

# **BASELINE RISK ASSESSMENT REPORT**

**DEL AMO SUPERFUND SITE  
LOS ANGELES, CALIFORNIA**

*Prepared for:*

**SHELL OIL COMPANY AND  
THE DOW CHEMICAL COMPANY**

*Prepared by:*



GeoSyntec Consultants  
924 Anacapa Street, Suite 4A  
Santa Barbara, California 93101  
(805) 897-3800

*With:*



URS Corporation  
130 Robin Hill Road, Suite 100  
Santa Barbara, California 93117  
(805) 964-6010 ♦ Fax (805) 964-0259

**September 7, 2006**

## TABLE OF CONTENTS

1.0 INTRODUCTION.....	1
1.1 BASELINE RISK ASSESSMENT APPROACH.....	1
2.0 SITE BACKGROUND AND REMEDIAL INVESTIGATION/FEASIBILITY STUDY .....	4
2.1 SITE BACKGROUND.....	4
2.2 REMEDIAL INVESTIGATION/FEASIBILITY STUDY (RI/FS).....	4
2.3 RI/FS DATA REVIEW .....	8
2.3.1 Surface Soil .....	9
2.3.2 Shallow Soil.....	10
2.3.3 Deep Soil .....	11
2.3.4 Shallow Soil Gas .....	12
2.3.5 Deep Soil Gas .....	12
2.3.6 Indoor Air .....	12
2.3.7 Groundwater .....	13
2.3.8 NAPL.....	13
3.0 DATA EVALUATION AND ANALYSIS.....	16
3.1 DATA VALIDATION AND SELECTION .....	16
3.2 DATA PROCESSING .....	17
3.2.1 Composite Samples .....	17
3.2.2 Fixed Laboratory and Mobile Laboratory Results .....	18
3.2.3 Soil Gas and Soil Matrix Conversion.....	18
3.3 EXPOSURE AREAS OF POTENTIAL CONCERN.....	20
3.4 IDENTIFICATION OF CHEMICALS OF POTENTIAL CONCERN.....	22
3.4.1 Background Analysis.....	22
3.4.1.1 <u>Metals</u> .....	22
3.4.1.2 <u>Carcinogenic Polynuclear Aromatic Hydrocarbons</u> .....	25
3.4.2 COPC Selection.....	26
3.4.2.1 <u>Organics</u> .....	26
3.4.2.2 <u>Metals</u> .....	26
3.5 EXPOSURE POINT CONCENTRATIONS .....	27
3.5.1 Alternatives for the Reasonable Maximum EPC.....	28
3.5.2 Procedure for Selection of Statistical Estimator .....	29
4.0 EXPOSURE ASSESSMENT.....	31
4.1 CONCEPTUAL SITE MODEL.....	31
4.2 EXPOSURE PATHWAY EVALUATION .....	32
4.2.1 Direct Exposure to COPCs in Soil .....	32
4.2.2 Inhalation of Airborne Dust Containing COPCs.....	33
4.2.3 Inhalation of Vapors in Indoor and Outdoor Air.....	33
4.2.4 Summary of Potentially Complete Exposure Pathways .....	33
4.3 ESTIMATING CHEMICAL INTAKE.....	34
4.3.1 Exposure Parameters .....	35
4.3.2 Exposure Point Concentrations .....	37
4.3.3 Calculation of Summary Intake Factors .....	37
4.3.3.1 <u>Incidental Soil Ingestion</u> .....	37

## TABLE OF CONTENTS

4.3.3.2	<u>Dermal Contact</u> .....	39
4.3.3.3	<u>Inhalation Exposures</u> .....	40
4.3.4	Inhalation of Fugitive Dust Containing COPCs.....	42
4.3.5	Inhalation of Outdoor Air Vapors.....	43
4.3.6	Inhalation of Indoor Air Vapors.....	45
4.3.6.1	<u>Correlation Analysis</u> .....	46
4.3.6.2	<u>Vapor Transport Modeling</u> .....	47
4.3.6.3	<u>Tier 2 Vapor Diffusive Transport Analysis</u> .....	49
4.4	ECOLOGICAL RISK ASSESSMENT.....	51
5.0	TOXICITY ASSESSMENT.....	54
5.1	HEALTH EFFECTS CRITERIA FOR POTENTIAL CARCINOGENS.....	54
5.2	HEALTH EFFECTS CRITERIA FOR POTENTIAL NONCARCINOGENS.....	56
5.3	DERMAL TOXICITY CRITERIA.....	57
5.4	HEALTH EFFECTS FROM LEAD.....	57
6.0	RISK CHARACTERIZATION.....	59
6.1	COMMERCIAL WORKER EXPOSURES.....	61
6.1.1	Surface and Shallow Soil Exposures – Outdoor Pathways.....	61
6.1.2	Indoor Air Exposures.....	63
6.1.2.1	<u>Modeled Indoor Air Exposures from Soil/Soil Gas Data</u> .....	63
6.1.2.2	<u>Indoor Air Exposures from Workplace Air Monitoring Data</u> .....	64
6.1.2.3	<u>Modeled Indoor Air Exposures from Groundwater Data</u> .....	65
6.2	HYPOTHETICAL FUTURE RESIDENTIAL EXPOSURES.....	66
6.2.1	Shallow Soil Exposures – Outdoor Pathways.....	66
6.2.2	Indoor Air Exposures.....	67
6.2.2.1	<u>Modeled Indoor Air Exposures from Soil/Soil Gas</u> .....	67
6.2.2.2	<u>Modeled Indoor Air Exposures from Groundwater</u> .....	68
6.3	TRENCH WORKER EXPOSURES.....	69
6.4	RISK CHARACTERIZATION SUMMARY.....	70
6.4.1	Commercial Worker.....	70
6.4.2	Hypothetical Future Resident.....	71
6.4.3	Trench Worker.....	72
7.0	ASSESSMENT OF UNCERTAINTY.....	73
7.1	COPC SELECTION AND ESTIMATION OF CHEMICAL CONCENTRATIONS.....	73
7.1.1	Random Sampling Uncertainty.....	74
7.1.2	Spatial Representation Uncertainty.....	74
7.1.3	Detection Limit Uncertainty.....	76
7.1.4	Distribution Assumption Uncertainty.....	77
7.1.5	Partitioning and Transport Modeling Uncertainty.....	77
7.1.6	Temporal Uncertainty.....	79
7.2	EXPOSURE ASSUMPTIONS.....	79
7.3	CHEMICAL TOXICITY.....	80
7.4	RISK CHARACTERIZATION.....	84
7.5	SUMMARY.....	85
8.0	SUMMARY AND CONCLUSIONS.....	86

**TABLE OF CONTENTS**

---

8.1 COMMERCIAL WORKER ..... 87  
    8.1.1 Surface and Shallow Soil Exposures – Outdoor Pathways ..... 87  
    8.1.2 Indoor Air Exposures..... 87  
8.2 HYPOTHETICAL FUTURE RESIDENT ..... 88  
    8.2.1 Shallow Soil Exposures – Outdoor Pathways ..... 88  
    8.2.2 Indoor Air Exposures..... 89  
8.3 TRENCH WORKER ..... 89  
8.4 UNCERTAINTY ..... 90  
9.0 REFERENCES ..... 91

## LIST OF TABLES

- Table 1 – Site-Wide Summary of Surface Soil Data (0 to 1' bgs)  
Table 2 – Site-Wide Summary of Shallow Soil Data (0 to 15' bgs)  
Table 3 – Site-Wide Summary of Deep Soil Data (>15' bgs)  
Table 4 – Site-Wide Summary of Shallow Soil Gas Data (0 -15' bgs)  
Table 5 – Site-Wide Summary of Deep Soil Gas Data (>15' bgs)  
Table 6 – Parcel-Specific Summary of Workplace Air Monitoring Data  
Table 7 – Workplace Air Monitoring Data – Background Ambient Air  
Table 8 – Site-Wide Summary of Groundwater VOCs  
Table 9 – Site-Wide Summary of Detected Chemicals  
Table 10 – Summary of Data Types Available and Sampling Rationale – Exposure Areas of Potential Concern  
Table 11 – Exposure Areas of Potential Concern  
Table 12a – Parcel-Specific COPCs and EPCs  
Table 12b – Parcel-Specific Soil-To-Indoor Air EPCs  
Table 12c – Groundwater-To-Indoor Air EPCs  
Table 13 – Exposure Parameter Values – Commercial Worker  
Table 14 – Exposure Parameter Values – Hypothetical Resident  
Table 15 – Exposure Parameter Values – Trench Worker  
Table 16 – Cancer Toxicity Data – Oral/Dermal  
Table 17 – Cancer Toxicity Data – Inhalation  
Table 18 – Noncancer Toxicity Data – Oral/Dermal  
Table 19 – Noncancer Toxicity Data – Inhalation  
Table 20 – Cumulative Risk and Hazard – Commercial Worker Exposure Scenario  
Table 21 – Cumulative Risk and Hazard – Hypothetical Residential Exposure Scenario  
Table 22 – Cumulative Risk and Hazard – Trench Worker Exposure Scenario  
Table 23 – COPCs Contributing to Majority of Risk and Hazard – Commercial Worker Exposure Scenario  
Table 24 – COPCs Contributing to Majority of Risk and Hazard – Hypothetical Residential Exposure Scenario  
Table 25 – Cumulative Risk and Hazard – Tier 1 Groundwater-to-Indoor Air – Commercial and Hypothetical Residential Exposure Scenarios  
Table 26 – Parcel-Specific Risk and Hazard – Tier 2 Groundwater-to-Indoor Air – Commercial Worker Exposure Scenario  
Table 27 – Parcel-Specific Risk and Hazard – Tier 2 Groundwater-to-Indoor Air – Residential Exposure Scenario  
Table 28 – Background Ambient and Indoor Air VOC Data  
Table 29A – Risk and Hazard Index Summary by Receptor and Pathway  
Table 29B – Risk and Hazard Index Summary by EAPC

## LIST OF FIGURES

- Figure 1 – Location Map - Del Amo Site
- Figure 2 – Current and Former Site Development
- Figure 3 – EAPCs and Groundwater Contamination Source Areas
- Figure 4 – VOC Sampling Locations in Surface Soil
- Figure 5 – SVOC/PAH Sampling Locations in Surface Soil
- Figure 6 – Pesticide/PCB Sampling Locations in Surface Soil
- Figure 7 – Metals Sampling Locations in Surface Soil
- Figure 8 – VOC Sampling Locations in Shallow Soil
- Figure 9 – SVOC/PAH Sampling Locations in Shallow Soil
- Figure 10 – Pesticide/PCB Sampling Locations in Shallow Soil
- Figure 11 – Metals Sampling Locations in Shallow Soil
- Figure 12 – VOC Sampling Locations in Deep Soil
- Figure 13 – Total VOC Concentrations at Shallow Soil Gas Sampling Locations
- Figure 14 – Total VOC Concentrations in Deep Soil Gas Samples
- Figure 15 – Indoor Air Monitoring Locations
- Figure 16 – Dissolved Benzene Concentrations - Water Table Zone
- Figure 17 – Potential and Known NAPL Areas
- Figure 18 – Conceptual Site Model
- Figure 19a – Data Use for the Outdoor Soil Pathway
- Figure 19b – Data Use for the Indoor Air Pathway
- Figure 20 – EAPCs with Elevated RME Risk for Outdoor Soil and Commercial Worker
- Figure 21 – EAPCs with Elevated RME Risk/Hazard for Indoor Air and Commercial Worker Using Modeling
- Figure 22 – EAPCs with Elevated RME Risk for Outdoor Soil and Hypothetical Resident
- Figure 23 – EAPCs with Elevated RME Risk for Indoor Air and Hypothetical Resident Using Modeling
- Figure 24 – EAPCs with Elevated RME Risk for Outdoor Soil and Trench Worker

## **LIST OF APPENDICES**

- Appendix A – Data Quality Review of Non-RI/FS Data
- Appendix B – Background Comparison For Metals
- Appendix C – Selection of Chemicals of Potential Concern
- Appendix D – Summary of Parcel-Specific and Chemical-Specific Risk and Hazard, with Supporting Documentation
- Appendix E – Tier 1 Vapor Modeling Parameters
- Appendix F – Tier 2 Vapor Modeling Results
- Appendix G – Toxicological Profiles
- Appendix H – Supporting Information for Indoor Air Pathway Evaluation
- Appendix I – Ecological Risk Assessment

## LIST OF ACRONYMS

95UCL	95% Upper Confidence Limit
ADD	Average Daily Dose
AFB	Air Force Base
AOC	Administrative Order on Consent
AST	Above Ground Storage Tank
ASTM	American Society for Testing and Materials
BAF	Bioaccumulation Factor
BAP-eq	Benzo(a)pyrene Equivalent
BCF	Bioconcentration Factor
bgs	Below Ground Surface
BRA	Baseline Risk Assessment
BTAG	Biological Technical Assistance Group
BTEX	Benzene, Toluene, Ethylbenzene, and Xylene
CAG	Cancer Assessment Guidelines
CalEPA	California Environmental Protection Agency
CARB	California Air Resources Board
cm	Centimeter
CNDDDB	California Natural Diversity Database
COI	Constituent Of Interest
COPCs	Chemicals of Potential Concern
cPAH	Carcinogenic Polynuclear Aromatic Hydrocarbon
CSF	Cancer Slope Factor
CSM	Conceptual Site Model
CT	Central Tendency
CV	Coefficient of Variation
DL	Detection Limit
dl	Deciliter
DLM	Dominant-Layer Model
DNAPL	Dense Non-aqueous Phase Liquid
DTSC	Department of Toxic Substances Control
EAPC	Exposure Area of Potential Concern
EPC	Exposure Point Concentration
ERA	Ecological Risk Assessment
FS	Feasibility Study
g	Gram
HEAST	Health Effects Assessment Summary Tables
HI	Hazard Index
HSAA	California Hazardous Substances Account Act
IRIS	Integrated Risk Information System

## LIST OF ACRONYMS

kg	Kilogram
LADD	Lifetime Average Daily Dose
LADWP	Los Angeles Department of Water and Power
LMM	Linearized Multistage Model
LNAPL	Light Non-aqueous Phase Liquid
LOAEL	Lowest Observable Adverse Effect Level
log K <sub>ow</sub>	Octanol-Water Partition Coefficient
m	Meter
MCL	Maximum Contaminant Level
MEK	Methyl Ethyl Ketone Peroxide
mg	Milligram
MSL	Mean Sea Level
NAPL	Non-aqueous Phase Liquid
NCEA	National Center for Environmental Assessment
NCP	National Oil and Hazardous Substances Pollution Contingency Plan
ND	Non-detection
NOAEL	No Observable Adverse Effect Level
OEHHA	Office of Environmental Health Hazard Assessment
OU	Operable Unit
PAHs	Polynuclear Aromatic Hydrocarbons
PCBs	Polychlorinated Biphenyl Compounds
PEF	Particulate Emission Factor
PEL	Permissible Exposure Levels
PM10	Particulate Matter Less Than 10 Microns in Diameter
ppmv	Parts Per Million by Volume
PRGs	Preliminary Remediation Goals
Q/C	Dispersion Coefficient
QAPP	Quality Assurance Project Plan
RAGS	Risk Assessment Guidance for Superfund
RCRA	Resources Conservation and Recovery Act
RELs	Reference Exposure Levels
RfCs	Reference Concentrations
RfDs	Reference Doses
RI	Remedial Investigation
RI/FS	Remedial Investigation / Feasibility Study
RME	Reasonable Maximum Exposure
ROD	Record of Decision
s	Second

## LIST OF ACRONYMS

---

SF	Seasonality Factor
SVOCs	Semi-volatile Organic Compounds
TPH	Total Petroleum Hydrocarbon
TRV	Toxicity Reference Value
UCL	Upper Confidence Limit
USEPA	United States Environmental Protection Agency
UST	Underground Storage Tank
UTL	Upper Tolerance Limit
VF	Volatilization Factor
VOCs	Volatile Organic Compounds

## **1.0 INTRODUCTION**

This Baseline Risk Assessment (BRA) for the Del Amo Superfund Site (site) is presented by Shell Oil Company and The Dow Chemical Company (the Respondents) pursuant to the Administrative Order on Consent (AOC; USEPA Docket Number 92-13) between the Respondents, the U.S. Environmental Protection Agency Region IX (USEPA) and the California Department of Toxic Substances Control (DTSC). This report was prepared based on regulatory agency comments on the previously submitted Draft BRA Report (URS, 2001a) and Revised Draft BRA Report (URS, 2005), and incorporates additional site data collected to address identified data gaps.

The site comprises approximately 280 acres in Los Angeles, California (Figure 1). For the purposes of the risk assessment, the site excludes the Del Amo Waste Pit Area that was addressed as a separate operable unit (OU) and for which a Record of Decision (ROD) has been previously issued by USEPA (USEPA, 1997WPROD). A separate ROD for the groundwater OU has also been previously issued (USEPA, 1999JGWROD), although a portion of the groundwater data is included for the purposes of the risk assessment, as explained later in this document. The risk assessment evaluates potential health risks to commercial workers, trench workers, and hypothetical future residents at the site associated with chemicals within the soil and non-aqueous phase liquid (NAPL) OU. This OU consists of vadose zone soils and areas of identified or suspected NAPL. Potential exposures to chemicals detected in surface and shallow soils have been evaluated for the direct contact pathways as well as inhalation of volatile chemicals in indoor and outdoor air and fugitive dust. The potential for volatile chemicals to migrate from the subsurface to indoor air was evaluated for deeper vadose zone soils and groundwater.

This BRA is primarily focused on evaluation of potential risks to human health due to the highly developed, urban nature of the site and limited habitat it provides for wild life species. The southern margin of the site is known to provide raptor habitat however, as confirmed by observations of an American kestrel (*Falco sparverius*). The BRA therefore includes an ecological risk assessment (ERA) that is focused on evaluation of risks to the single kestrel observed at the site. The complete ERA is presented in Appendix I and a summary of the ERA and findings is presented in Section 4.4.

### **1.1 BASELINE RISK ASSESSMENT APPROACH**

The overall approach used in the BRA is based on current USEPA guidance documents (USEPA, 1986, 1989; 1991a,b; 1992a,b; 1995b; 1997a; 1999a). The BRA is a predictive tool used to assess the potential human health risks associated with past releases of chemicals at

the site. The results of the risk assessment will be used in the remedial decision-making process during the feasibility study (FS) phase.

There are five steps involved in risk assessment:

- The **Data Review and Evaluation** selects a data set for use in the risk assessment and summarizes the nature and extent of environmental contamination at the site. Chemicals of Potential Concern (COPCs) are selected based on the risk assessment data set. The data review and evaluation are summarized in Sections 2.0 and 3.0 of this report.
- The **Exposure Assessment** evaluates the magnitude, frequency, duration, and routes of potential human exposure to site-related COPCs. The exposure assessment considers both current and potential future site uses under a range of potential exposure scenarios and is based on complete exposure pathways to either actual or hypothetical receptors (i.e., generalized groups that could come in contact with site-related COPCs). The exposure scenarios are summarized in the Conceptual Site Model (CSM; see Section 4.1), which includes the sources, affected media, release mechanisms, and exposure pathways for each identified receptor population. The exposure assessment is presented in Section 4.0.
- The **Toxicity Assessment** provides a review of available information to identify the nature and degree of toxicity, and to characterize the dose-response relationship (the relationship between magnitude of exposure and magnitude of potential adverse health effects on each receptor) for each COPC. A summary of the toxicity assessment is presented in Section 5.0.
- **Risk Characterization** is a synthesis of exposure and toxicity information to yield quantitative estimates of potential cancer risks and noncancer hazards to defined receptor populations. The risk characterization is presented in Section 6.0.
- The **Assessment of Uncertainty** identifies and characterizes the uncertainties associated with each of the four previous steps to assist decision-makers in evaluating the risk assessment results in the context of the assumptions and variabilities in the data used. The uncertainty analysis is presented in Section 7.0.

USEPA guidance documents were consulted as necessary to provide exposure values applicable to site receptors. These documents include Risk Assessment Guidance for Superfund (RAGS), Volume I, Human Health Evaluation Manual (USEPA, 1989); Supplemental Guidance: Standard Default Exposure Factors (USEPA, 1991a), and Exposure Factors Handbook (USEPA, 1997a). Both central tendency (CT) and high-end estimates of exposure were developed for the identified exposure scenarios. For high-end estimates of

exposure, reasonable conservative modeling assumptions (those which tend to overestimate exposure point concentrations) and upper bound (or high) default values for most exposure parameters were used. An evaluation of risk and hazard was additionally conducted using more realistic, site-specific, and refined modeling techniques. Specifically, vapor transport models incorporating biodegradation in the vadose zone were developed and applied using site-specific data and the dominant-layer model (DLM) of Johnson et al. (1999), as discussed in Section 4.3.6.3.

**2.0 SITE BACKGROUND AND REMEDIAL  
INVESTIGATION/FEASIBILITY STUDY****2.1 SITE BACKGROUND**

The Del Amo site is situated in the city of Los Angeles, in proximity to the cities of Torrance and Carson (Figure 1). For the purposes of the risk assessment, the site consists of approximately 280 acres defined by the boundaries of a former synthetic rubber manufacturing plant complex, but excluding an approximately four-acre Waste Pit Area in the southern portion of the plant.

The synthetic rubber plant was constructed between 1942 and 1943 by the U.S. Government. The plant site was used by a number of companies, including Shell Oil Company and The Dow Chemical Company, to produce feedstock chemicals and synthetic rubber under contract with the U.S. Government. The plant consisted of a butadiene plant, a styrene plant, and a copolymer plant. The butadiene and styrene produced at their respective plants were combined in the copolymer plant to form the synthetic rubber product.

The plant was decommissioned and dismantled in the early 1970s and the site has been redeveloped over the last thirty years as a commercial and industrial business park. Land use within the business park is currently characterized by light industrial/manufacturing activities, import/export warehouses, and commercial office space. The business park currently includes 65 parcels, 59 of which have been developed with occupied business buildings. Five parcels are unsuitable as building sites due to their very narrow width and/or their use as railroad spurs or as a public utility corridor with high voltage power transmission towers. The remaining site parcel is currently used as a storage yard and is undeveloped. With the noted exceptions, the entire land surface within the site is currently covered by buildings, parking areas, roadways, or landscaping. Land use surrounding the site includes commercial, industrial and residential developments. Figure 2 presents aerial photographs showing site conditions in 1971, just before the rubber plant was dismantled, and in 2004, after redevelopment of the site as a business park.

**2.2 REMEDIAL INVESTIGATION/FEASIBILITY STUDY (RI/FS)**

A broad spectrum of chemical data for various sample media was collected for the RI/FS, as summarized below:

<u>Sample Media</u>	<u>Data Types</u>
Soil	Volatile Organic Compounds (VOCs) Polynuclear Aromatic Hydrocarbons (PAHs) Semi-Volatile Organic Compounds (SVOCs) Pesticides Polychlorinated Biphenyls (PCBs) Metals Cyanide
Soil Gas	VOCs
Groundwater	VOCs SVOCs Pesticides PCBs Metals General Minerals
Indoor Air	VOCs

Data collection during the RI/FS was completed using the following guidelines:

- Data collection was focused in site areas where contamination was known to be present or judged most likely to occur. These areas included former rubber plant facility locations where chemicals were used, stored, transported, or disposed. The footprints of former plant site facilities/features that were targeted for sampling during the RI are indicated on Figure 4.
- Data were collected to adequately characterize exposure pathways and evaluate remedial alternatives in areas impacted by former rubber plant operations;
- Site history information and vadose zone soil and soil gas data were used to evaluate optimum locations for collection of groundwater data. This information collectively facilitated identification of groundwater contamination source areas. The investigation additionally worked upward from areas of known groundwater contamination to identify areas where releases of chemicals and vadose zone contamination were likely and additional investigation was needed; and
- The disruption to current businesses located within the site was limited by purposefully minimizing intrusive sampling within and under site buildings.

Further explanation of the above guidelines is provided below.

The scope of the RI/FS has been significantly shaped by the site layout, historical operations, and various facilities within the former synthetic rubber plant. The plant was characterized by multiple areas of densely packed chemical storage and processing areas, separated by large areas of open space, parking or administration facilities. Since demolition of the plant, the majority of the site has been redeveloped with closely spaced commercial-industrial buildings. These factors have resulted in sampling locations being concentrated in currently accessible areas where chemical facilities were formerly present.

The RI/FS sampling strategy also considered potential pathways of exposure to site-related chemicals. Data for soil ingestion, dermal contact, and particulate inhalation pathways were collected through surface (less than 1 foot below ground surface [bgs]) and shallow (0-15 feet bgs) soil and soil gas sampling in areas where historical information indicated a potential for chemical releases. Data for evaluation of the vapor inhalation pathway was collected through shallow and deep (>15 feet bgs) soil gas sampling, groundwater sampling, and workplace (indoor) air sampling.

Soil and groundwater samples were initially analyzed for a broad spectrum of COPCs, including VOCs, SVOCs, metals, pesticides, PCBs, and cyanide. Early site-wide data for groundwater and other sampled media indicated that elevated levels of VOCs, particularly benzene, were distributed across a greater area than other detected chemicals. Given this finding and the relative toxicity of benzene and related compounds, VOCs were judged to be the primary risk-driving compounds and chemicals of concern. Therefore, the RI/FS then focused on former plant site facility locations where VOCs were known to have been stored, transported, or used in plant process areas.

The lateral extent of groundwater contamination and characterization of areas/pathways where there could be an imminent human exposure were additionally evaluated as part of the RI/FS. Investigation typically progressed from areas of known contamination to areas of unknown conditions. Areas of known impact at the outset of the investigation included the MW-20 light non-aqueous phase liquid (LNAPL) area, the Waste Pit Area, and an area in the southern portion of the copolymer plant where surface staining was identified from historical aerial photographs. Data were also collected from three large undeveloped areas to evaluate the potential for exposure where extensive areas of surface soil were exposed. The majority of these exposed areas have recently been developed and incorporated into the industrial business park.

The RI/FS investigative strategy included the development and use of multiple lines of evidence to identify and evaluate areas where past chemical releases may have contributed to soil and groundwater contamination. The search for such areas proceeded in both a “top-down” and “bottom-up” fashion. The top-down process started with the findings of the site history investigation, which led to shallow soil and/or soil gas sampling and testing being focused in former process and chemical storage areas and in areas where persistent soil staining was visible on historical aerial photographs. Where elevated chemical concentrations were detected, additional step-out samples were collected and analyzed, and groundwater sampling was performed at locations downgradient of the identified soil contamination.

The bottom-up process started with an independent groundwater plume delineation program that included water table sampling along multiple transects across the site and along segments of the site perimeter. The detection of elevated concentrations of dissolved contaminants in groundwater led to additional groundwater, soil or soil gas sampling in upgradient areas where past releases were suspected to have occurred.

Using the described top-down/bottom-up approach, 12 areas were identified where contaminants were inferred to have migrated downward and impacted groundwater. These areas are referred to as “groundwater contamination source areas” with their locations indicated on Figure 3. Since operations at the former rubber plant complex ceased in approximately 1970, any potential impacts to groundwater would likely have already occurred. Consequently, the application of the independent top-down and bottom-up approaches provides confidence that the most significant areas of soil and groundwater contamination at the site have been identified and evaluated.

The following tasks were completed for the RI/FS:

- Site history review;
- Collection of physical and observational data necessary to develop a hydrostratigraphic model for the site;
- Evaluation of soil and soil vapor contaminants in shallow soil at former rubber plant facilities where VOCs, SVOCs/PAHs, pesticides/PCBs, or metals are suspected to have been used, stored, transported, or disposed;
- Evaluation of potential surface soil contamination in areas where the surface soil was exposed and where there was evidence of surficial spreading of process waste;

**SECTION 2.0**

---

- Focused investigations of the MW-20 LNAPL area and other suspected areas of LNAPL;
- Evaluation of contaminant concentrations in indoor air at current site buildings where soil contamination is known or suspected to extend beneath existing buildings; and
- Delineation of the lateral and vertical extent of groundwater contamination.

A Draft BRA (URS, 2001a) was prepared based on data generated from the above tasks and from investigations conducted outside the RI/FS process, provided they met specific QA/QC requirements, as further discussed in Section 3.1 and Appendix A. Following regulatory agency review of the Draft BRA, data gaps were identified; more specifically, former rubber plant facility locations were identified where additional shallow data were necessary to characterize the potential presence of contaminants. Based on the identified data gaps, additional shallow soil sampling and analyses were completed as part of an Addendum RI (URS, 2002). The primary focus of the Addendum RI was evaluation of the potential presence of PAHs, but to a lesser extent, sampling was also completed for VOCs, pesticides/PCBs, and metals. Following completion of the additional sampling, USEPA indicated that the data set was complete, and no further sampling was required (letter to N. Pasvantis from USEPA [USEPA, 2004a]).

The Revised Draft BRA was submitted in May, 2005.

**2.3 RI/FS DATA REVIEW**

A summary of the risk assessment data assessment is presented below and in Tables 1 through 10 and Figures 3 through 16. The data are segregated according to sample media and the depth range in feet bgs for discussion purposes, as follows:

- Surface soil (0-1 foot bgs)
- Shallow soil (0-15 feet bgs; includes all surface soil samples)
- Deep soil (>15 feet bgs)
- Shallow soil gas (0-15 feet bgs)
- Deep soil gas (>15 feet bgs to groundwater level)
- Indoor air
- Groundwater (limited to water table zone data).

**SECTION 2.0**

---

**2.3.1 Surface Soil**

RI surface soil samples were collected to evaluate the potential for exposure where relatively large areas of soil were exposed (no buildings, asphalt, concrete, or landscaping) and contact with soil would be most likely to occur. Most of these areas have subsequently been redeveloped and are now covered by buildings, pavement, or landscaping, significantly reducing the exposure potential. The areas of exposed soil where RI sampling occurred were (1) in the northwest corner of the former copolymer plant; (2) along the southern boundary of the former rubber plant; and (3) in the southern portion of the former butadiene plant.

Surface soil data include results for VOCs, SVOCs, metals, and pesticides/PCBs. The data are summarized in Table 1, and briefly discussed below.

**VOCs**

Surface soil sampling locations with VOC data are indicated on Figure 4 and are limited to two locations. VOC testing of surface soil samples was generally not completed as part of the RI/FS since volatilization would normally remove VOCs in surface soil over the more than 30 years that have passed since the rubber plant was demolished. VOC detections were limited to low concentrations (0.15 milligrams per kilogram [mg/kg] maximum) of ethylbenzene, acetone, Methyl Ethyl Ketone (MEK), and n-butylbenzene. VOCs were not detected at concentrations in excess of USEPA Region IX Preliminary Remediation Goals (PRGs) for a residential setting.

**SVOCs/PAHs**

Surface soil sampling locations where SVOCs were analyzed are indicated on Figure 5. The most commonly detected SVOCs were pyrene, phenanthrene, and fluoranthene. Detections of SVOCs/PAHs in excess of screening criteria (industrial PRGs) were limited to a single sampling location in the former copolymer plant, as shown on Figure 5.

**Pesticides/PCBs**

Pesticides/PCBs detections were limited to DDT and its isomers (DDD and DDE), dieldrin, Aroclor1260, and "total PCBs". Pesticides/PCBs detected at concentrations in excess of RI screening criteria (residential PRGs) were limited to DDT at three composite sampling locations near the southwest corner of the site, as indicated on Figure 6.

**Metals**

Metals are naturally occurring, and their detection alone does not necessarily indicate a contaminant release. A total of 19 metals were detected in surface soil samples from the site. Metals detected in excess of RI screening criteria (residential PRGs or background levels)

**SECTION 2.0**

were limited to arsenic at three composite sampling locations, two of which were located in the southwestern portion of the site and one in the former butadiene plancor (Figure 7).

**2.3.2 Shallow Soil**

Shallow soil incorporates the zone from the ground surface to 15 feet bgs. The previously discussed surface soil samples are therefore a subset of the shallow soil samples. Shallow soil sample data were used along with shallow soil gas data to evaluate conditions in the vadose zone. Results for shallow soil are summarized in Table 2 and further described below.

**VOCs**

Commonly detected VOCs in shallow soils include ethylbenzene, tetrachloroethene (PCE), benzene, and trichloroethene (TCE). Sampling locations where one or more VOCs were detected at concentrations in excess of RI screening criteria (residential PRGs) are indicated on Figure 8. VOCs detected at concentrations in excess of screening criteria include benzene (11 locations), ethylbenzene (11 locations), TCE (7 locations), styrene (1 location), and 1,2,4-trimethylbenzene (1 location). Elevated VOC concentrations were most prevalent in the tank farm and process areas of the former styrene plancor, at the pits and trenches feature in the southwest corner of the copolymer plancor, and adjacent to a former benzene pipeline in the southern butadiene plancor.

**SVOCs/PAHs**

The most commonly detected SVOCs/PAHs were phenanthrene, pyrene, and fluoranthene. One or more SVOCs/PAHs were detected at concentrations in excess of RI screening criteria (industrial PRGs) at 13 sampling locations, as indicated on Figure 9. The following compounds were detected in excess of the RI screening criteria:

<u><b>Compound</b></u>	<u><b>No. of samples with PRG Exceedances</b></u>
Benzo(a)pyrene	11
Benzo(a)anthracene	3
Benzo(b)fluoranthene	2
Benzo(k)fluoranthene	3
Dibenzo(ah)anthracene	3
Indeno(1,2,3-cd)pyrene	1
Nitrosodiphenylamine	1

Four of the 13 sampling locations with PRG exceedances are located in the vicinity of the copolymer plancor laboratory, while the remaining exceedances are single occurrences located sporadically throughout the site.

**Pesticides/PCBs**

Detections of pesticides/PCBs in shallow soil were limited to DDT and its isomers (DDD and DDE), dieldrin, and the PCB Aroclor 1260. Compounds for which there were screening criteria (residential PRG) exceedances were limited to DDT derivatives (four composite sampling locations) and Aroclor 1260. Figure 10 indicates the sampling locations where screening criteria exceedances occurred. A cluster of three composite samples with DDT exceedances is located near the southwestern corner of the site.

**Metals**

Many metals are naturally present in soil, and thus their detection alone does not indicate a contaminant release. Many metals were detected in samples at or near a frequency of 100%. Shallow soil sampling locations where one or more metals were detected in excess of screening criteria are indicated on Figure 11. Metals detected at concentrations in excess of screening criteria included arsenic (nine samples), copper (three samples), thallium (three samples) and lead (one sample). Clusters of locations with metal exceedances occurred in the vicinity of the former copolymer plant laboratory and near wastewater treatment facilities in the northern butadiene plant. Residential PRGs served as the screening concentrations for these metals except for arsenic. Arsenic is naturally present at concentrations above the PRG, and therefore the maximum background level of 10 mg/kg served as the screening criteria for this metal. The background evaluation for metals is summarized in Appendix B.

**2.3.3 Deep Soil**

Deep soil data were collected in a limited number of locations, typically where there was evidence of overlying shallow soil contamination. Exposure pathways for deep soil contaminants are limited to upward migration of volatilized contaminants to the surface. For this reason, deep soil data summarized here are limited to VOCs. Deep soil VOC data are summarized in Table 3.

**VOCs**

The most frequently detected VOCs in deep soil were benzene, toluene, ethylbenzene and styrene. VOCs detected at concentrations in excess of RI screening criteria (residential PRGs) were limited to benzene (34 samples) and ethylbenzene (two samples). The majority of sampling locations with elevated benzene occur near a former underground benzene pipeline in the southeast corner of the former butadiene plant and near the western site boundary, near an area of known benzene NAPL (Figure 12).

**SECTION 2.0**

---

**2.3.4 Shallow Soil Gas**

Shallow soil gas data were collected along with shallow soil data to evaluate vadose zone conditions. Shallow soil gas data is limited to VOCs and is available for approximately 875 soil gas sampling locations at the site. The shallow soil gas VOC data are summarized in Table 4.

For the VOCs routinely analyzed for, the most frequently detected VOCs were PCE and BTEX compounds. Figure 13 presents a map indicating the distribution of sampling locations with various ranges of total VOC concentrations. Sampling locations with the more elevated total VOC concentrations tend to be clustered in the tank farm and process areas of the former styrene plancor, in the southwest corner of the former copolymer plancor, in the vicinity of a former laboratory in the butadiene plancor, and near a former benzene pipeline at the southern end of the butadiene plancor.

**2.3.5 Deep Soil Gas**

Deep soil gas VOC data are limited to 12 locations in the former styrene plancor (Figure 14) in the vicinity of known NAPL. Sampling depths for all locations were between 47 and 59 feet bgs, immediately above the water table at the time of sample collection.

Deep soil gas VOC data are summarized in Table 5. While 10 different VOCs were detected, benzene concentrations were far higher than other VOCs at all locations, with concentrations ranging from 1,760 to 30,800 parts per million by volume (ppmv). It is inferred that the source of the elevated deep soil gas concentrations is volatilization of benzene dissolved in groundwater, as further explained in the Revised Draft Soil and NAPL RI report (in progress).

**2.3.6 Indoor Air**

Indoor (workplace) air monitoring was conducted to evaluate the potential for worker exposure to VOCs (URS, 2001b). Sampling was performed at 13 site buildings (Figure 15) that overlie or are immediately adjacent to (within 25 feet of) areas where vadose zone VOC contamination was either known or suspected to be present.

Workplace air monitoring data are summarized in Table 6. In addition to indoor air samples, ambient (outside) monitoring data were collected during the study to establish ambient background levels for the site. These data are summarized in Table 7.

**SECTION 2.0**

Detected indoor air VOC concentrations were all less than OSHA permissible exposure levels (PELs) and PEL/20 (i.e., 5% of the PEL) evaluation criteria, indicating that no immediate health risk existed at the time of the sampling. Seven compounds (benzene, chloroform, ethylbenzene, methylene chloride, PCE, TCE, and xylenes) were detected at concentrations above the ambient air PRGs. Additionally, comparison of the indoor air and outdoor air measurements presented in Tables 6 and 7 indicated that it was likely the ambient outdoor air quality had a significant impact on the indoor air concentrations.

While the detected concentrations of the above compounds exceeded ambient air PRGs, it is uncertain whether this was the result of subsurface sources. More specifically, the elevated VOC concentrations may have been associated with ambient (background) air, activities conducted within the sampled buildings, the upward migration of subsurface contamination, or some combination of these. The relative contribution from these possible sources is unknown.

**2.3.7 Groundwater**

Groundwater data relevant to this BRA are limited to VOC concentrations, by which risks for upward migration of vapor from the water table can be evaluated. Potential health risks associated with direct exposures to groundwater have been previously evaluated in the Groundwater Risk Assessment (McLaren Hart and Dames & Moore, 1998).

Groundwater data used for this BRA are limited to VOC data from the water table for the July 2000 groundwater monitoring event. Data from this monitoring event are summarized in Table 8, and indicate that benzene was detected at concentrations in excess of its screening level (drinking water maximum contaminant levels [MCLs]) at a far greater frequency than other VOCs. Other VOCs commonly detected in excess of their MCLs included TCE, and chlorobenzene. Individual VOC plumes are typically subsidiary to the benzene plume, and the distribution of benzene is therefore used as an indicator of the extent of dissolved VOC contaminants in the water table, as presented on Figure 16.

**2.3.8 NAPL**

Areas of non-aqueous phase liquids (NAPL) at the site are discussed in the RI report with respect to whether they are light (LNAPL) or dense (DNAPL), and the following three categories:

- (A) **Areas where NAPL is potentially present, but has never been observed or measured.** NAPL is judged to be “potentially present” based on dissolved

**SECTION 2.0**

---

concentrations at a significant fraction of applicable solubility limits (5% for LNAPL components, 1% for DNAPL components) and associations with former or existing facilities where VOCs are known or likely to have been used, stored, or disposed.

- (B) **Areas where NAPL is present, but at residual (non-mobile) saturations**, as evident from soil core jar testing, laboratory measurements of hydrocarbon saturations, and the lack of any direct observation of NAPL accumulation at groundwater monitoring locations. Saturations of less than 16% are inferred to be indicative of residual levels, based on data presented in the MW-20 Pilot Program (URS, 2003). All areas meeting these criteria lie entirely within a larger, potential NAPL area, as described in “A” above.
- (C) **Areas where NAPL accumulations have been observed or measured within a monitoring well or temporary well point.** This occurrence is distinguished from A and B above in that remediation by direct NAPL removal techniques (NAPL pumping or bailing) can be considered as part of the FS process. NAPL accumulations may be due to zones of higher NAPL saturations within the larger NAPL area. Zones of NAPL saturation above residual levels are not an indication of active NAPL migration. Based on the results of the MW-20 Pilot Program (URS, 2003), NAPL in areas of high saturation do not appear to be mobile. NAPL accumulations within monitor wells may also be associated with the physical disturbance of soil that occurs during well drilling, which may trigger NAPL migration from soil pore spaces into a well, even where present only at residual saturations.

Site areas corresponding to the above categories are indicated on Figure 17. The primary contaminants present in the LNAPL site areas are benzene and ethylbenzene. The LNAPL in the laboratory and pipeline area near the eastern site boundary is inferred to be a complex of BTEX, styrene, and numerous other VOCs, SVOCs, and unidentified compounds in the C10-C23 range.

The NAPL areas presented in Figure 17 are applicable to the water table zone. NAPL is also known to be present in the vadose zone, but at residual saturations, at the VOC tank farm, laboratory and pipeline, and benzene feedstock pipeline areas. NAPL cannot migrate under natural conditions when at or below residual saturation levels.

No areas of DNAPL accumulations are known to exist at the site. The areas of potential DNAPL presented along the western site boundary on Figure 17 are both adjacent to known offsite source areas. The more northerly potential DNAPL areas is associated with TCE and

## **SECTION 2.0**

## **SITE BACKGROUND AND REMEDIAL INVESTIGATION/FEASIBILITY STUDY**

---

PCE, with the maximum dissolved concentration occurring in offsite wells located on the American Polystyrene Co. and PACCAR properties. The potential DNAPL area near the southwest corner of the site is associated with chlorobenzene, which was used extensively for DDT production at the nearby Montrose property.

**3.0 DATA EVALUATION AND ANALYSIS**

Review and analysis of the data involved the following processes: (1) data validation and selection for use in the risk assessment; (2) data processing related to composite samples, fixed laboratory and mobile laboratory results for the same sample location, and conversion between soil gas and soil matrix samples; (3) selection of Exposure Areas of Potential Concern (EAPCs); (4) selection of COPCs; and (5) calculation of Exposure Point Concentrations (EPCs) for use in calculating cancer risk and noncancer hazard estimates. These five processes are described below.

**3.1 DATA VALIDATION AND SELECTION**

The Data Quality Objectives and work plan rationale for the RI/FS Work Plan (Dames & Moore, 1993) describe the fundamental strategy underlying all of the RI tasks, including the data collection. Investigations were focused to characterize potential soil and/or groundwater contamination in locations where site history indicated the potential for contamination from specific operations or facilities formerly present at the site. Thus, the RI/FS strategy implied a fundamental assumption that contamination would be highest in the vicinity of former chemical facilities, as discussed in Section 2.0.

The data set is inherently conservatively biased for estimating exposure concentrations because the sampling effort was focused on the areas where contamination was most likely. For example, data were collected along a former pipeline suspected or known to have carried chemicals. These data will produce a conservatively biased estimate of exposure point concentration, and therefore risk, in an exposure area surrounding the data.

All RI/FS data were collected and validated according to the Quality Assurance Project Plan (QAPP) for the RI/FS. The RI/FS data that were deemed acceptable through the validation process were carried forward into the risk assessment. A comprehensive presentation and discussion of the RI/FS data set will be presented in the forthcoming RI report for the Soil and NAPL operable unit.

A substantial volume of laboratory analytical data for samples from the former plant complex has been generated outside of the current RI being conducted by URS Corporation (formerly Dames & Moore) for the Respondents. These data typically originate from investigations conducted on behalf of individual property owners, and were completed by numerous investigators, including Dames & Moore and other consulting firms. In some cases, the data have been independently submitted to USEPA.

A review was undertaken to determine which, if any, of the non-RI/FS data described above could be used in the RI and risk assessment. The minimum acceptance criteria were based on a subset of principles given in the USEPA National Functional Guidelines (USEPA, 1999a) as well as supporting information, as described in Appendix A. Data for 112 of the 254 non-RI/FS samples (44%) were found to be suitable for use in the risk assessment, 91 (36%) were qualitatively acceptable, and 51 (20%) were unacceptable. The 112 non-RI/FS samples considered suitable were incorporated into the database and used quantitatively in the risk assessment. Rejected data were deemed unreliable and excluded from use in the risk assessment in a similar manner to data generated from the RI/FS that was rejected based on criteria specified in the National Functional Guidelines.

The final data set used in the Del Amo BRA was comprised of all RI/FS and non-RI/FS data that were retained after the data validation process. A summary of the chemicals detected at the site is presented in Table 9. A summary of data types available for each parcel is provided in Table 10. The parcels included in Table 10 are limited to the site EAPCs, the selection of which is described in Section 3.3.

## **3.2 DATA PROCESSING**

The parcel-specific data sets were put through three data processing steps. The first step ensured appropriate treatment of composite samples. The second step involved an algorithm to select the appropriate data when both fixed laboratory and mobile laboratory results were available for the same sampling location. Lastly, VOC concentrations from soil gas samples were converted to equivalent soil matrix concentrations, and vice versa, so that the final soil matrix data sets include values estimated from soil gas concentrations, and the final soil gas data sets include values estimated from soil matrix concentrations. Data presentations in this report reflect the described data processing and combine the soil gas and soil matrix data. This was done to make the maximum use of the available data for each parcel. Details regarding each of the three data processing steps are provided below.

### **3.2.1 Composite Samples**

The soil data included results for multiple composite samples, with each sample composed of between two to six discrete samples. Some of the composites were made up of individual samples that were collected across parcel boundaries. Each composite result was treated as a mean concentration for the individual samples contributing to the composite, and was assigned to each of the individual sample locations for the purposes of the risk assessment. The average is the main population parameter of interest and, therefore, use of the composite results in this way does not have a substantial impact on the calculation of the average exposure point concentration. Furthermore, any impact is unbiased on the average. However,

the use of composite data may decrease the variance in the data set and therefore cause a decrease in the 95UCL calculations. The maximum detected concentrations were used in the reasonable maximum exposure (RME) risk calculations for EAPCs with composite samples to account for this.

### 3.2.2 Fixed Laboratory and Mobile Laboratory Results

A majority of the soil gas data was generated by an onsite mobile laboratory. The mobile laboratory data were determined to be sufficiently reliable to be used in the risk assessment, as documented in a previously submitted Technical Memorandum (URS, 2000).

Split soil gas samples were collected for approximately 12% of the soil gas sampling locations, with one portion being analyzed for VOCs by the onsite mobile laboratory, and the second portion analyzed for VOCs by an offsite, fixed laboratory. The following algorithm was applied for all samples where both mobile and fixed laboratory results were available to determine which results would be used for the risk assessment:

- If both results were detects, then the highest detection was used;
- If one result was a detection and the other a non-detect, then the detected value was used;
- If both results were non-detects, then the lowest detection limit was used.

### 3.2.3 Soil Gas and Soil Matrix Conversion

Soil gas results were used to estimate soil matrix concentrations to support risk calculations involving direct soil exposure pathways. Soil matrix results were additionally used to estimate soil gas concentrations to support risk calculations for indirect exposure via inhalation. The sample concentrations were converted using the site-specific soil properties to calculate a soil matrix concentration corresponding to the soil gas concentration, and vice versa.

The following equation (Feenstra et al., 1991) was used to convert between the two types of results:

$$C_t = C_g \left[ \left( \frac{\theta_w}{K_H} \right) + (\theta_t - \theta_w) + \left( \frac{P_b K_d}{K_H} \right) \right] * (1/P_b) * (CF_1) * (CF_2) * (CF_3)$$

where

$$C_t = \text{total soil concentration (mg/kg)}$$

- $C_g$  = vapor concentration ( $\mu\text{g}/\text{m}^3$ )  
 $K_H$  = Henry's law constant (chemical-specific; unitless)  
 $K_d$  = soil-organic carbon distribution coefficient (where  $K_d$  = fraction organic carbon (foc) x organic partition coefficient ( $K_{oc}$ )) ( $\text{cm}^3/\text{g}$ )  
 $\theta_w$  = soil moisture content (unitless)  
 $\theta_t$  = total porosity (unitless)  
 $P_b$  = soil bulk density ( $\text{g}/\text{cm}^3$ )  
 $CF_1$  = conversion factor ( $\mu\text{g}/\text{m}^3$  to  $\text{mg}/\text{L}$ ;  $1\text{E}-06$ )  
 $CF_2$  = conversion factor ( $\text{L}/\text{cm}^3$ ,  $1\text{E}-03$ )  
 $CF_3$  = conversion factor ( $\text{g}/\text{kg}$ ,  $1\text{E}+03$ )

The chemical-specific parameters used in this equation (the  $K_{oc}$  for  $K_d$  calculation and  $K_H$ ), were obtained from three literature sources, in the following order of preference. The first source was the Region IX PRG tables (USEPA, 2002a). If a chemical was not listed in this source, the next source used was the Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites Soil Screening Guidance: Users Guide, Appendix C (USEPA, 1996 and 2002c), followed by the Johnson and Ettinger model spreadsheet (USEPA's version of the Johnson and Ettinger Model, 2000a). Data were not found in a few cases. Typically, these chemicals do not have toxicity criteria; therefore, they were assigned surrogate chemicals and were carried forward into the quantification of cancer risk and noncancer hazard. The default value of 0.006 from the Soil Screening Guidance (USEPA, 2002c) was used for the fraction organic carbon (foc).

Site-specific soil parameters were developed based on soil physical testing results from the RI/FS. The values used in the conversion equation were derived to represent mean conditions in the 0 to 15 feet depth bgs, and were therefore assigned as the geometric mean of the 17 physical property samples collected between 0 to 15 feet bgs and analyzed for soil physical parameters. These soil samples had a geometric mean bulk density of 1.63, a total porosity of 38%, and a water-filled porosity of 25%. These geometric mean soil property values were used in the conversion equation (see Table D-1).

The table below presents an example of the chemical-specific inputs and results for conversion of soil gas concentrations to soil concentrations, and is specific to PCE and TCE for a sample from parcel 7351-034-015,-050,-056:

Chemical	$K_{oc}$	$K_d$	$K_H$	Soil Gas Concentration ( $\mu\text{g}/\text{m}^3$ )	Converted Soil Concentration (mg/kg)
Tetrachloroethene	160	9.3E-01	7.5E-01	1.4E+06	2.2
Trichloroethene	170	1.0E+00	4.2E-01	2.9E+06	8.1

The table below presents an example of the chemical-specific inputs and results for conversion of soil matrix detections of benzene and styrene in Parcel 7351-034-015,-050,-056 to their estimated soil gas concentrations:

COPC	K <sub>oc</sub>	K <sub>d</sub>	K <sub>H</sub>	Soil Concentration (mg/kg)	Converted Soil Gas Concentration (µg/m <sup>3</sup> )
Benzene	58.9	3.5E-01	2.3E-01	0.928	4.1E+05
Styrene	780	4.7E+00	1.1E-01	15000	3.4E+08

The above approach was used to maximize the use of the large amount of soil gas data collected at the site. While this approach provides for a larger data set for the risk assessment, the partitioning relationship used to correlate soil and soil gas concentrations has limitations and may over- or under-estimate the calculated concentrations. These limitations include:

- The soil-organic carbon distribution coefficient is dependent on the fractional organic carbon content of the soils. This property will vary from sample to sample; however, a typical value suggested in the USEPA Soil Screening Guidance (0.006; USEPA, 2002c) was used for the fraction organic carbon (foc) for these calculations.
- Studies comparing the relationship between soil and soil gas concentrations from data collected across the country do not demonstrate a good correlation (USEPA, 2002b and CalEPA, 2005). This may be due to the variability of VOC concentrations in soil samples.
- The partitioning equation does not consider the presence of a residual NAPL phase. If residual NAPL is present, then the estimated soil gas concentrations converted from soil concentrations may be biased high and the estimated soil concentrations converted from soil gas concentrations may be biased low. However, based on the RI/FS results and field observations, the presence of NAPL in the vadose zone soils at the site is thought to be limited in extent.

### 3.3 EXPOSURE AREAS OF POTENTIAL CONCERN

The site EAPCs are defined to reflect the current use of the site as an industrial park and are based on parcel boundaries assigned by the Los Angeles Land Assessors Office. Each parcel generally includes the land and any associated building(s). The designation of selected parcels as exposure areas is a reasonable approach in that they not only encompass what a

typical industrial lot size would be in the area, but also provide information for use in later remedial decisions for the site. Furthermore, a hypothetically exposed receptor would only be exposed to contaminants within the building and land parcel where they work. Thus, each parcel should be evaluated as a separate exposure area.

A parcel was selected as an EAPC if it met one or more of the following criteria:

- (1) The parcel overlapped one or more of the 12 groundwater contamination source areas defined in the Soil and NAPL RI Report (URS, 2004), as presented on Figure 3. Groundwater contamination source areas typically encompass areas of elevated VOCs in soil and/or soil gas samples associated with an underlying groundwater contaminant plume.
- (2) One or more VOCs, SVOCs, pesticides, or PCBs were detected in samples from the parcel at levels exceeding their respective Region IX or CAL-Modified PRGs for residential soil (Figures 4, 5, 6, 8, 9, 10 and 12). This includes soil gas samples that were converted into equivalent soil matrix values.
- (3) One or more metals were detected at the parcel at a level above their respective ambient breakpoints, as discussed in Section 3.4, and above their respective Region IX or CAL-Modified PRGs (Figures 7 and 11).
- (4) The parcel was surrounded by other parcels that were selected as EAPCs.

A parcel was not selected as an EAPC if it did not meet any of the above criteria, and it is inferred that no significant health risks are associated with such parcels.

EAPCs are listed in Table 11 along with the rationale for their selection. EAPC locations are presented on Figure 3. As indicated on the figure, the EAPCs include two street segments.

Two of the potential receptors associated with each EAPC were trench workers and hypothetical future residents. Exposure areas for these receptors are not parcels, but are defined by the size and location of a typical trench and residential lot, respectively. A typical trench could be located in any outdoor area at the site. In general, smaller exposure areas will have a higher EPC, because a receptor may be exposed to only a small hot spot area. If a receptor is exposed to a larger area that includes a small hot spot, their exposure to the hot spot is more limited because they are assumed to also spend time in the other parts of the area. Risks for the trench worker and residential scenarios were estimated based on the maximum concentrations at each parcel since risk estimates for every conceivable trench or hypothetical future residential lot location is impractical. This is a very conservative, worst-

case approach, since it is unlikely a trench worker or hypothetical future resident would contact the maximum concentration of each chemical for the full duration of exposure assumed in this assessment.

### 3.4 IDENTIFICATION OF CHEMICALS OF POTENTIAL CONCERN

COPCs were selected for each EAPC and each environmental medium (surface soil, shallow soil, shallow soil gas, deep soil, deep soil gas, and groundwater). All chemicals detected in at least one sample in the site data set were considered in the COPC selection process, including those chemicals not known to be associated with former rubber plant operations. Chemicals were selected as COPCs based on their individual prevalence and concentration relative to Region IX or CAL-modified PRGs, as explained in the following subsections. Metals were additionally screened based on their concentrations relative to background, as explained below.

#### 3.4.1 Background Analysis

Background concentrations are those chemical concentrations, either native or anthropogenic, that are present in the environment but not associated with any site activities. Background concentrations can exceed concentrations that equate to cancer risks of  $10^{-5}$  and higher for some chemicals, such as arsenic and carcinogenic polynuclear aromatic hydrocarbons (cPAHs). Therefore, the selection of COPCs in a risk assessment only includes those chemicals that have concentrations above their respective background concentration. Since metals are naturally occurring in the environment, background concentrations are generally applied to metals detected at a site. For this risk assessment, the background comparison for metals used a “weight of evidence” approach based on CalEPA guidance (CalEPA, 1997), but modified to consider background level data for some metals from Southern California (CalEPA, 1996).

Background concentration information for cPAHs is also discussed in this section, as cPAHs are found in Southern California soils. CalEPA has approved a methodology and regional data set for comparing site cPAHs. Background concentration of cPAHs was not used in the COPC selection process, but was used for the purpose of discussing potential risks associated with cPAHs in the risk characterization section.

##### 3.4.1.1 Metals

A site-wide background comparison for metals was conducted following the approach put forth by CalEPA (1997) for inorganic chemicals. This approach evaluates whether the data distributions reflect single lognormal or normal populations, or contain multiple populations

that would indicate contamination in addition to ambient levels. CalEPA recognizes that the data set analyzed might include data from areas that are possibly contaminated with only a single or a few metals, and refers to this data set as the “expanded data set”. CalEPA recommends a “weight of evidence” approach where three indicators of whether background is exceeded are considered. The three criteria include: (1) the degree to which the site data distributions are fit by a lognormal or normal distribution, (2) a graphical assessment (probability plot against the normal or lognormal distribution) to identify breaks or nonlinearity indicative of more than a single population; and (3) the skewness of the data as indicated by the coefficient of variation ( $CV = \text{standard deviation/average}$ ) and the data range (order of magnitude difference between the maximum and minimum concentrations). A fourth criterion was additionally included in this evaluation based on a comparison between the site metals concentrations and background concentrations for Southern California (CalEPA, 1996). The advantage of using a site-wide data set over smaller localized data sets is that statistical power (the power to discriminate an elevated concentration above background) will theoretically be increased because of the increase in sample size.

The background comparison for metals is described in Appendix B. The background analysis was used to establish concentrations that individual sample results could be compared to in the EAPC selection process. The analysis focused on surface and shallow soil samples that were collected before the RI sampling that was conducted in 2003 (i.e., the 2003 RI data was not included in the background metals analysis). However as discussed below, the 2003 RI data were reviewed with respect to pre-2003 data and only one metal, lead, was detected in 2003 at higher concentrations that warranted consideration in the background analysis. Ambient versus non-ambient breakpoints were developed by evaluating the Q-Q plots in Appendix B for each of the nine metals that were determined to have concentrations above background, with the exception of thallium, for which there were only two detects. For thallium, the breakpoint was defined to be the mean of the regional background data. For the remaining metals, Q-Q plots were used to identify the breakpoint above which a second (non-ambient) population was indicated. These breakpoints are as follows: arsenic (10 mg/kg), cadmium (2 mg/kg), chromium (60 mg/kg), copper (150 mg/kg), nickel (25 mg/kg), manganese (450 mg/kg), thallium (0.3 mg/kg), vanadium (65 mg/kg), and zinc (170 mg/kg). Figures 7 and 11 highlight each location where the concentration exceeded both the ambient/non-ambient breakpoint and the residential soil PRG.

### Surface Soil

The Shapiro-Wilk test found a significant discrepancy with a normal and lognormal distribution fit for cadmium, chromium, manganese, nickel, vanadium, and zinc in surface soil. In general, these discrepancies were corroborated by either or both of the Q-Q plot evaluation or range and CV evaluation. These metals were also found to exceed regional background levels for Southern California with the exception of manganese, for which

background data were not available. Including manganese, these metals were identified as being present at the site at concentrations above background, and therefore remained in the COPC selection process. Although the Shapiro-Wilk test found that arsenic and copper data fit a lognormal distribution, these metals were identified as above background based on the bend apparent in each of their graphical evaluations as well as their exceedance of a second criterion. The CV exceeded one (1) in the case of arsenic and the site distribution exceeded the Southern California background distribution in the case of copper.

The remaining metals were identified as being within background concentrations and were screened out of the COPC selection for surface soils. The Q-Q plot for cobalt indicated a number of low-precision samples that consisted of detections having the same value for some unknown reason (possibly truncation of significant figures). The anomaly created by the horizontal spread of these points is obvious and results in an interference of the Shapiro-Wilk tests. However, based on a visual evaluation of the Q-Q plot, the fit for cobalt was classified as “ok”. Because of the Q-Q plot assessment and the low range and CV, cobalt was excluded from the list of metals above background. Interference due to the low frequency of detections made the Shapiro-Wilk test also unreliable for silver and mercury in surface soils. Silver had 2 detections out of 15 samples, while mercury had 6 detections out of 15 samples. The ranges and CVs calculated from the data sets, as well as visual inspection of the Q-Q plots, implied that the underlying distributions had low skewness for silver and moderate skewness for mercury. In the case of mercury, the CV of 1.6 provided evidence for multiple distributions above ambient. However, the Wilcoxon test found no significant difference between the site and the regional background data sets for mercury and silver. Therefore, these metals were concluded to be within background and excluded from the COPC selection process.

The Wilcoxon test is robust to different distribution assumptions, and therefore it can provide approximate results, even for cases where the distribution type is unknown. Therefore, the Wilcoxon Two Sample test was performed for silver and mercury. The Wilcoxon test indicated that both silver and mercury could not be distinguished from the background data set. The P values of the Wilcoxon test for these two metals are considered approximate due to their high frequency of non-detected values. However, because the detection limits of these data sets are relatively low, and very close to the detection limits of the associated background data set, the overall outcome of significance or non-significance is considered robust and reliable.

### **Shallow Soil**

Results for shallow soils were similar but not identical to those for surface soils. Cadmium, chromium, copper, nickel, manganese, vanadium, and zinc were again identified above background due to their lack of fit to the two distribution types (i.e., Shapiro-Wilk P value < 0.03 and breaks or outliers in the Q-Q plot). Cadmium and vanadium also had data

distributions which were significantly above the regional background level, while chromium, manganese, nickel, and zinc did not exceed background. Thallium was detected at concentrations that were above the mean regional background concentration. Therefore, thallium was identified as above background for shallow soils on the site.

The remaining metals were well fit by normal or lognormal distributions, were not significantly above the regional background data set, and were therefore excluded from the list of metals above background in shallow soil. Lead was identified as within background previously, however higher concentrations of lead were observed in shallow soil in the RI data collected in 2003 than observed previously. Therefore, lead was evaluated as a potential COPC for shallow soil in parcels where maximum concentrations in the 2003 data set were higher than previously observed. Other metal concentrations were generally the same as observed previously so their background status was not changed. In the case of arsenic, the shallow soil data included many more low concentrations than the surface data. These low concentrations resulted in a nearly linear Q-Q plot. Therefore, arsenic was excluded from the list of metals above background for the shallow soil layer. In the case of silver and mercury, the P values of the Wilcoxon test are again considered approximate due to the high frequency of non-detected values. However, because the detection limits of these data sets are relatively low, and very close to the detection limits of the associated background data set, the overall outcome of significance or non-significance is again considered robust and reliable.

#### **3.4.1.2 Carcinogenic Polynuclear Aromatic Hydrocarbons**

A site-wide background comparison for cPAHs was conducted based on the benzo(a)pyrene equivalent (BAP-eq), calculated for each sample. All soil samples collected from the site, including the 2003 RI data set, were evaluated in this background evaluation of cPAHs. If at least one cPAH was detected in a given sample, the BAP-eq was calculated based on summing the products of the CalEPA Toxicity Equivalency Factors and the concentrations of detected cPAHs, or one half the detection limits for non-detected cPAHs. The BAP-eq values are presented in Table D-2 in Appendix D.

Southern California background levels of cPAHs have been documented in ENVIRON (1998). Concentrations of BAP-eq in the background data set range up to 4.05 mg/kg with a 95UCL of 0.24 mg/kg. An upper tolerance limit (UTL) of 0.9 mg/kg was developed to assist in remedial decision making for individual sample comparisons. Concentrations of BAP-eq below the UTL can be considered within background. When comparing the site BAP-eq results with the background data set, one sample (19 mg/kg in SBL0337) out of 298 samples analyzed in shallow soils exceeded the maximum background value of 4.05 mg/kg. Five additional samples exceeded the UTL of 0.9 mg/kg, with the two highest concentrations

occurring in SBL0309 (3.44 mg/kg) and SBL0330 (3.51 mg/kg). The remaining three samples had BAP-eq concentrations of 1 mg/kg or less.

### 3.4.2 COPC Selection

COPCs were selected for each EAPC and each soil and soil gas layer (surface, shallow and deep) based on three criteria: (1) comparison to background for inorganic chemicals; (2) prevalence for organic chemicals; and (3) comparison to a toxicity threshold value for both inorganic and organic chemicals. COPCs were independently selected for the surface (0-1 foot bgs) and shallow (0-15 feet bgs) soil layers because surface soil contamination may reflect much different source and transport processes than subsurface contamination. COPCs were also selected for shallow ( $\leq 15$  feet bgs) and deep ( $>15$ ) soil gas layers. COPCs were selected for each EAPC and for each of these layers as follows.

#### 3.4.2.1 Organics

Organic chemicals were selected as COPCs for a given EAPC and soil layer, if:

- (1) The percentage of positive detections for that EAPC and soil layer exceeded a 5% prevalence screen (i.e., the chemical was positively detected in at least 5% or more of the samples) and
- (2) The maximum concentration for that EAPC and soil layer exceeded the toxicity threshold of 1/10 the PRG (residential soil PRG for VOCs, SVOCs, pesticides, and PCBs, and the ambient air PRG for VOCs detected in soil gas [USEPA, 2002a]).

The above criteria are judged to be adequately protective of additive risk (adding risk from multiple chemicals) while focusing the risk assessment on the most prevalent chemicals.

#### 3.4.2.2 Metals

Metals were considered for COPC status if they were detected on the site and were above background, as previously discussed in Section 3.4.1.1. Additional screening was then conducted to select inorganic COPCs on an EAPC- and soil layer-specific basis. A metal was selected as a COPC for a given EAPC and soil layer if the maximum detected concentration in that EAPC and soil layer exceeded the toxicity threshold of 1/10 of the residential soil PRG or where available, the CAL-Modified PRG (USEPA, 2002a).

The results of the COPC selection process for organics and metals are presented in Appendix C.

### 3.5 EXPOSURE POINT CONCENTRATIONS

Estimates of chemical concentrations at points of potential human exposure are necessary for evaluating chemical intakes by potentially exposed individuals. Development of long-term exposure point concentrations from point-in-time data requires an underlying assumption about the representativeness of the data with respect to current and future exposure conditions for a receptor at the site. The issues of temporal and spatial representativeness of the data are discussed below.

The concentration used to estimate the RME in modeling a receptor's spatial and temporal integration exposure is defined by USEPA (1992b) as the 95% upper confidence limit (95UCL) of the arithmetic mean or the maximum observed concentration, whichever is lower. The arithmetic mean reflects the assumption that exposure is averaged by the receptor as they traverse an area over time. The intent of the RME scenario is to focus the assessment on a conservative exposure that is the maximum exposure that is reasonably expected to occur (USEPA, 1989). Because of the multiple conservative assumptions used in the risk assessment process, the RME is often a high-end estimate of exposure and risk. For the Del Amo site, exposure point concentrations (EPCs) for a commercial exposure scenario were calculated for each EAPC as a 95UCL of the arithmetic mean (using one of three formulae) or the maximum observed concentration, whichever is lower. EPCs for the hypothetical future resident and trench worker were based on the EAPC maximum concentrations of COPCs (maximum detect concentrations or  $\frac{1}{2}$  the detection limit for non-detects, whichever was higher).

The three formulae used for calculating the EPC included: (1) the t-based confidence interval; (2) the Land confidence interval for lognormal distributions; and (3) the Chebychev conservative confidence interval, which is expected to be conservatively biased for most skewed distributions. The selection between these three types of 95UCLs and the fourth alternative, the maximum concentration, was performed objectively based on a selection algorithm, as discussed below. Exposure point concentrations for each exposure area were then used to calculate intake for each COPC, which in turn was used to calculate hazard and cancer risk. For comparison, a central tendency estimate of risk was also calculated based on the arithmetic average concentration for each EAPC.

Exposure point concentrations were calculated within each EAPC for commercial, hypothetical future residential and trench worker exposures under the assumption that environmental concentrations would remain constant for an indefinite period of time at levels

reflected in the data. No abiotic or biotic degradation mechanisms, which reduce the concentrations of COPCs over time, are assumed to occur. This general assumption of steady-state conditions also applies to sources and chemical release mechanisms and results in a conservative estimation of long-term exposure concentrations. Biodegradation was assumed to occur as chemicals migrate from the source area for exposure pathways in which site-specific modeling was conducted (indoor air pathway). While this results in reduced exposure point concentrations for the indoor air pathway over time, the source term was assumed to be constant through time.

Site soil data were divided into two categories: (1) surface soil data (less than 1 foot bgs and (2) shallow soil data (0 to 15 feet bgs). Surface soil data were evaluated for a commercial worker since they are the current receptors of concern present at the site and may be exposed to chemicals in surface soils. Site soils may be graded and moved around the site in the future, resulting in soils deeper than 1 foot bgs to be brought to the surface. The shallow soil data collected between 0 and 15 feet bgs were evaluated for a commercial worker, hypothetical future resident and a trench worker to account for this potential future change.

Exposure point concentrations were calculated based on the data set resulting from the processing steps described in Section 3.2, and an additional step wherein non-detect results were replaced with a concentration equal to half the associated detection limit. For commercial exposures, the RMEs for each COPC were based on the 95UCL concentration for all sampling locations within an EAPC. For hypothetical future resident and trench worker exposures, EAPC maximum concentrations (maximum detect concentrations or  $\frac{1}{2}$  the detection limit for non-detects, whichever was higher) were used as the EPCs. The use of the maximum value for each COPC for these two exposure scenarios is based on the assumption that a house or trench can be placed in the area of maximum contamination. This is a very conservative use of the data set, as it is unlikely that all of the maximum concentrations are co-located. More likely, they are spread throughout a portion of the EAPC, thus precluding the simultaneous exposure of a hypothetical future resident or trench worker to all the data maximums. The parcel-specific EPCs used in the risk assessment are presented in Table 12a.

### 3.5.1 Alternatives for the Reasonable Maximum EPC

USEPA guidance suggests the RME estimate of the EPC be based on the t-based confidence interval and the Land confidence interval for estimating the upper 95% confidence limit, as well as the data maximum. In light of the limitations of the Land UCL (discussed in USEPA technical papers as described below), the Chebychev theorem in classical statistics (USEPA, 1997c) was also incorporated in RME calculations. The Chebychev formula provides an estimate of the 95UCL which is known to be conservative (offering higher than 95% confidence on average), regardless of distribution shape, when used with either the true or

estimated standard deviation of the population. The formula has a form similar to the more familiar t-based confidence interval, but with a higher multiplier applied to the standard error term:

$$95^{\text{th}}\% \text{ UCL} = \bar{\mu} + (4.47 * \bar{\sigma} / \sqrt{N})$$

$\bar{\mu}$  and  $\bar{\sigma}$  are either arithmetic estimators of the mean and standard deviation (unbiased for any distribution type) or estimators based on lognormal theory. This and other estimators for the 95UCL are discussed in more detail in USEPA 1997c.

### 3.5.2 Procedure for Selection of Statistical Estimator

The EPCs were selected from the four alternative EPCs (t-based, Land, Chebychev, and data maximum) that were calculated for every EAPC, chemical, and soil layer. The number of chemicals and exposure areas make a case-by-case selection of EPCs impractical to conduct, document, and review for the Del Amo site. Therefore, a set of decision rules was applied as discussed below. Applying a set of rules has the advantage of making decisions explicit and repeatable, but has the disadvantage of possibly oversimplifying decisions that require statistical judgment. The decision rules were applied in an automated fashion for all the cases to address these issues, but supplementary information has also been provided. This supplementary information was evaluated as part of the uncertainty analysis described in Section 7.0.

The first step in selecting an EPC was to evaluate the distribution fit for both a normal and lognormal distribution using the Shapiro-Wilk goodness-of-fit test. If the fit to one of these distributions was adequate (defined as  $p\text{-value} \geq 0.05$ ), then the UCL formula associated with the best fitting distribution type (t-based or Land) was selected as the “preferred estimator”. If neither distribution fit, but sample size was adequate, the Chebychev formula was chosen as the preferred estimator. If the preferred estimator was lower than the data maximum, it was assigned to be the EPC. If the preferred estimator exceeded the data maximum, or if the data did not fit a normal or lognormal distribution and the sample size was less than five, then the data maximum was assigned to be the EPC. Because of the variation in the data detection limits, in some cases the maximum  $\frac{1}{2}$  detection limit value exceeded the maximum detected value. In these cases, to account for data uncertainty, the higher of the two was used as a conservative EPC. This approach was used for commercial exposures for the RME scenario. For CT exposures to a worker and hypothetical future resident, the average concentration for each chemical was used. For RME exposures to a hypothetical future resident and trench worker, the data maximum concentration was used. The various exposure and EPC types for each receptor population are summarized below.

Receptor Population	Exposure Scenario	EPC Type
Commercial Worker	RME	95UCL or data maximum, whichever is lower
	CT	Average
Hypothetical Future Resident	RME	Maximum
	CT	Average
Trench Worker	RME	Maximum

The COPCs and EPCs for each parcel that were used in the calculation of exposure and risk are presented in Table 12a.

## 4.0 EXPOSURE ASSESSMENT

This section describes the receptors and exposure pathways that were evaluated in the risk assessment. The objectives of the exposure assessment were to estimate the magnitude, frequency, duration and routes of potential human exposures to COPCs at the site. Potential receptor groups are identified in the exposure assessment and estimates of exposure or chemical intake are calculated based on assumptions regarding exposure pathways and exposure parameters. The exposure assessment focuses on the COPCs detected in soil, soil gas, groundwater and indoor air at the site. The primary routes of potential human exposure to chemicals detected at the site include incidental ingestion, dermal contact, inhalation of fugitive dust and inhalation of vapors in indoor and outdoor air.

The end product of the exposure assessment is a measure of chemical intake as an average daily dose (ADD) that integrates the exposure parameters for the receptors of concern (e.g., contact rates, exposure frequency, and duration) with exposure point concentrations for the media of concern. These ADDs are then used in conjunction with chemical-specific toxicity values (e.g., reference doses and cancer slope factors) to arrive at an estimate of potential health risks.

### 4.1 CONCEPTUAL SITE MODEL

The Conceptual Site Model (CSM) identifies potential chemical sources, release mechanisms, transport media, routes of chemical migration through the environment, exposure media, and potential receptors. Receptors that may be potentially exposed to site-related chemicals are identified and the likelihood of their potential exposures assessed through consideration of the current and the anticipated future use of the site. The CSM for the Del Amo site, illustrated in Figure 18, shows all potentially complete exposure pathways for human exposures.

Current land use zoning precludes pure residential development within the Del Amo site, with the exception of a dwelling occupied by a single worker serving as a watchman or caretaker where industrial development is present. Additionally, the possible application of other institutional control mechanisms that could enhance existing controls and prevent inappropriate land uses inconsistent with current zoning at the site in the future is being evaluated as part of the FS. Nevertheless, a hypothetical future residential exposure scenario has been included in this risk assessment at the request of USEPA and DTSC. Residential exposure pathways are indicated as only potentially complete in the CSM due to the hypothetical nature of this pathway. Future residential development is unlikely based on current zoning.

The majority of the site is covered by pavement, buildings, and landscaping, which significantly limit the potential for exposure to current workers. However, exposures may occur if excavation takes place during construction of foundations, pipelines or other types of maintenance activities (referred to herein as a trench worker exposure). Future commercial/industrial worker exposure could additionally occur at the site if areas were redeveloped and bare soil exposed. The potential also exists for vapor migration into indoor and outdoor air both currently and in the future.

## **4.2 EXPOSURE PATHWAY EVALUATION**

An exposure pathway describes a specific environmental mechanism by which an individual (receptor) can be exposed to COPCs present at or originating from a source. A complete exposure pathway comprises the following five elements:

- A chemical source
- A mechanism for chemical release to the environment
- An environmental transport medium (e.g., soil or air)
- A point of potential human contact with the medium
- A means of entry (i.e., intake route) into the body (e.g., ingestion).

There must be a complete exposure pathway from the source of chemicals in the environment (i.e., from soil, air, or groundwater) to human receptors for chemical intake to occur. If all exposure pathways are incomplete for human receptors, no chemical intake occurs and hence, no human health effects are associated with site-related COPCs. Potential pathways at the site examined for completeness include:

- Incidental ingestion of COPCs in soil;
- Contact with soil and absorption of COPCs through the skin;
- Inhalation of dust generated from soil;
- Inhalation of vapors emanating from soil or soil gas into outdoor air; and
- Inhalation of vapors emanating from soil, soil gas, or groundwater into indoor air.

Complete and potentially complete exposure pathways identified based on the characteristics of the COPCs and the site conditions are presented in the CSM on Figure 18. These exposure pathways are described in more detail below.

### **4.2.1 Direct Exposure to COPCs in Soil**

Onsite workers and hypothetical future residents can potentially come into contact with chemicals in onsite soils via dermal absorption and incidental ingestion. Chemicals may be

absorbed through the skin to the degree that soil adheres to an individual's skin or clothing. A certain minor fraction of such soil also tends to be ingested (USEPA, 1989).

#### **4.2.2 Inhalation of Airborne Dust Containing COPCs**

COPCs such as metals and SVOCs can adhere to soil particles, thus potential exposure to these COPCs may occur via inhalation of fugitive dust. Therefore, a relationship must be estimated between the chemical concentration in surface soil and the concentration in air (secondary media) due to fugitive dust emissions (see Section 4.3.4).

The generation of dust resulting from wind erosion depends on the surface roughness, soil moisture, vegetative cover, wind velocity, and the amount of the soil surface exposed to the eroding force. An individual may also disturb surface soil and create airborne dust during work activities or by walking over the surface of the site. These types of activities are more transient in nature and were not evaluated in this risk assessment.

#### **4.2.3 Inhalation of Vapors in Indoor and Outdoor Air**

VOCs were detected in soil and soil gas samples collected at the site. Because these compounds are volatile, humans could potentially be exposed to vapors migrating through the soil to the surface. Therefore, both indoor and outdoor air exposures were evaluated for VOCs detected in soil and soil gas. VOCs were also detected in groundwater beneath the site and migration of vapor into indoor air from groundwater was evaluated. Direct exposures to groundwater were not evaluated in the BRA due to an incomplete pathway of exposure and previous completion of a Groundwater Risk Assessment (McLaren Hart and Dames & Moore, 1998) and ROD (USEPA, 1999JGWROD).

#### **4.2.4 Summary of Potentially Complete Exposure Pathways**

The exposure pathways for the site under current and future land use conditions considered in this risk assessment are presented in the following table:

Receptor Population	Exposure Medium	Exposure Pathways
Commercial Worker	Surface Soil ( $\leq 1$ foot bgs)	<ul style="list-style-type: none"> <li>● Incidental Ingestion</li> <li>● Dermal Contact</li> <li>● Fugitive Dust Inhalation</li> </ul>
	Shallow Soil/Soil Gas ( $\leq 15$ feet bgs)	<ul style="list-style-type: none"> <li>● Incidental Ingestion</li> <li>● Dermal Contact</li> <li>● Fugitive Dust and Vapor Inhalation</li> <li>● Vapor Inhalation in Indoor Air</li> </ul>
	Deep Soil/Soil Gas ( $> 15$ feet bgs) Groundwater	<ul style="list-style-type: none"> <li>● Vapor Inhalation in Indoor Air</li> </ul>
	Workplace Air	<ul style="list-style-type: none"> <li>● Vapor Inhalation in Indoor Air</li> </ul>
Hypothetical Future Resident	Shallow Soil/Soil Gas ( $\leq 15$ feet bgs)	<ul style="list-style-type: none"> <li>● Incidental Ingestion</li> <li>● Dermal Contact</li> <li>● Fugitive Dust and Vapor Inhalation</li> <li>● Vapor Inhalation in Indoor Air</li> </ul>
	Deep Soil/Soil Gas ( $> 15$ feet bgs) Groundwater	<ul style="list-style-type: none"> <li>● Vapor Inhalation in Indoor Air</li> </ul>
Trench Worker	Shallow Soil/Soil Gas ( $\leq 15$ feet bgs)	<ul style="list-style-type: none"> <li>● Incidental Ingestion</li> <li>● Dermal Contact</li> <li>● Fugitive Dust and Vapor Inhalation</li> </ul>

### 4.3 ESTIMATING CHEMICAL INTAKE

The next step in the exposure assessment is to quantify the magnitude, frequency, and duration of chemical intakes (daily dose) by receptor populations. This step is conducted in two stages: (1) estimation of EPCs, and (2) estimation of the ADD or “Lifetime Average Daily Dose” (LADD) of COPCs for each exposure pathway under consideration. ADDs and LADDs are calculated using guidelines in the Risk Assessment Guidance for Superfund (USEPA, 1989), Exposure Factors Handbook (USEPA, 1997a), site-specific information, and professional judgment, as appropriate.

Daily intakes are classified as being either ADDs (for noncarcinogens) or LADDs (for carcinogens; USEPA, 1989). They differ primarily in the length of time over which the effects of the chemical are assumed to be averaged. ADDs and LADDs are expressed as the amount of a substance taken into the body per unit body weight per unit time, or mg/kg-day. The LADD is averaged over a lifetime (70 years) for carcinogens, and the ADD is averaged over the expected exposure duration for noncarcinogens. The duration of exposure is assumed to vary depending on whether exposure occurs to a working population or a residential population. LADDs and ADDs are calculated from the concentration of the chemical at the exposure point, the daily intake rate, the exposure frequency (i.e., number of

times during a week or year), the exposure duration (i.e., the number of days, weeks, or years the exposure persists), and the physical characteristics of the receptor (such as body weight). Default values are assumed for each of these parameters that are appropriate for the expected habits of the potentially exposed population.

USEPA RAGS recommends that LADDs and ADDs be estimated for both average and RME conditions. LADDs and ADDs under RME conditions are calculated by combining exposure factors so that the result is the maximum exposure that is reasonably expected to occur (USEPA, 1989). Because of the multiple conservative assumptions used in the risk assessment process, the RME is often a high-end estimate of exposure and risk. USEPA 1999b, however, also recommends consideration of a more probable case. Therefore, a CT exposure was evaluated to represent more “typical” or average exposure conditions. For the CT exposure, the average chemical concentration was used in combination with exposure factors that represent the 50th percentile of exposure. The equations and variables used to calculate LADDs and ADDs are presented in Section 4.3.3.

RMEs are intended to place conservative upper bounds on the potential risks, meaning that each risk estimate is unlikely to be underestimated, and therefore are likely to be overestimated. The RMEs of dose for a given pathway were derived in this study by combining the upper bound estimate of the concentration for each chemical (maximum or 95UCL) with reasonable maximum values for the extent, frequency, and duration of exposure (USEPA, 1989).

#### 4.3.1 Exposure Parameters

Exposure parameter values were selected based on values presented in the following USEPA guidance documents: Supplemental Guidance: Standard Default Exposure Factors (1991a); Dermal Exposure Assessment: Principles and Applications (1992a and 2000b); Supplemental Guidance for Dermal Risk Assessment (2004c), Superfund's Standard Default Exposure Factors for The Central Tendency and Reasonable Maximum Exposure (1993); and Exposure Factors Handbook (1997a). Several exposure parameters are briefly discussed below and are presented in Tables 13 (Commercial Worker), 14 (Hypothetical Resident) and 15 (Trench Worker).

Commercial workers were assumed to be exposed to COPCs for 8 hours per day, 250 days per year (five days per week for 50 weeks, accounting for a two-week vacation) for 25 years for the RME scenario (USEPA, 1991a). The exposure duration was assumed to be 6.6 years for the CT exposure scenario, consistent with the average time a person works at one location (USEPA, 1997a).

For trench workers, exposures were assumed to occur during repair of pipelines or utilities within a parcel. A preliminary pipeline survey was conducted by telephone to determine the repair frequency of pipelines/utilities that traverse the site. This information was to be used to identify the exposure frequency and duration for the trench worker scenario. Ten pipeline owners were contacted to determine if pipelines in the Del Amo site area had been repaired and how often. Some quantitative information was provided by three owners: Chevron, the Southern California Gas Company, and Dow USA. Chevron has one gasoline pipeline under the site that was installed in 1976. The pipeline has not been repaired in the site vicinity since it was installed. The gas company representative wasn't sure if their lines traversed the site, but said that pipeline repairs are rare and don't occur more than once every 10 years. Dow Chemical has one pipeline in the area that has been idle since 1992-1993. It was repaired for a leak 7 to 8 years ago; typically repairs are infrequent, not more than once every ten years. This information indicates that pipeline repairs are very infrequent. In addition, repairs to other smaller utility lines associated with the buildings such as water and sewer lines would be relatively infrequent and of short duration. Furthermore, it is unlikely the same individual would be conducting repairs at this same location during his/her occupational tenure if the repair frequency is greater than one event every 10 years.

Based on the above information, trench workers were assumed to be exposed to COPCs for 20 days per year, but for only one year exposure duration. This exposure frequency is believed to represent a reasonable time for trench worker exposure, and coupled with the use of the maximum detected concentration for each chemical as the EPC, satisfies the RME concept of exposure. Based on site data, it is highly unlikely that an individual would be exposed to the maximum concentration of each chemical and, as a result, this assumption likely results in an over estimation of risk.

Hypothetical future residents were assumed, for the RME scenario, to be exposed to COPCs 350 days per year (allowing 15 days per year for vacations and holidays) for 24 years for adults, and 6 years for children (USEPA, 1991a). For the CT residential scenario, the exposure duration was assumed to be 7 years for adults and 2 years for children, consistent with the average residence time of 9 years at one location (USEPA, 1993). The division between the child and adult exposure duration for the CT scenario is based on the assumptions used for the RME scenario, where an individual is assumed to be a child for 20% of the time (6 years) and an adult for 80% of the time (24 years) for a 30-year exposure. Therefore, for a 9-year CT exposure duration, this equates to 2 years as a child and 7 years as an adult. An average time of 25,550 days, based on lifetime exposure duration of 70 years, was used to model exposure to carcinogens. An averaging time equal to the exposure duration (in years) multiplied by 365 days per year was used to model exposures to noncancer COPCs (USEPA, 1989).

Several exposure parameters (e.g., exposure duration, body weight, and averaging times) have general application in all chemical intake estimations, regardless of the exposure pathway and are presented in Tables 13, 14, and 15.

### 4.3.2 Exposure Point Concentrations

Potential cancer risk or noncancer hazard are calculated by first estimating the EPC to which an individual is exposed. EPCs are the COPC concentrations in environmental media (soil/air/groundwater) at the point of contact. A detailed discussion on how the EPCs were calculated for the site was presented in Section 3.5. The COPCs and EPCs that were used in the calculation of potential exposure and risk for each parcel are presented in Table 12a. Modeled EPCs from soil/soil gas to indoor air, and from groundwater to indoor air, are presented in Tables 12b and 12c, respectively.

An overview of the types of EPCs used for each receptor type, exposure type, and media is presented on Figures 19a (outdoor soil pathway) and 19b (indoor air pathway). Figure 19b also explains how the data were used for vapor modeling, which is discussed in more detail in Section 4.3.6.

### 4.3.3 Calculation of Summary Intake Factors

The intake factor is a value that combines the site-specific and receptor-specific assumptions for a given exposure pathway and is expressed as the amount of medium (e.g., soil) taken into the body per unit concentration of chemical in the medium. Multiplying the intake factor by the selected EPC yields the ADD in mg/kg-day for that receptor population and exposure pathway. The following is a generic equation used to calculate the daily dose:

$$\text{ADD or LADD (mg/kg-day)} = \text{Selected EPC} \times \text{Summary Intake Factor}$$

Separate intake factors are calculated for each complete exposure pathway. The values and assumptions used to calculate each intake factor are dependent on the exposure pathway and receptor population being evaluated. A more detailed description of the values used for the intake calculations is presented below. The exposure parameters and intake factor equations used in this risk assessment are summarized in Tables 13, 14, and 15 for the commercial, residential, and trench worker scenarios, respectively.

#### 4.3.3.1 Incidental Soil Ingestion

The rate of soil ingestion is based on the amount of soil a child or adult inadvertently swallows in a given day from all sources. Exposures to COPCs via incidental ingestion of

soil are estimated using the following variables: (1) the rate of soil ingestion; (2) the fraction of ingested soil that is contaminated; and (3) the frequency and duration of exposure. Individuals may ingest soil through incidental contact of the mouth with hands and clothing. The following is the equation used to calculate the LADDs and ADDs of COPCs (units of mg/kg-day) via incidental ingestion of soil:

$$\text{ADD or LADD} = \frac{C_s \times IR_s \times ABS \times EF \times ED \times CF}{BW \times AT}$$

Where:

$C_s$	=	chemical concentration in soil (mg/kg)
$IR_s$	=	ingestion rate of soil (mg/day)
ABS	=	percent absorption (assumed to be 100%)
EF	=	exposure frequency (days/year)
ED	=	exposure duration (years)
CF	=	conversion factor for soil ( $10^{-6}$ kg/mg)
BW	=	body weight (kg)
AT	=	averaging time (days)
		cancer effects: 70 years x 365 days = 25,550 days
		noncancer effects: ED x 365 days

The following soil ingestion rates were used in this risk assessment:

Commercial workers: RME of 100 mg/day and a CT of 50 mg/day

Hypothetical future adult residents: RME of 100 mg/day and a CT of 50 mg/day

Hypothetical future child residents: RME of 200 mg/day and a CT of 100 mg/day

Trench workers: RME of 330 mg/day

Cancer risks are calculated for the hypothetical future residential exposure scenario using an age-adjusted factor for the soil ingestion pathway since ingestion rates are different for children and adults in their first 30 years of life. The age-adjusted factor approximates the integrated exposure for small children and adults by combining their intake rates, body weights, and exposure frequency and duration. The following equation was used to estimate the age-adjusted ingestion factor:

$$\text{IngF}_{\text{adj}} = \frac{ED_c \times IR_c}{BW_c} + \frac{ED_a \times IR_a}{BW_a}$$

Where:

$\text{IngF}_{\text{adj}}$	=	ingestion age-adjusted factor (mg-year)/(kg-day)
a	=	adult exposure factor
c	=	child exposure factor

The  $\text{IngF}_{\text{adj}}$  is used in the following equation to calculate the lifetime average residential daily dose via ingestion of COPCs:

$$\text{LADD}_{\text{ing - adj}} = \frac{C_s \times \text{IngF}_{\text{adj}} \times \text{ABS} \times \text{EF} \times \text{CF}}{\text{AT}}$$

Adjusting for age is also considered for the other exposure pathways of concern (see sections below). A child exposure of 6 years was assumed for noncancer hazard for the hypothetical future residential exposure scenario.

Details on the exposure parameters used to calculate intake of COPCs via incidental soil ingestion are provided in Tables 13, 14, and 15 along with references.

#### 4.3.3.2 Dermal Contact

COPCs in soil may come into contact with skin, and then absorb across the skin into the bloodstream. The amount of absorption into the body depends upon the amount of soil in contact with the skin, COPC concentrations in soil, the skin surface area exposed, and the potential for the chemical to be absorbed across the skin.

The following equation was used to calculate the steady-state dose absorbed across the skin:

$$\text{ADD or LADD} = \frac{C_s \times \text{SA} \times \text{SAF} \times \text{EF} \times \text{ED} \times \text{CF} \times \text{DAF}}{\text{BW} \times \text{AT}}$$

Where:

$C_s$	=	chemical concentration in soil (mg/kg)
SA	=	skin surface area exposed to soil per day (cm <sup>2</sup> /day)
SAF	=	soil-skin adherence factor (mg/cm <sup>2</sup> )
CF	=	conversion factor (10 <sup>-6</sup> kg/mg)
DAF	=	dermal absorption factor (unitless, chemical-specific)

An approximate skin surface area of 5,700 cm<sup>2</sup> was used to represent exposures to the head, hands, forearms, and lower legs for hypothetical future adult residents, while a skin surface area of 3,300 cm<sup>2</sup> was used for commercial and trench workers under the RME scenario

(USEPA, 2000b). An area of 5,000 cm<sup>2</sup> was assumed for hypothetical future adult residents under the average, central tendency scenario (USEPA, 1997a). The skin surface area used in this risk assessment for a hypothetical future child resident was 2,800 cm<sup>2</sup> under the RME scenario (USEPA, 2000b) and 2,000 cm<sup>2</sup> under the CT scenario (USEPA, 1992a). Chemical-specific dermal absorption factors were used based on agency guidance documents such as the Dermal Assessment Guidance (USEPA, 2000b and CalEPA, 1999). The soil adherence factor for a hypothetical future adult resident of 0.07 mg/cm<sup>2</sup>, for a trench worker of 0.3 mg/cm<sup>2</sup>, and 0.2 mg/cm<sup>2</sup> for a commercial worker and hypothetical future child resident, were assumed for RME exposures. The soil adherence factors of 0.04 mg/cm<sup>2</sup> for a hypothetical future child resident, 0.01 mg/cm<sup>2</sup> for a hypothetical future adult resident, and 0.02 mg/cm<sup>2</sup> for a commercial worker were assumed for CT exposures (USEPA, 2004c).

Cancer risks under the hypothetical future residential exposure scenario were calculated using an age-adjusted factor for the dermal soil contact pathway since dermal exposure factors are different for children and adults in their first 30 years of life. The equation used to estimate the age-adjusted dermal factor is as follows:

$$DF_{adj} = \frac{ED_c \times SA_c \times EF_c}{BW_c} + \frac{ED_a \times SA_a \times EF_a}{BW_a}$$

Where:

$DF_{adj}$  = dermal contact age-adjusted factor (mg/kg)

The  $DF_{adj}$  was used in the following equation to calculate the lifetime average residential daily dose of COPCs via dermal absorption:

$$LADD_{abs - adj} = \frac{C_s \times DF_{adj} \times SAF \times DAF \times CF}{AT}$$

Details on the exposure parameters used to calculate intake of COPCs via dermal absorption are provided in Tables 13, 14, and 15.

#### 4.3.3.3 Inhalation Exposures

Inhalation of vapors and fugitive dust-containing COPCs in outdoor air is a consideration for surface and shallow soil exposures for a commercial worker and trench workers, as well as for a hypothetical future resident. Inhalation of indoor air vapor is an additional consideration for a commercial worker and a hypothetical future resident. The potential dose for inhalation of vapors and fugitive dust is calculated using the following equation:

$$\text{ADD or LADD} = \frac{C_A \times \text{IR}_A \times \text{ABS} \times \text{EF} \times \text{ED}}{\text{BW} \times \text{AT}}$$

Where:

$C_A$	=	concentration in air ( $\text{mg}/\text{m}^3$ )
$\text{IR}_A$	=	inhalation rate ( $\text{m}^3/\text{day}$ )
ABS	=	percent absorption (assumed to be 100%)

The COPC concentrations in soil,  $C_S$ , were adjusted by a particulate emission factor (PEF) for non-VOCs or by a volatilization factor (VF) for VOCs, to calculate an outdoor air concentration ( $C_A$ ) in  $\text{mg}/\text{m}^3$  for the outdoor air pathway. For the non-VOCs,  $C_A$  is calculated by dividing the soil concentration by the PEF. The VFs for the commercial and residential receptors are defined differently than those for the trench worker. For the commercial and residential receptors,  $C_A$  is calculated by dividing the soil VOC concentration by the VF as defined in the USEPA Soil Screening Level Guidance (USEPA, 2002c). For the trench worker, the chemical-specific volatilization factors,  $\text{VF}_{\text{ss,amb}}$ , included in the ASTM Risk Based Corrective Action Standard (ASTM, 1998) were multiplied by VOC concentrations in soil to determine  $C_A$ . Use of the PEFs and VFs in the risk calculations is described further in Sections 4.3.4 and 4.3.5 below. Workplace monitored air data were used as the indoor air concentration ( $C_A$ ) in the risk calculations for commercial workers, if available. Additionally, vapor transport model-predicted indoor air concentrations were used as the  $C_A$  to calculate the ADD or LADD (see Table 12b, Section 4.3.6 and Appendices E and F). The following inhalation rates were used:

- Commercial workers: RME of  $15 \text{ m}^3/\text{day}$  (USEPA, 1991a) and CT of  $10.8 \text{ m}^3/\text{day}$ , which is the average inhalation rate for adult males and females performing a mixture of light and moderate activities during an 8-hour workday;
- Hypothetical future adult residents: RME of  $20 \text{ m}^3/\text{day}$  and a CT of  $13.25 \text{ m}^3/\text{day}$ ;
- Hypothetical future child residents: RME of  $10 \text{ m}^3/\text{day}$  and a CT of  $7.4 \text{ m}^3/\text{day}$ ; and
- Trench workers: RME of  $20 \text{ m}^3/\text{day}$  (USEPA, 2002c)

Cancer risks under the hypothetical future residential exposure scenario were calculated using an age-adjusted factor for the inhalation pathway since inhalation rates are different for children and adults in their first 30 years of life. The following equation was used to estimate the age-adjusted inhalation factor:

$$\text{InhF}_{\text{adj}} = \frac{\text{ED}_c \times \text{IRAc}}{\text{BW}_c} + \frac{\text{ED}_a \times \text{IRa}_a}{\text{BW}_a}$$

Where:

$$\text{InhF}_{\text{adj}} = \text{inhalation age-adjusted factor (m}^3\text{-year)/(kg-day)}$$

The  $\text{InhF}_{\text{adj}}$  is used in the following equation to calculate the lifetime average residential daily dose of COPCs via inhalation:

$$\text{LADD}_{\text{inh - adj}} = \frac{C_A \times \text{InhF}_{\text{adj}} \times \text{ABS} \times \text{EF}}{\text{AT}}$$

Details on the exposure parameters used to calculate intake of COPCs via inhalation are provided in Tables 13, 14, and 15.

#### 4.3.4 Inhalation of Fugitive Dust Containing COPCs

Inorganic and semi-volatile organic compounds (metals and SVOCs) were detected in soil at the site. These compounds can adhere to soil particles with subsequent wind erosion resulting in airborne, COPC-containing dust. Exposure to these chemicals may then occur via inhalation of the dust. Inhalation exposure to non-volatile compounds is typically minor in fugitive dust when compared to direct ingestion exposure (USEPA, 2002c). Nevertheless, a relationship must be estimated between the chemical concentration in soil and the concentration in air (secondary media) due to fugitive dust emissions from surface soil.

Potential exposure to airborne dust is estimated using a PEF that relates the concentration of soil contaminant to the concentration of dust particles in air. The PEF represents an annual average emission rate based on wind erosion. The PEF equation (Equation 4-5: Derivation of the PEF) can be found in the Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites (USEPA, 2002c). The emissions part of the PEF equation is based on the “unlimited reservoir” model developed to estimate particulate emissions (particulate matter less than 10 microns in diameter [PM10]) due to wind erosion (Cowherd et al., 1985). The dispersion part of the PEF equation includes a dispersion coefficient (Q/C) in units of grams per square meter-second per kilogram per cubic meter ( $\text{g/m}^2\text{-s per kg/m}^3$ ). The Q/C term was generated using the Industrial Source Complex model and varies depending on the source area, city, and climatic zone.

The PEF was derived using the following equation (USEPA, 2002c):

$$\text{PEF} = [(Q/C \times 3600) / (0.036 \times (1-V) \times (U_m/U_t)^3 \times F_x)]$$

Where:

Q/C	=	inverse of mean concentration at center of 5-acre-square source (g/m <sup>2</sup> -s per kg/m <sup>3</sup> )
V	=	fraction of vegetative or other cover (unitless)
U <sub>m</sub>	=	mean annual wind speed (m/s)
U <sub>t</sub>	=	equivalent threshold value of wind speed at 7 meters (m/s)
F <sub>x</sub>	=	function dependent on U <sub>m</sub> /U <sub>t</sub> derived using Cowherd et al. (1985) (unitless)

The selected Q/C value of 45.95 g/m<sup>2</sup> -s per kg/m<sup>3</sup> is the inverse of the mean concentration at the center of a 5-acre square source in Los Angeles, California (USEPA, 2002c) based on an average parcel size of 4.7 acres for the site. Using this Q/C term for the commercial and hypothetical future residential scenarios and the assumption that only 50 percent of the site is covered by vegetation, a PEF value of 6.66 x 10<sup>+8</sup> m<sup>3</sup>/kg was calculated. This PEF was used in this risk assessment by dividing the COPC soil concentration (C<sub>s</sub>) by the PEF to arrive at an outdoor air concentration (C<sub>A</sub>) in mg/m<sup>3</sup>. This is a conservative assumption for the commercial worker scenario, as most parcels currently are entirely covered with buildings, parking areas, and landscaping. The PEF of 6.66 x 10<sup>+8</sup> m<sup>3</sup>/kg was used for both commercial and hypothetical future residential exposures. For the trench worker exposures, an assumption of zero (0) percent cover was used along with the Q/C value of 68.18 g/m<sup>2</sup> -s per kg/m<sup>3</sup> for a 0.5 acre area in Los Angeles, which resulted in a PEF of 4.94 x 10<sup>+8</sup> m<sup>3</sup>/kg. The derivation of PEFs is presented in Appendix D.

#### 4.3.5 Inhalation of Outdoor Air Vapors

VOCs were detected in soil and soil gas samples collected at the site. Because these compounds are volatile, receptors could potentially be exposed to vapors emanating from the subsurface to the surface. Therefore, outdoor air exposures were evaluated for VOCs detected in soil and soil gas, as discussed below.

Potential migration of vapors from soil to outdoor air for commercial and hypothetical future residential exposures was estimated using the volatilization factor presented in the Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites (USEPA, 2002c, Section 4.2.3). Default parameters for the Los Angeles area were used (e.g., the Q/C term as discussed above for the PEF formula). In this risk assessment, the COPC concentrations in soil (C<sub>s</sub>) were divided by their respective VF to arrive at an air concentration (C<sub>A</sub>) in mg/m<sup>3</sup> with the VF term incorporating the dispersion factor.

The chemical-specific VF for commercial and residential exposures was derived using the following equation (USEPA, 2002c):

$$VF = Q/C \times [(3.14 \times Da \times T)^{1/2} \times 10^{-4}] / (2 \times Pb \times Da)$$

Where:

Q/C	=	inverse of mean concentration at center of 5-acre-square source (g/m <sup>2</sup> -s per kg/m <sup>3</sup> )
Da	=	apparent diffusivity (cm <sup>2</sup> /s)
T	=	exposure interval (s)
Pb	=	dry soil bulk density (g/cm <sup>3</sup> )

And where:

$$D_a = \frac{(\theta_a^{3.33} D_{air} H_{eff} + \theta_w^{3.33} D_{water}) / \theta_T^2}{Pb Kd + \theta_w + \theta_a H_{eff}}$$

Where:

D <sub>air</sub>	=	molecular diffusion coefficient in air (cm <sup>2</sup> /s)
D <sub>water</sub>	=	molecular diffusion coefficient in water (cm <sup>2</sup> /s)
θ <sub>air</sub>	=	soil air content (cm <sup>3</sup> -air/cm <sup>3</sup> -soil)
θ <sub>water</sub>	=	soil water content (cm <sup>3</sup> -water/cm <sup>3</sup> -soil)
θ <sub>T</sub>	=	soil porosity (cm <sup>3</sup> -air/cm <sup>3</sup> -soil) = θ <sub>air</sub> + θ <sub>water</sub>

For the trench worker scenario, VOC emissions into a trench and subsequent mixing in air were estimated using the volatilization factor for transport of chemicals from soil to outdoor air from the American Society for Testing and Materials (ASTM) Standard Guide For Provisional Risk-Based Corrective Action (ASTM, 1998, Table X.3.4). A conservative wind speed of 0.255 meters per second was assumed based on 1/10 of the average wind speed for the Los Angeles area (NCDC, 2004). This speed represents the reduced airflow expected in a shallow trench. Conservative assumptions regarding the size of the trench were also used (assumed area of two side-walls and bottom area of trench was approximately 1.1 x 10<sup>+6</sup> cm<sup>2</sup>, length and depth of trench of 9.14 meters and 4.57 meters, respectively). The chemical-specific VF<sub>ss,amb</sub> for trench worker exposures was derived using the following equation (ASTM, 1998):

$$VF_{ss,amb} = (Pb/DF_{amb}) \times [(4 \times D_{eff} \times H_{eff}) / (3.14 \times T \times K_{sw} \times Pb)]^{1/2}$$

Where:

- $VF_{ss,amb}$  = volatilization factor, surficial soils to ambient air (g-soil/cm<sup>3</sup>-air)  
 $D_{eff}$  = effective diffusion coefficient for vadose-zone soils (cm<sup>2</sup>/s)  
 $DF_{amb}$  = dispersion factor for ambient air (cm/s)  
 $H_{eff}$  = effective Henry's law coefficient (cm<sup>3</sup>-water/cm<sup>3</sup>-air)  
 $K_{sw}$  = soil to water partition coefficient (cm<sup>3</sup>-water/g-soil)  
 $P_b$  = dry soil bulk density (g/cm<sup>3</sup>)  
 $T$  = averaging time for surface emission vapor flux (s)

And where:

$$K_{sw} = \frac{\theta_w + \theta_a H_{eff} + P_b K_d}{P_b}$$

$$D_{eff} = [((D_{air} \times (\theta_{air}^{3.33} / \theta_T^2)) + ((D_{water} / H_{eff}) \times (\theta_{water}^{3.33} / \theta_T^2)))]$$

$$DF_{amb} = (U_{air} \times W \times H) / A$$

Where:

- $U_{air}$  = ambient air velocity in mixing zone (cm/s)  
 $W$  = width of source-zone area (cm)  
 $H$  = mixing zone height (cm)  
 $A$  = source-zone area (cm<sup>2</sup>)

The derivation of VFs is presented in Appendix D.

### 4.3.6 Inhalation of Indoor Air Vapors

Evaluation of the indoor air pathway at the site is complex, since workplace indoor air monitoring, shallow soil and soil gas, deep soil and soil gas, and groundwater data all required consideration. It is important to consider all potential contributing sources of contamination when evaluating indoor air exposures, including both surface and subsurface sources. Surface sources include indoor chemical use and chemicals in outdoor (ambient) air. For the purposes of this risk assessment, the contribution from subsurface sources is most relevant since that is where residual contamination from the former rubber plant would primarily occur.

Indoor air data provide the most direct evaluation of risk, but may incorporate contaminant contributions from sources unrelated to the former rubber plant, including chemical facilities and operations within the current buildings, and ambient air. Evaluation using shallow subsurface data has the advantage of minimizing contributions from surface sources that are

unrelated to the former rubber plant, but this method is also less direct, since indoor air concentrations must be mathematically derived from the subsurface concentrations through vapor transport modeling.

#### 4.3.6.1 Correlation Analysis

In view of the above, a correlation analysis was completed to evaluate whether subsurface data should be used to estimate indoor air exposure from vapor intrusion of former rubber plant contamination. The analysis compared indoor air concentrations with shallow subsurface concentrations for each of the EAPCs where indoor air sampling was conducted. A positive correlation between shallow subsurface concentrations and indoor air concentrations (i.e., higher subsurface concentrations paired with higher indoor air concentrations) would indicate that subsurface sources were a contributor to the indoor air concentrations and suggest that direct use of the measured indoor air data was the preferred method of evaluating risk for the pathway. Alternatively, a poor or nonexistent correlation would suggest that vapor intrusion from the subsurface is not the primary factor controlling the measured indoor air concentrations in this case. Under this outcome, vapor transport modeling of the subsurface data would be the preferred approach for evaluating the vapor intrusion pathway for the site.

The correlation analysis consisted of both graphical and nonparametric statistical evaluations. The graphical evaluation consisted of plotting average indoor air concentrations versus average soil concentrations for both benzene and PCE, the primary chemicals of interest for the indoor air pathway. Plots of concentration rank for benzene and PCE in soil and indoor air were additionally prepared. The plots are presented in figures H-1 through H-4 of Appendix H and the corresponding data points for the charts are identified in Table H-1. The EAPC number for each data point is labeled on the charts. EAPCs where NAPL is present are also identified on the plots.

The graphical analyses show wide scatter in the rank graphs and clustering of data along the y-axis in the concentration graphs, indicating there is no correlation between subsurface impacts and indoor air concentrations. Similar results are indicated on charts plotting mean EAPC groundwater concentrations versus indoor air concentrations (figures H-5 through H-8; Table H-2).

Further support for the above conclusion is provided through the non-parametric statistical analysis. Correlation coefficients can be useful metrics to assist in the graphical evaluation of the association between variables. For parametric analysis, these tests measure the linear relationship between variables with a correlation coefficient ( $r^2$ ) indicating the fit of a line drawn through the data points. For non-parametric data, the equivalent Spearman's Rank

correlation coefficient is used to measure the degree of association. Values can range from -1 (negative correlation) to +1 (positive correlation) with values close to 0 suggesting no correlation. For example, positive correlation coefficients indicate that an increase in shallow soil gas concentration has a corresponding increase in indoor air concentration. Mirroring the conclusions from the graphical analysis, correlation coefficients for site subsurface and measured indoor data were low (-0.2 to +0.2), indicating no correlation.

The data were also subjected to a statistical test to determine if the observed association was statistically significant. In this case, the non-parametric Spearman's rho correlation test was performed, evaluating the probability that the observed association between the variables could have arisen by chance alone. A statistically significant correlation is indicated when the probability of incorrectly identifying that an association exists is less than 5% ( $p < 0.05$ ). P-values for the site data ranged from 0.47 to 0.76, meaning that it is highly likely that the observed association is the result of chance. The results of the statistical analysis are also presented in Appendix H.

Based on the correlation analysis results described above, modeling of subsurface data collected at the site was judged to be the preferred method by which to evaluate risks from the vapor intrusion pathway. Modeling of subsurface data has an additional benefit over the direct use of indoor air data in that this approach can be used for almost all of the EAPCs. This is not possible with direct use of the measured indoor air data, since these data are only available for 13 of the 37 EAPCs. While the lack of a positive correlation between subsurface and indoor air data supports the preferential use of the modeling approach over measured indoor air data, it does not rule out the possibility that subsurface sources have contributed to indoor air concentrations. Measured indoor air concentrations are influenced by many factors, including building-specific ventilation and air-exchange rates that could potentially mask soil vapor intrusion contributions.

#### **4.3.6.2 Vapor Transport Modeling**

Shallow soil or soil gas rather than deeper data were typically used for modeling of vapor intrusion from the subsurface into indoor air. This is because the shallow soil zone represents the contamination that is closest to the building slab, whether it be from a shallow source or emanating from sources in deeper soils or groundwater. In the few instances where shallow zone data were not available, deeper soil/soil gas or groundwater was considered.

Two tiers of analysis were conducted to evaluate vapor intrusion from soil, soil gas and groundwater, as described in the following sections

**Tier 1 Modeling**

The indoor air exposure was evaluated in Tier 1 primarily through application of the Johnson & Ettinger vapor transport model (J&E; 1991) series of spreadsheets developed by USEPA (USEPA, 2000a) to the shallow soil, deep soil, and groundwater data to predict indoor air concentrations. This method is commonly employed by USEPA and CalEPA DTSC. Tier 1 vapor modeling was constructed assuming a worst case scenario; no biodegradation was assumed and maximum EPCs were used in the model. With this approach, if a chemical or media did not pose a significant risk, then other, less intensive exposure conditions could be considered even less significant.

The J&E vapor model input parameters and calculations are presented in Appendix E. Site-specific soil properties, including bulk density, total porosity, air-filled porosity, and water-filled porosity were used when possible. Additional chemical transport parameters were selected from ASTM (1998) for chemicals under soil conditions similar to this site. The predicted indoor air concentrations from the model were subsequently used as EPCs in the calculation of risk and hazard. The site-specific soil properties are the primary parameters that result in vapor intrusion attenuation factors lower than default values in USEPA and DTSC guidance documents.

The following subsections describe the major assumptions used in each of the J&E modeled pathways.

**Vapors Emanating from Shallow Soil to Indoor Air**

Exposure to indoor air vapor from shallow soil (0 to 15 feet bgs) was evaluated for commercial and hypothetical future residential scenarios. The depth below ground surface to the top of contamination was assumed to be 7.5 feet, the center of the shallow soil layer. The source layer was assumed to have infinite thickness. Default building parameters (USEPA, 2000a) were used for hypothetical future residential exposures. For commercial exposures, the smallest onsite building dimensions for the enclosed space length, width, and height were used to be conservative. Additional building parameters such as enclosed space floor thickness, the floor-wall crack width, and indoor air exchange rate were selected from USEPA (2000a). The building parameters and soil properties (including bulk density, total porosity, air-filled porosity, water-filled porosity, and  $f_{oc}$ ), that were used in the model are presented in Appendix E. Calculated vapor intrusion attenuation factors for shallow soil to indoor air range from  $3.3 \times 10^{-5}$  to  $2.7 \times 10^{-4}$  for the residential scenario and from  $1.4 \times 10^{-5}$  to  $1.1 \times 10^{-4}$  for the commercial scenario.

**Vapors Emanating from Deep Soil to Indoor Air**

Exposure to indoor air vapors from deep soil was evaluated for commercial and hypothetical future residential scenarios. The depth below ground surface to top of contamination was

estimated at 30 feet based on the average minimum depth to deep soil contamination at the site. The source layer was assumed to have infinite thickness in the model. The building parameters that were used in the shallow soil analysis were also used in the deep soil analysis. The parameter values used in the modeling are presented in Appendix E. Calculated vapor intrusion attenuation factors for deep soil to indoor air range from  $2.6 \times 10^{-5}$  to  $4.6 \times 10^{-5}$  for the residential scenario and from  $1.1 \times 10^{-5}$  to  $2.0 \times 10^{-5}$  for the commercial scenario.

### **Vapors Emanating from Groundwater to Indoor Air**

Exposure to indoor air vapor from groundwater was evaluated for commercial and hypothetical future residential scenarios. Indoor air exposures were evaluated by first using the maximum detected chemical concentrations in groundwater on a site-wide basis (highest concentrations found at the whole site) in the Tier 1 vapor model. The parameter values used in the modeling are presented in Appendix E and the resulting EPCs are presented in Table 12c. The Tier 1 risk results for the groundwater to indoor air pathway are presented in Appendix D.

The depth to water table was estimated at approximately 47 feet (based on the time of sample collection) for the groundwater to indoor air modeling, and site-specific soil properties, including bulk density, total porosity, air-filled porosity, and water-filled porosity were also used in this modeling. A constant (i.e., steady-state) groundwater source concentration was assumed. The building parameters were the same as used for shallow and deep soil. The parameter values used in the modeling are presented in Appendix E. Calculated vapor intrusion attenuation factors for groundwater to indoor air range from  $9.2 \times 10^{-6}$  to  $5.7 \times 10^{-5}$  for the residential scenario and from  $3.9 \times 10^{-6}$  to  $2.4 \times 10^{-5}$  for the commercial scenario.

#### **4.3.6.3 Tier 2 Vapor Diffusive Transport Analysis**

Tier 2 vapor modeling was conducted for EAPCs in which the Tier 1 modeling for shallow soil/soil gas resulted in a cancer risk greater than  $1 \times 10^{-6}$  or a noncancer hazard greater than 1. Tier 2 vapor modeling was also conducted for select EAPCs and chemicals where the risk and/or hazard associated with underlying groundwater contamination was elevated based on Tier 1 modeling. Tier 2 modeling provided a more refined estimate of exposure by considering BTEX biodegradation during vapor migration from the subsurface contaminant source to the surface. The Tier 2 analysis was limited to BTEX because these compounds were frequently detected and their tendency to biodegrade in the vadose zone is well documented.

The Dominant Layer Model (DLM) developed by Johnson et al. (1999) was used for the Tier 2 analysis. The DLM is an extension of the Johnson and Ettinger 1991 model that

incorporates biodegradation in the vadose zone. Biodegradation parameters included in the model were determined by calibrating the model to measured soil gas concentrations within the Del Amo Study Area (Dames & Moore, 1999a). Conservative model inputs to characterize the aerobic biodegradation along with the input parameters from the Tier 1 vapor intrusion modeling were used to estimate vapor attenuation factors.

Tier 2 modeling of shallow soil/soil gas data was preferred over Tier 2 modeling of groundwater data due to the proximity of the data to potential receptors and relative abundance. Therefore, Tier 2 modeling of groundwater data was limited to those EAPCs where the shallow soil/soil gas data were judged to be limited based on the number and distribution of soil/soil gas samples relative to the historical rubber plant facility locations. Typically, vapor intrusion from groundwater was evaluated for any EAPC with fewer than six shallow soil/soil gas sampling locations with VOC data. As a result, the following 13 EAPCs were selected for Tier 2 evaluation of the groundwater to indoor air pathway for both the commercial and hypothetical future residential scenarios:

Parcel	EAPC No.
7351-31-20	2
7351-31-7	4
7351-33-30	10
7351-33-40	12
7351-33-900	15
7351-34-45	20
7351-34-66	25
7351-34-68	27
7351-34-73	31
7351-34-76	32
7351-34-803	33
7351-34-901	34
Magellan Drive	35

The Tier 2 modeling of groundwater data followed the same approach used for modeling with the soils/soil gas data to calculate attenuation factors and predict indoor air concentrations for the 13 selected EAPCs. The model-predicted indoor air concentrations were subsequently provided as input to the risk calculations. The Tier 2 modeling of groundwater data focused on the primary risk drivers identified in the Tier 1 analysis, benzene, PCE and TCE. A detailed discussion of the Tier 2 vapor diffusive transport

analysis, including the Tier 2 model input parameters and calculated EPCs, is presented in Appendix F.

#### 4.4 ECOLOGICAL RISK ASSESSMENT

The ecological risk evaluation for the site was originally limited to a qualitative screening evaluation due to the highly developed, urban setting of the site and lack of any sensitive habitats or special status species. However, the open grassy area located along the southern boundary of the site may serve as a raptor habitat, as confirmed by on-site observations of an American kestrel (*Falco sparverius*). Based on these sightings, EPA requested a quantitative ERA be completed, but stipulated that the assessment could be limited to the individual kestrel at the site. EPA further acknowledged that the habitat value was limited in terms of supporting viable populations of wildlife species and that special status species are not likely to be present.

A summary of the ERA and its conclusions is presented below. The complete ERA is presented in Appendix I.

A tiered approach was used to assess the potential for risk to the local kestrel population based on the sightings of an individual kestrel inferred to be residing within an approximately 24-acre undeveloped area. This 24-acre area is referred to as the Total Habitat. Approximately 15 acres of the Total Habitat are located within the Superfund site and referred to as the Onsite Habitat, with the balance of the area located immediately south of the Superfund site. The kestrel was assumed to consume soil invertebrates and incidentally ingest soil exclusively from the Onsite and Total Habitats.

Conservative assumptions were incorporated into a Tier 1 ERA regarding the soil depth to which kestrels could be exposed (0 to 6 feet bgs), and avian toxicity reference values (TRVs) not specific to the kestrel were used for all constituents of interest (COIs). In Tier 2, these assumptions were refined, and exposure was assumed to be limited to soils from ground surface to 1.5 feet bgs. The Tier 2 ERA also used kestrel-specific TRVs for DDT metabolites. The results of the Tier 2 ERA are judged to be more representative of actual site conditions and are therefore preferred over the Tier 1 results.

Low and high TRVs were applied in both Tier 1 and 2 to generate a range of HQs for each COI. These TRVs correspond to chronic no-observable-adverse-effect levels and lowest-observable-adverse-effect levels (NOAELs and LOAELs), or in the case of the high TRVs from the DTSC Biological Technical Assistance Group (BTAG), a mid-range level of effects. Typically, TRVs derived from chronic NOAELs representative of a sensitive endpoint, such

as reproduction, are appropriate for protection of individuals. Depending on the study, TRVs derived from chronic LOAELs may be adequately protective of populations. Because the kestrel was the only target receptor included in the evaluation and this bird is not a state or federally threatened or endangered species, the assessment endpoint for this ERA focuses on protection at the population level. Based on this objective, risk management decisions for the site may emphasize the HQs calculated from the Tier 2 high TRVs.

The following table summarizes the COIs with elevated HQs or HIs based on the Tier 2, High and low TRV evaluations for the Onsite and Total Habitats:

Analyte Class	Chemical of Interest	HQ or HI (Low TRV)		HQ or HI (High TRV)	
		Onsite Habitat	Total Habitat	Onsite Habitat	Total Habitat
Inorganics	Cadmium	4	3	HQ ≤ 1	HQ ≤ 1
	Chromium	(not a COI)	2	(not a COI)	HQ ≤ 1
	Lead	3	3	HQ ≤ 1	HQ ≤ 1
	Mercury	(not a COI)	2	(not a COI)	HQ ≤ 1
	Nickel	4	13	HQ ≤ 1	HQ ≤ 1
	Zinc	(not a COI)	4	(not a COI)	HQ ≤ 1
	Cadmium + Lead + Mercury	7	9	HI ≤ 1	2
Pesticides/ PCBs	4,4'-DDD	7	5	HQ ≤ 1	HQ ≤ 1
	4,4'-DDE	21	14	2	HQ ≤ 1
	4,4'-DDT	67	44	7	4
	Total Pesticides/PCBs	95	62	9	6

The highest Tier 2 HQs are for DDT metabolites and the DDT HQs are higher for the Onsite Habitat compared to the Total Habitat. DDT is not known to have been used at the former synthetic rubber plant.

The HQs for inorganics are generally lower for the Onsite Habitat than for the Total Habitat. With the exception of mercury, risk-driving inorganic detections (the area of maximum concentrations) occurred in a limited off-site area within the Total Habitat. This area is approximately 13 feet in diameter and is associated with location P1-G. If the detections of nickel in the P1-G area were omitted from the Tier 2 risk evaluation, the Total Habitat Tier HQs based on the low TRV would decrease from 13 to 4. While not accounted for in the risk calculations, this limited area in which elevated inorganic concentrations typically occur further reduces the likelihood of significant exposure by the kestrel, since the majority of kestrel foraging area contains significantly lower concentrations of inorganics.

The ERA results likely overestimate risk due to the conservative assumptions used in the exposure assessment, including area use and seasonality factors of 1.0. The available kestrel foraging resources present at the site are limited by the relatively small open area and highly urbanized setting. The area is zoned for commercial/industrial/residential use and is covered with infrastructure and small residential lots. However, the Total Habitat may support at least one kestrel that could defend this small foraging area throughout most of the year, except possibly during the breeding season.

Although adverse effects to an individual kestrel may occur from exposure to pesticides in surface soils from the Onsite Habitat, effects to the population are expected to be negligible. A less conservative assessment that incorporates more site-specificity regarding the actual bioavailability and bioaccumulation potential of constituents in soil, and site use and seasonal population variations, would reduce the risk estimates and would likely demonstrate a low potential for adverse effects to populations, and possibly even to individual kestrels. No further evaluation of the kestrel is expected to be necessary for the site.

## 5.0 TOXICITY ASSESSMENT

Toxicity assessment characterizes the relationship between the magnitude of exposure to a COPC and the nature and magnitude of resulting adverse health effects. Adverse health effects are classified as carcinogenic and noncarcinogenic. Toxicity criteria are generally developed based on the threshold approach for noncarcinogenic effects and the non-threshold approach for carcinogenic effects. This division relates to the currently-held scientific opinion that the mechanism of action for these endpoints differ (Johnson, 1991). It is assumed that there is no level of exposure that does not have a finite possibility of causing cancer for carcinogens (i.e., there is no threshold dose for carcinogenic effects). That is, a single exposure of a carcinogen, at any level, results in an increased probability of developing cancer.

It is believed that organisms have protective mechanisms that must be overcome before the toxic endpoint results (i.e., there is a threshold dose) for noncarcinogens. For example, if a large number of cells perform the same or similar functions, it would be necessary for significant damage or depletion of these cells to occur before a toxic effect could be seen. As a result, a range of exposures exists from zero to some finite value that can be tolerated by the organism with essentially no chance of expression of adverse effects (USEPA, 1989). Some chemicals may elicit both carcinogenic and noncarcinogenic effects.

Chronic toxicity criteria were selected, in order of preference, from the following sources:

- (1) Office of Environmental Health Hazard Assessment (OEHHA) Toxicity Criteria Database, online (CalEPA, 2004);
- (2) USEPA's Integrated Risk Information System (IRIS) (USEPA, 2004b);
- (3) USEPA Health Effects Assessment Summary Tables (HEAST; USEPA, 1997b); and/or
- (4) USEPA-NCEA Superfund Health Risk Technical Support Center.

Toxicological profiles for those chemicals that contributed significantly to cancer risk or noncancer hazard are presented in Appendix G.

### 5.1 HEALTH EFFECTS CRITERIA FOR POTENTIAL CARCINOGENS

Potential carcinogenic effects resulting from human exposure to chemicals are generally estimated quantitatively using oral cancer slope factors (CSFs) or inhalation unit risk factors. Oral CSFs are expressed in units of (mg/kg-day)<sup>-1</sup>. Inhalation unit risk factors were

converted, when needed, from units of  $(\mu\text{g}/\text{m}^3)^{-1}$  to  $(\text{mg}/\text{kg}\text{-day})^{-1}$  by assuming an inhalation rate of  $20 \text{ m}^3$  per day, body weight of 70 kg, and that absorption is equivalent by either route (USEPA, 1989) to characterize potential carcinogenic risks.

Oral and inhalation CSFs are derived by CalEPA and USEPA from the results of chronic animal bioassays, human epidemiological studies, or both. Animal bioassays are usually conducted at dose levels that are much higher than those likely to be produced by human exposure to environmental media. These high dose levels are used to detect possible adverse effects in the relatively small test populations used in the studies.

CSF data are extrapolated using mathematical models since humans are generally exposed at lower doses. The linearized multistage model is most commonly used to estimate the largest possible linear slope (95UCL) at low extrapolated doses that is consistent with the data (Crump et al., 1976). The 95UCL slope of the dose-response curve is subjected to various adjustments, and an interspecies scaling factor is usually applied to derive a CSF for humans. Dose-response data derived from human epidemiological studies are fitted to dose-time-response curves on an ad hoc basis.

Conservative (i.e., health protective) assumptions are generally applied to the models to provide rough estimates of the upper limits on potential carcinogenic potency. The actual risks associated with exposure to a potential carcinogen quantitatively evaluated on the basis of its CSF are not likely to exceed the risks estimated and may be much lower or even zero.

CSFs available for carcinogenic COPCs are presented in Tables 16 and 17. When available, CalEPA CSFs were also identified.

USEPA assigns weight-of-evidence classifications to potential carcinogens in addition to deriving a quantitative estimate of cancer potency. Chemicals are classified as either Group A, Group B1, Group B2, Group C, Group D, or Group E, as defined below:

- **Group A** chemicals (known human carcinogens) are agents for which there is sufficient evidence to support the causal association between exposure to the agents in humans and cancer.
- **Group B1** chemicals (probable human carcinogens) are agents for which there is limited evidence of carcinogenicity in humans.
- **Group B2** chemicals (probable human carcinogens) are agents for which there is sufficient evidence of carcinogenicity in animals, but inadequate evidence or lack of evidence of carcinogenicity in humans.

- **Group C** chemicals (possible human carcinogens) are agents for which there is limited evidence of carcinogenicity in animals and inadequate or lack of human data.
- **Group D** chemicals (not classifiable as to human carcinogenicity) are agents with inadequate human and animal evidence of carcinogenicity or for which no data are available.
- **Group E** chemicals (evidence of noncarcinogenicity in humans) are agents for which there is no evidence of carcinogenicity from human or animal studies, or both.

Weight-of-evidence classifications are also presented in Tables 16 and 17.

## 5.2 HEALTH EFFECTS CRITERIA FOR POTENTIAL NONCARCINOGENS

Potential noncarcinogenic effects resulting from human exposure to chemicals are estimated quantitatively using reference doses (RfDs) for ingested chemicals and reference concentrations (RfCs) for inhaled chemicals. RfDs and RfCs are only available for oral and inhalation exposures, as was the case for the CSFs. The oral RfDs are used to evaluate the dermal route of exposure in the absence of criteria specific to the dermal exposure pathway.

The RfD, expressed in units of milligrams of chemical intake per kilogram of body weight per day (mg/kg-day), is an estimate of the maximum human exposure level that can be present without an appreciable risk of deleterious effects during a designated time. The RfC is expressed in units of milligrams of chemical per cubic meter of air (mg/m<sup>3</sup>) and is an estimate of the maximum air concentration that can be present without an appreciable risk of deleterious effects. RfCs assume a human body weight of 70 kg and an inhalation rate of 20 m<sup>3</sup>/day.

RfDs and RfCs are developed by the USEPA RfD/RfC workgroup on the basis of a wide array of noncarcinogenic health effects. They are usually derived from either human studies involving workplace exposures or from animal studies, and are adjusted using generic uncertainty factors, as described by Barnes et al. (1987). The RfD and RfC provide benchmarks against which human intakes of chemicals resulting from exposure to contaminated environmental media are compared.

Chronic Reference Exposure Levels (RELs) are similar to RfCs, and have been developed for inhalation exposure by CalEPA for the Air Toxics Hot Spots program. When available, these values were used in the risk assessment.

The duration of exposure is considered in the development of RfDs and RfCs. Exposure duration is divided into three categories following RAGS (USEPA, 1989):

- **Acute** refers to exposures for short durations measured in seconds, minutes, or hours and to effects which appear promptly after exposure.
- **Subchronic** generally refers to exposures of intermediate duration from two weeks up to seven years, and to effects that develop within the same time frame.
- **Chronic** refers to prolonged or repeated exposures with duration of days, months, or years, and describes on-going exposures and effects that develop only after exposures from seven years to a lifetime.

Chronic RfDs and RfCs have been conservatively selected to evaluate risks in this assessment, as most potential exposures at the site are assumed to be occurring for greater than seven years. The noncancer toxicity criteria and associated uncertainty factors used in their derivation are presented in Tables 18 and 19.

### 5.3 DERMAL TOXICITY CRITERIA

CalEPA and USEPA have only developed CSFs and RfDs for the oral and inhalation routes of exposure at the present time. The oral factors are used for the dermal toxicity factors in the absence of specific dermal route values, as recommended by USEPA until more suitable dermal toxicity values become available. While there are uncertainties involved in presuming a chemical is equally toxic across different exposure routes, the increased uncertainty created by this default assumption is much less than if it is presumed that this pathway is without risk and is therefore not quantitatively evaluated in the risk assessment (i.e., greater uncertainty exists in the risk estimate if a route-to-route extrapolation from oral to dermal was not conducted).

### 5.4 HEALTH EFFECTS FROM LEAD

The traditional RfD approach to the evaluation of chemicals is not applied to lead because most human health effects data are based on blood lead concentrations, rather than external dose (CalEPA, 1992). Blood lead concentration is an integrated measure of internal dose, reflecting total exposure from site-related and background sources. A clear no observed effects level (NOEL) has not been established for such lead-related endpoints as birth weight, gestation period, heme synthesis and neurobehavioral development in children and fetuses, and blood pressure in middle-aged men. Dose-response curves for these endpoints appear to extend down to 10 micrograms per deciliter ( $\mu\text{g}/\text{dl}$ ) or lower (ATSDR, 1993).

The DTSC has developed a methodology for evaluating exposure and the potential for adverse health effects resulting from exposure to lead in the environment (CalEPA, 1992), and has provided a spreadsheet (LeadSpread) based on its guidance for evaluating lead toxicity (CalEPA, 1993). DTSC's LeadSpread presents an algorithm for estimating blood lead concentrations in children and adults based on a multi-pathway analysis. The output of the LeadSpread Model is presented in Appendix D. The USEPA adult lead model (USEPA, 2003a) is used for assessing risks associated with nonresidential adult exposure to lead in soils. The default parameters in the USEPA adult lead model are modified to reflect the exposure assumptions for the construction worker and trench worker. The output from the USEPA adult lead model is also presented in Appendix D. The DTSC lead model was used for residential exposures, while the USEPA lead model was used for commercial worker and trench worker exposures.

## 6.0 RISK CHARACTERIZATION

Risk characterization integrates the results of the toxicity assessment (Section 5.0) and the exposure assessment (Section 4.0) to estimate potential carcinogenic risks and adverse noncarcinogenic health effects associated with exposure to chemicals detected at the site. This integration provides quantitative estimates of risk and noncancer hazard that are then compared to acceptable standards.

The acceptable standards or acceptable risk levels have been established by regulatory agencies. For example, the USEPA has established an acceptable risk range for Superfund sites. The National Contingency Plan (NCP; 40 CFR 300) indicates that lifetime incremental cancer risks posed by a site should not exceed a range of one in one million ( $1 \times 10^{-6}$ ) to one hundred in one million ( $1 \times 10^{-4}$ ) and noncarcinogenic chemicals should not be present at levels expected to cause adverse health effects (i.e., a hazard index [HI] greater than 1). Other relevant guidance (USEPA, 1991b) additionally states that sites posing a cumulative cancer risk of less than  $10^{-4}$  and hazard indices less than unity (1.0) for noncancer endpoints are generally not considered to pose a significant risk warranting remediation. The California Hazardous Substances Account Act (HSAA) incorporates the NCP by reference, and thus also incorporates the acceptable risk range set forth in the NCP. The Resource Conservation and Recovery Act (RCRA) Corrective Action program incorporates this same range of potential health risks as the “acceptable risk range” for determining whether corrective action is warranted at RCRA facilities and for closure purposes. Finally, The Safe Drinking Water and Toxic Enforcement Act of 1986 (California Proposition 65) regulates chemical exposures to the general population and is based on an acceptable risk level of  $1 \times 10^{-5}$ .

The maximum acceptable risk level for a site is between  $10^{-4}$  and  $10^{-6}$ , and is selected on a case-by-case basis by USEPA. The risk range between  $10^{-4}$  and  $10^{-6}$  is commonly called the “discretionary risk range.” For the purposes of this section, estimated commercial and trench worker risks greater than a  $5 \times 10^{-5}$  benchmark are discussed. This benchmark was selected by EPA rather than the  $10^{-4}$  level to provide an additional factor of safety when distinguishing EAPCs that *warrant* remedial action (typically those where risk  $> 10^{-4}$  or HI is  $> 1$ ) from those that *may warrant* remedial action. Risks estimated for hypothetical future residents are compared to both the midpoint of the discretionary risk range ( $1 \times 10^{-5}$ ) and the upper-bound of the discretionary risk range ( $1 \times 10^{-4}$ ) to highlight those parcels where the risk is greatest. Both commercial/trench worker and residential risks are also compared to the  $10^{-6}$  level.

The process of risk assessment is an iterative process where factual site, receptor, and chemical-specific data are used when available. When specific data are not available, conservative (i.e., health protective) assumptions are utilized. The use of repeated, conservative assumptions can lead to overly conservative estimations of risk, but certainly

provides an upper bound estimate of the actual risk. Thus, for any site, the estimated risk level reflects an upper bound estimate of the most probable risk. The most probable risk is likely to be much less, perhaps as low as zero, and probably not measurable in the potentially exposed population.

Risks estimated in this assessment are of two types, CT (an average condition) and RME, in accordance with USEPA guidance (USEPA, 1989). Estimates of exposure under RME conditions are calculated by combining exposure factors so that the result is the maximum exposure that is reasonably expected to occur (USEPA, 1989). RMEs are intended to place conservative upper bounds on the potential risks, meaning that each risk estimate is unlikely to be underestimated, but likely to be overestimated. The RMEs for a given pathway are derived in this study by combining the upper bound estimate of the concentration for each chemical (either maximum or 95UCL) with reasonable maximum values describing the extent, frequency, and duration of exposure, as discussed in Section 4.0, Exposure Assessment.

USEPA (1999b), however, also recommends consideration of a more probable case, the CT exposure, which represents more “typical” or average exposure conditions. For the CT exposure, the average chemical concentration is used in combination with exposure factors that represent the 50th percentile of exposure. The risk estimates for the CT exposure in this study are based on the average concentrations of COPCs and exposure parameters.

Excess cancer risk is estimated by multiplying the LADD by the chemical carcinogenic toxicity criteria or CSF in the risk characterization step of the risk assessment. The equation used to estimate the excess cancer risk is:

$$\text{Excess Cancer Risk} = \text{LADD} \times \text{CSF}$$

Chemical-specific hazard quotients are estimated by calculating the ratio of the ADD to the corresponding chronic RfD for noncarcinogenic effects. The equation used to estimate the hazard quotient is:

$$\text{Hazard Quotient} = \frac{\text{ADD}}{\text{RfD}}$$

The hazard quotients are then summed to form a HI, which is compared to an acceptable hazard level. HIs less than the benchmark HI of 1 indicate that no adverse health effects are predicted from exposure to COPCs at the site.

The cancer risks and noncancer hazards were calculated for the EAPCs identified in Section 3.0. Potential exposures have been evaluated for the three receptor types (commercial worker,

hypothetical future resident, and trench worker) for each EAPC. The various exposure media and pathways evaluated for each receptor type were previously summarized in the table presented in Section 4.2.4.

Potential vapor migration from the subsurface was evaluated using environmental fate and transport modeling in two tiers of analysis, as discussed in Section 4.3.6. Summaries of the estimated cumulative risk and hazard at each EAPC for commercial workers, hypothetical future residents, and trench workers are presented in Tables 20, 21, and 22, respectively. The chemical- and pathway-specific risks for each EAPC that comprise the summary risk estimates shown on these tables are presented in Appendix D.

Tables 23 and 24 present information regarding the risk-driving chemicals for each media and pathway of concern for commercial worker and hypothetical future residential exposures, respectively. For the indoor air pathway, these tables present the risk estimates from soil and soil gas, as well as the indoor air measurements. The type of EPC (maximum or 95UCL) selected for input into the calculations are also presented in Tables 23 and 24 to illustrate the conservatism of the input concentration for the calculation of dose and risk. The risk and hazard values for the groundwater-to-indoor air scenario are presented in Tables 25 through 27.

The following sections summarize the significant cancer risk and HI estimates for the three receptor populations.

## 6.1 COMMERCIAL WORKER EXPOSURES

Estimates of cancer risk and noncancer hazard were calculated for commercial workers assuming they are exposed to chemicals via incidental soil ingestion, dermal contact with soil, outdoor inhalation of fugitive dust/vapors, and indoor inhalation of vapors.

### 6.1.1 Surface and Shallow Soil Exposures – Outdoor Pathways

Cancer risk estimates from potential RME exposures to chemicals in surface and shallow soils exceeded  $5 \times 10^{-5}$  for EAPCs 2 and 16, both with risk estimates of approximately  $1 \times 10^{-4}$  (see Table 20 and Figure 20). For EAPC 2, the primary chemicals that contributed to risk were benzo(a)pyrene at  $1.0 \times 10^{-4}$  and indeno(1,2,3-cd)pyrene at  $1.5 \times 10^{-5}$  in shallow soil via the incidental ingestion and dermal contact pathways (Table 23). The EPCs for these chemicals are based on the maximum detected concentrations from the soil samples collected from this parcel. The one soil sample with the maximum concentration had the only elevated cPAH concentrations that produced a BaP-equivalent concentration greater than the maximum Southern California background value of 4.05 mg/kg (see Section 3.4.1.2).

The primary chemicals that contributed to potential risk at EAPC 16 were benzo(a)pyrene ( $5.8 \times 10^{-5}$ ) and dibenzo(a,h)anthracene ( $3.9 \times 10^{-5}$ ) via the incidental ingestion and dermal contact pathways, and benzene ( $1.1 \times 10^{-5}$ ) via inhalation of volatile chemicals in outdoor air (Table 23). The elevated risk contributions from these chemicals are due to elevated detection limits at sampling locations SBL0036 and SBL0069 rather than actual detections at EAPC 16. More specifically, benzo(a)pyrene and dibenzo(a,h)anthracene were not detected at SBL0036 and SBL0069, but had elevated detection limits due to interference from high concentrations of one or more other compounds. As a result, the detection limits were substantially above levels of concern and the associated risk contribution (based on  $\frac{1}{2}$  the respective detection limits; see Section 3.5) was also elevated. In short, the estimated potential risk is elevated even though the compounds of concern may not even be present in the soil samples.

The 95UCL values and maximum detected concentrations for each risk-driving chemical at EAPC 16 are presented below to illustrate the substantial difference in EPCs that occur due to the elevated detection limits:

<u>Chemical</u>	<u>95UCL (mg/kg)</u>	<u>Maximum Detected Concentration (mg/kg)</u>
Benzo(a)pyrene	7.43	0.043
Dibenzo(a,h)anthracene	14.7	0.012
Benzene	11.2	0.928

If the maximum detected concentrations for the above compounds were used instead of  $\frac{1}{2}$  their detection limit in the risk calculation, the associated potential risks would be reduced to  $3.4 \times 10^{-7}$ ,  $3.2 \times 10^{-8}$ , and  $9.5 \times 10^{-7}$  respectively. The cumulative risk for EAPC 16 would decrease from  $1 \times 10^{-4}$  using the UCL approach to  $6 \times 10^{-6}$  using maximum detected concentrations for those chemicals influenced by elevated detection limits.

The estimated RME risks for outdoor shallow soil exposures are based on the assumption that bare soil is available for contact when, in fact, almost the entire site (including EAPCs 2 and 16) is covered with buildings, parking lots, and landscaping, eliminating exposure. In addition, elevated risks are based on chemical data from only a few sample locations. Any commercial worker exposure to these locations would be of much shorter exposure frequency and duration than assumed in the risk assessment. All cancer risk estimates were below  $5 \times 10^{-5}$  for CT exposures (Table 20).

Noncancer HI estimates for all EAPCs under the RME and CT exposure scenarios did not exceed the benchmark level of 1 (Table 20), indicating that the soil exposure pathways for

surface soil and shallow soil do not pose an unacceptable noncancer hazard for commercial workers. The chemical-specific risk and hazard estimates for this scenario are presented in Appendix D.

Lead concentrations in soil at the site were below background for all EAPCs except EAPC 2. Based on the EPC of 492 mg/kg for lead in soil at EAPC 2, the USEPA lead model predicted a blood lead level for a commercial worker that did not exceed the blood lead goal of 10 µg/dl (Table D-3a in Appendix D).

### 6.1.2 Indoor Air Exposures

Parcel-specific indoor air exposures were evaluated for commercial workers using: (1) Tier 1 plus Tier 2 (Tier 2 considered only BTEX) modeled indoor air concentrations from shallow and deep soils; (2) Workplace (indoor) air monitoring data; and (3) Tier 1 plus Tier 2 modeled indoor air concentrations from groundwater. Results for each of the three approaches are presented in the subsections below. Results from modeling of shallow soil are given greater credence than those from indoor air monitoring based on results from the correlation analysis, as described in Section 4.3.6.1. Results based on modeling of groundwater concentrations are limited to 13 EAPCs where there were limited shallow soil data to provide additional information.

#### 6.1.2.1 Modeled Indoor Air Exposures from Soil/Soil Gas Data

Indoor air risk estimates for commercial workers using Tier 1 and Tier 2 modeling of shallow soil/soil gas and deep soil/soil gas data at each EAPC are presented in Table 20. The combined Tier 1 (for non-BTEX compounds) and Tier 2 (for BTEX) cancer risk estimates from potential RME exposures to chemicals detected in shallow soil/soil gas exceeded  $5 \times 10^{-5}$  for EAPC 16 ( $4 \times 10^{-4}$ ) and EAPC 23 ( $1 \times 10^{-4}$ ). The combined Tier 1 and Tier 2 risk estimates due to the deep soil/soil gas data exceed  $5 \times 10^{-5}$  only for EAPC 24 ( $2 \times 10^{-4}$ ). However, the risk estimate from the shallow soil/soil gas data are more representative than the deep soil/soil gas data for evaluation of the vapor intrusion pathway and the cancer risk calculated from the shallow soil/soil gas data for EAPC 24 is  $5 \times 10^{-5}$ . Cancer risk estimates under the CT exposure scenario did not exceed  $5 \times 10^{-5}$  for any EAPC.

Noncancer HI estimates for all EAPCs under the RME and CT exposure scenarios did not exceed the benchmark level of 1, except for EAPC 16, which had a HI of 3 under the RME scenario (Table 20).

### 6.1.2.2 Indoor Air Exposures from Workplace Air Monitoring Data

When considering indoor air exposures resulting from subsurface contamination, it is important to consider contributing sources of contamination other than those that may be present in the subsurface. These other sources include indoor chemical use and chemicals in outdoor (ambient) air. Based on the results of the correlation analysis (Section 4.3.6.1), indoor chemical use and/or ambient air are inferred to be the primary sources of indoor air contaminants rather than vapor intrusion from the subsurface. Appendix H presents supporting information regarding other sources of indoor air contamination.

VOCs are emitted by a wide array of indoor products numbering in the thousands. Examples include: petroleum fuels, solvents, paints and lacquers, paint strippers, cleaning supplies, pesticides, building materials and furnishings, office equipment such as copiers and printers, correction fluids and carbonless copy paper, graphics and craft materials including glues and adhesives, permanent markers, and photographic solutions.

Outdoor air contamination is also an important consideration when evaluating indoor air exposures. The VOCs driving the indoor air risk and hazard estimates are commonly found in background ambient air throughout the Los Angeles area due to a variety of sources such as automobile emissions, industrial and manufacturing facility emissions and commercial chemical use.

Ambient concentrations of VOCs were detected in outdoor air at the site during the Workplace Air Monitoring Program (Table 7), as well as at the nearest ambient air monitoring station in Long Beach, which is operated by the California Air Resources Board (Table 28). The local background risk and hazard using the outdoor air data from the Workplace Air Monitoring Program and RME assumptions are  $5 \times 10^{-5}$  and 0.2, respectively, as presented in Table 20. Indoor air studies conducted for California additionally indicate that benzene concentrations can range up to  $0.13 \text{ mg/m}^3$  in indoor air (see Table 28 and references therein). This concentration is above the maximum detected concentration for benzene of  $0.0958 \text{ mg/m}^3$  detected in all indoor air samples collected at the Del Amo site (Table 6).

Table 20 presents the cancer risk and noncancer hazard estimates for the parcels where workplace (indoor) air monitoring was conducted. Cancer risk estimates from potential RME exposures to chemicals detected during the Workplace Air Monitoring Program exceeded the risk level of  $5 \times 10^{-5}$  for 12 EAPCs. The highest risks were estimated for EAPCs 4 and 19 (both at  $3 \times 10^{-4}$ ), EAPC 22 ( $2 \times 10^{-4}$ ), and EAPCs 5, 18, and 28 ( $1 \times 10^{-4}$ ). Benzene and PCE were the primary risk drivers for the indoor air pathway (Table 23). Cancer risk estimates for all EAPCs under the CT exposure scenario did not exceed  $5 \times 10^{-5}$ .

Noncancer HI estimates for all EAPCs under the RME and CT exposure scenarios did not exceed the benchmark level of 1 (Table 20), indicating that the indoor air pathway using monitoring data does not pose an unacceptable noncancer hazard for commercial workers.

The indoor air risks and HIs based on the data from the Workplace Air Monitoring Program overestimate the risk due to vapor intrusion because the contribution from local background (ambient) air and indoor sources associated with normal buildings' facilities and operations bias high the measured indoor air concentrations (see Appendix H). Excluding the contribution due to background concentrations in outdoor air, the estimated RME risks for those EAPCs with elevated risk driven by indoor air data are within the NCP discretionary risk range, with the exception of EAPC 19.

### 6.1.2.3 Modeled Indoor Air Exposures from Groundwater Data

Potential indoor air exposures associated with vapors emanating from groundwater were initially evaluated using maximum COPC concentrations for groundwater from the entire site and the Tier 1 vapor model. The Tier 1 chemical-specific risk estimates for site-wide exposures to groundwater vapors in indoor air are presented in Table 25 and Appendix D. The site-wide maximum benzene concentration in groundwater resulted in an estimated risk for commercial worker exposures above the upper bound of the NCP discretionary risk range ( $10^{-4}$ ). The site-wide maximum detected concentration of PCE resulted in an estimated risk of  $1 \times 10^{-5}$  for commercial worker exposures. All other chemical-specific risks for commercial worker exposures were below  $1 \times 10^{-5}$ . Noncancer hazard estimates greater than the benchmark level of 1 were due to benzene. The table below summarizes the risks and noncancer hazard estimates for commercial worker exposures using the Tier 1 analysis.

Groundwater COPCs	Commercial Exposure Scenario	
	Cancer Risk	Hazard Index
Benzene	1.0E-02	1.7E+01
Tetrachloroethene	1.4E-05	1.8E-01
Trichloroethene	2.8E-06	6.5E-03
All Other Chemicals	6.7E-07	7.8E-01

Chemicals with an elevated risk for commercial worker exposures (i.e., cancer risk greater than  $1 \times 10^{-6}$  or an HI greater than 1) based on the site-wide maximum groundwater concentration and Tier 1 modeling were then evaluated on an EAPC-specific basis. EAPC-specific evaluations were conducted for benzene, PCE and TCE, and limited to the 13 EAPCs where there were only limited or no shallow soil gas data (see Section 4.3.6.2 and Appendix F). EPCs for PCE and TCE were derived using the Tier 1 model, and the Tier 2 model was used for benzene.

Individual and cumulative risk estimates for PCE, TCE and benzene at the 13 EAPCs are presented in Table 26. None of the cancer risk estimates for the 13 EAPCs exceeded  $5 \times 10^{-5}$  under the RME exposure scenario. The highest calculated cumulative risk for the three risk drivers was  $2 \times 10^{-5}$  at EAPC 15. This parcel corresponds to a Los Angeles Department of Water and Power utility corridor, where occupied buildings are not present and are unlikely for the foreseeable future. The estimated noncancer HIs for the 13 EAPCs did not exceed the benchmark level of 1. Cumulative risks and HIs from compounds other than benzene, PCE and TCE are not significant and would not alter risk results for the groundwater to indoor air pathway.

## 6.2 HYPOTHETICAL FUTURE RESIDENTIAL EXPOSURES

Estimates of cancer risk and noncancer hazard were calculated for hypothetical future residents assuming they are exposed to chemicals via incidental soil ingestion, dermal contact with soil, outdoor inhalation of fugitive dust/vapors, as well as indoor inhalation of vapors. However, current land use zoning precludes pure residential development within the Del Amo site, with the exception of a dwelling occupied by a single worker serving as a watchman or caretaker where industrial development is present. Application of other institutional control mechanisms that could enhance existing controls and prevent inappropriate land uses at the site in the future are being evaluated as part of the FS. Residential exposure pathways are indicated as only potentially complete in the CSM (Section 4.1) due to the existing zoning and hypothetical nature of this pathway. Future residential development is unlikely based on current zoning. The results of the risk assessment for the hypothetical future residential scenario are discussed further below.

### 6.2.1 Shallow Soil Exposures – Outdoor Pathways

Cancer risk estimates from potential RME exposures to chemicals in shallow soils ( $\leq 15$  feet bgs) exceeded the midpoint of the discretionary risk range ( $10^{-5}$ ) for 16 EAPCs (Table 21). Three of these EAPCs (2, 16, and 23) have risk estimates for direct contact and outdoor air inhalation exposures exceeding the upper bound of the risk range ( $10^{-4}$ ). The highest risks were estimated for EAPCs 16 ( $2 \times 10^{-3}$ ) and 23 ( $9 \times 10^{-4}$ ). For EAPC 16, the primary chemicals that contributed to risk (chemical-specific risk  $>10^{-4}$ ) were benzo(a)pyrene, dibenzo(a,h)anthracene, and PCE via direct contact exposures (incidental soil ingestion and dermal contact), as well as benzene and PCE via outdoor inhalation of volatile chemicals (Table 24). For EAPC 23, the primary risk-driving chemical was benzene via outdoor air inhalation. The calculated risk for EAPC 16 was influenced by elevated detection limits. The estimated cumulative risk for EAPC 16 would decrease from  $2 \times 10^{-3}$  using the UCL

approach to  $2 \times 10^{-5}$  using maximum detected concentrations for those chemicals influenced by elevated detection limits.

Noncancer HI estimates from potential RME exposures exceeded the benchmark level of 1 for seven EAPCs (Table 21). The highest noncancer hazards were estimated for EAPC 14 (HI = 15), EAPC 16 (HI = 12) and EAPC 23 (HI = 7). Copper contributed the majority to the HI for EAPC 14, ethylbenzene and PCE for EAPC 16, and benzene and ethylbenzene for EAPC 23 (Table 24).

All cancer risk estimates were at or below  $1 \times 10^{-5}$  under the CT exposure scenario, and the noncancer HIs were below the benchmark level of 1, with the exception of EAPC 14, which had an HI of approximately 2 (Table 21). The chemical-specific risk and hazard estimates for this scenario are presented in Appendix D.

Based on the maximum EPC of 586 mg/kg for lead detected in soil at EAPC 2, the DTSC lead model predicted 95th and 99th percentile blood lead levels that exceed the blood lead goal of 10 µg/dl for a hypothetical future child resident (Table D-3b in Appendix D). All other lead concentrations detected in soils at the site were within background.

## 6.2.2 Indoor Air Exposures

Indoor air exposures are presented for shallow soil/soil gas, deep soil and groundwater media in this section. Shallow soil/soil gas is the preferred medium as it represents the contamination that is closest to the building slab. Vapor transport modeling of groundwater data was completed to estimate indoor air EPCs and risks for the 13 EAPCs where shallow soil/soil gas data were limited, as previously explained in Section 4.3.6.3. The final indoor air risk value for these 13 EAPCs was conservatively selected as the maximum value from the shallow soil/soil gas, deep soil, and groundwater evaluations.

### 6.2.2.1 Modeled Indoor Air Exposures from Soil/Soil Gas

Indoor air risk estimates for hypothetical future residents using Tier 1 and Tier 2 modeling of shallow soil/soil gas and deep soil/soil gas data at each EAPC are presented in Table 21. Benzene and PCE were the primary risk drivers for the Tier 1 analysis (Table 24). Estimated indoor air concentrations and risks were much less for several of the EAPCs where benzene was the primary risk driver when the Tier 2 site-specific calibrated model is used to account for biodegradation (Table 21).

Risks were typically estimated based on the EAPC-specific shallow soil/soil gas data, except for EAPC 15, where only limited shallow data was available and therefore the deep data were

relied upon. The risks and noncancer hazards for all EAPCs are presented in Table 21. Chemicals that contribute significantly to the risk are presented in Table 24.

Estimated cancer risks were above  $1 \times 10^{-4}$  for seven EAPCs, with the highest estimated risks at EAPC 16 ( $4 \times 10^{-2}$ ), EAPC 23 ( $1 \times 10^{-2}$ ), and EAPC 24 ( $7 \times 10^{-4}$ ). Benzene and PCE were the most common chemicals that contributed predominantly to risk via the indoor air pathway. Noncancer hazard estimates were above 1 for seven EAPCs, with the highest estimates at EAPC 16 (HI = 520) and EAPC 23 (HI = 140). Chloroform, PCE, and TCE contributed predominantly to the hazard at EAPC 16, whereas chloroform and PCE contributed the majority to the hazard at EAPC 23.

The elevated residential risk estimates for some EAPCs were primarily due to the use of elevated detection limits as the exposure point concentration (Table 12a and 24). These included PCE (EAPCs 5, 23, 24, and 35), TCE (EAPCs 5, 16, 23, and 24), and chloroform (EAPCs 16 and 23). The indoor air risk of  $4 \times 10^{-2}$  for EAPC 16 would decrease about two orders of magnitude to  $4 \times 10^{-4}$  if the maximum detected concentrations of these chemicals were used instead of the EPCs that are biased high due to elevated detection limits.

Risk and hazard estimates under the CT exposure scenario were generally approximately an order of magnitude lower than the RME estimates (Table 21). CT cancer risk estimates were still above  $10^{-5}$  at EAPCs 16 and 23, while noncancer hazards were still above the benchmark level of 1 at EAPCs 7 and 16.

The primary risk drivers for the residential indoor air pathway, benzene and PCE, are both detected in ambient air within the Los Angeles area, as shown in Table 28. According to the California Air Resources Board (CARB) (2001), estimated risks at the North Long Beach station for exposures to the general population associated with VOCs similar to those at the Del Amo site range from  $2 \times 10^{-7}$  (TCE) to  $1 \times 10^{-4}$  (benzene). The local background risk and hazard for a hypothetical future resident using the outdoor air data from the Workplace Air Monitoring Program and RME assumptions are  $3 \times 10^{-4}$  and 2, respectively (Table 21), similar in magnitude as regional background.

#### **6.2.2.2 Modeled Indoor Air Exposures from Groundwater**

The Tier 1 vapor model was used to evaluate hypothetical future residential exposures to indoor air vapors emanating from groundwater using site-wide, maximum detected chemical concentrations in groundwater. The Tier 1 chemical-specific risk estimates for site-wide residential exposures to groundwater vapors in indoor air are presented in Table 25, as well as in Appendix D. The results indicate that benzene was the primary risk driver in groundwater, with an estimated risk above the upper bound of the NCP discretionary risk

range ( $10^{-4}$ ). The maximum detected concentration of PCE resulted in an estimated risk of  $7 \times 10^{-5}$ . All other chemical-specific risks were near or below the midpoint of the risk range ( $10^{-5}$ ). Noncancer hazard estimates greater than 1 were due to benzene and toluene (Table 25).

Tier 2 modeling was completed for 13 EAPCs, as previously described for the commercial receptor (see Section 4.3.6 and Appendix F). Taking into account the Tier 2 results for benzene and the Tier 1 results for the other risk drivers (TCE and PCE), the cumulative cancer risk for the groundwater to indoor air pathway exceeded  $1 \times 10^{-5}$  for EAPCs 15 and 34 (Table 27). The estimated noncancer HIs for all EAPCs did not exceed 1.

### 6.3 TRENCH WORKER EXPOSURES

Estimates of cancer risk and noncancer hazard were calculated for trench workers assuming they are exposed to chemicals via incidental soil ingestion, dermal contact with soil, and outdoor inhalation of fugitive dust/vapors. Estimated RME risks and hazards for all pathways and EAPCs were below  $5 \times 10^{-5}$  and 1, respectively (Table 22). CT estimates of exposure and risk were not developed for the trench worker since the RME results were within acceptable levels. The chemical-specific risk and hazard estimates for this scenario are presented in Appendix D.

Cancer risk and noncancer hazard were re-calculated for a trench worker using the alternative HERD-recommended soil adherence factor (AF) of  $0.8 \text{ mg/cm}^2$  for several parcels with the highest risks to evaluate whether risk would increase significantly. The resulting risk and hazard increases were relatively slight and did not result in any changes to the EAPC risk groupings based on the  $10^{-6}$ ,  $5 \times 10^{-5}$  and  $10^{-4}$  benchmarks. The table below presents a comparison between the highest risks using the USEPA-recommended AF and the corresponding risks estimated using the HERD-recommended AF:

Parcel	EAPC No.	Cumulative Results			
		USEPA AF = $0.3 \text{ mg/cm}^2$		HERD AF = $0.8 \text{ mg/cm}^2$	
		Cancer Risk	Hazard Index	Cancer Risk	Hazard Index
7351-031-020	2	1.2E-06	2.0E-02	1.7E-06	2.3E-02
7351-034-015,-050,-056	16	5.3E-06	5.1E-01	6.5E-06	5.6E-01
7351-034-058	24	3.1E-07	4.5E-04	4.6E-07	5.6E-04
Magellan Drive	35	2.8E-07	3.1E-03	3.8E-07	3.4E-03

Based on the maximum EPC of 586 mg/kg for lead detected in soil at EAPC 2, the USEPA lead model predicted a blood lead level for a trench worker that did not exceed the blood lead goal of 10 µg/dl (Table D-3c in Appendix D). All other lead concentrations detected in soils at the site were within background.

#### 6.4 RISK CHARACTERIZATION SUMMARY

Cancer risks and noncancer hazards were presented in the risk characterization for each EAPC for commercial worker, hypothetical future resident, and trench worker receptors. Exposure to chemicals in surface soil (0 to 1 foot bgs), shallow soil and soil gas (0 to 15 feet bgs), deep soil and soil gas (>15 feet bgs), and groundwater were evaluated.

##### 6.4.1 Commercial Worker

Risk assessment findings for the commercial worker receptor are summarized below, in accordance with the groups defined by the various risk and HI benchmarks.

Pathway	EAPCs in Risk / Hazard Index Benchmark Groups Using Reasonable Maximum Exposures			
	Risk $\leq 10^{-6}$ and HI $\leq 1$	$10^{-6} < \text{risk} \leq 5 \times 10^{-5}$ and HI $\leq 1$	$5 \times 10^{-5} < \text{risk} \leq 10^{-4}$ and HI $\leq 1$	Risk $> 10^{-4}$ and/or HI $> 1$
Outdoor Soil	1, 5, 8, 9, 10, 14, 17, 18, 19, 20, 21, 22, 25, 26, 27, 31, 37 (17 of 37)	3, 4, 6, 7, 11, 12, 13, 15, 23, 24, 28, 29, 30, 32, 33, 34, 35, 36 (18 of 37)	2, 16* (2 of 37)	(0 of 37)
Indoor Air (Tier 1/Tier 2 Modeling)	1, 2, 3, 4, 8, 9, 10, 12, 13, 14, 17, 18, 19, 20, 21, 22, 25, 26, 27, 28, 29, 30, 31, 32, 33, 34, 36, 37 (28 of 37)	5, 6, 7, 11, 15, 24, 35 (7 of 37)	23* (1 of 37)	16* (1 of 37)

\* The risk at this EAPC is driven by elevated detection limits. Risk would decrease to less than  $5 \times 10^{-5}$  if the exposure point concentration was calculated using the maximum detected concentrations rather than 1/2 the detection limit for the risk-driving COPCs

Cancer risk estimates from direct contact and outdoor inhalation exposures to chemicals in surface and shallow soils under the RME scenario exceeded  $5 \times 10^{-5}$  for EAPCs 2 and 16, both with risk estimates of approximately  $1 \times 10^{-4}$ . Almost the entire site is currently covered (including EAPCs 2 and 16) with buildings, parking lots and landscaping, essentially eliminating these pathways of exposure for current commercial workers. All cancer risk

estimates under the CT scenario were below  $5 \times 10^{-5}$ . Noncancer HI estimates for all EAPCs under the RME and CT exposure scenarios did not exceed the benchmark level of 1.

Lead concentrations in soil at the site were below background for all EAPCs except EAPC 2. The predicted blood lead level for a commercial worker did not exceed the blood lead goal of  $10 \mu\text{g}/\text{dl}$  for EAPC 2.

The RME cancer risk estimates for indoor air exposures based on modeling exceeded  $5 \times 10^{-5}$  at two EAPCs (EAPCs 16 and 23). The cancer risks were estimated at  $1 \times 10^{-4}$  for EAPC 23 and at  $4 \times 10^{-4}$  for EAPC 16. Benzene, PCE and chloroform contributed the majority to the risk estimates for both EAPCs. Under the CT exposure scenario, cancer risk estimates did not exceed  $5 \times 10^{-5}$  at any EAPC. Noncancer HI estimates under the RME and CT exposure scenarios did not exceed the benchmark level of 1 at any EAPCs except EAPC 16, with a noncancer HI of 3 (primarily due to PCE).

The RME cancer risk estimates for the groundwater to indoor air pathway did not exceed the risk level of  $5 \times 10^{-5}$  for any EAPCs.

#### 6.4.2 Hypothetical Future Resident

Risk assessment findings for the hypothetical resident are summarized below, in accordance with the groups defined by the various risk and HI benchmarks.

Pathway	EAPCs in Risk / Hazard Index Benchmark Groups Using Reasonable Maximum Exposures			
	Risk $\leq 10^{-6}$ and HI $\leq 1$	$10^{-6} < \text{risk} \leq 5 \times 10^{-5}$ and HI $\leq 1$	$5 \times 10^{-5} < \text{risk} \leq 10^{-4}$ and HI $\leq 1$	Risk $> 10^{-4}$ and/or HI $> 1$
Outdoor Soil	1, 19, 20, 22, 37  (5 of 37)	3, 4, 5, 8, 9, 12, 13, 15, 17, 30, 32, 33, 36  (13 of 37)	6, 7, 11, 24, 35  (5 of 37)	2, 10, 14, 16, 23, 28, 29, 34  (8 of 37)
Indoor Air (Tier 1/Tier 2 Modeling)	1, 2, 3, 10, 12, 13, 14, 18, 21, 25, 26, 29, 30, 31, 32, 33, 34, 36, 37  (19 of 37)	8, 9, 17, 19, 20, 22  (7 of 37)	11  (1 of 37)	5, 6, 7, 15, 16, 23, 24, 28, 35  (8 of 37)

Chemicals detected in shallow soils pose a risk above the upper bound risk range ( $10^{-4}$ ) for the direct contact and outdoor air inhalation pathway under the hypothetical future residential RME scenario at EAPCs 2, 16, and 23. HI estimates from potential RME exposures exceeded

the benchmark level of 1 for seven EAPCs, with the highest estimated noncancer hazards at EAPCs 14, 16, and 23. All cancer risk estimates were at or below  $1 \times 10^{-5}$  under the CT exposure scenario, and the noncancer HIs were below the benchmark level of 1, with the exception of EAPC 14.

Cancer risk estimates for the soil to indoor air pathway were above  $10^{-4}$  for seven EAPCs (5, 6, 15, 16, 23, 24, 35), with the highest estimated risks at EAPCs 16, 23, and 24. Benzene and PCE were the most common chemicals that contributed predominantly to risk via the indoor air pathway. Noncancer hazard estimates were above 1 for seven EAPCs with the highest estimates at EAPCs 16 and 23. Chloroform, PCE, and TCE contributed predominantly to the hazard at EAPC 16; whereas benzene, chloroform, and PCE contributed the majority to the hazard at EAPC 23. Risk and hazard estimates for the soil to indoor air pathway under the CT exposure scenario were generally approximately an order of magnitude lower than the RME estimates. CT cancer risk estimates were still above  $10^{-5}$  at EAPCs 16 and 23, while noncancer hazards were still above the benchmark level of 1 at EAPCs 7 and 16.

The RME cancer risk estimates for the groundwater to indoor air pathway exceeded  $1 \times 10^{-5}$  for EAPCs 15 and 34. EAPC 2 was the only parcel with a detection of lead above background. The predicted blood lead level using the maximum detected lead concentration from this parcel (586 mg/kg) and the DTSC lead model for the hypothetical future child resident exceeded the blood lead goal of 10  $\mu\text{g}/\text{dl}$ . All other lead concentrations detected in soils at the site were within background.

### 6.4.3 Trench Worker

Estimated RME risks for trench workers were below the risk level of  $5 \times 10^{-5}$  for all EAPCs. Noncancer hazard estimates for all EAPCs were below the benchmark level of 1. The predicted blood lead level for a trench worker did not exceed the blood lead goal of 10  $\mu\text{g}/\text{dl}$  for EAPC 2.

## 7.0 ASSESSMENT OF UNCERTAINTY

The methodology used in this risk assessment is consistent with USEPA and CalEPA risk assessment guidance and prior communications with agency personnel. However, the procedures used in any quantitative risk assessment are conditional estimates given the many assumptions that must be made about exposure and toxicity. Major sources of uncertainty in risk assessment include: (1) natural variability (e.g., differences in body weight or sensitivity in a group of people); (2) incomplete knowledge of basic physical, chemical and biological processes (e.g., the affinity of a chemical for soil, degradation rates); (3) model assumptions used to estimate key inputs (e.g., exposure, dose-response models, fate and transport models); and (4) measurement error primarily with respect to sampling and laboratory analysis. Site-specific factors, which this assessment incorporates, decrease uncertainty, although significant uncertainty persists in even the most site-specific risk assessments. This inherent uncertainty in quantitative risk assessment methodology affects the level of confidence which can be placed in the final results; however, because the assumptions used tend to be health-protective and conservative in nature, the estimated risks are likely to exceed the most probable risk posed to potential receptors at the site.

The uncertainty associated with the COPC selection and estimation of chemical concentrations is discussed in Section 7.1. A qualitative analysis of these uncertainties for each risk driving EPC estimate is also presented in Section 7.1. The uncertainty associated with the exposure assessment and toxicity assessment is discussed in Sections 7.2 and 7.3. The overall uncertainty in the risk characterization is discussed in Section 7.4.

### 7.1 COPC SELECTION AND ESTIMATION OF CHEMICAL CONCENTRATIONS

Multiple factors contribute to uncertainty in the COPC selection and estimation of EPCs and associated risk estimates. As discussed in Section 3.0, the risk assessment approach involved the use of validated data pertaining to the exposure concentration for a given parcel, media, and chemical. This approach involved several data processing and analysis steps that have contributed to the overall uncertainty about the EPCs. These processing and analysis steps are discussed below, followed by a qualitative analysis of the degree to which uncertainties in these steps impact the confidence in the EPC estimates for risk-driving chemicals at EAPCs where cancer risk and HI estimates are above  $1 \times 10^{-5}$  and 1, respectively.

Uncertainty is prevalent and multi-layered in complex risk assessments. However, many of the recognized elements of uncertainty are traditionally incorporated into the variables and risk estimates in a conservative manner with respect to health protection. Therefore, it is important to distinguish between an uncertainty that has been conservatively incorporated as a bias (e.g., using an upper 95UCL to represent a mean exposure) and an uncertainty that is

“free floating” (i.e., an uncertainty that implies a value could be either substantially higher or lower than the derived value). These types of uncertainty are discussed with respect to the elements of the risk assessment in the sections below.

### 7.1.1 Random Sampling Uncertainty

Sampling uncertainty typically refers to the uncertainty resulting from the use of a limited sample to represent the entire underlying population. An example of this is using the sample mean as an estimate of the true population mean. As defined here, “random sampling uncertainty” describes expected uncertainty due to random or representative sampling. The term “spatial representation uncertainty” is used to describe any known and unknown sources of bias due to sampling patterns that are not spatially representative, as discussed in the following subsection.

Random sampling uncertainty is incorporated quantitatively, based on statistical theory, as a conservative bias in the risk calculations by the use of the 95UCL estimate of the mean rather than the sample mean to characterize each EPC. For example, the uncertainty from a relatively low sample size, say 5, requires that a greater amount of conservatism be added to the estimated mean, while a high sample size of 100 generally requires that very little conservatism be added to the estimated mean. In either case, a sample size of 5 or 100, we can say that, based on random sampling uncertainty, there is only a five percent chance that the true mean exceeds the 95UCL. Therefore, the 95UCL is considered health protective.

### 7.1.2 Spatial Representation Uncertainty

Another component of sampling uncertainty is the spatial representation (representativeness) across an individual EAPC that results from the sampling strategies employed. This component brings in known and potentially unknown sources of bias into the EPC calculation.

Historical site operations were utilized to shape the scope and approach to the RI/FS. The plant site layout was characterized by multiple areas of densely packed chemical storage and processing areas separated by large areas of open space and parking or administration facilities. The majority of the site has been redeveloped with closely spaced commercial and industrial buildings since the demolition of the plant. This resulted in RI/FS sampling locations which are concentrated in accessible areas where the potential for contamination was judged to be highest, including areas of known contamination such as former facility locations where chemicals were stored, used, transported, or disposed. The data set is thought to be biased high for most EAPCs, since data collection was focused on such potential source areas. As a result, there is a conservative bias for the EPC and associated risk estimates; that is, overestimates of exposure and risk.

The former location of some rubber plant facilities of interest is substantially or completely within the footprint of existing, active business buildings. Subsurface sampling was not conducted in these areas due to access difficulties and the associated disruption to the businesses. As a result, uncertainty has been introduced into the risk assessment. Risks may be over or underestimated at EAPCs where a significant portion of a former facility is located beneath a building.

Uncertainty is also associated with the 0-15 foot shallow soil depth interval assumed in the risk assessment. The 15-foot limitation on shallow soil was based on the maximum depth to which the subsurface would be penetrated during normal maintenance activities such as utility repair. This assumption differs from the Cal-EPA assumption of 0 to 10 feet bgs.

For hypothetical future resident and trench worker RMEs, EAPC maximum concentrations (maximum detect concentrations or  $\frac{1}{2}$  the detection limit for non-detects, whichever was higher) were used as the EPCs. Therefore, the inclusion of the additional soil between 10 to 15 feet bgs (compared to the Cal-EPA assumption of 0-10 feet bgs) would not impact the EPC values for these two scenarios. For commercial exposures, the RMEs for each COPC were based on the 95UCL concentration for all sampling locations within an EAPC. Review of the shallow soil data indicates that the majority of it is from depths of 10 feet bgs or less. Therefore, inclusion of the additional soil data from 10 to 15 feet bgs would not significantly affect the final exposure point concentrations calculated.

Spatial representation uncertainty also arises from the use of soil data from composite samples, for which a limited number of metals, SVOCs, PCBs, and pesticide analyses were completed. Composite samples represent local averages rather than discrete concentrations. Composite samples are considered to provide useful information for estimating the exposure concentration because the main parameter of interest in risk assessment is an area mean. For this reason, composite sample results were combined with results for discrete samples. Therefore, while the composite sampling is not in itself a source of bias, potential bias is introduced by the combined use of composite data and individual samples.

The statistical treatment of composite samples potentially introduced a limited amount of bias into the prevalence screening during the selection of the COPCs, although the direction of this bias could be either positive or negative for different cases. In short, the effect of composite samples on prevalence was a potential overestimation of prevalence for high-prevalence chemicals and a potential underestimation of prevalence for low-prevalence chemicals. The prevalence estimation was positively biased upward when the composite result was a detected concentration (more likely for prevalent chemicals); the N contributing samples are represented as N detections from N samples, when the true prevalence could

have been as low as 1 of the N samples. The opposite bias occurs if the composite sample was not associated with a detected concentration (more likely for low-prevalence chemicals); the samples are represented as having 0 detections in all of N samples, even though it is possible that one or two of the individual sample concentrations could have been detected if analyzed separately. The true prevalence in this case could have been as high as 1 or 2 of N instead of 0 of N.

The treatment of composite samples together with individual samples introduced a negative bias when they were incorporated into the EPC calculation. The single composite result was assigned to each of the contributing sample locations to combine the individual and composite sample data. This method does not impart a bias to the estimation of the mean since the expected concentration at each contributing location is equal to the composite concentration. However, the uncertainty component that is added to the sample mean estimate to derive the 95UCL used for the EPC can be underestimated by this approach, because the true individual variability within the contributing samples is lost when they are composited. This results in a nonconservative bias for the 95UCL or exposure point concentration, which was addressed by using the maximum concentrations as the EPC in the RME evaluation for those parcels where composites were a part of the data set.

### 7.1.3 Detection Limit Uncertainty

The limitations of the chemical analytical methods (i.e., detection limits) introduced substantial uncertainty into the COPC selection and EPC estimation processes in some cases. Sample data recorded as non-detected or below a detection limit provide precise information whenever the detection limits are low relative to population concentrations for a given parcel *or* low relative to the toxicity screening level (PRG based). Detection limits imply substantial uncertainty when they are high relative to both the population of concentrations (implying low percentage detection) and to the toxicity screening levels.

The minimum and maximum detection limits associated with non-detected results for each case are identified in Table C-1 of Appendix C, which describes the COPC selection process. It is possible to identify cases from this table where there is low uncertainty associated with the elimination of a COPC (the detection limits are below the toxicity screening level), and cases where there is high uncertainty associated with the elimination of a COPC (the detection limits are above the toxicity screening level).

The second source of uncertainty implied by elevated detection limits is introduced during the data analysis step (discussed below), where it is assumed that the value of the sample concentrations is adequately represented by one-half the value of the detection limit. This

assumption is consistent with conventional methods and generally considered to add conservative bias to the calculation of the EPC, for most cases.

Much of the site VOC data was soil gas data analyzed by a mobile laboratory. Duplicate summa canister samples analyzed by a fixed laboratory were collected for at least 10% of the mobile laboratory sampling locations. When a detection was observed by both the mobile lab and the fixed lab analyses, the paired results were well correlated. The mobile lab results are therefore believed to provide adequate data for characterizing the exposure concentration and were used in the risk assessment.

#### **7.1.4 Distribution Assumption Uncertainty**

The last step of the EPC estimation process involves calculation of the EPC based on an assumed distribution type. Although the formulae for the EPC calculations are statistically exact for particular distribution types, there are numerous cases in which the distribution type is unclear as indicated by the Shapiro-Wilk test. There is data analysis uncertainty for these cases in the choice of which EPC formula is most accurate. This type of uncertainty is highest for cases where the distribution type was uncertain, and where different distribution assumptions resulted in widely varying estimates of the EPC. Cases in which distribution type was uncertain, but the values of the different EPCs were similar, were by definition, less sensitive to the distribution assumption and therefore more certain with respect to this type of uncertainty.

#### **7.1.5 Partitioning and Transport Modeling Uncertainty**

The sampling strategy was focused on VOC data from soil gas and to a lesser extent, soil. The soil gas data were converted into soil matrix concentrations prior to calculating the exposure point concentration and dose to quantify risks and hazards to these chemicals via direct exposures (i.e., soil ingestion and dermal contact). Thus, partitioning uncertainty was introduced into the EPC calculation for volatiles when soil gas concentrations were converted to soil matrix concentrations, and vice versa. Partitioning uncertainty is a result of sample specific variability in soil properties (e.g., fractional organic carbon content) and VOC concentrations in soil. The partitioning calculations assume that equilibrium conditions exist. This is a reasonable assumption given that contaminants associated with the former rubber plant must have been released during the period of plant operation and that there has been sufficient time to reach steady-state. Additionally, the presence of residual NAPL may bias the soil gas and soil matrix conversion results. The presence of NAPL may result in over-estimation of soil gas concentrations converted from soil matrix analytical results and under-estimation of soil matrix concentrations converted from soil gas analytical results. Uncertainty in the partitioning equation is generally considered low and outweighed by the increased amount of data available for the EPC determination resulting from using the soil and soil gas data together.

Transport modeling uncertainty was introduced into the indoor air exposure concentrations with the use of the Johnson and Ettinger model. The transport modeling for vapor migration from shallow soils assumed the source depth to be 7.5 feet bgs. Samples from this zone may have been collected from depths less than or greater than this assumed depth. A sensitivity analysis on the affect of sample depth on the vapor intrusion pathway demonstrates that the assumed 7.5 feet bgs source will result in over-estimation of risks for samples deeper than this assumed level and under-estimation of risks for samples less than 7.5 feet bgs. For sample depths ranging from 5 to 15 feet, this variability in sample depths will result in changes to the calculated risk by a factor ranging from 0.5 to 1.5. Similarly, for the vapor migration from deep soils evaluation, samples collected from depths less than or greater than the assumed depth of 30 feet bgs will result in over-estimation and under-estimation of risks.

The air exchange rate value (0.9 exchanges per hour) used for the evaluation of vapor intrusion to commercial buildings is commonly assumed for vapor intrusion assessments and is consistent with American Society of Heating, Refrigerating and Air-Conditioning Engineers standards (ASHRAE, 1999) and USEPA, DTSC, California Regional Water Quality Control Board, San Francisco Bay Region (CRWQCB-SFB), and ASTM guidance (USEPA, 2002b; CalEPA, 2003, 2005; ASTM 1995, 1998). This suggests that the air exchange rate used in the vapor intrusion calculations is a reasonable, conservative value for the buildings at the site. However, as with many other parameters used in the vapor intrusion modeling, the value used for the air exchange rate is general and not specific to the site buildings. Literature values for commercial building air exchange rates range from approximately 0.2 to 2 exchanges per hour; however, values less than 0.5 exchanges per hour for commercial buildings with operating HVAC systems are not expected to be common. The vapor intrusion attenuation factors (and consequently, the risk estimates) are expected to be inversely proportional to the air exchange rate. For example, if the air exchange rate increases by a factor of two, the vapor intrusion attenuation factor and risk will decrease by a factor of two. Higher air exchange rates will result in lower attenuation factors and risk estimates and lower air exchange rates will result in higher attenuation factors and risk estimates.

This Tier 1 model is considered to have moderate to high uncertainty; however, this uncertainty generally is incorporated into the model in a conservative manner. The Tier 2 model, which was performed only for BTEX compounds, is considered to have relatively low to moderate uncertainty due to the incorporation of site-specific model assumptions. The Tier 2 modeling evaluation (see Appendix F) suggests the conservative bias in the Tier 1 results for BTEX. Therefore, in cases where BTEX compounds are driving risk or hazard based on vapor modeling, uncertainty could be considered moderate. The level of uncertainty associated with the Tier 2 modeling evaluation using current site data is acceptable for the

site BLRA. However, Tier 2 model uncertainty may be reduced if additional site characterization data (e.g., sub-slab soil gas samples) are collected.

### 7.1.6 Temporal Uncertainty

The EPCs used in the risk assessment are assumed to be non-varying. This is a conservative assumption and is consistent with stable groundwater plumes and constant soil and soil gas concentrations. This assumption will over-estimate risks in areas where soil, soil gas or groundwater concentrations are decreasing due to either abiotic or biotic degradation.

## 7.2 EXPOSURE ASSUMPTIONS

The exposure assumptions used for the RME approach are considered conservative and likely lead to overstating the most probable estimate of potential risk. For example, the RME exposure scenario assumes a hypothetical future resident will remain at the site from birth through age 30 years for 350 days per year, or a commercial worker will work at the site for 25 years. The CT approach reduces some of this conservatism by incorporating more realistic assumptions regarding exposure duration and exposure frequency (e.g., 6.6 years and 250 days/year for commercial worker) based upon the average parameter distributions. Regarding the adherence factor assumption, an evaluation was conducted for the Trench Worker scenario using the higher HERD recommended adherence factor. As presented in Section 6.3, use of the HERD value did not significantly increase the estimated risks.

Intake parameters for the various exposure pathways (soil ingestion, dermal contact, inhalation) were conservatively assumed to be upper bound estimates (e.g., 5700 cm<sup>2</sup> of exposed skin exposed every day—regardless of the weather conditions—or ingestion of 100 mg of soil each day over the exposure period for adults, etc.) for the RME approach. A key area of uncertainty associated with exposure is the bioavailability of the chemicals present in soil and movement of the chemicals into the bloodstream, i.e., dermal penetration and gastrointestinal absorption. It is well established that lipophilic chemicals present in soil for long periods of time become less bioavailable than the same chemicals freshly added to soils. The basis for this is uncertain, but is believed to be attributable to a time-limited desorption and diffusion process, partitioning into organic carbon in soil or deep into micropores, or the formation of bonds between the chemicals and soil moieties. Regardless of the basis, studies using chemicals freshly added to a soil carrier over-predict the oral or dermal absorption potential for aged or weathered chemicals present in soil. As a result, due to the anticipated aged condition of chemicals in site soils, dermal or oral absorption of chemicals is expected to be much lower than assumed in this BRA, which used absorption factors based on laboratory testing using freshly added chemicals.

Lastly, both the RME and CT approach fail to account for future environmental degradation of the organic chemicals present in the source areas. Significant degradation of the chemicals is likely to occur over a 30-year exposure duration due to microbial degradation, photolysis, hydrolysis, and other processes which over time reduce the concentrations of chemicals present in soil. These degradation processes are recognized as relevant considerations for purposes of estimating potential health risks to hypothetical receptors. For example, degradation is included in the CalEPA CalTOX framework to provide more realistic estimates of potential risks to such receptors (CalEPA, 1994). In non-NAPL areas of the site, risks are likely to be substantially overstated based upon the assumption that no attenuation will occur over a 30-year exposure duration.

### 7.3 CHEMICAL TOXICITY

Estimating the toxicity potential of COPCs represents one of the greatest areas for uncertainty in the risk assessment. Neither the RME nor the CT approach account for any of the uncertainty associated with estimating the toxicity potential for the COPCs. This uncertainty may arise from one or a combination of factors, depending on the specific chemicals, the methodology, and data used to derive estimates of potential chemical toxicity and associated toxic endpoint(s). Several key areas of uncertainty associated with estimating the toxic potential of COPCs are:

- Animal to human extrapolation
- High to low dose extrapolation
- Upper bound cancer slope factors
- Biological mechanism: threshold versus non-threshold carcinogens
- Dose-response assessment for noncarcinogenic endpoints
- Multiple chemical exposures.

Generally, toxicity data developed based upon exposure of laboratory test animals to chemicals serve as the basis for predicting the toxicity potential in humans. This data is then used as input into mathematical models to estimate the possible human exposure response to chemicals at environmental levels far below those tested in animals. These models contain several limitations that are considered when interpreting the estimates of potential health risks, determining whether remediation is warranted, and in deriving risk-based cleanup levels. Primary among these limitations is the uncertainty in extrapolation of results obtained in animal research to humans, and the shortcomings in extrapolating responses obtained from high-dose animal research studies to estimate human responses at very low environmental doses.

Most of the values for the cancer potency estimates were derived from the linearized multistage model (LMM). The LMM typically uses high-dose animal carcinogenicity data to derive cancer potency estimates for extremely low-dose exposures to humans. The resulting potency value is a number which may not be an accurate estimate of potential toxicity in humans at low environmental doses due to variety of reasons. The proposed revised USEPA Cancer Assessment Guidelines (CAG) address some of this uncertainty by using an alternative to estimate cancer potency through the derivation of the dose, giving an estimated 10% response or ED10, and use of a Margin of Exposure approach to obtain an estimate of cancer potency (USEPA, 1995a).

The LMM may not accurately predict cancer potency in humans because absorption and metabolism of the chemical may be different in humans due to inherent physiological differences or differences attributable to high dose effects. For example, humans are typically exposed to environmental chemicals at levels that are less than a thousandth of the lowest dose tested in animals. Such doses may be easily degraded or eliminated by physiological internal mechanisms that are present in humans (Ames, 1987). Thus, there are recognized limitations to using the results of standard rodent bioassays to understand the human biological hazard or cancer risks posed by routine levels of exposure (Crump, et al., 1976; Sielken, 1985; USEPA, 1995a).

The cancer potency values, often referred to as slope factors, are considered to be plausible upper bounds of risk at a 95% confidence level. Thus, there is a 95% probability that the true cancer potency does not exceed these levels, and the most probable cancer potency estimate and corresponding health risks are likely to be much lower. The CAG states that the use of the linearized multistage model and upper bound risk estimates is appropriate, but that the lower limit of risk may be as low as zero. The proposed CAG addresses this issue by including the concept of threshold and nonlinear carcinogens (USEPA, 1995a, 2003b). When biological factors are considered, the best estimates of the toxicity potential and corresponding risks are likely to be at significantly lower levels, often near zero.

As stated previously, the USEPA modeling approach to determine carcinogenic potency for the COPCs present at the site includes the assumption that there is no threshold for carcinogenicity. Thus, even a single molecule of a potential carcinogen is assumed to produce an increase in potential cancer risk. However, the evidence indicates that thresholds are likely to exist for the majority of carcinogens. Many toxicologists and physicians believe that the assumption of no carcinogen thresholds is no longer defensible (Butterworth, 1987; Marcus and Rispin, 1990). If such thresholds do exist for carcinogenic COPCs present at the site, and the LADD is less than this threshold, then the cancer risk is substantially overstated. However, the uncertainty of extrapolating information from high-dose animal studies to low-dose human exposures is not applicable to chemicals that are known human carcinogens and

for which the cancer potency has been derived from epidemiological studies. This is the case for benzene, one of the COPCs at the site. In addition, the CalEPA cancer slope factor for benzene was used in the risk assessment. If the USEPA inhalation cancer slope factor was used in this risk assessment, all benzene inhalation cancer risk estimates would be approximately 3.5 times lower.

An additional source of uncertainty is the toxicity value for TCE which is currently under evaluation by USEPA. A draft TCE reassessment document entitled Trichloroethylene Health Risk Assessment: Synthesis and Characterization dated August 2001 reported a range of cancer slope factors from  $0.0003$  to  $7$   $(\text{mg/kg-day})^{-1}$  with most between  $0.02$  and  $0.4$   $(\text{mg/kg-day})^{-1}$ . The value of  $0.4$   $(\text{mg/kg-day})^{-1}$ , has been used in some cases as a provisional toxicity value. Many of the cancer slope factors derived in the reassessment, including the upper end value of  $0.4$   $(\text{mg/kg-day})^{-1}$ , are based on oral exposure rather than inhalation exposure. Inhalation exposure is typically the most sensitive exposure route, especially when considering the vapor intrusion pathway at a site. USEPA is currently in consultation with the National Academy of Sciences to provide advice on scientific issues related to the draft assessment.

The risk assessment used the CalEPA TCE cancer slope factor of  $0.007$   $(\text{mg/kg-day})^{-1}$  in the estimate of cancer risk for inhalation exposures. This value is based on liver and lung tumor incidence in rodent inhalation studies and is within the range of the cancer slope factors presented in the USEPA draft reassessment. This value is considered appropriate for use, as it is based on inhalation exposures and is not a provisional value. The risk assessment also used a CalEPA toxicity value in the estimate of cancer risk for incidental ingestion and dermal contact exposures. The oral cancer slope factor of  $0.013$   $(\text{mg/kg-day})^{-1}$  was used for TCE.

There are protective mechanisms in the body which must be overcome before a chemical can exert a harmful effect on human health for chronic noncancer effects. Thus, the approach is to identify an upper bound safe dose, or "reference dose" for this tolerance range which will protect the most sensitive persons. The final value for the reference dose incorporates uncertainty factors indicating the degree of confidence which can be assigned to the animal experimental data from which the reference dose was derived. Virtually all of the uncertainty adjustments for noncancer oral RfDs and inhalation RfCs have incorporated large "safety factors" wherein no-effect dose levels in animal experiments have been lowered by several arbitrary factors of 3 to 10 (often totaling 1,000 to 10,000 fold) to account for uncertainty (e.g., variations in human sensitivity, animal-to-human extrapolations, deficiencies in available animal data). The reference dose levels are thought by USEPA to be uncertain over perhaps an order of magnitude or more (USEPA, 1989). Therefore, the RfD or RfC is not strictly a scientifically based demarcation between what is a safe level and a toxic level. However, due to the degree of conservatism employed in setting RfDs and RfCs, the net result is likely to be an

overestimate of the potential noncancer health effects, which leads to a lower estimate of the level of chemical which could remain in the soil with little or no potential for causing adverse human health effects.

USEPA has developed statements of confidence (high, medium, or low) for the RfDs reflecting the stability of the value in addition to the uncertainty factors. High confidence is often given to RfDs that are based on human data. Low confidence indicates that the data supporting the RfD may be of limited quality and/or quantity.

Reference doses are frequently derived from animal studies which have little quantitative bearing on potential adverse effects in humans. Some of this uncertainty may be reduced if the absorption, distribution, metabolic fate, and excretion parameters of a chemical are known. Because the fate and mechanism of action of a chemical may differ in animals and humans, effects observed in animals may not be observed in humans, resulting in uncertainty in the potential for adverse health effects.

An additional area of uncertainty is exposure to multiple chemicals. Toxicological criteria are developed for individual chemicals. Potential interactions between chemicals could occur, leading to uncertainty in the risk estimates for multiple-chemical exposures. The risk assessment assumes that toxicity is additive across chemicals. This assumption would underestimate risk for chemicals that are synergistic or potentiometric with regard to toxicity, and overestimate risk for chemicals that are antagonistic with regard to toxicity. In addition, if chemical toxicological mechanisms differ or affect different organ systems, the assumption of additivity is conservative.

Information on the toxicity criteria used in the risk assessment can aid in identifying the uncertainty that may be associated with an estimate of risk or hazard due to the toxicity assessment. For example, benzene and arsenic are known human carcinogens (Class A) and the cancer slope factors are derived from human data. These chemicals have a low contribution to the uncertainty in the risk estimate since there is a relatively high degree of confidence in the toxicity criteria. For other carcinogenic chemicals such as PCBs, benzo(a)pyrene, and DDT, the cancer slope factors are based on animal to human extrapolation (Class B2) with little or no data for human exposures. Therefore, the toxicity criteria used for these chemicals are more uncertain and contribute more to the uncertainty analysis. The uncertainty factor and degree of confidence in the reference doses provides an indication of the level of confidence and uncertainty associated with the reference doses used in the risk assessment. For example, the uncertainty factor used for ethylbenzene and toluene is 1,000, indicating a relatively lower uncertainty in the original toxicology study as compared to sec-butylbenzene where the uncertainty factor is 10,000.

Another source of uncertainty in this risk assessment involves changing toxicity criteria. Over time, as more scientific information becomes available, toxicity criteria and cancer classifications may change. The toxicity information for a few of the COPCs has changed since the risk assessment calculations were performed. For carcinogens, these changes include the arsenic Cal-EPA oral cancer slope factor increasing from 1.5 to 9.5 (mg/kg-day)<sup>-1</sup>, chloromethane's cancer slope factor being withdrawn, and naphthalene being designated as a carcinogen by Cal-EPA with a cancer slope factor of 0.12 (mg/kg-day)<sup>-1</sup>. For non-carcinogens, the most significant changes were for n-hexane, with an increase in the oral reference dose from 0.06 to 11 mg/kg-day, and for 1,3 dichlorobenzene, with an increase in the oral reference dose from 0.0009 to 0.03 mg/kg-day. Estimated risks would be correspondingly higher if the new cancer slope factor were used for those cases where the cancer slope factor increased. The estimated noncancer hazard quotients would be correspondingly lower if the new reference dose were used for those cases where the noncancer reference dose increased.

#### 7.4 RISK CHARACTERIZATION

Uncertainties in the EPC estimation, exposure assessment, and toxicity assessment affect the degree of confidence in the chemical-specific risks. If the uncertainty in the EPC is low and the risk-driving chemical is a known human carcinogen (Class A), the corresponding uncertainty in the risk characterization is considered low. For cases where the EPC uncertainty is low, but the toxicity criteria is more uncertain, the corresponding uncertainty is considered low to moderate. Finally, if the EPC uncertainty is moderate to high, then the corresponding uncertainty in the risk characterization is considered moderate to high.

Combining the upper bound exposure assumptions, upper bound toxicity assumptions, and upper bound exposure concentrations, as in the RME approach, is a conservative approach typically utilized in risk assessment. This approach assumes, for example, that individuals who are most sensitive to the potential cancer effects of a chemical will also have a breathing rate and exposure duration (e.g., time at one residence) that exceeds most of the population. With numerous upper bound exposure assumptions combined, the risk is typically overestimated for the population. The corollary is that virtually all potentially exposed individuals will have a much lower level of potential risk than that which is estimated by the conservative assumptions employed in this assessment. One method to account for the uncertainty introduced by such conservatism is to estimate a CT exposure or use a probabilistic approach using Monte Carlo analysis. A CT approach was utilized to estimate a more average exposure, and account for some of the uncertainty attributable to the repeated use of conservative estimates associated with exposure parameters in this assessment.

**7.5 SUMMARY**

There are a variety of factors that contribute to the uncertainty in risk estimates presented in this risk assessment. The use of site-specific factors can decrease uncertainty, but it persists in even the most site-specific risk assessments. This inherent uncertainty affects the level of confidence which can be placed in the final results; however, because the assumptions used in the exposure and toxicity assessments tend to be health-protective and conservative in nature, the estimated risks are likely to exceed the most probable risk posed to potential receptors at the site.

An important source of uncertainty is the degree to which the available EAPC data are representative and adequate for providing estimates of risk and hazard. Therefore, a primary focus of the uncertainty analysis was a review of the uncertainty in the EPC used in the risk assessment.

There is uncertainty with respect to the EPCs and risk estimates for many of the parcels where elevated risk (greater than  $5 \times 10^{-5}$ ) was identified for the commercial worker. Specifically, the elevated outdoor soil and/or indoor air risks identified for EAPCs 16 and 23 are driven by elevated detection limits rather than actual detections.

While there is uncertainty in the existing site data, the data are judged sufficient for completion of the risk assessment and proceeding with the FS. Depending upon the outcome of USEPA's remedy selection process, additional limited and focused sampling at some parcels may be appropriate in the future to reduce the above uncertainties.

## 8.0 SUMMARY AND CONCLUSIONS

This risk assessment presents results of an evaluation of potential human health risks for the Soil and NAPL OU at the Del Amo site. The Soil and NAPL OU consists of the vadose zone soils and areas of identified or suspected NAPL within the area of the former rubber plant complex. The risk assessment addresses potential exposures to commercial workers, hypothetical future residents, and trench workers. Potential exposures to chemicals detected in surface and shallow soils have been evaluated for direct contact pathways, as well as inhalation of volatile chemicals in indoor and outdoor air and fugitive dust in outdoor air. The potential for volatile chemicals to migrate from the subsurface to indoor air was also evaluated for deeper vadose zone soils and contaminated groundwater at a few parcels where shallow data were not available.

Review and analysis of the site data involved the following processes: (1) data validation and selection for use in the risk assessment; (2) data processing related to composite samples, fixed laboratory and mobile laboratory results for the same sample location, and conversion of results between soil gas and soil matrix; (3) selection of EAPCs; (4) selection of COPCs; and (5) calculation of EPCs for use in calculating cancer risk and noncancer hazard estimates.

Cancer risks and noncancer hazards were presented in the risk characterization for each EAPC and for the commercial worker, hypothetical future resident, and trench worker receptor populations. Exposure to chemicals in surface soil (0 to 1 foot bgs), shallow soil and soil gas (0 to 15 feet bgs), deep soil and soil gas (>15 feet bgs), and groundwater were evaluated. A RME scenario was evaluated to represent an upper bound estimate of exposure and risk. The intent of the RME scenario was to focus the assessment on a conservative exposure that is the maximum exposure that is reasonably expected to occur (USEPA, 1989). Because of the multiple conservative assumptions used in the risk assessment process, the RME is often a high-end estimate of exposure and risk. A CT exposure scenario was evaluated to represent more “typical” or average exposure conditions. A CT estimate of exposure and risk was not developed for the trench worker since RME results were within acceptable levels.

The primary conclusions from the risk assessment for the commercial worker, hypothetical future resident, and trench worker receptors are summarized below. Table 29A presents a summary of the risk assessment results organized by receptor type and risk group. Table 29B summarizes the results according to EAPC and receptor. Risk assessment results are additionally summarized on site maps, as presented on Figures 20 through 24. The indoor air pathway results discussed below and presented in the summary tables and figures are based on Tier 1 plus Tier 2 modeling of shallow soil/soil gas data, except where specifically noted otherwise. These results were preferred over those from workplace (indoor) air monitoring

data based on the correlation analysis described in Section 4.3.6.1. For 13 EAPCs where shallow soil/soil gas data were limited or unavailable, the final indoor air risk value was conservatively selected as the maximum value derived through vapor transport modeling of shallow soil/soil gas, deep soil, and groundwater data.

## 8.1 COMMERCIAL WORKER

### 8.1.1 Surface and Shallow Soil Exposures – Outdoor Pathways

Risk assessment results for the commercial worker potentially exposed via direct contact and outdoor inhalation under the RME scenario are summarized on Figure 20. Cancer risk estimates from potential RME exposures to chemicals in surface and shallow soils exceeded the risk level of  $5 \times 10^{-5}$  for EAPCs 2 and 16, both with risk estimates of approximately  $1 \times 10^{-4}$ . As shown on the figure, the elevated risk at EAPC 2 is driven by the detected benzo(a)pyrene concentration at a single sampling location. Risk at EAPC 16 is driven by two sampling locations where benzo(a)pyrene and dibenzo(a,h)anthracene were not detected, but had elevated detection limits. The estimated risk at EAPC 16 decreases from  $1 \times 10^{-4}$  to  $6 \times 10^{-6}$  when using the maximum detected concentrations for those chemicals.

All cancer risk estimates were below  $5 \times 10^{-5}$  for CT exposures. Noncancer HI estimates for all EAPCs under the RME and CT exposure scenarios did not exceed the benchmark level of 1, indicating that the soil exposure pathways for surface soil and shallow soil do not pose an unacceptable noncancer hazard for commercial workers.

For the single parcel where lead was detected in soil above background (EAPC 2), the predicted blood lead level using the USEPA lead model for a commercial worker did not exceed the blood lead goal of 10 µg/dl.

The estimated RME and CT risks for outdoor shallow soil exposures are conservative in that bare soil is assumed to be available for contact, when in fact; almost the entire site is covered with buildings, parking lots, and landscaping, eliminating exposure. For example, there is 100% coverage of surface soil at EAPCs 2 and 16. In addition, elevated risks are based on chemical data from a few sample locations. Any commercial worker exposure to these locations would be of much shorter exposure frequency and duration than assumed in the risk assessment.

### 8.1.2 Indoor Air Exposures

Cancer risk estimates from potential RME exposures to the model-predicted indoor air concentrations exceeded the risk level of  $5 \times 10^{-5}$  at EAPCs 16 ( $4 \times 10^{-4}$ ) and 23 ( $1 \times 10^{-4}$ )

(Figure 21). The elevated risks at both EAPCs are driven by elevated detection limits for chloroform and PCE, which occur at two distinct sampling locations at both EAPCs.

Cancer risk estimates for all EAPCs under the CT exposure scenario did not exceed the risk level of  $5 \times 10^{-5}$ . Noncancer HI estimates for all EAPCs under the RME and CT exposure scenarios did not exceed the benchmark level of 1, except for EAPC 16, which had a HI of 3 under the RME scenario. This exceedance was driven by elevated detection limits for PCE at two sampling locations.

## 8.2 HYPOTHETICAL FUTURE RESIDENT

Current land use zoning precludes pure residential development within the Del Amo site, with the exception of a dwelling occupied by a single worker serving as a watchman or caretaker where industrial development is present. Application of other institutional control mechanisms that could enhance existing controls and prevent inappropriate land uses at the site in the future are being evaluated as part of the FS. Residential exposure pathways are indicated as only potentially complete in the CSM due to the existing zoning and hypothetical nature of this exposure scenario. Future residential development is unlikely based on current zoning.

### 8.2.1 Shallow Soil Exposures – Outdoor Pathways

Figure 22 summarizes the risk assessment findings with respect to hypothetical future residents and outdoor soil pathways. Chemicals detected in shallow soils pose a risk above the upper bound of the NCP discretionary risk range ( $>10^{-4}$ ) for the direct contact and outdoor air inhalation pathway under the hypothetical RME residential scenario at EAPC 2 ( $5 \times 10^{-4}$ ), EAPC 16 ( $2 \times 10^{-3}$ ), and EAPC 23 ( $9 \times 10^{-4}$ ). The primary chemicals that contributed to risk (chemical-specific risk  $>10^{-4}$ ) for EAPC 16 were benzo(a)pyrene, dibenzo(a,h)anthracene, and PCE via direct contact exposures (incidental soil ingestion and dermal contact), as well as benzene and PCE via outdoor inhalation of volatile chemicals. The primary risk-driving chemical was benzene via outdoor air inhalation for EAPC 23. The calculated risk of  $2 \times 10^{-3}$  for EAPC 16 was influenced by elevated detection limits and would decrease by two orders of magnitude to  $2 \times 10^{-5}$  if the maximum detected concentrations are used. All cancer risk estimates were at or below  $1 \times 10^{-5}$  under the CT exposure scenario.

HI estimates from potential RME exposures were above the benchmark level of 1 for EAPCs 10, 14, 16, 23, 28, 29, and 34. The highest noncancer hazards were estimated for EAPCs 14 (HI = 15), 16 (HI = 12) and 23 (HI = 7). Copper contributed the majority to the HI for EAPC 14, ethylbenzene and PCE for EAPC 16, and benzene and ethylbenzene for EAPC 23. All

noncancer HIs were below the benchmark level of 1 under the CT exposure scenario, with the exception of EAPC 14, which had an HI of approximately 2.

EAPC 2 was the only parcel with a detection of lead above background. The predicted blood lead levels using the maximum detected lead concentration from this parcel (586 mg/kg) and the DTSC lead model for the hypothetical future child resident exceeded the blood lead goal of 10 µg/dl. All other lead concentrations detected in soils at the site were within background.

### 8.2.2 Indoor Air Exposures

Figure 23 shows the locations of parcels within each risk group for the indoor air pathway. The RME cancer risk estimates for the indoor air pathway were above the upper bound of the NCP discretionary risk range ( $>10^{-4}$ ) for seven EAPCs (5, 6, 15, 16, 23, 24, and 35), with the highest estimated risks at EAPC 16 ( $4 \times 10^{-2}$ ), EAPC 23 ( $1 \times 10^{-2}$ ), and EAPC 24 ( $7 \times 10^{-4}$ ). Benzene and PCE were the most common chemicals that contributed predominantly to risk via the indoor air pathway. Noncancer hazard estimates were above 1 for seven EAPCs (5, 7, 16, 23, 24, 28, and 35), with the highest HI estimates at EAPC 16 (500) and EAPC 23 (100). Chloroform, PCE and TCE contributed predominantly to the HI estimate for EAPC 16; whereas chloroform and PCE contributed the majority to the HI for EAPC 23. The risks and noncancer hazard estimates for EAPCs 16 and 23 were influenced by elevated detection limits. The indoor air risk of  $4 \times 10^{-2}$  for EAPC 16 would decrease about two orders of magnitude to  $4 \times 10^{-4}$  if the maximum detected concentrations of these chemicals were alternatively used instead of the EPCs that are biased high due to elevated detection limits.

Risk and hazard estimates under the CT exposure scenario were generally approximately an order of magnitude lower than the RME estimates. CT cancer risk estimates were still above the midpoint risk level of  $1 \times 10^{-5}$  at EAPCs 16 ( $1 \times 10^{-4}$ ) and 23 ( $3 \times 10^{-5}$ ), while noncancer hazards were still above the benchmark level of 1 at EAPCs 7 (HI = 2) and 16 (HI = 5).

The cancer risk estimates for the groundwater to indoor air pathway exceeded the risk level of  $1 \times 10^{-5}$  for EAPCs 15 ( $9 \times 10^{-5}$ ) and 34 ( $3 \times 10^{-5}$ ). These two EAPCs are within the LADWP utility corridor. The estimated noncancer HIs for all EAPCs did not exceed the benchmark level of 1.

## 8.3 TRENCH WORKER

Estimates of potential cancer risk and noncancer hazard were calculated for trench workers assuming they are exposed to chemicals via incidental soil ingestion, dermal contact with soil, and outdoor inhalation of fugitive dust/vapors. Risk results for the trench worker

scenario are summarized on Figure 24. The maximum potential cancer risk was  $5 \times 10^{-6}$  (EAPC 16). All noncancer HIs were less than 1.

EAPC 2 was the only parcel with a detection of lead above background. The predicted blood lead level using the USEPA lead model for a trench worker did not exceed the blood lead goal of 10  $\mu\text{g}/\text{dl}$ . All other lead concentrations detected in soils at the site were within background.

#### 8.4 UNCERTAINTY

There are a variety of factors that contribute to the uncertainty in risk estimates presented in this risk assessment. The use of site-specific factors can decrease uncertainty, but it persists in even the most site-specific risk assessments. This inherent uncertainty affects the level of confidence which can be placed in the final results; however, because the assumptions used in the exposure and toxicity assessments tend to be health-protective and conservative in nature, the estimated risks are likely to exceed the most probable risk posed to potential receptors at the site.

An important source of uncertainty is the degree to which the available data provide for representative estimates of risk and hazard. In particular, there is uncertainty with respect to the EPCs and risk estimates for many of the parcels where elevated risk (greater than  $5 \times 10^{-5}$ ) was identified for the commercial worker. Specifically, the elevated outdoor soil and/or indoor air risks identified for EAPCs 16 and 23 are driven by elevated detection limits rather than actual detections. Additional uncertainty regarding the subsurface vapor intrusion contribution to indoor air risk results from the subsurface data being limited to the area outside of the building footprints.

While there is uncertainty in the existing site data, the data are judged sufficient for completion of the risk assessment and proceeding with the FS. Depending upon the outcome of USEPA's remedy selection process, additional limited and focused sampling at some parcels may be appropriate in the future to reduce the above uncertainties. Future sampling, as necessary, would potentially include subslab sampling at selected buildings to further evaluate the potential contribution of soil vapor migration to indoor air concentrations. Such sampling is recommended prior to planning or implementation of any significant active remediation measures to insure that such measures are both necessary and effective.

---

**9.0 REFERENCES**

- Agency for Toxic Substances and Disease Registry (ATSDR), 1993. Toxicity Profile for Lead. U.S. Department for Health and Human Services. U.S. Public Health Service. PB93-182475.
- American Society for Testing and Materials (ASTM), 1995. Standard Guide for Risk-Based Corrective Action Applied at Petroleum Release Sites. E 1739-95.
- ASTM, 1998. Standard Provisional Guide for Risk-Based Corrective Action. PS 104-98.
- American Society of Heating, Refrigeration, and Air Conditioning Engineers (ASHRAE), 1999. Ventilation for Acceptable Indoor Air Quality, Standard 62-1999.
- Ames B., R. Magaw, and L. Gold, 1987. Ranking Possible Carcinogenic Hazards. *Science* 236-71-273.
- Barnes, D., et al., 1987. "Reference dose (RfD): description and use in health risk assessment." Appendix A in IRIS Supportive Documentation Volume 1. Office of Health and Environmental Assessment, USEPA, Washington, D.C. EPA 600/8-86/032a.
- Butterworth, B., 1987. Nongenotoxic Carcinogens. *CIIT Activities*. 7:1-4
- California Environmental Protection Agency (CalEPA), 1992. Supplemental Guidance for Human Health Multimedia Risk Assessments of Hazardous Waste Sites and Permitted Facilities. Department of Toxic Substances Control. July. (Corrected and reprinted August 1996).
- CalEPA, 1993. Addendum to Chapter 7 of DTSC Supplemental Guidance for Human Health Multimedia Risk Assessments of Hazardous Waste Sites and Permitted Facilities, Memorandum to Interested Parties. Department of Toxic Substances Control. June.
- CalEPA, 1994. CalTOX User's Guide. Office of Scientific Affairs, Department of Toxic Substances Control. August 1994.
- CalEPA, 1996. Office of Scientific Affairs. Background Levels of Trace Elements in Southern California Soils. Department of Toxic Substances Control. May 1996.

- CalEPA, 1997. Selecting Inorganic Constituents as Chemicals of Potential Concern at Risk Assessments at Hazardous Waste Sites and Permitted Facilities. Final Policy. Human and Ecological Risk Division. Department of Toxic Substances Control. February 1997.
- CalEPA, 1999. Preliminary Endangerment Assessment Guidance Manual. State of California Department of Toxic Substances Control. January 1994, Second Printing June 1999.
- CalEPA, 2003. Screening for Environmental Concerns at Sites with Contaminated Groundwater, California Regional Water Quality Control Board, San Francisco Bay Region, Interim Final - July 2003 (updated 9/4/03).
- CalEPA, 2004. *Cancer Potency Factors - Toxicity Criteria Database*. Office of Environmental Health Hazard Assessment. Website address: <http://www.oehha.ca.gov/risk/chemicalDB/index.asp>.
- CalEPA, 2005. Interim Final Guidance for the Evaluation and Mitigation of Subsurface Vapor Intrusion to Indoor Air. State of California Department of Toxic Substances Control. December 15, 2004. Revised February 7, 2005.
- California Air Resources Board (CARB), 2001. CARB N. Long Beach Station, California Ambient Air Quality Data 1980-1996, CD Number TSD-97-008-CD. Tables 9-3, 9-13, and 9-14.
- CH2M Hill, 1993. Draft Risk Assessment Strategy Report for the Del Amo Site. Los Angeles, California. October 6, 1993.
- CH2M Hill, 1998a. Joint Groundwater Risk Assessment, Montrose and Del Amo Sites, February 24, 1998, by McLaren Hart and Dames & Moore.
- CH2M Hill, 1998b. Final Joint Groundwater Feasibility Study for the Montrose and Del Amo Sites, May 18, 1998.
- Cowherd, C., G. Muleski, P. Engelhart, and D. Gillete, 1985. Rapid Assessment of Exposure to Particulate Emissions from Surface Contamination. Prepared for USEPA Office of Health and Environmental Assessment. USEPA/600/8-85/002.
- Crump, K., D. Hoel, C. Langley, and R. Peto, 1976. Fundamental Carcinogenic Process and their Implications for Low Dosage Risk Assessment. *Cancer Res.* 36:29975-2979

- Dames & Moore (D&M), 1991. Baseline Health Risk Characterization. Del Amo Site. April 1, 1991.
- D&M, 1993. Remedial Investigation/Feasibility Study Workplan. Del Amo Site. February 25, 1993.
- D&M, 1998. Final Groundwater Remedial Investigation Report. Del Amo Study Area. May, 1998.
- D&M, 1999a. Vapor Transport Modeling Report. Del Study Area, Los Angeles, California. November 4, 1999.
- D&M, 1999b. Baseline Risk Assessment Workplan. Del Amo Site. March, 1999.
- ENVIRON, 1998. A Methodology for using Background PAHs to Support Remediation Decisions. March 6, 1998.
- Feenstra, S., D. M. Mackay, and J.A. Cherry, 1991. A method for assessing the presence of residual NAPL based on organic chemical concentrations in soil samples, *Ground Water Monitoring Review*, 11(2), 128-136, Spring 1991.
- Geraghty & Miller, 1998. Phase II Environmental Site Assessment, Prentiss Properties Site, Los Angeles, CA. July 2, 1998.
- Johnson, E.F., 1991. "A Partnership Between the Dioxin Receptor and a Basic Helix-Loop-Helix Protein." *Science* 252: 924-925.
- Johnson, P.C. and R.A. Ettinger (J&E), 1991. Heuristic Model for Predicting the Intrusion Rate of Contaminant Vapors into Buildings: *Environmental Science & Technology*, Vol. 25, p. 1445-1452.
- Johnson, P.C., M.W. Kemblowski, and R.L. Johnson, 1999. Assessing the Significance of Subsurface Contaminant Vapor Migration to Enclosed Spaces: Site-Specific Alternatives to Generic Estimates, *Journal of Soil Contamination*, 8(3) 389-421.
- Marcus, W., and A. Rispin, 1990. Threshold Carcinogenicity Using Arsenic As an Example. In: *Advances in Modern Toxicology: Risk Assessment and Risk Management of Industrial and Environmental Chemicals*. Prin Sci. Pub. Co., Inc., Princeton, NJ.

- McLaren Hart and Dames & Moore, 1998. Joint Groundwater Risk Assessment, Montrose and Del Amo Sites. February.
- National Climatic Data Center (NCDC), 2004. Department of Commerce, National Oceanic and Atmospheric Administration (NOAA), and the National Environmental Satellite, Data and Information Service (NESDIS). Last Update 23-June.
- Sielken, R., 1985. Some Issues in The Quantitative Modeling Portion of Cancer Risk Assessment. *Required Toxicol. Pharm.* 5:175-181.
- URS Corporation (URS), 2000. Technical Memorandum of Information for Risk Assessment Conference Call. December 15, 2000.
- URS, 2001a. Draft Baseline Risk Assessment Report, Del Amo Site, Los Angeles California. September 28, 2001.
- URS, 2001b. Workplace Air Monitoring Program Report, Del Amo Study Area, Los Angeles, California. November 16, 2001.
- URS, 2002. Remedial Investigation/Feasibility Study Work Plan Addendum, Supplemental Shallow Soil Sampling. December 4, 2002.
- URS, 2003. MW-20 Pilot Program Summary Report, Del Amo Study Area. Los Angeles, CA. June 30, 2003.
- URS, 2004. Draft Remedial Investigation Report, Soil and NAPL Operable Unit, Del Amo Superfund Site. Los Angeles, CA. April 7, 2004.
- URS, 2005. Revised Draft Baseline Risk Assessment Report, Del Amo Site, Los Angeles, CA. May 19, 2005.
- U.S. Environmental Protection Agency (USEPA), 1986. Guidelines for Health Risk Assessment of Chemical Mixtures. *Federal Register*, Vol. 51, No. 185, pp. 34014-34025. September 24.
- USEPA, 1989. Risk Assessment Guidance for Superfund Manual Part A. Interim Final. OSWER Directive 9285.701A. Office of Solid Waste and Emergency Response, Washington, DC.

- USEPA, 1991a. Risk Assessment Guidance for Superfund, Volume 1, Human Health Evaluation Manual, Supplemental Guidance Standard Default Exposure Factors, Draft Final, March 25, 1991, OSWER Directive 9285.6-03, Office of Solid Waste and Emergency Response, Washington, DC.
- USEPA, 1991b. Role of Baseline Risk Assessment in Superfund Remedy Selection Decisions. Office of Solid Waste and emergency Response. PB91-921359. Washington, D.C.
- USEPA, 1992a. Dermal Exposure Assessment: Principles and Applications. USEPA/600/8-91/011.
- USEPA, 1992b. Supplemental Guidance to RAGS: Calculating the Concentration Term. Office of Solid Waste and Emergency Response, Washington, D.C. Publication 9285.7-081. May.
- USEPA, 1993. Superfund's Standard Default Exposure Factors For The Central Tendency and Reasonable Maximum Exposure. Preliminary Review Draft. May 5.
- USEPA, 1995a. Proposed and Interim Guidelines for Carcinogen Risk Assessment. NCEA-I-024 Review Draft, July, 1995.
- USEPA, 1995b. Guidance for Risk Characterization. U.S. USEPA Science Policy Council. Online. February.
- USEPA, 1997a. Exposure Factors Handbook. Volumes I-III. An Update to Exposure Factors Handbook USEPA/600/8-89/043 March 1989. USEPA/600/P-95-002Fa. August.
- USEPA, 1997b. Health Effects Assessment Summary Tables (HEAST). Annual FY-1997. OHEA-ECAO-CIN-821. March.
- USEPA, 1997c. The Lognormal Distribution in Environmental Applications. Office of Solid Waste and Emergency Response. EPA/600/R-97/006. Washington D.C.
- USEPA, 1997WP ROD. Record of Decision for the Del Amo Waste Pits Operable Unit Del Amo Facility Proposed Superfund Site. Los Angeles, CA. September 5, 1997.
- USEPA, 1999a. CLP National Functional Guidelines for Organic Data Review. October, 1999.

- USEPA, 1999b. Risk Assessment Guidance for Superfund. Volume 3 (Part A, Process for Conducting Probabilistic Risk Assessment), Draft. Office of Emergency and Remedial Response, Washington, D.C. 1999.
- USEPA, 1999JGWROD. Record of Decision for Dual Site Groundwater Operable Unit, Montrose Chemical and Del Amo Superfund Sites. March 30, 1999.
- USEPA, 2000a. User's Guide for the Johnson and Ettinger (1991) Model for Subsurface Vapor Intrusion into Buildings (Revised). Prepared by Environmental Quality Management, Inc. for submittal to USEPA, Office of Emergency and Remedial Response. December.
- USEPA, 2000b. Risk Assessment Guidance for Superfund. Volume I: Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment), Interim Guidance. EPA/540/R-99/005. Office of Solid Waste and Emergency Response, Washington, DC. PB99-963312. OSWER 9285.7-02EP. September.
- USEPA, 2002a. Region IX Preliminary Remedial Goal (PRG) Table. October.
- USEPA, 2002b. Draft Guidance for Evaluating the Vapor Intrusion to Indoor Air Pathway from Groundwater and Soils (Subsurface Vapor Intrusion Guidance). Office of Solid Waste and Emergency Response. OSWER. November.
- USEPA, 2002c. Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites. Office of Solid Waste and Emergency Response. OSWER 9355.4-24.
- USEPA, 2003a. USEPA Recommendations of the Technical Review Workgroup for Lead for an Approach to Assessing Risks Associated with Adult Exposure to Lead in Soil. January 2003, version May 19, 2003.
- USEPA, 2003b. Draft Final Guidelines for Carcinogen Risk Assessment (External Review Draft, February 2003). U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC, 2003.
- USEPA, 2004a. Letter to N. Pasvantis, Shell Oil Company, Re: Del Amo Superfund Site, Administrative Order on Consent, Docket No. 92-13 Supplemental Shallow Soil Investigation. March 12, 2004.
- USEPA, 2004b. Integrated Risk Information System (IRIS) Substance File Database.

USEPA, 2004c. Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual (Part E, Supplemental Guidance for Dermal Risk Assessment) Final. EPA/540/R-99/005. Office of Solid Waste and Emergency Response, Washington, DC. PB99-963312. OSWER 9285.7-02EP. July.