criteria:

- A source and mechanism for chemical release
- An environmental transport medium (e.g., air, water, soil)
- A point of potential human contact with the medium
- A route of exposure (e.g., inhalation, ingestion, dermal contact).

As defined in the site-specific discussions presented in Sections 3.4.2, 4.4.2, 5.4.2, 6.4.2, and 7.4.3, the primary pathways of potential exposure to the site areas of interest include incidental ingestion of soil, dermal contact with soil, inhalation of particulate dust, inhalation of vapors, and ingestion of groundwater.

### 2.2.3 Exposure Scenarios

Exposure scenarios describe the way in which potential human receptors could be exposed to COPCs at a site. As recommended by EPA, two separate exposure scenarios were evaluated for each receptor: an average exposure scenario and a reasonable maximum exposure (RME) scenario. It is important to note that although attempts are made to represent true average and RME exposures, all exposure scenarios presented here likely overestimate potential risk at these sites because of the uncertainty inherent in the assumptions used.

The exposure scenarios used in each BRA were based on the predicted future use of each site area. Table 2.3 summarizes the receptors selected for quantitative evaluation in the BRAs for the four RI sites. The detailed discussion for each site includes a thorough review of all potential human receptors. Only the most sensitive potential receptors were selected for quantitative evaluation to estimate the baseline risks for each RI site.

**2.2.4 Estimation of Exposure (Dose)**

This section describes the methods used to estimate the chemical intake (dose) for the exposure scenarios described in Section 2.2.3. Dose is defined as the amount of chemical absorbed by the body over a given period of time. For noncarcinogenic effects, the dose is averaged over the period of exposure and is referred to as the average daily dose (ADD). For carcinogenic effects, the dose was averaged over a lifetime and is referred to as the lifetime average daily dose (LADD). Consistent with current EPA guidance (1989b), the following general equation was used to assess the dose for each exposure pathway considered in this assessment:

$$\text{Dose} = \frac{C \times IR \times EF \times ED \times FI \times AF}{BW \times AT}$$

Where:

- Dose = ADD or LADD in milligrams per kilogram per day (mg/kg-day)
- C = Chemical concentration in environmental medium (mg/kg)
- IR = Intake rate in milligrams per day (mg/day)
- EF = Exposure frequency in days per year (days/year)
- ED = Exposure duration (years)
- FI = Fraction of intake (unitless)
- AF = Absorption factor (unitless)
- BW = Body weight in kilograms (kg)

- AT = Averaging time (days): for noncarcinogenic effects, AT = Exposure duration x 365 days/year; for carcinogenic effects, AT = Lifetime (70 years) x 365 days/year

To evaluate the relative sensitivity of each exposure pathway, receptor- and pathway-specific intake factors (IFs) were estimated using the general dose equation presented. An IF is a nonchemical-specific term that incorporates information on medium contact rate (e.g., milligrams of soil ingested per day), exposure times, and other receptor- and pathway-specific assumptions. Receptor- and pathway-specific ADDs and LADDs were then estimated for each chemical, receptor, and exposure pathway by multiplying the IF for each receptor and pathway by the chemical concentration term (C x AF). The chemical concentration term was the measured or modeled concentration of the chemical in the appropriate medium multiplied by a chemical-specific absorption factor (AF) for some pathways of exposure as shown in this equation:

$$\text{Dose} = \text{IF} \times (C \times \text{AF})$$

Where:

- Dose = ADD or LADD (mg/kg-day)
- IF = Intake factor
- C = Chemical concentration in environmental medium
- AF = Absorption factor

The format of the IFs used in the BRAs for the Fort Ord RI sites are consistent with the standard dose equations recommended by EPA (1989b). The pathway-specific equations used to estimate IFs are presented in the following sections. The exposure assumptions used to estimate IFs are presented in Section 2.2.5.

**2.2.4.1 Incidental Ingestion of Soil**

Incidental ingestion of soil was evaluated using the exposure point concentration (EPC) of the chemical in soil, a chemical-specific absorption factor, and the soil ingestion IF. The EPCs and absorption factors for compounds in soil are presented in subsequent sections. The equation for the IF for ingestion of soil is estimated as follows:

$$IF_{ing-s} = \frac{IR \times CF \times EF \times ED \times FI}{BW \times AT}$$

Where:

- IF<sub>ing-s</sub> = Intake factor for incidental ingestion of soil in kilograms of soil per kilogram of body weight per day (kg<sub>soil</sub>/kg<sub>body weight</sub>-day)
- IR = Soil ingestion rate (mg/day)
- CF = Conversion factor of one millionth of a kilogram per milligram (10<sup>-6</sup> kg/mg)
- EF = Exposure frequency (days/year)
- ED = Exposure duration (years)
- FI = Fraction of intake (unitless)
- BW = Body weight (kg)
- AT = Averaging time (period over which exposure is averaged in days)

**2.2.4.2 Dermal Contact with Soil**

Dermal exposure to chemicals present in soil was evaluated using the EPC of the chemical in soil, chemical-specific absorption factors, and the dermal IF. The EPCs and dermal absorption factors for COPCs in soil are presented in subsequent sections. The equation for the IF for dermal contact with soil is as follows:

$$IF_{der-s} = \frac{SA \times AF \times CF \times EF \times ED \times FI}{BW \times AT}$$

Where:

- IF<sub>der-s</sub> = Intake factor for dermal contact with soil (kg<sub>soil</sub>/kg<sub>body weight</sub>-day)
- SA = Surface area of exposed skin in square centimeters (cm<sup>2</sup>)
- AF = Soil to skin adherence factor in milligrams per square centimeter per day (mg/cm<sup>2</sup>-day)
- CF = Conversion factor (10<sup>-6</sup> kg/mg)
- EF = Exposure frequency (days/year)
- ED = Exposure duration (years)
- FI = Fraction of intake (unitless)
- BW = Body weight (kg)
- AT = Averaging time (period over which exposure is averaged in days)

**2.2.4.3 Inhalation of Dust Entrained In Air**

Exposures to chemicals via inhalation of particulates, or dust, in air were evaluated using an EPC for dust and the particulate IF. Airborne dust EPCs are presented in Section 2.2.7. The equation for the IF for inhalation of particulates is as follows:

$$IF_{inh-p} = \frac{IR \times ET \times EF \times ED}{BW \times AT}$$

Where:

- IF<sub>inh-p</sub> = Intake factor for the inhalation of particulates in cubic meters per kilogram per day (m<sup>3</sup>/kg-day)
- IR = Inhalation rate in cubic meters per hour (m<sup>3</sup>/hr)
- ET = Exposure time in hours per day (hr/day)

- EF = Exposure frequency (days/year)  
 ED = Exposure duration (years)  
 BW = Body weight (kg)  
 AT = Averaging time (period over which exposure is averaged in days)

**2.2.4.4 Inhalation of Vapors from Groundwater**

Inhalation exposure resulting from the volatilization of chemicals in groundwater and subsequent release to air at the soil surface was evaluated using airborne chemical concentrations (EPCs) predicted by vapor flux modeling (Section 2.2.8), and the vapor inhalation IF. The equation for the IF for inhalation of vapors is as follows:

$$IF_{inh-v} = \frac{IR \times ET \times EF \times ED}{BW \times AT}$$

Where:

- IF<sub>inh-v</sub> = Intake factor for the inhalation of volatile chemicals (m<sup>3</sup>/kg-day)  
 IR = Inhalation rate (m<sup>3</sup>/hr)  
 ET = Exposure time (hr/day)  
 EF = Exposure frequency (days/year)  
 ED = Exposure duration (years)  
 BW = Body weight (kg)  
 AT = Averaging time (period over which exposure is averaged in days)

Inhalation exposure resulting from the volatilization of chemicals in groundwater during domestic use of groundwater (i.e., showering) was evaluated using a generic model from EPA guidance which assumes that the dose from inhalation of VOCs while showering is approximately equivalent to the dose from ingestion of 2 liters per day of the same water

(EPA, 1989o). Inhalation-specific toxicity values were used to characterize potential risks and noncancer health effects from inhalation exposures (Section 2.4).

**2.2.4.5 Ingestion of Water**

Ingestion of groundwater as drinking water was evaluated using the EPC for the chemical in groundwater and the IF for the ingestion of water. The EPCs for groundwater are presented in subsequent sections. The IF for water ingestion was calculated using the following equation:

$$IF_{ing-w} = \frac{IR \times EF \times ED}{BW \times AT}$$

Where:

- IF<sub>ing-w</sub> = Intake factor for ingestion of water in liters per kilogram per day (l/kg-day)  
 IR = Ingestion rate in liters per day (l/day)  
 EF = Exposure frequency (days/year)  
 ED = Exposure duration (years)  
 BW = Body weight (kg)  
 AT = Averaging time (period over which exposure is averaged in days)

**2.2.5 Exposure Assumptions Used to Estimate Intake Factors (IFs)**

Some of the exposure assumptions used to estimate IFs via the potential exposure pathways presented in Section 2.2.4, are described below and summarized in Tables 2.4 and 2.5. The remaining assumptions used to estimate IFs are presented in the discussions of exposure scenarios for each site.

**2.2.5.1 Soil Ingestion Rate**

A soil ingestion rate of 50 mg/day was used to estimate potential doses and risks in the average case scenarios for all receptors. Numerous investigations have provided data for incidental soil ingestion rates, the most accurate of which are those using tracer elements. The results suggest that soil ingestion rates range from 9 to 40 mg/day for young children (*Calabrese, Barnes et al., 1989*). Later work by some of the same investigators confirmed this range and concluded that the data were normally distributed with a geometric mean of 20.5 mg/day and a standard deviation of 87 mg/day (*Calabrese and Stanek, 1991a, b*). This range has been used in published risk assessments as the basis for characterizing a probability distribution for soil ingestion (*Copeland et al., 1993; Finley and Paustenbach, 1994*). Estimates of soil ingestion rates for older children and adults, based on studies in adults (*Calabrese, Gilbert et al., 1990*), range from 1 to 10 mg/day (*Paustenbach, Jernigan et al., 1992; Paustenbach, Wenning et al., 1992*). The upper-bound value for the probability distribution developed from the Calabrese and Stanek data (*1991a, b*) of 50 mg/day was selected as the average exposure value for the BRAs. This value is also suggested as the appropriate upper-bound value for a commercial/industrial worker (*EPA, 1991b*).

For the RME scenarios, EPA-recommended age-specific soil ingestion rates were used in the estimation of potential doses and risks via incidental ingestion of soil. For onsite resident and nearby resident receptors aged 0 to less than 6 years, a soil ingestion rate of 200 mg/day was used (*EPA 1989b, 1991b*). EPA's default soil ingestion rate for ages 6 and above, 100 mg/day (*EPA, 1989b, 1991b*), was used for the onsite resident (6 to less than 30 years), student resident, nearby resident (6 to less than 30 years), park ranger, habitat management worker, and nearby resident trespasser receptors. EPA's default onsite commercial worker soil ingestion rate of 50 mg/day (*EPA, 1991b*) was used for the evaluation of potential doses and risks via incidental ingestion of soil for the commercial worker receptor. EPA's default soil ingestion rate of 480 mg/day for construction/excavation

scenarios was used for the construction worker and utility worker receptors (*EPA, 1991b*).

**2.2.5.2 Surface Area of Exposed Skin**

The skin surface areas used in the estimation of potential doses and risks via dermal contact with soil represent the average surface area values for certain body parts in the particular age category being evaluated. Fiftieth percentile data points for both males and females in the appropriate age category were taken from EPA's *Exposure Factors Handbook* (*EPA, 1990b*). For average exposure scenarios, the skin surface area for face, neck, and both hands was calculated for each receptor assumed to contact soil. Skin surface areas of 1,420 cm<sup>2</sup> and 1,635 cm<sup>2</sup> were calculated for receptors aged 0 to less than 6 years (onsite resident) and receptors aged 6 to less than 9 years (onsite resident and nearby resident trespasser), respectively. A skin surface area of 2,109 cm<sup>2</sup> was calculated for adult receptors evaluated in the average scenarios: commercial worker, utility worker, student resident, construction worker, and park ranger.

For RME scenarios, the skin surface area for face, neck, both arms, and both hands were calculated for each receptor assumed to contact soil. Skin surface areas of 2,348 cm<sup>2</sup> and 3,764 cm<sup>2</sup> were calculated for receptors aged 0 to less than 6 years (onsite resident and nearby resident) and receptors aged 6 to less than 18 years (onsite resident, nearby resident, and nearby resident trespasser), respectively. A skin surface area of 4,714 cm<sup>2</sup> was calculated for adult receptors evaluated in the RME scenarios: commercial worker, onsite resident, utility worker, student resident, construction worker, park ranger, and nearby resident.

**2.2.5.3 Soil to Skin Adherence Factor**

In the estimation of potential doses and risks via dermal contact with soil, a soil to skin adherence factor (AF) was used to estimate the volume of soil that adheres to each square centimeter of exposed skin during the assumed exposure period. This value was assumed to be 0.2 mg/cm<sup>2</sup>-day for the average exposure

scenarios, as recommended in EPA's dermal absorption guidance (EPA, 1992m).

An AF of 0.4 mg/cm<sup>2</sup>-day was used to evaluate RME scenarios. EPA's 1.0 mg/cm<sup>2</sup>-day default value for estimating upper-bound exposure was not used because not all exposed skin was assumed to be exposed at upper-bound levels. The RME AF was developed by assuming that the heaviest soiling would occur on the palms of the hands and inner forearms, and that the balance of the arms, and the face and neck would be less exposed. EPA's upper-bound AF of 1.0 mg/cm<sup>2</sup> was used to estimate exposure at the most heavily soiled skin areas, and the EPA's default average AF of 0.2 mg/cm<sup>2</sup>-day was used to estimate exposure to other skin areas. These AFs were used with 50<sup>th</sup> percentile values for the areas of the surfaces considered to develop an area-weighted AF of 0.4 mg/cm<sup>2</sup>-day (See Table below).

Description	Adherence		Adhered Soil (AS) (mg)
	Factor (AF) (mg/cm <sup>2</sup> )	Area (A) (cm <sup>2</sup> )	
hands	1	420	420
hands (back)	0.2	420	84
forearms (front)	1	570	570
forearms (back)	0.2	570	114
upper arms	0.2	1430	286
face and neck (head)	0.2	1180	236
Sum		4590	1710

Area-weighted AF = 0.37

Area values obtained from *Exposure Factors Handbook* (EPA 1990b) Table 4-1

AS = AF x A

Area-weighted AF = Sum of areas divided by the sum of adhered soil

This approach provides a conservative AF for RME because day-to-day exposure generally involves a variety of different activities; activities resulting in heavy soiling are unlikely to occur at every exposure opportunity.

#### 2.2.5.4 Inhalation Rate

The inhalation rates used in the estimation of potential doses and risks via inhalation of particulate dust in air and of vapors from groundwater were derived from age-specific and activity level-specific data presented in EPA's *Exposure Factors Handbook* (EPA, 1990b). Inhalation rates were estimated for each receptor age group based on outdoor activity data presented in EPA's *Exposure Factors Handbook*. The inhalation rate calculated for child receptors aged 0 to less than 6 for both average and RME scenarios was based on the inhalation rate for a child, age 6 (1.24 m<sup>3</sup>/hour). The inhalation rate calculated for receptors aged 6 to less than 9 years for the average scenarios was based on the average reported inhalation rate for a child of age 6 and a child of age 10 (1.56 m<sup>3</sup>/hour). The inhalation rate calculated for receptors aged 6 to less than 18 years for the RME scenarios was based on the reported inhalation rate for child, age 10 (1.87 m<sup>3</sup>/hour).

Inhalation rates for adult receptors were also taken from data presented in *Exposure Factors Handbook*. The inhalation rate used in the average exposure scenario for commercial worker, utility worker, student resident, park ranger, and habitat management worker receptors was 0.83 m<sup>3</sup>/hour, and that for construction worker receptors was 1.4 m<sup>3</sup>/hour. These average exposure rates were based on the average inhalation rates for adults. The inhalation rate used in the RME scenario for commercial worker, onsite resident, utility worker, student resident, nearby resident adult, park ranger, and habitat management worker receptors was 1.25 m<sup>3</sup>/hour, and that for construction worker receptors 3.0 m<sup>3</sup>/hour. These RME values were based on upper-bound inhalation rates for adults.

#### 2.2.5.5 Water Ingestion Rate

The drinking water ingestion rates used in this assessment were age-specific ingestion rates

based on data presented in EPA's *Exposure Factors Handbook*. Weighted ingestion rates were calculated for onsite resident receptors aged 0 to less than 6 years, 6 to less than 9 years, and 6 to less than 18 years. The drinking water ingestion rates used for both average exposure and RME are 0.4 l/day for receptors aged 0 to less than 6 years, 0.5 l/day for receptors aged 6 to less than 9 years, and 0.6 l/day for receptors aged 6 to less than 18 years.

Drinking water ingestion rates for onsite resident and student resident receptors were also taken from EPA's *Exposure Factors Handbook*, which presents an upper-bound adult tap water ingestion rate of 1.5 l/day. This water ingestion rate was selected for the average exposure. EPA's default ingestion rate of 2 l/day, which represents an upper-bound volume of beverages consumed per day, was selected as the water ingestion rate for the RME scenario.

### 2.2.5.6 Body Weight

The body weights used in the estimation of potential doses and risks for all pathways represent the average body weights of males and females in the particular age category being evaluated. Fiftieth percentile data for both males and females in the appropriate age category were taken from EPA's *Exposure Factors Handbook*. The same body weight data was used to evaluate both average and RME scenarios.

The average body weight of adults was reported to be 70 kg. This weight was used in the evaluation of adult commercial worker, resident, utility worker, student resident, construction worker, nearby resident, park ranger, and habitat management worker receptors.

The average body weight of male and female children ages 0 to less than 6 years was reported to be 14 kg. This weight was used in the evaluation of onsite, nearby, and offsite resident receptors. The average body weight of children ages 6 to less than 9 years was reported to be 24.2 kg and was used to evaluate average exposures of onsite, nearby, and offsite resident, and nearby resident trespasser receptors. The average body weight for 6 to less than 18 year old males and females was reported to be 41.6 kg and

was used to evaluate potential RME exposures of onsite, nearby, and offsite resident, and nearby resident trespasser receptors.

### 2.2.5.7 Exposure Time

Exposure time is the number of hours that a receptor is assumed to inhale air containing COPCs each day that they are on the site. The values used for exposure time are described in detail in the exposure scenarios for each site presented in Sections 3.4.3, 4.4.3, 5.4.3, 6.4.3, and 7.4.4.

### 2.2.5.8 Fraction of Intake

The *Risk Assessment Guidance for Superfund (EPA, 1989b)* describes a fraction of intake term (FI) that accounts for the fact that only some of the soil that a receptor potentially contacts in 1 day comes from the site. The EPA recommends that the FI term should reflect chemical location and population activity patterns. This evaluation assumed that most receptors are likely to ingest and contact soils at both onsite and offsite locations on the days they are exposed to chemicals in soil at the Fort Ord sites. In the equations to estimate intake via incidental ingestion and dermal contact with soil, the FI factor represents the proportion of soil ingested and contacted that come from the site on a given day of exposure.

Although only a fraction of the total soil ingested or contacted on a given day is likely to come from the site, the RME scenarios conservatively assumed that 100 percent of the soil ingested and contacted on a given day came from the Fort Ord site being evaluated. Because many receptors are assumed to be on the site for only a portion of the given days of exposure, this assumption overestimates overall risks from ingestion and dermal contact exposures to soil.

The average exposure scenarios for all receptors, except the onsite resident evaluated at Sites 2 and 12, assumed that 50 percent of the total soil ingested and contacted on a given day came from the site being evaluated (i.e., FI equals 50 percent). Because it was also assumed that onsite residents spend the majority of their time on the site, an FI of 75 percent was assumed for

the average scenario for this receptor (evaluated for Sites 2 and 12). These values likely overestimate average exposure.

FIs selected for receptors assumed to be exposed to more than one discrete area on a site (e.g., the student resident evaluated for Sites 16 and 17 who was assumed to be exposed to soils in three discrete areas), are described in detail in the site-specific discussions about exposure scenarios.

### 2.2.5.9 Exposure Frequency

Exposure frequency is the number of days in a year an individual may contact chemicals at the site. The receptor-specific exposure frequencies used in this assessment are described in the exposure scenarios for each site in Sections 3.4.3, 4.4.3, 5.4.3, 6.4.3, and 7.4.4.

### 2.2.5.10 Exposure Duration

Exposure duration is the length of time in years an individual may contact the media of interest at a site. The values used for exposure duration are described in detail in the exposure scenarios for each site in Sections 3.4.3, 4.4.3, 5.4.3, 6.4.3, and 7.4.4.

### 2.2.6 Chemical-Specific Absorption Factors

As described in Section 2.2.4, ADDs and LADDs were estimated by multiplying the receptor- and pathway-specific intake factors by the EPC and by a chemical-specific absorption factor for certain pathways of exposure. This assessment used only one oral absorption factor (OAF) to estimate doses and risks. An OAF of 43 percent for chlorinated dibenzodioxins and dibenzofurans (CDDs and CDFs) was used to evaluate incidental ingestion of soil. This value, used in many published risk assessments, is based on the study conducted by *Shu, Paustenbach et al. (1988)*, who reported a range of 39 to 49 percent and

mean of 43 percent for tetrachlorodibenzo-p-dioxin (TCDD). This range is consistent with data reported by other investigators (*Lucier et al., 1986; Umbreit et al., 1986; Birnbaum and Couture, 1988*). The mean value was selected as representative for all 2,3,7,8 congeners of CDDs and CDFs because the penta- through octa-congeners exhibit reduced absorption due to higher chlorination (*Couture et al., 1988*).

Chemical-specific dermal absorption factors (DAFs) were derived for all COPCs in soil for the evaluation of dermal contact exposures following guidelines presented in Cal/EPA's *Preliminary Endangerment Assessment Guidance Manual (Cal/EPA, 1994)* and verbal recommendations from the California Department of Toxic Substances Control. The DAFs used in this assessment are presented in Table 2.6. DAFs for inorganic metals given to HLA by Dr. John Christopher from DTSC include 0.1 percent for cadmium, 3 percent for arsenic, and 1 percent for all other metals (meeting among U.S. EPA, DTSC, RWQCB, COE, Army, and HLA representatives, March 26, 1993). For all organic except explosives and CDDs and CDFs, chemical- or class-specific DAFs were derived using the recommendations presented in Cal/EPA's guidance.

In the absence of chemical- or class-specific DAFs for explosives, a DAF of 100 percent was used to conservatively estimate the uptake of explosive compounds from soils.

A DAF of 1 percent was used to conservatively estimate uptake of CDDs and CDFs from soils, based on information presented in *Dermal Exposure Assessment: Principles and Applications (EPA, 1992m)*. The EPA's (1992m) document presents four DAF estimates based on the findings of three separate studies evaluating dermal uptake of dioxins from soil:

Value (percent)	Basis
2.5	A 1991 EPA rat study using <i>in vivo</i> administration, corrected to reflect differences between dermal absorption <i>in vivo</i> in rats and humans observed in the same study
0.2	A 1988 rat study by Shu, et al., using <i>in vivo</i> administration, corrected to reflect differences between dermal absorption <i>in vivo</i> in rats and humans observed in the 1991 EPA study. EPA's (1992m) discussion of the correction calculation indicates that the this value should be 0.33 percent
1	A 1980 rat study by Poiger and Schlatter using <i>in vivo</i> administration, corrected to reflect differences between dermal absorption <i>in vivo</i> in rats and humans observed in the 1991 EPA study
0.45	A 1991 EPA study which used <i>in vitro</i> administration to (human) cadaver skin, corrected to reflect differences between dermal absorption <i>in vitro</i> in rats and humans observed in the same study

The four experimentally-derived DAF values presented above were all based on soil with low organic carbon content, consistent with the conditions at Fort Ord, and were based on conservative interpretations of the experimental data. The DAF value of 1 percent used to estimate exposure represents the average of the four values above (computed using either 0.2 or 0.33 percent for the data from Shu et al.).

The 1 percent DAF value is based on soils representative of conditions at Fort Ord, is within the 0.1 to 3 percent range recommended by EPA (1992m), is consistent with a 0.5 percent value predicted by McCone (1990) using a dermal fugacity model for TCDD, and is in the range

described by the probability distribution developed by Copeland, et al. (1993).

### 2.2.7 Exposure Point Concentrations (EPCs)

The concentrations of COPCs at the assumed points of human exposure (i.e., EPCs) were estimated under two separate exposure conditions for each receptor. For the RME scenario, the lower of the maximum detected concentration and the upper 95th confidence level of the arithmetic mean concentration for a chemical was selected as the EPC for each area. For the average exposure the arithmetic average concentration of a chemical in each area was selected as the EPC. As recommended in EPA guidance, one half the reporting limit value was used as a proxy concentration for each nondetected (ND) sample.

The EPCs for direct exposure to soil and water were based on the measured site sample data discussed in Section 2.1. The EPCs for exposure to airborne dust were estimated as described in Section 2.2.8 Fate and Transport Modeling. The EPCs for volatile compounds in air from groundwater were modeled using a compartment fate and transport model as described in Section 2.2.8.

The potential toxicity of certain groups of compounds is characterized by extensive toxicological information available for only one or a few compounds in the group. This is the case with the potential carcinogenic toxicity of CDDs and CDFs and polynuclear aromatic hydrocarbons (PAH). For each of these groups, EPA has developed toxicity equivalent factors (TEFs) for many of the compounds within these groups. TEFs are used to rank the relative toxicity of the compounds for which little toxicity information is available using one or a few compounds for which extensive toxicity information is available.

Samples analyzed for CDDs and CDFs were converted to 2,3,7,8-tetrachlorodibenzo-p-dioxin toxic equivalents (TCDD-TEs) using TEFs. EPA TEFs (EPA, 1989b) are shown in Table 2.7 and were used as follows: The concentration of each detected CDD and CDF congener was multiplied

by its respective TEF; then these factors were summed for each sample in an area of interest. Summary statistics were then calculated for the TCDD-TE samples in each area of interest to yield one EPC concentration for CDDs and CDFs in each area where either dioxins or furans were detected.

The EPA-recommended TEFs used to evaluate the potential carcinogenic effects of PAH are based on benzo(a)pyrene (B[a]P ([EPA, 1993f])). Table 2.8 presents the B(a)P TEFs used to evaluate PAH in this assessment. The EPCs used for the evaluation of the potential carcinogenic effects of PAH in the BRAs were estimated for each area as follows. One half the reporting limit was used as a surrogate value for all samples for compounds with at least one detect in the area of interest; compounds not detected in the area of interest were omitted from the analysis.

TEFs for B(a)P were then multiplied by the measured or surrogate value for each compound. These products were then summed to yield a single concentration of B(a)P toxic equivalents (B[a]P-TEs) for each sample. Summary statistics were then calculated for each group of samples in an area of interest as for all other detected compounds.

B(a)P-TE concentrations are appropriate for the evaluation of potential carcinogenic health effects, but these adjusted concentrations of potentially carcinogenic PAH are not appropriate for the evaluation of the potentially noncarcinogenic effects of these compounds. As discussed in Section 2.3.1, no noncarcinogenic toxicity criteria are available for any of the individual carcinogenic PAH. Therefore, the potential noncarcinogenic effects of carcinogenic PAH were evaluated in this assessment as pyrene, on the basis of structural similarities. Because it was assumed that all potentially carcinogenic PAH act similarly, (i.e., benz(a)anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[k]fluoranthene, chrysene, dibenzo[a,h]anthracene, and indeno (1,2,3-cd) pyrene; EPA, 1993f; 1994), their potential noncarcinogenic effects are evaluated using total carcinogenic PAH (cPAH) concentrations. Total cPAH concentrations were derived by taking one

half the reporting limit for all ND samples for compounds with at least one detect in the area of interest and then summing the measured and surrogate concentrations to yield a total cPAH concentration for each sample in the area of interest. Summary statistics were then calculated for each group of samples in an area of interest as for all other detected compounds.

The EPCs and other summary statistics for all chemicals in all media and areas evaluated in the BRAs are presented in the site-specific discussions.

### 2.2.8 Fate and Transport Modeling

Fate and transport modeling is required when exposure is anticipated to occur at a point for which no measured data are available. Measured data are available for soil and groundwater. Measured data are not available for either vapors or particulate dust in air. This section describes the methods used to estimate EPCs in air.

EPCs for compounds in air volatilizing from groundwater were estimated using a vapor flux model developed by the U.S. Department of the Army (Army model) and reviewed by W.A. Jury, W.W. Nazaroff, and V.C. Rogers (*Army, 1991*) and a box dispersion model (*Wadden and Sheaff, 1983*). The Army model was based on publications by Jury et al. that describe the behavior of volatile chemicals in soil and groundwater (*Jury et al., 1983; 1984a, b, c; Jury, Russo et al., 1990*). The Army model was selected because it was specifically designed to evaluate possible vapor emissions from groundwater. The Army model is described in detail in Appendix D.

EPCs for airborne dust concentrations were derived by multiplying the EPCs in soil in milligrams of chemical per kilogram of soil ( $\text{mg}_{\text{chemical}}/\text{kg}_{\text{soil}}$ ) by the concentration of respirable particles with a mean diameter of less than or equal to 10 microns ( $\text{PM}_{10}$ ) for the Monterey County area in micrograms of soil per cubic meter of air ( $11.5 \mu\text{g}_{\text{soil}}/\text{m}^3_{\text{air}}$ ), and a units conversion factor in kilograms per micrograms ( $\text{kg}/\mu\text{g}$ ). The Monterey County  $\text{PM}_{10}$  value was obtained from Monterey Bay Unified Air

Pollution Control District (MBUAPCD) representative Mr. John Fear on April 6, 1994 via fax to Mr. Craig Nichols of HLA. This approach will result in conservative estimates of potential particulate inhalation exposures because much of the site is covered with vegetation, buildings, or pavement and there is little potential for dust generation in such areas.

### 2.2.9 Evaluation of Lead

Due to the complex toxicokinetics of lead in the body, standard exposure assessment methods used in risk assessment are not appropriate for the evaluation of exposures to lead. Both EPA and Cal/EPA use pharmacokinetic models to evaluate lead exposure; both of these models estimate doses as blood-lead concentrations related to specific chemical doses. EPA's Uptake Biokinetic Model (UBK) Version 0.6 (EPA, 1990e), and Cal/EPA's LEADSPREAD (Cal/EPA, 1992a) exposure models (computer programs) were developed separately to estimate blood-lead levels in children ages 0 to 6 and adults, respectively.

The UBK model was used in this evaluation to evaluate lead exposures in children (0 to 6 years old) because it incorporates current toxicokinetic (chemical uptake and distribution) data for lead in a child's body over time. Because the UBK model is limited to children 6 years old or younger, the LEADSPREAD model was used to evaluate lead exposures in all receptors over 6 years old.

#### 2.2.9.1 Methods for the Uptake Biokinetic Model

The UBK model was used to estimate a blood-lead level for the possible exposures of children to soil containing lead. The UBK model addresses possible exposure to lead via inhalation of airborne dust, ingestion of drinking water, incidental ingestion of soil and dust, incidental ingestion of paint containing lead, and maternal contribution to infant body burdens. The UBK model considers background exposures (i.e., exposures that occur due to our daily activities), site media concentrations, default exposure assumptions, and empirically derived toxicokinetic relationships to estimate the blood-lead concentrations from all sources for children

at 1 year intervals from 6 months through 6 years of age. EPA has established some default lead contributions from each of the background sources and has preprogrammed these into the model. Default contributions were replaced with site-specific soil concentrations and site-specific dust in air concentrations where appropriate. Because a target blood-lead level has been established by EPA and the model considers contributions to blood-lead concentrations from background sources (i.e., lead in drinking water, mother's milk, etc.), the higher the background contributions of lead, the lower are the permissible lead exposures from any of the Fort Ord sites.

The EPA's default UBK model exposure assumptions and estimated blood-lead concentrations for child (0 to 6 years) receptors for both the average and RME scenarios are presented in Appendix F. As a conservative health-protective measure, the highest estimated blood-lead level predicted for any age group (0.5 to 1, 1 to 2, 2 to 3, 3 to 4, 4 to 5, or 5 to 6 years) was selected to represent blood-lead levels in child receptors.

#### 2.2.9.2 Methods for the LEADSPREAD Model

The UBK model does not address lead exposures for receptors over 6 years old. Cal/EPA's LEADSPREAD model was used in this evaluation to estimate blood-lead concentrations of all receptors over 6 years old. As does the UBK model, the LEADSPREAD model incorporates background exposures, user-defined media concentrations, default and user-defined assumptions, and empirically derived toxicokinetic relationships to estimate blood-lead concentrations. The LEADSPREAD model addresses possible exposure to lead via inhalation of airborne dust, incidental ingestion of soil, direct dermal contact with soil, ingestion of drinking water, and ingestion of food.

The EPCs for average and RME scenarios used in the LEADSPREAD exposure analysis are presented in the exposure assessment section for each site. All receptors were assumed to be exposed to lead at background levels of home-grown or purchased produce. In addition, all

receptors were assumed to be exposed to background lead concentrations in drinking water of 15 microliters of lead per liter of water as estimated by Cal/EPA.

Exposure to lead via ingestion of site-grown produce was evaluated for longer-term onsite resident receptors (i.e., at Site 12).

The intake assumptions for the LEADSPREAD model and the blood-lead levels estimated for each receptor are presented in Appendix F.

### 2.3 Toxicity Assessment

The purpose of the toxicity assessment is to identify the types of adverse health effects a COPC may potentially cause and to define the relationship between the dose of a chemical and the likelihood of an adverse effect (response). Adverse effects are characterized by EPA as carcinogenic or noncarcinogenic. Dose-response relationships are defined by EPA for oral exposure and for exposure by inhalation. Oral dose-response values were used to evaluate dermal exposures because EPA has not yet developed values for dermal exposure. Oral dose-response values were also used to evaluate inhalation exposures for some compounds lacking inhalation dose-response values. Combining the results of the dose-response assessment with information on the magnitude of potential human exposure provides an estimate--usually very conservative--of potential risk.

The majority of information available about the dose-response relationship for a given chemical is based on data collected from animal studies (usually rodents) and theoretical predictions about what might occur in humans. When available, human exposure data are also considered and given more weight. When animal data are considered, mathematical models are used to estimate the possible response in humans at exposure levels far below those tested in animals. These models contain conservative assumptions that should be considered when the resulting risk estimates are evaluated. Conservatism arises in animal models because of the uncertainty in extrapolating results obtained in animal research to humans and extrapolating

responses obtained from high-dose studies to estimate responses at very low doses. For example, humans are typically exposed to chemicals in the environment at levels that are less than one thousandth of the lowest dose tested in animals. Such doses may be easily handled by the myriad of biological protective mechanisms in humans (*Ames et al., 1987*). This means that while the results of standard recent bioassays may be used to understand the human biological hazard or cancer risk posed by typical exposure levels, this understanding is considered to be very limited (*Crump et al., 1976; Sielkin, 1985*).

The EPA and Cal/EPA have used dose-response data to establish "maximally acceptable" levels of daily human exposure for noncarcinogenic chemicals. For carcinogenic chemicals, regulatory policy assumes a potential carcinogenic response at any dose. Carcinogenic potency is a measure of the relationship between dose and tumor incidence.

EPA's Integrated Risk Information System (IRIS), an on-line database, contains dose-response criteria currently approved by EPA; and EPA's Health Effects Assessment Summary Tables (HEAST), an annual report, tabulates EPA-approved dose-response information. For the Fort Ord BRAs, dose-response values were taken from IRIS (*EPA, 1994*) when available. HEAST (*EPA, 1993e, 1992b*) was used as a secondary source if dose-response values were not available on IRIS. Dose-response values from Cal/EPA (*1992e*) were used in place of EPA values if Cal/EPA values were more conservative. The following sections discuss the noncarcinogenic and carcinogenic risk dose-response values selected for the COPCs at Fort Ord RI sites; noncarcinogenic and carcinogenic risk dose-response values are presented in Table 2.9.

#### 2.3.1 Possible Noncancer Health Effects

It is widely accepted that noncarcinogenic effects from chemical substances occur after a threshold dose is reached. To establish health risk criteria for noncarcinogenic effects, the threshold dose is usually estimated from the no-observed adverse effect level (NOAEL) or the lowest observed

adverse effect level (LOAEL) determined in chronic animal exposure studies. The NOAEL is defined as the highest dose at which no adverse effects appear. The LOAEL is defined as the lowest dose at which adverse effects begin to appear.

NOAELs and LOAELs derived from human or animal studies are used by the EPA to establish oral and inhalation reference doses (RfDs). An RfD is a maximal daily dose that is not expected to cause adverse health effects. Uncertainty factors are used to establish RfDs in an attempt to account for limitations in the quality or quantity of available data. If the estimated dose for a given set of conditions is less than the chemical-specific RfD, then it is appropriate to conclude that no significant health hazard exists under the defined set of conditions.

As summarized in Table 2.9, either an oral or an inhalation RfD, or a surrogate value, exists for all of the COPCs at Fort Ord except 1,2-dichloroethane, B[a]P, 4,4'-DDE, 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-TCDD), and lead. 1,2-Dichloroethane, 4,4'-DDE, and 2,3,7,8-TCDD are considered carcinogenic and no applicable RfD is available to evaluate the potential noncarcinogenic effects of these compounds. The potential adverse health effects associated with lead are evaluated as described in Section 2.2.9.

Surrogate values are selected on the basis of structural similarities to other chemicals when chemical-specific RfDs are not available. The RfD for pyrene is used as a surrogate value for B(a)P to evaluate the potential noncarcinogenic effects of potentially carcinogenic PAH (Total cPAH). The RfD for trinitrotoluene is used as a surrogate value for 2-amino-dinitrotoluene and 4-amino-dinitrotoluene.

For this assessment, the oral RfD was used to represent the inhalation RfD for any chemical lacking an inhalation RfD (i.e., oral-to-inhalation route-to-route extrapolation was performed). Although chemical toxicity may vary substantially with route of uptake, this extrapolation was performed to reduce possible underestimation of health risks due to the absence of toxicity values.

### **2.3.2 Possible Cancer Effects**

Regulatory agencies have generally assumed that carcinogenic agents should be treated as if they do not have thresholds. In other words, the dose-response curve for carcinogens used for regulatory purposes allows for zero risk only at zero dose (i.e., for any dose, some risk is assumed to be present). To estimate a theoretically plausible response at low environmental doses, various mathematical models are used to extrapolate response at low-dose levels from high-dose data. The EPA generally uses the linearized multistage model for extrapolation to low doses. This model assumes that the effect of the carcinogenic agent on tumor formation is linear. The cancer slope factor (SF) quantitatively defines the relationship between dose and response. The chemical-specific SF represents the upper-bound estimate of the probability of a carcinogenic response per unit intake of a chemical over a 70-year lifetime.

The EPA classifies chemicals into Groups A through E: Group A is designated "human carcinogen" and Group E is designated "noncarcinogen" (with "probable," "possible," and "not classifiable" as Groups B, C, and D, respectively). Quantitative carcinogenic risk assessments are performed for chemicals in Groups A and B and may be performed for those in Group C on a case-by-case basis (EPA, 1989b).

Of the COPCs considered in this assessment, the following have been determined by the EPA and/or Cal/EPA to possess carcinogenic potential (i.e., group A, B1, or B2): carbon tetrachloride, chlordane, 1,2-dichloroethane, methylene chloride, B(a)P, bis(2-ethylhexyl) phthalate, 4,4'-DDE, 4,4'-DDT, 2,3,7,8-TCDD, arsenic, beryllium, cadmium, lead, and nickel.

The cancer slope factor for B(a)P was used to evaluate the potential carcinogenic effects of B(a)P-TE. The cancer slope factor for 2,3,7,8-TCDD was used to evaluate the potential carcinogenic effects of TCDD-TE. The cancer slope factors used in this assessment are summarized in Table 2.9.

### 2.3.3 Possible Effects of Lead

EPA (1994) assigns lead to weight of evidence group B2, but neither EPA nor Cal/EPA has published RfDs or SFs for lead. The BRA therefore used different methods to evaluate possible effects of exposure to lead. As discussed in Section 2.2.9, the exposure assessment used the UBK and LEADSPREAD models to estimate receptor blood-lead concentrations. A target blood-lead concentration of 10 micrograms of lead per deciliter of blood ( $\mu\text{g}/\text{dl}$ ) was used to evaluate possible exposures to lead. This target blood-lead concentration reflects the findings of the Agency for Toxic Substances and Disease Registry (ATSDR) that 10  $\mu\text{g}/\text{dl}$  represents a lowest observed adverse effect level (LOAEL) associated with lead exposure, based on hypertension as the toxic effect (ATSDR, 1990b).

## 2.4 Risk Characterization

This section presents the methods used to quantify potential human health risks for each BRA. Subsections 2.4.1 and 2.4.2 describe the noncancer and cancer health risk estimates for all COPCs except lead. Section 2.4.3 describes the methods used to evaluate the potential risks associated with exposures to lead.

### 2.4.1 Possible Noncancer Health Effects

The estimates of receptor-specific noncancer health effects are represented by a hazard index (HI). The HI is determined for each receptor by summing the hazard quotient (HQ), for each chemical in each exposure pathway. The HQ is the fraction of the RfD represented by the average daily dose (ADD). This approach to estimating noncancer health effects is conservative and was used where the HIs were less than one. For receptors with HIs exceeding one, separate HIs were developed for chemicals that act on the same target organs (i.e., respiratory tract, liver, etc.). HIs were calculated separately for each receptor age-group for each BRA. If the HI is greater than 1, there may be potential adverse noncancer health effects associated with the pathway being evaluated according to EPA's definition (1989b).

### 2.4.2 Possible Cancer Risk

The estimates of potential upper-bound cancer risks are estimated for each receptor by summing the age-specific cancer risks for all pathways for that receptor. The potential cancer risk estimates are represented by the product of the lifetime average daily dose (LADD) and the cancer slope factor (SF). EPA has defined a target range of cancer risk estimates for Superfund sites as one in one million ( $1 \times 10^{-6}$ ), to one in ten thousand ( $1 \times 10^{-4}$ ). Cancer risk estimates falling below this target range do not typically trigger remedial action to reduce the estimated risks. Cancer risk estimates falling within this target range may trigger remedial action at some sites, and estimates above this range typically require some remedial action to reduce potential risk to within or below this range.

### 2.4.3 Evaluation of Blood Lead

Possible effects of exposure to lead were evaluated by comparing the receptor blood-lead levels estimated using the UBK and LEADSPREAD models described in Section 2.2.9 with the target blood-lead concentration of 10  $\mu\text{g}/\text{dl}$  identified in Section 2.3.3. Estimated blood-lead levels less than the target blood-lead concentration were considered acceptable.