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FINAL

ADDITIONAL DATA COLLECTION REPORT AND SCREENING LEVEL ECOLOGICAL RISK ASSESSMENT AND STEP 3A OF BASELINE ECOLOGICAL RISK ASSESSMENT AT SWMUs 1 AND 2

NAVAL STATION ROOSEVELT ROADS
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H Equilibrium Partitioning Approach
I Identification of Bioaccumulative Chemicals

LIST OF ACRONYMS AND ABBREVIATIONS

AET	Apparent Effects Threshold
AOCs	Areas of Concern
AQUIRE	Aquatic Toxicity Information Retrieval
AUF	Area Use Factor
AVS	Acid Volatile Sulfide
BAF	Bioaccumulation Factor
BAF_{sed}	Sediment-to-fish BAF
BAF_{sw}	Surface water-to-fish BAF
BCF	Bioconcentration Factor
BCF_{sw}	Measured surface water-to-fish BCF (L/kg)
bgs	Below ground surface
BTAG	Biological Technical Assistance Group
BW	Body Weight
BW_j	Mean body weight for receptor j (kg, wet weight)
BW_r	Body weight of receptor species (kg)
BW_t	Body weight of test species (kg)
C_{sed}	Maximum sediment concentration (mg/kg)
C_{sw}	Maximum surface water concentration (mg/L)
C_{xf}	Concentration of chemical x in whole-body fish (mg/kg)
CCC	Criteria Continuous Concentration
CCME	Council of Ministers of the Environment
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CMS	Corrective Measures Study
CN-	Cyanide Ion
CNO	Chief of Naval Operations
COPCs	Chemicals of Potential Concern
CTO	Contract Task Order
D	duplicate
DI_x	Dietary intake for chemical x (mg chemical/kg body weight/day)
DoN	Department of the Navy
DRMO	Defense Reutilization and Marketing Office
EC_{50}	Median Effective Concentration
ECOTOX	Ecotoxicology Database
EqP	Equilibrium Partitioning
ER	Equipment Rinsate
ER-L	Effect Range Low
ER-M	Effect Range Median
ERA	Ecological Risk Assessment
FB	Field Blank
FCM	Food Chain Multiplier
FCV	Final Chronic Value
FC_{xi}	Maximum concentration of chemical x in food item i (mg/kg, dry weight)
FIR	Food ingestion rate (kilograms per day [kg/day], dry-weight)
FIR_j	Mean food ingestion rate for receptor j (kg/day, dry-weight)

LIST OF ACRONYMS AND ABBREVIATIONS
(Continued)

GPS	Global Positioning System
HCN	Hydrogen cyanide
HQ	Hazard Quotient
IAS	Initial Assessment Study
IC_{50}	Median Inhibition Concentration
IR	Installation Restoration
K_d	Adsorption Coefficient
K_{oc}	Organic Carbon Partition Coefficient
K_{ow}	Octanol-Water Partitioning Coefficient
LANTDIV	Atlantic Division, Naval facilities Engineering Command
LC_{50}	Median Lethal Concentration
LD_{50}	Median Lethal Dose
LEL	Low Effect Level
LOAEL	Lowest Observed Adverse Effect Level
LOEC	Lowest Observable Effect Concentration
LOEL	Lowest Observable Effect Level
Log B_v	Log soil-to-plant BCF
Log K_{ow}	Log octanol-water partitioning coefficient
MATC	Maximum Acceptable Toxicant Concentration
MDL	Method Detection Limit
µg/kg	micrograms per kilogram
µg/L	micrograms per liter
mg/kg	milligrams per kilogram
mg/kg-BW/day	milligrams per kilogram of body weight per day
MS/MSD	Matrix Spike/Matrix Spike Duplicate
MRL	Method Reporting Limit
NAWQC	National Ambient Water Quality Criteria
NEESA	Naval Energy and Environmental Support Activity
NFESC	Naval Facilities Engineering Command
NOAA	National Oceanic and Atmospheric Administration
NOAEL	No Observed Adverse Effect Level
$NOAEL_{ij}$	Ingestion-based screening value for chemical i applied to receptor j (mg chemical/kg body weight/day)
$NOAEL_r$	NOAEL of the receptor species (mg/kg-BW/day)
$NOAEL_t$	NOAEL of the test species (mg/kg-BW/day)
NOEC	No Observed Effect Concentration
NPL	National Priorities List
NSRR	Naval Station Roosevelt Roads
OP	Organo Phosphorus
OU	Operable Unit

LIST OF ACRONYMS AND ABBREVIATIONS
(Continued)

PAHs	Polyaromatic Hydrocarbons
PCBs	Polychlorinated Biphenyls
PDF_i	Proportion of diet composed of food item i (mg/kg, dry weight)
PDF_{ij}	Proportion of receptor j diet composed of food item i (mg/kg, dry weight) (Equation 4-6)
PDS	Proportion of diet composed of surface soil/sediment (dry weight basis)
PDS_j	Proportion of receptor j diet composed of surface soil (dry weight basis) (Equation 4-6)
PEC	Probable Effect Concentration
PEL	Permissible Effect Level
QA/QC	Quality Assurance/Quality Control
RAGS	Risk Assessment Guidelines
RCRA	Resource Conservation and Recovery Act
RFA	RCRA Facility Assessment
RFI	RCRA Facility Investigation
SCV	Secondary Chronic Value
SC_x	Maximum concentration of chemical x in surface soil/sediment (Equation 4-4)
SC_x	Concentration of chemical x in surface soil resulting in a mean exposure dose greater than or equal to the NOAEL-based screening value (mg/kg, dry weight) (Equation 4-6)
SEL	Severe Effect Level
SQG	Sediment Quality Guideline
SQUIRTs	Screening Quick Reference Tables
SSI	Supplemental Site Investigation
SSL	Soil Screening Level
SVOCs	Semivolatile Organic Compounds
SWMU	Solid Waste Management Unit
TEL	Threshold Effects Level
TOC	Total Organic Carbon
TPH	Total Petroleum Hydrocarbons
UCL	Upper Confidence Limit
USACE	United States Army Corps of Engineers
USEPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
USTs	Underground Storage Tanks
VOCs	Volatile Organic Compounds

1.0 INTRODUCTION

This report presents results from the Additional Data Collection Field Investigation performed in July 2003 and the Additional Data Collection Effort performed in October 2004 in support of the Screening-level ecological risk assessment (ERA) and Step 3A of the baseline ERA for Solid Waste Management Unit (SWMU) 1 (Army Cremator Disposal Site), SWMU 2 (Langley Drive Disposal Site), and the Estuarine Wetland System Background located at Naval Station Roosevelt Roads (NSRR), Ceiba, Puerto Rico. It should be noted that the report developed following the July 2003 field investigation [Draft Additional Data Collection Report and Screening Level Ecological Risk Assessment and Step 3a of Baseline Ecological Risk Assessment at SWMUs 1 and 2 (Baker, 2004a)] identified the need for additional data collection at SWMUs 1 and 2 and recommended additional sampling. The Additional Data Collection Effort performed in October 2004 was conducted to address these identified needs. This report has been prepared under the Corrective Action provisions of the NSRR's Resource Conservation and Recovery Act (RCRA) Permit No. PR2170027203. This additional data collection report has been prepared under contract to the Atlantic Division, Naval Facilities Engineering Command (LANTDIV), Contract Number N62470-95-D-6007, Contract Task Order (CTO) 271.

1.1 Objective of the Additional Data Collection Field Investigation Report in Support of the Ecological Risk Assessment

The objective of the Additional Data Collection Field Investigation was to perform additional sampling of surface water and sediment at SWMUs 1, 2, and the Estuarine Wetland System Background to address the data gaps presented in the Draft Screening Level Ecological Risk Assessment Problem Formulation (Step 1) and Exposure Estimate for SWMUs 1 and 2 (Baker, 2001). This objective was met with the performance of the field investigation conducted in July 2003.

The objective of the 2004 Additional Data Collection Effort is listed in the following bulleted items:

- Establish nature and extent,
- Establish and/or verify that maximum concentrations have been captured,
- Reduce uncertainties with existing analytical data,
- Establish a final list of potential ecological drivers for evaluation in the baseline ERA, and
- Collect data for evaluation of chemical bioavailability.

The objectives of this report are as follows:

- Present the data collected during the additional data collection field investigation, as well as to present the revised Step 3a of the ERA incorporating the new data collected.

- Make a determination whether or not this site will move forward to Step 3b of the ERA or continue in the Corrective Measures Study (CMS) planning stage.

1.2 Facility and Site Description

This section contains a description of the physical layout and a background history of NSRR, as well as a description of the physical layouts of the Estuarine Wetland System Background, as well as SWMUs 1 and 2.

The subsections that follow provide a brief facility description, as well as a site history and descriptions of the Estuarine Wetland System Background and SWMUs 1 and 2.

1.2.1 Facility Description

NSRR occupies over 8,600 acres on the northern side of the east coast of Puerto Rico, along Vieques Passage with Vieques Island lying to the east about 10 miles off the harbor entrance. The north entrance to NSRR is about 35 miles east along the coast road (Route 3) from San Juan. The closest large town is Fajardo (population approximately 37,000), which is about 10 miles north of NSRR off Route 3. Ceiba (population approximately 17,000) adjoins the west boundary of NSRR (see Figure 1-1).

NSRR was commissioned in 1943 as a Naval Operations Base, and redesignated a Naval Station in 1957. The current primary mission of NSRR is limited. NSRR is currently preparing for closure.

1.2.2 Estuarine Wetland System Background

The Estuarine Wetland System Background was identified at NSRR from areas unlikely to have been impacted by base operations. This background area lies within the Los Machos National Forrest north of the fire station along Forrestal Drive as shown on Figure 1-2. The previous field sampling was performed on sediment and surface water. The analysis conducted for the previous field sampling was limited to RCRA Appendix IX volatile organic compounds (VOCs), semivolatile organic compounds (SVOCs), metals, and polychlorinated biphenyls (PCBs). Since this area has not been impacted by base operations, it is ideally suited for further background sampling.

Samples located in this area during this most recent investigation were labeled by the sample location first (i.e., BGEWSSD05). The sample numbering for all "BG" samples coincided with the previous background sampling conducted for SWMU 9 (9SW/SD05, 9SW/SD08 through 9SW/SD12). The actual site and sample identifications were changed for this investigation to further indicate that this is a background area that has not been impacted by any site or base operations.

1.2.3 SWMU 1 Description

SWMU 1 lies east of the Navy Lodge and encompasses an area of roughly 116 acres (see Figure 1-2). The site is bounded to the north by Kearsage Road leading to the US Customs Pier, Ensenada Honda to the east, estuarine wetlands to the south, and the Navy Lodge and Bowling Alley to the west. Based on previous reports, the Army Cremator Disposal Site operated from the early 1940s until the early 1960s and was the main station landfill during this period. Waste material was disposed of by piling, burning, and compacting (A.T. Kearney, Inc., 1988). According to the Naval Energy and Environmental Support Activity (NEESA), an estimated 100,000 tons of waste including scrap metal, inert ordnance, batteries, tires, appliances, cars, cables, dry cleaning solvent cans, paint cans, gas cylinders, construction debris, dead animals, and residential waste was disposed of at this site (NEESA, 1984). No reliable information exists regarding the amounts of material present in the disposal area that could be hazardous; however, in 1984, an Initial Assessment Study (IAS) team estimated that as much as 1,000 tons of hazardous material could be present (NEESA, 1984).

1.2.4 SWMU 2 Description

SWMU 2 is located along Langley Drive approximately 1,000 feet northeast of the Navy Commissary and encompasses an area of roughly 28 acres (see Figure 1-2). The site extends from Langley Drive to the estuarine wetland system. This site operated as a landfill from approximately 1939 to 1959 and is documented as having been used for the disposal of both hazardous and non-hazardous wastes (NEESA, 1984). Debris noted during the IAS included partially buried metal and concrete objects, old fuel lines, flexible metal hoses, sample containers containing pellets, steel cables, hardened tar, rubble, and ten to fifteen 55-gallon drums that were corroded. The drum contents, generally consisting of a whitish solid with a green outer crust, were exposed (NEESA, 1984). The IAS team estimated the volume of disposed waste to be approximately 1,700 cubic yards, of which approximately 20,000 pounds could be hazardous.

The additional data collection field investigation was designed to address the surface water and sediment data gaps presented in the Draft Screening-Level Ecological Risk Assessment Problem Formulation (Step 1) and Exposure Estimate for SWMUs 1 and 2 (Baker, 2001).

1.3 Regulatory Framework and Site Status

In 1943, NSRR was commissioned as a Naval Operations Base. NSRR continued in this status until 1957 when it was redesignated a naval station with the mission of providing full support for Atlantic Fleet weapons training and development activities. Until 1993 all environmental operations, with the exception of underground storage tanks (USTs), were conducted under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) regulations as part of the Department of the Navy's (DoN) Installation Restoration (IR) Program. Naval Station Roosevelt Roads submitted a RCRA Part B Permit application for the storage of hazardous waste on the Base. Recognizing that corrective action would apply to unpermitted waste management units, the Navy performed a Supplemental Site Investigation (SSI) at a variety of units (including SWMUs 1 and 2) to provide additional site characterization information to the United States Environmental Protection Agency (USEPA) to assist in their permitting decisions. On October 20, 1994, a Final RCRA Part B permit was issued by the USEPA Region II to the Defense Reutilization and Marketing Office (DRMO) of NSRR as RCRA/HSWA Permit No. PR2170027203. The corrective action provisions of the permit contained specific requirements for investigation, and potentially, remediation at SWMUs 1 and 2, as well as required RCRA Facility Investigation (RFI) activities at 25 SWMUs and 3 Areas of Concern (AOCs). Two additional SWMUs (53 and 54) were identified during May of 2000 bringing the total to 27 SWMUs and 3 AOCs.

RCRA regulations provide a procedure to investigate and remediate areas that may have been affected by a release of hazardous wastes. The first steps for investigating a site are the RCRA Facility Assessment (RFA) and the RFI. These assessments and investigations are studies on a property to determine if there has been a release of hazardous waste and to quantify any releases that have occurred. If these studies determine that a release has occurred, a CMS is performed to identify the most appropriate corrective measure for a given site.

A RFA was performed in 1988 and updated in 1993 by A.T. Kearney, Inc. for the USEPA to identify SWMUs and AOCs, and to assess the potential for the release of hazardous constituents from any areas or units. The RFA identified 52 SWMUs and 4 AOCs, and recommended additional investigation at 25 of the SWMUs and three of the AOCs.

The RCRA Part B permit required a full RFI for SWMUs 1 and 2. RFI Work Plans (Baker, 1995) were developed for NSRR that included SWMUs 1 and 2. The work plan provided the framework for site characterization activities; its scope was guided by the results of the SSI. The field investigations for SWMUs 1 and 2 proposed in this work plan were conducted during October/November 1996 and September/October 1997. The Revised Draft RCRA Facility Investigation Report for Operable Unit 3/5 (including SWMUs 1 and 2) was submitted in April 1999 (Baker, 1999), and EPA approved in a September 28, 1999 letter.

Based on the recommendations presented in the Revised Draft RFI Report for OU 3/5 (Baker, 1999), a Revised Final II CMS Work Plan was submitted on July 14, 2000 (Baker, 2000), and EPA approved on May 4, 2001. A Draft Screening-Level Ecological Risk Assessment Problem Formulation (Step 1) and Exposure Estimate, as well as the Draft Additional Data Collection Work Plan in Support of Ecological Risk Assessment for SWMUs 1 and 2 were submitted on August 10, 2001 (Baker, 2001). EPA approved the above documents in their letter dated October 4, 2001. The Navy submitted a letter to the EPA stating the lack of funding to perform the work associated with SWMUs 1 and 2. The Navy submitted a response to EPA's comment letter dated October 4, 2001, as well as submitted an Addendum to the Draft Screening-Level Ecological Risk Assessment Problem Formulation (Step 1) and Exposure Assessment for SWMUs 1 and 2 on May 14, 2003. The EPA approved the above addendum on June 10, 2003. The field

investigation associated with the above-mentioned EPA approved Screening-Level ERA plan was initiated and completed in July 2003. This additional data collection report and Step 3a of the baseline ERA focus on the objectives found in Section 1.1 of this report.

1.4 Previous Investigations

Sampling activities at SWMUs 1 and 2 have previously been conducted under a Confirmation Study in 1985 (Round 01) and 1987 (Round 2) (SWMUs 1 and 2), a Supplemental Investigation in 1993 (SWMUs 1 and 2), a Relative Risk Ranking in 1995 (SWMUs 1 and 2), and a RFI field investigation in 1996 (SWMUs 1 and 2) and 1997 (SWMU 1) as mentioned in the Draft Screening-Level Ecological Risk Assessment Problem Formulation (Step 1) and Exposure Estimate (Baker, 2001).

Environmental media collected during the various field investigations at the Estuarine Wetland System Background included surface water and sediment as presented in Table 1-1, while the media collected during the various field investigations at SWMUs 1 and 2 included surface soil, subsurface soil, groundwater, surface water, and sediment as presented in Tables 1-2 and 1-3, respectively. During the 1993 Supplemental Investigation (Baker, 1994), a geophysical investigation (electromagnetic terrain profiling and magnotometry) was performed at SWMU 1 based on a review of photographs and map analysis as discussed in the Revised Draft RFI Report for OU 3/5 (Baker, 1999). Correlation was found to be high between the disposal features noted by the photo-interpretation, and disposal indications found during the land-clearing activities, as presented on Figure 1-3. It should be noted that the geophysical investigation only covered the horizontal extent of the disposal area. All sampling locations that have been conducted at SWMU 1, along with the geophysical and aerial photo interpretation results to date are presented on Figure 1-3 to assist with interpretation. The field investigation and associated analytical data for the Estuarine Wetland System Background data were presented and discussed in the Final CMS Investigation Report for SWMU 9 (Baker, 2003). The field investigations and associated analytical data for SWMUs 1 and 2 were presented and discussed in the EPA-approved Revised Draft RFI for Operable Unit (OU) 3/5 (Baker, 1999). The reader is referred to these documents for a detailed description of sampling activities and analytical data.

1.5 Current Site Conditions

The following is a description of the site conditions identified during the July 2003 field investigation. The disposal area at SWMU 1 was identified to be located in the upland area of the SWMU. The SWMU 1 upland area is heavily overgrown with secondary growth vegetation with no unobstructed access to the site. There are no maintained access ways to the SWMU. The estuarine wetland area is made up of a mangrove forest extending out to the open water located east of the SWMU.

The disposal area at SWMU 2 was identified to be located in the upland area of the SWMU. The SWMU 2 upland area is consistent with that identified at SWMU 1 consisting of heavily overgrown secondary growth vegetation with no unobstructed access to the site. There are no maintained access ways to the SWMU. The estuarine wetland area is made up of a mangrove forest extending out to the open water located southeast of the SWMU. The field crew observed metal and wood scattered on the surface in isolated areas of the estuarine wetland of SWMU 2.

1.6 Report Organization

This Additional Data Collection Report in Support of the ERA at SWMUs 1 and 2 is organized into six sections. Section 1.0, the Introduction, is designed to introduce the reader to the objective

of the additional data collection investigation, a description of the base, as well as the Estuarine Wetland System Background and SWMUs 1 and 2, a regulatory framework established at this site, a discussion of the previous investigations, and current site conditions at the above listed locations. Section 2.0 provides the methodologies utilized during the field investigation, while Section 3.0 describes the results from the additional data collection field investigation. The refined screening-level ERA and Step 3a of the baseline ERA is described in Section 4.0. Section 5.0 provides the conclusions and recommendations based on the results obtained from the additional data collection field investigation, and findings presented in the refined ERA. Section 6.0 provides the references cited in this report.

2.0 INVESTIGATION METHODOLOGIES

The additional data collection field investigations at the Estuarine Wetland System Background, as well as at SWMUs 1 and 2, consisted of the collection of surface water and sediment for the purpose of addressing data gaps presented in the Draft Screening-Level Ecological Risk Assessment Problem Formulation (Step) 1 and Exposure Estimate for SWMUs 1 and 2 (Baker, 2001) and the collection of surface soil, subsurface soil, and sediment for the purpose of addressing additional needs identified in the Draft Additional Data Collection Report and Screening Level Ecological Risk Assessment and Step 3a of Baseline Ecological Risk Assessment at SWMUs 1 and 2 (Baker, 2004a). The methods and procedures utilized during the field investigation are presented in the following subsections.

2.1 Sampling Procedures

All the investigation tasks described in subsequent sections of this report were performed in accordance with the techniques and methodologies provided in the original USEPA approved work plan (Baker, 1995). Therefore, only the work elements themselves are discussed in the sections that follow.

2.1.1 Surface Water

The surface water samples were collected using the direct dip method with a sample container. A dedicated non pre-preserved glass sample container was utilized to collect and pour the surface water into the laboratory prepared sample containers. The sample containers were labeled and kept in coolers on ice and under strict chain-of-custody until delivered to the laboratory. Chain-of-custody forms for the samples collected are provided as Appendix A. The following surface water field parameters: pH, dissolved oxygen, temperature, specific conductance, conductivity, and salinity as presented in Tables 2-1 through 2-3 were measured in the field. The field notes taken during this investigation are provided in Appendix B.

Samples for the Estuarine Background were shipped to a fixed based laboratory for analysis of Appendix IX pesticides/PCBs, chlorinated herbicides, organo phosphorous (OP) pesticides, dioxins/furans, low level polyaromatic hydrocarbons (PAHs), and cyanide as presented in Table 2-4.

Samples for SWMUs 1 and 2 were shipped to a fixed based laboratory for analysis of Appendix IX VOCs, SVOCs, pesticides/PCBs, chlorinated herbicides, OP-pesticides, dioxins/furans, low level PAHs, total and dissolved metals, and cyanide as presented in Tables 2-5 and 2-6.

2.1.2 Sediment

Sediment samples were collected from the top six inches of sediment using a disposable sediment liner and/or a new stainless steel spoon. Sediment obtained with the sediment liners was composited in aluminum pie pans prior to placement on the sample container. Care was taken to remove any rocks or twigs if present prior to the placement of the sample into the laboratory supplied container. Samples were labeled and kept in coolers on ice and under strict chain-of-custody until delivered to the laboratory. Chain-of-custody forms for the samples collected are provided as Appendix A, and the field notes taken during this investigation are provided in Appendix B.

Samples for the Estuarine Background were collected during the Additional Data Collection Investigation (July 2003) and were shipped to a fixed based laboratory for analysis of Appendix

IX pesticides/PCBs, chlorinated herbicides, OP-pesticides, dioxins/furans, low level PAHs, total organic carbon (TOC), grain size, and cyanide as presented in Table 2-4.

Samples collected during the Additional Data Collection Investigation (July 2003) for SWMUs 1 and 2 were shipped to a fixed based laboratory for analysis of Appendix IX VOCs, SVOCs, pesticides/PCBs, chlorinated herbicides, OP-pesticides, dioxins/furans, low level PAHs, grain size, TOC, total metals, and cyanide as presented in Tables 2-5 and 2-6.

Samples collected during the Additional Data Collection Effort (October 2004) for SWMUs 1 and 2 were shipped to a fixed based laboratory for analysis of select SVOCs, select pesticides, select metals, TOC, and AVS/SEM as presented in Tables 2-5a and 2-6a.

2.1.3 Surface Soil

Surface soil samples were collected from 0 to 1 foot bgs utilizing a stainless steel hand auger and/or stainless steel spoon. Surface soil was composited in aluminum pie pans prior to placement in the sample container. Care was taken to remove any rocks or twigs if present prior to the placement of the sample into the laboratory supplied container. Samples were labeled and kept in coolers on ice and under strict chain-of-custody until delivered to the laboratory. Chain-of-custody forms for the samples collected are provided as Appendix A, and the field notes taken during this investigation are provided in Appendix B.

Samples collected during the Additional Data Collection Effort (October 2004) for SWMUs 1 and 2 were shipped to a fixed based laboratory for analysis of select pesticides, select metals, and dioxins/furans as presented in Tables 2-5a and 2-6a.

2.1.4 Subsurface Soil

Subsurface soil samples were collected from a depth of 1 to 2 feet bgs utilizing a stainless steel hand auger and stainless steel spoon. Subsurface soil was composited in aluminum pie pans prior to placement in the sample container. Care was taken to remove any rocks or twigs if present prior to the placement of the sample into the laboratory supplied container. Samples were labeled and kept in coolers on ice and under strict chain-of-custody until delivered to the laboratory. Chain-of-custody forms for the samples collected are provided as Appendix A, and the field notes taken during this investigation are provided in Appendix B.

Samples collected during the Additional Data Collection Effort (October 2004) for SWMUs 1 and 2 were shipped to a fixed based laboratory for analysis of select pesticides and select metals as presented in Tables 2-5a and 2-6a.

2.1.5 Quality Assurance/Quality Control Samples

Quality Assurance/Quality Control (QA/QC) samples were collected during the additional data collection investigation and the additional data collection effort at the above-mentioned sites. These samples were obtained to:

(1) ensure that the new stainless steel spoons were free of contamination (i.e., equipment rinsate blank);
(2) verify that the lab grade deionized water used to collect the equipment rinsate blank samples was free of contamination (i.e., field blank);
(3) evaluate field methodology (i.e., duplicate samples); and,

(4) Evaluate whether cross-contamination occurred during sampling and/or shipping (i.e., trip blanks – SWMUs 1 and 2 only).

Several types of field QA/QC samples were collected and analyzed including duplicate samples, equipment rinsate samples, field blanks, and matrix spike/matrix spike duplicate (MS/MSD).

These QA/QC samples are defined below:

- Duplicate Sample (D): Two samples collected simultaneously into separate containers from the same source under identical conditions. One duplicate sample was collected for every 10 environmental samples collected for each media type.

- Equipment Rinsate Sample (ER): Sample obtained by running laboratory supplied deionized water over/through sample collection equipment after it was decontaminated and/or new disposable equipment. Two equipment rinsate samples (1ER01 and 1ER02) were taken during the Additional Data Collection Investigation (July 2003) by running deionized water over a new disposable stainless steel spoon and through a sediment liner, respectively. Four equipment rinsate samples (2004ER01 through 2004ER04) were collected during the Additional Data Collection Effort (October 2004) by running deionized water over a new disposable stainless steel spoon, through a stainless steel hand auger, over an aluminum pie pan, and through a sediment core liner, respectively These samples were collected to determine if the sampling equipment was free of contamination.

- Field Blank Sample (FB): Sample obtained by collecting laboratory grade deionized water used in the collection of equipment rinsate samples. Two field blank samples (2003FB01 and 2004FB01) were collected to determine if the laboratory grade deionized water was free of contamination.

- Trip Blank (TB): Trip blanks were prepared at the laboratory and shipped with the sample containers. Trip blanks were packaged for shipment with the other VOC samples and sent for analysis. At no time after preparation were the trip blank sample containers opened before they reached the laboratory. At least one trip blank per shipping cooler containing samples requiring VOC analysis was sent to the laboratory for VOC analysis.

- Matrix Spike/Matrix Spike Duplicate (MS/MSD): MS/MSDs are not field samples but are laboratory derived, and are collected to evaluate the matrix effect of the sample upon the analytical methodology. An MS and MSD must be performed for each group of samples of a similar matrix. MS/MSD samples were collected at a frequency of five percent.

2.2 Surface Water Investigation

The surface water investigation conducted during the Additional Data Collection Investigation consisted of surface water from the Estuarine Wetland System Background, and the Estuarine Wetland System and Open Water Marine environments of SWMU 1 and the Open Water marine environment of SWMU 2 only. Surface water was not collected at the Open Water Marine environment from SWMU 2 due to the rationale presented in Section 2.2.3.1. The site-specific surface water investigations are presented in the following subsections. All surface water

samples collected during this investigation were identified in the field using a global positioning system (GPS) unit.

2.2.1 Estuarine Wetland System Background

A total of six surface water samples (BGEWSSW05, BGEWSSW08 through BGEWSSW12), including one duplicate sample, were collected at the same locations as the previous investigations conducted at the Estuarine Wetland System Background during the Phase III RFI and the CMS Investigation conducted at SWMU 9 (Baker, 2003). As mentioned in the Additional Data Collection Work Plan (Baker, 2003), the samples were collected at these locations to build a more definitive base-wide background database. As presented in Section 2.1.1, a set of field parameters, including pH, dissolved oxygen, temperature, specific conductance, conductivity and salinity, were collected at each surface water location in this area as presented in Table 2-1.

Figure 2-1 presents the locations of the Estuarine Wetland System Background samples collected during the July 2003 field investigation.

2.2.2 SWMU 1

The surface water samples collected at SWMU 1 during the July 2003 field investigation were located in two environments (Estuarine Wetland System and Open Water Marine).

2.2.2.1 Estuarine Wetland System

A total of nine surface water samples (01EWSW01 through 01EWSW09) including one duplicate sample, were collected from site locations 1EW01 through 1EW09 from the SWMU 1 Estuarine Wetland System during the July 2003 field investigation as presented on Figure 2-2. A set of field parameters, including pH, dissolved oxygen, temperature, specific conductance, conductivity, and salinity, were measured in-situ at each surface water location in this area as presented in Table 2-2.

2.2.2.2 Open Water Marine

A total of nine surface water samples (01OWSW01 through 01OWSW09), including one duplicate, sample were collected from site locations 1OW01 through 1OW09 from the SWMU 1 Open Water Marine environment during the July 2003 field investigation as presented on Figure 2-2. A set of field parameters, including pH, dissolved oxygen, temperature, specific conductance, conductivity and salinity, were measured in-situ at each surface water location in this area as presented in Table 2-2.

2.2.3 SWMU 2

The surface water samples collected at SWMU 2 during the July 2003 field investigation were only located in the Open Water Marine environment.

2.2.3.1 Estuarine Wetland System

A total of nine surface water samples were to be collected from SWMU 2 estuarine wetland as outlined in the Additional Data Collection Work Plan (Baker, 2001). Due to minimal amounts of surface water available in the estuarine wetland system at SWMU 2, no surface water samples

were collected. Surface water samples were obtained from the open water environment as described in the following section.

2.2.3.2 Open Water Marine

A total of nine surface water samples (02OWSW01 through 02OWSW09), including one duplicate sample, were collected from site locations 2OW01 through 2OW09 from the SWMU 2 Open Water Marine environment during the July 2003 field investigation as presented on Figure 2-3. A set of field parameters, including pH, dissolved oxygen, temperature, specific conductance, conductivity, and salinity, were measured in-situ at each surface water location as presented in Table 2-3.

2.3 Sediment Investigation

The sediment investigation conducted during the Additional Data Collection Investigation consisted of the collection of sediment from the Estuarine Wetland System Background, and the Estuarine Wetland System and Open Water Marine environments of SWMUs 1 and 2. The sediment investigation conducted during the Additional Data Collection Effort (October 2004) consisted of the collection of sediment from the Estuarine Wetland System of SWMUs 1 and 2 and the Open Water Marine environment of SWMU 1. The site-specific sediment investigations are presented in the following subsections. All sediment samples collected during these investigations were identified in the field using a GPS unit.

2.3.1 Estuarine Wetland System Background

A total of six sediment samples (BGEWSSD05, BGEWSSD08 through BGEWSSD12), including one duplicate sample, were collected at the same locations as the previous investigations conducted at this site as mentioned for the Estuarine Wetland System Background surface water. These samples were collected at these locations to build a more definitive base-wide background database as mentioned in the Additional Data Collection Work Plan (Baker, 2003). Figure 2-1 presents the locations of the Estuarine Wetland System Background samples collected during the July 2003 field investigation.

2.3.2 SWMU 1

2.3.2.1 Additional Data Collection Investigation – July 2003

The sediment samples collected at SWMU 1 during the July 2003 field investigation were located in two separate environments (Estuarine Wetland System and Open Water Marine).

Estuarine Wetland System

A total of nine sediment samples (01EWSD01 through 01EWSD09), including one duplicate sample, were collected from site locations 1EW01 through 1EW09 from the SWMU 1 Estuarine Wetland System during the July 2003 field investigation as presented on Figure 2-2. These samples were collected using stainless steel spoons.

Open Water Marine

A total of nine sediment samples (01OWSD01 through 01OWSD09), including one duplicate sample, were collected from site locations 1OW01 through 1OW09 from the SWMU 1 Open Water Marine environment during the July 2003 field investigation as presented on Figure 2-2.

These samples were collected using disposable sediment liners.

2.3.2.2 Additional Data Collection Effort – October 2004

The sediment samples collected at SWMU 1 during the October 2004 field investigation were located in two separate environments (Estuarine Wetland System and Open Water Marine).

Estuarine Wetland System – October 2004

A total of eleven sediment samples (01EWSD10 through 01EWSD20), including two duplicate samples, were collected from site locations 1EW10 through 1EW20 from the SWMU 1 Estuarine Wetland System during the October 2004 field investigation as presented on Figure 2-4. These samples were collected using stainless steel spoons.

Open Water Marine – October 2004

A total of three sediment samples (01OWSD10 through 01OWSD12), including one duplicate sample, were collected from site locations 1OW10 through 1OW12 from the SWMU 1 Open Water Marine environment during the October 2004 field investigation as presented on Figure 2-5. These samples were collected using disposable sediment liners.

2.3.3 SWMU 2

2.3.3.1 Additional Data Collection Investigation – July 2003

The sediment samples collected at SWMU 2 during the July 2003 field investigation were located in two separate environments (Estuarine Wetland System and Open Water Marine).

Estuarine Wetland System

A total of nine sediment samples (02EWSD01 through 02EWSD09), including one duplicate sample, were collected from site locations 2EW01 through 2EW09 from the SWMU 2 Estuarine Wetland System during the July 2003 field investigation as presented on Figure 2-3. These samples were collected using stainless steel spoons.

Open Water Marine

A total of nine sediment samples (02OWSD01 through 02OWSD09), including one duplicate sample, were collected from site locations 2OW01 through 2OW09 from the SWMU 2 Open Water Marine environment during the July 2003 field investigation as presented on Figure 2-3. These samples were collected using disposable sediment liners.

2.3.3.2 Additional Data Collection Effort – October 2004

The sediment samples collected at SWMU 2 during the October 2004 field investigation were located in the Estuarine Wetland System environment.

A total of ten sediment samples (02EWSD10 through 02EWSD19), including one duplicate sample, were collected from site locations 2EW10 through 2EW19 from the SWMU 2 Estuarine Wetland System during the October 2004 field investigation as presented on Figure 2-6. These samples were collected using stainless steel spoons.

2.4 Surface and Subsurface Soil Investigation

The soil investigation conducted during the Additional Data Collection Effort (October 2004) consisted of the collection of surface and subsurface soils from SWMUs 1 and 2. The site-specific soil investigations are presented in the following subsections. All soil samples collected during these investigations were identified in the field using a GPS unit.

2.4.1 SWMU 1

A total of eleven surface soil samples (1SS09 through 1SS19) including two duplicate samples and three subsurface soil samples (1SB15-01 through 1SB17-01) were collected from the SWMU 1 soils. Figure 2-7 presents the locations of the SWMU 1 soil samples collected during the October 2004 field investigation.

2.4.2 SWMU 2

A total of twelve surface soil samples (2SS01 through 2SS05, 2SS07, and 2SS09 through 2SS14) including three duplicate samples and five subsurface soil samples (2SS01-01 through 2SS03-01, 2SS05-01, and 2SS07-01) were collected from the SWMU 2 soils. Figure 2-8 presents the locations of the SWMU 2 soil samples collected during the October 2004 field investigation. It should be noted that three of the subsurface and two of the surface soil samples to be collected at SWMU 2 were not collected. Two of the locations 2SS06 and 2SS08 were unable to be obtained due to their locations being in the middle of the estuarine wetland habitat. The other location 2SS04-01 was not able to be collected due the water table being present at one-foot bgs.

2.5 Quality Assurance/Quality Control Samples

Two equipment rinsate samples (1ER01 and 1ER02) were collected during the July 2003 investigation by running lab grade deionized water over a stainless steel spoon, as well as a sediment liner. One field blank sample (2003FB01) was collected during the July 2003 investigation by collection of a sample of laboratory grade deionized water used to collect the equipment rinsate samples. These samples were analyzed for Appendix IX VOCs, SVOCs, Pesticides/PCBs, chlorinated herbicides, OP-pesticides, dioxins/furans, low level PAHs, total metals, and cyanide as presented in Table 2-7.

Four trip blank samples (1TB01, 1TB02, 2TB01, and 2TB02) were sent to the laboratory and analyzed for Appendix IX VOCs as presented in Table 2-7.

Four equipment rinsate samples (2004ER01 and 2004ER04) were collected during the October 2004 investigation by running lab grade deionized water over a stainless steel spoon, stainless steel hand auger, aluminum pie pan, as well as a sediment core liner. One field blank sample (2004FB01) was collected during the October 2004 investigation by collection of a sample of laboratory grade deionized water used to collect the equipment rinsate samples. These samples were analyzed for select SVOCs, select pesticides, dioxins/furans, and total metals as presented in Table 2-7a.

2.6 Laboratory Analyses

Surface soil, subsurface soil, surface water and sediment samples were submitted to the mainland laboratory for the analysis mentioned in the above sections. The same firm (STL Savannah Laboratories) was retained for this investigation that performed the laboratory analysis for the majority of the field investigations that have taken place at NSRR. This ensured a consistency of

techniques for analysis of the samples. Specific analytical methods utilized in the analysis process are presented in Table 2-8.

2.7 Data Validation

All mainland laboratory data generated by the investigation was subjected to independent, third party, validation. The USEPA Region II Data Validation Standard Operating Procedures were followed. The same firm (Heartland Environmental Services, Inc.) was retained for the July 2003 investigation that performed data validation for many of the previous investigations that have taken place at NSRR. The Salex Group, Inc. performed the data validation for the October 2004 investigation.

3.0 INVESTIGATION RESULTS

This section presents an overview of chemical analytical results obtained from samples taken during the Additional Data Collection Field Investigation (July 2003) at the Estuarine Wetland System Background, SWMUs 1 and 2 and the Additional Data Collection Effort field investigation (October 2004) at SWMUs 1 and 2. The data from the two investigations have been combined together into one complete data set. The data presented below was obtained through sample collection and analysis of surface water, sediment, and surface and subsurface soil samples. The analytical results for environmental and QA/QC samples also are included in this section. Appendix C contains the complete set of analytical results obtained from these investigations, including sediment, surface water, and surface and subsurface soil while Appendix D provides the complete set of QA/QC analytical results. Appendix E provides the data validation report narratives for the analytical results provided in this section.

The data reported for the Estuarine Wetland System Background, as well as for SWMUs 1 and 2 was not screened against any criteria in this section of the report. However, this data was screened against criteria in the Ecological Risk Assessment section of this report (Section 4.0). Please refer to Section 4.0 for any risks associated with the detections that will be mentioned in the subsections that will follow.

3.1 Estuarine Wetland System Background

Surface water and sediment samples were collected from the Estuarine Wetland System Background during the Additional Data Collection Investigation. The surface water and sediment sample results for the estuarine wetland system background are discussed in the subsections that follow.

3.1.1 Surface Water

Six surface water samples and one duplicate sample were collected from the Estuarine Wetland System Background during this investigation. There were no positive detections of any analysis requested as presented in Appendix C.1.

3.1.2 Sediment

Six sediment samples and one duplicate sample were collected from the Estuarine Wetland System Background during this investigation as shown on Figure 2-1. Of the analysis requested, only positive detections of pesticides were present. The TOC and grain size values for these samples are also provided in this section.

Four pesticides (heptachlor epoxide, 4,4'-DDE, chlorobenzilate, and 4,4'-DDD) were positively detected in the sediment samples collected from the Estuarine Wetland System Background as presented in Table 3-1. 4,4'-DDE was detected in all of the samples (ranging from 3.8J to 460 ug/kg) while 4,4'-DDD was detected in 4 of the 6 samples (ranging from 4.2J to 500 ug/kg). These detections are most likely attributed to anthropogenic usage of pesticides at the base. Heptachlor epoxide and chlorobenzilate were only detected in one sample BGEWSSD10.

The TOC in the seven sediment samples collected from the Estuarine Wetland System Background ranged from 28,000 milligrams per kilogram (mg/kg) in sample BGEWSSD11, to 160,000 mg/kg in sample BGEWSSD05.

There were no detections in the remaining analyses for the following parameters: PCBs, low-level PAHs, op-pesticides, chlorinated herbicides, dioxins/furans, and cyanides.

One sample was obtained from each location and tested to determine the grain size from the background estuarine wetland system. The makeup of the sediments in this environment consists of 88.1% fines, 11.8% sands, and 0.1% gravel on average for the five samples. Table 3-2 presents the results from each sample along with the ranges of the soil classification by percent of total sample. Appendix C presents the entire grain size data package.

3.2 SWMU 1

Surface water and sediment samples were collected from the Estuarine Wetland System and Open Water Marine environments during the Additional Data Collection Investigation for SWMU 1. Surface and subsurface soil samples were collected from the upland habitat and sediment samples were collected from the Estuarine Wetland System and Open Water Marine environments during the Additional Data Collection Effort for SWMU 1. The surface water, sediment, and surface and subsurface soil sample results for this site are discussed in the subsections that follow.

3.2.1 Surface Water

The surface water results from the Estuarine Wetland System and Open Water Marine environments for SWMU 1 are discussed separately by environment in the subsections that follow.

3.2.1.1 Estuarine Wetland System

A total of nine surface water samples along with one duplicate sample were collected from the Estuarine Wetland System during the Additional Data Collection Investigation for SWMU 1 as presented in Table 2-5. Of the analysis requested, only positive detections of VOCs, low level PAHs, and metals (total and dissolved fractions) were present.

A total of four volatiles: acetone, benzene, toluene, and ethylbenzene were positively detected in the surface water samples collected as shown in Table 3-3. The positive detections of acetone ranged from 4.4J to 11J ug/L and are most likely associated with the laboratory and not the site. The other three volatiles were detected at low levels ranging from 0.12J (toluene) to 0.21J ug/L (benzene) from only one (1EWS05) of the nine sample locations. This sample was located at 1EWS05 in the north-central portion of the site adjacent to Kearsage Road as presented on Figure 2-2. It is likely that these BTEX constituents are attributed to rainfall runoff from the adjacent roadway.

A total of three low level PAHs: naphthalene, 2-methylnaphthalene, and benzo(b)flouranthene were also detected in the surface water as shown in Table 3-2. 2-Methylnaphthalene was detected in the majority (7 out of 9 locations) of the samples ranging in concentrations of 0.031J to 0.035J ug/L. Naphthalene was detected in three of the nine locations with the maximum detections of 0.042J ug/L in sample 01EWSSW05D. Benzo(b)flouranthene was detected in only one sample 01EWSSW06 at a concentration of 0.55 ug/L. The locations of these detections can be located on Figure 2-2.

There were no detections listed for the remaining analysis requested including: semivolatiles, pesticides/PCBs, op-pesticides, chlorinated herbicides, and dioxins/furans.

A total of fourteen different metals were detected in the total fraction of the surface water samples as shown in Table 3-4, including antimony, arsenic, barium, chromium, cobalt, copper, lead, nickel, selenium, silver, thallium, tin, vanadium, and zinc. Half of the maximum detections found in the total metals were found in sample 01EWSSW02. Although this sample was located at 1EWS02 in the central portion of the site, it was the southern most surface water sample collected at SWMU 1 from the Estuarine Wetland System as presented on Figure 2-2. The remaining maximum detections were found in samples 01EWSSW01 (selenium, thallium, and tin), 01EWSSW04 (chromium), 01EWSSW06 (copper and zinc), and 01EWSSW09 (vanadium).

A total of twelve different metals were detected in the dissolved fraction of the surface water samples as shown in Table 3-5, including antimony, arsenic, barium, chromium, cobalt, copper, lead, nickel, selenium, tin, vanadium, and zinc. All of the metals detected in the total fraction were detected in the dissolved fraction except for silver and thallium. Half of the maximum detections found in the dissolved metals were detected in sample 01EWSSW02, whereas the remaining maximum detections were found in samples 01EWSSW01 (chromium and tin), 01EWSSW03 (copper nickel and zinc), 01EWSSW06 (zinc), 01EWSSW07 (selenium), and 01EWSSW09 (vanadium).

3.2.1.2 Open Water Marine

A total of nine surface water Open Water Marine samples along with one duplicate sample were collected during the Additional Data Collection Investigation for SWMU 1 as presented in Table 2-5. Of the analysis requested, only isolated positive detections of VOCs, SVOCs, and low level PAHs were present.

A total of four volatiles: chloroethane, acetone, 2-butanone (methyl ethyl ketone), and bromoform were detected in the surface water samples collected as shown in Table 3-6. Acetone was detected in four of the nine samples ranging from 3J to 8.8J ug/L and is most likely attributed to laboratory contamination. Positive detections of 2-Butanone ranged from 1.5J to 2J from three of the nine sample locations. Chloroethane and bromoform were only detected in one sample each at 7.2 and 0.24J ug/L, respectively. Figure 2-2 shows the sample locations associated with the SWMU 1 open water environment.

A total of three semivolatiles: isophorone, di-n-butylphthalate, methapyrilene were detected in the surface water as shown in Table 3-6. Isophorone and di-n-butylphthalate were detected in only one sample (01OWSW07) at concentrations of 14 and 5.9J ug/L, respectively. Methylpyriline was also only detected in one sample (01OWSW08) at 3.8J ug/L. It should be noted that the data validator due to continuing calibrations containing compounds with RRFs less than 0.050 rejected four of the non-detected results for methylpyriline.

One low level PAH (naphthalene) was detected in five of the nine surface water sample locations as shown in Table 3-6 ranging from 0.035J to 0.048J. The maximum detection of this constituent was found in samples 01OWSW03 and 01OWSW05 as presented on Figure 2-2.

There were no detections listed for the remaining analysis requested including: pesticides/PCBs, op-pesticides, chlorinated herbicides, and dioxins/furans.

A total of twelve different metals were detected in the total fraction of the surface water samples collected as shown in Table 3-7, including antimony, arsenic, barium, cobalt, copper, lead, nickel, selenium, thallium, tin, vanadium, and zinc. More than half of these metals were detected in all of the sample locations. Arsenic and thallium were detected in only two of the nine sampling locations.

A total of eleven different metals were detected in the dissolved fraction of the surface water samples collected as shown in Table 3-8, including antimony, arsenic, barium, cobalt, copper, nickel, selenium, thallium, tin, vanadium, and zinc. Lead is the only metal that was detected in the total fraction that was not detected in the dissolved fraction. The same trend of number of positive detections identified in the total fraction follows suit with the dissolved fraction.

3.2.2 Sediment

The sediment results from the Estuarine Wetland System and Open Water Marine environments for SWMU 1 are discussed separately by environment in the subsections that follow.

3.2.2.1 Estuarine Wetland System

A total of nine sediment samples along with one duplicate sample were collected from the Estuarine Wetland System during the Additional Data Collection Investigation for SWMU 1 as presented in Table 2-5. A total of eleven sediment samples along with two duplicate samples were collected from the Estuarine Wetland System during the Additional Data Collection Effort for SWMU 1 as presented in Table 2-5a. Of the analysis requested, positive detections of VOCs, low level PAHs, pesticides, chlorinated herbicides, and metals were present. The AVS/SEM, TOC, and grain size values for these samples are also provided in this section. Figures 2-2 and 2-4 provide the spatial reference of the samples obtained from the SWMU 1 estuarine wetland system for the two investigations.

A total of seven volatiles: acetone, carbon disulfide, 2-butanone (methyl ethyl ketone), benzene, toluene, chlorobenzene, and ethylbenzene were detected in the sediment samples as shown in Table 3-9. Toluene and chlorobenzene were only detected in two of the samples at maximum concentrations of 3.7J and 5.8J, respectively. Benzene and ethylbenzene were only detected in three of the nine samples ranging from 1.2J to 31 ug/kg. The detections of carbon disulfide and 2-butanone were observed more consistently across the site. Acetone was detected in every sample and is most likely attributed to the analytical laboratory. Sample 01EWSSD08 was rejected for acetone due to the continuing calibration containing compounds with %Ds greater than 90% or RRFs less than 0.050.

Two low level PAHs: fluoranthene and pyrene were detected in two of the nine sediment samples as presented in Table 3-9. The detections of flouranthene ranged from 4.1J to 8.3J ug/kg while the detections of pyrene ranged from 8.8J to 220 ug/kg. These detections were isolated to the southern boundary of the site.

A total of two pesticides: 4,4'-DDE and 4,4'-DDD were detected in the sediment as shown in Table 3-9. The detections of 4,4'-DDE were from nine of the sixteen sample locations located along the southern boundary of the site at concentrations ranging from 24J to 440J ug/kg. 4,4'-DDD was detected in seven of the sixteen samples also located along the southern boundary of the site at concentrations ranging from 12J to 430 ug/kg. It is likely that these detections are due to anthropogenic usage of pesticides at the base.

Two chlorinated herbicides: 2,4-D and 2,4,5-T were detected at low levels in three of the nine sediment samples as presented in Table 3-9. 2,4-D was detected in two of the nine samples at concentrations ranging from 4J to 9.6J ug/kg. 2,4,5-T was detected in one sample at a concentration of 2.5J ug/kg.

The TOC in the fifteen sediment samples collected from the Estuarine Wetland System at SWMU 1 ranged from 25,000 mg/kg in sample 01EWSSD05D, to 230,000 mg/kg in sample 01EWSD14.

There were no detections listed for the remaining analysis requested including: semivolatiles, PCBs, op-pesticides, and dioxins/furans.

A total of seventeen different inorganics were detected in the sediment samples as shown in Table 3-10, including antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, cyanide, lead, mercury, nickel, selenium, silver, tin, vanadium, and zinc. All of the inorganics detected except for cadmium, cyanide, and silver were detected in all of the samples. Silver was detected in half of the samples while cadmium was only detected in two samples and cyanide in only one sample. The inorganics detected in the sediment samples from SWMU 1 were consistently detected across the site.

One sample was obtained from each location during the Additional Data Collection Effort for AVS/SEM analysis. AVS was detected in every sample ranging in concentrations from 0.5 to 21.8 µmole/g as shown in Table 3-10. SEM metals (cadmium, copper, nickel, lead, and zinc) were detected in the majority of the samples. The total SEM results ranged from 0.083 to 1.1582 µmole/g as shown in Table 3-10.

One sample was obtained from each location during the Additional Data Collection Investigation to determine the grain size from the SWMU 1 estuarine wetland system. The makeup of the sediments in this environment consists of 78.3% fines, 21.6% sands, and 0.1% gravel on average for the nine samples. These averages are similar to those obtained from the estuarine wetland background samples as shown in Table 3-11. The results from each sample along with the ranges of the soil classification by percent of total sample are presented in Table 3-11. Appendix C presents the entire grain size data package.

3.2.2.2 Open Water Marine

A total of nine sediment Open Water Marine samples along with one duplicate sample were collected during the Additional Data Collection Investigation for SWMU 1 as presented in Table 2-5. A total of three sediment Open Water Marine samples along with one duplicate sample were collected from the Open Water Marine during the Additional Data Collection Effort for SWMU 1 as presented in Table 2-5a. Of the analysis requested, VOCs, SVOCs, low level PAHs, and chlorinated herbicides were present. The TOC and grain size values for these samples are also provided in this section. Figures 2-2 and 2-5 provide the spatial reference of the samples obtained from the SWMU 1 open water marine system for the two investigations.

Three volatiles: acetone, carbon disulfide, and 2-butanone (methyl ethyl ketone) were detected in all of the samples with the maximum concentrations from 1OWSD04. Acetone concentrations ranged from 59J to 310 ug/kg and are likely to originate from the laboratory. Low levels of carbon disulfide and 2-butanone were detected ranging from 3.3J to 20 ug/kg and 11J to 43J ug/kg, respectively. Ethylbenzene was detected in only one of the nine samples at a concentration of 1.6J ug/kg. Table 3-12 provides all the positive organic detections from the SWMU 1 open water marine sediment samples while Figure 2-2 provides the spatial locations of the samples.

A total of seven semivolatiles: phenol, 2-nitroaniline (o-nitroaniline), 2,4-dinitrotoluene, 4-chlorophenylphenyl ether, 4-nitroaniline (p-nitroaniline), di-n-butylphthalate, and di-n-octylphthalate were detected in the sediment as shown in Table 3-12. The maximum detection of all seven constituents, with the exception of di-n-butylphthalate, was found from sample location

01OW5. This location is in an inlet just south of the US Customs Pier, as presented on Figure 2-2. No SVOCs were detected in the additional samples obtained from this area during the Additional Data Collection Effort.

A total of eight low level PAHs: phenanthrene, fluoranthene, pyrene, benzo(a)anthracene, chrysene, benzo(k)fluoranthene, benzo(a)pyrene, benzo(g,h,i)perylene were detected in the sediment as shown in Table 3-12. More than half of the maximum detections of the constituents listed above were found in sample 01OWSD03. The remaining maximum detections were found in samples 01OWSD08 and 01OWSD01.

One chlorinated herbicide (2,4,5-T) was detected during this investigation of the Open Water Marine sediment at SWMU 1 from only one of the nine samples locations as presented in Table 3-12. The location of this low level detection of 2.2J ug/kg of 2,4,5-T was found in sample 01OWSD01. This sample was collected from sample location 1OW01, which was the southern most Open Water Marine sediment sample collected at SWMU 1 as presented on Figure 2-2.

The TOC in the thirteen sediment samples collected from the Estuarine Wetland System at SWMU 1 ranged from 14,000 mg/kg in sample 01OWSD08, to 110,000 mg/kg in sample 01OWSD10.

There were no detections listed for the remaining analysis requested including: pesticides/PCBs, op-pesticides, and dioxins/furans.

A total of seventeen different total metals were detected in the sediment samples as shown in Table 3-13, including antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, silver, thallium, tin, vanadium, and zinc. All of the inorganics except for beryllium, cadmium, mercury, and thallium were detected in all of the SWMU 1 open water marine sediment samples. Cadmium and mercury were detected in all of the samples except for 01OWSD04 while cadmium and thallium were only detected from two of the sample locations. All of the maximum detections of the metals were found in sample 01OWSD03 that was collected from sample location 1OW03. The remaining maximum detections were found in samples 01OWSD04, 01OWSD05D, 01OWSD08 and 01OWSD09.

One sample was obtained from each location during the July 2003 investigation to determine the grain size from the SWMU 1 open water marine. The makeup of the sediments in this environment consists of 39.1% fines, 60% sands, and 0.9% gravel on average for the nine samples. The results from each sample along with the ranges of the soil classification by percent of total sample are presented in Table 3-11. Appendix C presents the entire grain size data package.

3.2.3 Surface and Subsurface Soil

A total of eleven surface soil samples along with two duplicate samples were collected from the upland habitat during the Additional Data Collection Effort for SWMU 1 as presented in Table 2-5a. Of the analysis requested, positive detections of the three pesticides analyzed for along with the positive detections of the dioxin/furans and select metals are discussed below.

All three pesticides (4,4'-DDT, 4,4'-DDE, and 4,4'-DDD) were detected in all but one of the samples as shown in Table 3-25. The majority of the detections were low level. 4,4'-DDT was detected in eleven of the twelve samples ranging from 1.4J to 43,000J ug/kg. 4,4'-DDE was detected in eleven of the twelve samples ranging from 5.5 to 28,000 ug/kg. 4,4'-DDD was detected in eleven of the twelve samples ranging from 0.9J to 13,000 ug/kg. The maximum

concentrations were obtained from sample 1SS16 located near the center of the area were the sampling occurred in the southwest portion of the site as shown on Figure 2-7.

Three of the seven samples had positive detections of the majority of dioxin/furans. The remaining samples had limited detections of the dioxin furans as shown in Table 3-25. The maximum concentrations were from either sample 1SS13 or 1SS10 located in the southwest portion of the site.

All eight metals analyzed for were detected in all of the samples as shown in Table 3-26 with the maximum detections coming from samples 11SS13 or 1SS10 located in the southwest portion of the site.

A total of three subsurface soil samples were collected from the upland habitat during the Additional Data Collection Effort for SWMU1 as presented in Table 2-5a and analyzed for three pesticides (4,4'-DDT, 4,4'-DDE, and 4,4'-DDD). All three pesticides were detected in two of the three samples with the maximum concentrations coming from sample 1SB16-01 as shown in Table 3-27. It should be noted that this is the subsurface soil beneath the surface soil sample collected with the maximum detections. The maximum detections in the subsurface soil samples were two orders of magnitude below those obtained in the surface soils.

3.3 SWMU 2

Surface water and sediment samples were collected from the Estuarine Wetland System and sediment samples were obtained from the Open Water Marine environments during the Additional Data Collection Investigation for SWMU 2. Surface and subsurface soil samples were collected from the upland habitat and sediment samples were collected from the Estuarine Wetland System during the Additional Data Collection Effort for SWMU 2. The surface water, sediment, and surface and subsurface soil sample results for this site are discussed in the subsections that follow.

3.3.1 Surface Water

A total of nine surface water samples along with one duplicate sample were collected from the Open Water Marine environment during the Additional Data Collection Investigation for SWMU 2 as presented in Table 2-6. Of the analysis requested, only positive detections of VOCs, one SVOC, and one pesticide were present and are discussed below.

A total of five volatiles: acetone, 2-butanone (methyl ethyl ketone), benzene, toluene, and styrene were detected at low levels in the surface water samples collected as shown in Table 3-15. Acetone had the highest frequency of VOC detections in three of the nine sample locations and ranged from 2.5J to 3J ug/L and is most likely a laboratory artifact. Toluene was detected in two of the nine samples ranging from 0.19J to 0.22J ug/L. The remaining three VOCs (2-butanone, benzene, and styrene) were detected at low levels from one sample location. As presented under the location of maximum detection column, three of the five maximum detections were found in sample 02OWSW01 in the southern most Open Water Marine surface water sample collected at SWMU 2 as presented on Figure 2-3. The remaining maximum detections were found in samples 02OWSW07 and 02OWSW08.

One semivolatile (butylbenzylphthalate) was detected in the Open Water marine surface water as shown in Table 3-15. The only detection of this constituent was found in sample 02OWSW05D, with a concentration of 0.88J micrograms per liter (µg/L). This sample was located at sample

location 2OW05, which is the northern most Open Water Marine surface water sample collected at SWMU 2 as presented on Figure 2-3.

One pesticide (beta-BHC) was detected at a low level concentration of 0.0072J ug/L in the Open Water Marine surface water as shown in Table 3-15. The maximum detection of this constituent was found in sample 02OWSW02 as presented on Figure 2-3.

There were no detections listed for the remaining analysis requested including: low level PAHs, PCBs, op-pesticides, chlorinated herbicides, and dioxins/furans.

A total of fourteen different metals were detected in the total fraction of the surface water samples collected as shown in Table 3-16, including antimony, arsenic, barium, chromium, cobalt, copper, lead, nickel, selenium, silver, thallium, tin, vanadium, and zinc. Ten of the fourteen metals detected were from all of the sampling locations. Thallium and silver were only detected in four and two of the sample locations respectively.

A total of thirteen different metals were detected in the dissolved fraction of the surface water samples collected as shown in Table 3-17, including antimony, arsenic, barium, chromium, cobalt, copper, lead, nickel, selenium, thallium, tin, vanadium, and zinc. Silver is the only metal that was detected in the total fraction that was not detected in the dissolved fraction. The same trend of number of positive detections identified in the total fraction follows suit with the dissolved fraction.

3.3.2 Sediment

The sediment results from the Estuarine Wetland System and Open Water Marine environments for SWMU 2 are discussed separately by environment in the subsections that follow.

3.3.2.1 Estuarine Wetland System

A total of nine sediment samples along with one duplicate sample were collected from the Estuarine Wetland System during the Additional Data Collection Investigation for SWMU 2 as presented in Table 2-6. A total of ten sediment samples along with one duplicate sample were collected from the Estuarine Wetland System during the Additional Data Collection Effort for SWMU 2 as presented in Table 2-6a. Of the analysis requested, VOCs, SVOCs, low-level PAHs, pesticides and metals were present. The TOC and grain size values for these samples are also provided in this section. Figure 2-3 provides the spatial reference of the samples obtained from the SWMU 1 estuarine wetland system.

A total of seven volatiles: acetone, carbon disulfide, methylene chloride, 2-butanone, isobutanol, benzene, and chlorobenzene were detected in the sediment samples collected as shown in Table 3-18. Isobutanol, benzene and chlorobenzene were only detected in one of the nine sample locations at concentrations of 97J, 2.1J and 3.9J ug/kg, respectively. The data validator due to continuing calibrations contained compounds with RRFs less than 0.050 rejected all non-detections of isobutanol. Methylene chloride and 2-butanone were each detected in four of the nine sample locations at low levels. Carbon disulfide detected in five of the nine sample locations at concentrations ranging from 0.38J to 3.2J ug/kg. Acetone, a common laboratory contaminant was detected in all but two of the locations ranging in concentration from 5.1J to 110J. More than half of the maximum detections of the constituents listed above were found in sample 02EWSSD03, whereas the remaining maximum detections were found in samples 02EWSSD06 and 02EWSSD09, respectively.

A total of eight semivolatiles: pentachlorophenol, di-n-butylphthalate, n-nitrosomethylethylamine, methyl methanesulfonate, n-nitrosomorpholine, 5-nitro-o-toluidine, p-(dimethylamino)azobenzene, and 2-acetylaminofluroene were detected in the sediment samples as shown in Table 3-18. Seven of these constituents were only detected in one sample while the detection of di-n-butylphthalate was from two different sample locations. Seventy five percent of the SVOC detections were from sample 2EWSSD09. The remaining maximum detections were found in samples 02EWSSD02 and 02EWSSD05, respectively.

A total of five low level PAHs: fluoranthene, pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, and benzo(a)pyrene were detected in the sediment as shown in Table 3-18. Four of these PAHs were only detected in one sample. The detection of benzo(k)flouranthene was identified in four of the nine sample locations at concentrations ranging from 2J to 7.8J. Three of the five maximum detections were found in sample 02EWSSD06 as presented on Figure 2-3. The remaining maximum detections were found in samples 02EWSSD01 and 02EWSSD02.

One pesticide (4,4'-DDE) was detected at low levels in the sediment samples collected as shown in Table 3-18, with a concentration of 1.3J µg/kg found in sample 02EWSSD02.

The TOC in the nine sediment samples collected from the Estuarine Wetland System at SWMU 2 ranged from 26,000 mg/kg in sample 02EWSSD08, to 93,000 mg/kg in sample 02EWSSD06.

There were no detections listed for the remaining analysis requested including: PCBs, op-pesticides, chlorinated herbicides, and dioxins/furans.

A total of seventeen different metals were detected in the sediment samples collected as shown in Table 3-19, including antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, silver, thallium, tin, vanadium, and zinc. Fourteen of the seventeen metals detected were from each sample location at SWMU 2. Silver was detected in all but two of the sampling locations. Cadmium and thallium were the least frequently detected from only three of the nine sample locations. The majority of the maximum detections were from sample 02EWSSD01 in the southern most sediment sample collected in the Estuarine Wetland System at SWMU 2 as presented on Figure 2-3. The remaining maximum detections were found in samples 02EWSSD02, 02EWSSD03, and 02EWSSD05. Copper, lead and mercury were detected in all the samples from the Additional Data Collection Effort with the maximum concentrations for these three metals coming from sample 2EWSD12 located near the middle of the sediment sampling locations as shown on Figure 2-6. The maximum concentrations for these constituents were 280J mg/kg for copper, 170 mg/kg for lead, and 1.3 mg/kg for mercury.

One sample was obtained from each location during the Additional Data Collection Effort for AVS/SEM analysis. AVS was detected in all but one sample ranging in concentrations from 0.4 to 74.9 µmole/g as shown in Table 3-19. SEM metals (cadmium, copper, nickel, lead, and zinc) were detected in all of the samples. The total SEM results ranged from 0.0652 to 6.3099 µmole/g as shown in Table 3-19.

One sample was obtained from each location to determine the grain size from the SWMU 2 estuarine wetland system. The makeup of the sediments in this environment consists of 78.6% fines and 21.4% sands on average for the nine samples. These averages are similar to those obtained from the estuarine wetland background samples as shown in Table 3-20. The results from each sample along with the ranges of the soil classification by percent of total sample are presented in Table 3-20. Appendix C presents the entire grain size data package.

3.3.2.2 Open Water Marine

A total of nine sediment Open Water Marine samples along with one duplicate sample were collected during the Additional Data Collection Investigation for SWMU 2 as presented in Table 2-6. Of the analysis requested, VOCs, SVOCs, low level PAHs, and pesticides were present. The TOC and grain size values for these samples are also provided in this section. Figure 2-3 provides the spatial reference of the samples obtained from the SWMU 2 open water marine system.

A total of five volatiles: acetone, carbon disulfide, methylene chloride, 2-butanone, and isobutanol, were detected in the sediment samples collected as shown in Table 3-21. Methylene chloride and isobutanol were only detected in two of the nine sample locations at concentrations ranging from 0.82J to 1.2J ug/kg and 120J to 170J ug/kg, respectively. The data validator due to continuing calibrations contained compounds with RRFs less than 0.050 rejected all non-detections of isobutanol. 2-butanone was detected in seven of the nine sample locations at low levels. Carbon disulfide detected in all of the sample locations at concentrations ranging from 1.6J to 6.6J ug/kg. Acetone, a common laboratory contaminant was detected in all but two of the locations ranging in concentration from 31J to 91J.

Three semivolatiles: phenol, n-nitrosodi-n-propylamine, and di-n-butylphthalate were detected in the sediment at a low frequency of detection as shown in Table 3-21. Phenol and n-nitrosodi-n-propylamine were only detected in one of the nine sample locations at concentrations of 320J ug/kg. Di-n-butylphthalate was detected in four of the nine sample locations at concentrations ranging from 460J to 610J ug/kg.

A total of twelve low level PAHs: acenaphthylene, phenanthrene, anthracene, fluoranthene, pyrene, benzo(a)anthracene, chrysene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, indeno(1,2,3-cd)pyrene, and benzo(g,h,i)perylene were detected in the sediment samples as shown in Table 3-17. The majority of these detections were from the four northernmost sample locations from SWMU 2. Low-level PAHs were identified in five of the nine sample locations. The majority of the maximum detections were from sample 02OWSD05. This sample is located at sample location 2OW05, which is the northern most sediment sample collected in the Open Water Marine environment at SWMU 2, as presented on Figure 2-3.

One pesticide (methoxychlor) was detected in one sample during this investigation in the Open Water Marine sediment at SWMU 2 as presented in Table 3-21. Methoxychlor was detected at a concentration of 5.2J ug/kg from sample 02OWSD07.

The TOC in the nine sediment samples collected from the Open Water Marine environment at SWMU 2 ranged from 18,000 mg/kg in sample 02OWSD08, to 55,000 mg/kg in sample 02OWSD03.

There were no detections listed for the remaining analysis requested including: PCBs, op-pesticides, chlorinated herbicides, and dioxins/furans.

A total of seventeen different total metals were detected in the sediment samples collected as shown in Table 3-22, including antimony, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, silver, thallium, tin, vanadium, and zinc. Twelve of the metals were detected in all nine of the sample locations. Beryllium and mercury were detected in eight of the nine sample locations while silver was detected in seven of the nine sample locations. Thallium was only detected in two of the locations and cadmium was detected in only one sample location at a concentration of 0.091J mg/kg.

One sample was obtained from each location to determine the grain size from the SWMU 2 open water marine environment. The makeup of the sediments in this environment consists of 67.4% fines and 32.6% sands on average for the nine samples. The results from each sample along with the ranges of the soil classification by percent of total sample are presented in Table 3-23. Appendix C presents the entire grain size data package.

3.3.3 Surface and Subsurface Soil

A total of twelve surface soil samples along with three duplicate samples were collected from the upland habitat during the Additional Data Collection Effort for SWMU 2 as presented in Table 2-6a. Of the analysis requested, positive detections of the dioxin/furans and the five metals that were analyzed for are discussed below.

One of the five samples had positive detections of the majority of dioxin/furans detected. The remaining samples had limited detections of the dioxin furans as shown in Table 3-28. The maximum concentrations were from either sample 2SS01 or 2SS03 located near the center of the site as shown on Figure 2-8.

All five metals analyzed for were detected in all of the samples as shown in Table 3-29.

A total of five subsurface soil samples were collected from the upland habitat during the Additional Data Collection Effort for SWMU 2 as presented in Table 2-6a and analyzed for five metals (antimony, copper, lead, mercury, and zinc). All five metals were detected in all of the samples as shown in Table 3-30.

3.4 QA/QC Samples

A total of seven QA/QC samples including four trip blanks (1TB01, 1TB02, 2TB01, and 2TB02), one field blank (2003FB01), and two equipment rinsate samples (1ER01, and 1ER02) were collected during the Additional Data Collection Investigation for SWMUs 1 and 2 as presented in Table 2-7. A total of five QA/QC samples including one field blank (2004FB01) and four equipment rinsate samples (2004ER01 through 2004ER04) were collected during the Additional Data Collection Effort for SWMUs 1 and 2 as presented on Table 2-7a.

A total of two VOCs were detected in the trip blank samples that were collected, including toluene and styrene as presented in Table 3-24. The range of detections for toluene was 0.19J (2TB02), to 0.23J µg/L (1TB01). The range of detections for styrene was 0.17J µg/L (1TB01, 2TB01, and 2TB02), to 0.23J µg/L (1TB02). No detections of VOCs were identified in the field blank or equipment rinsate samples.

Two semi volatile constituents were positively detected (di-n-octylphthalate and benzo(a)pyrene) in the equipment rinsate samples 1ER01 and 1ER02 as presented in Table 3-24. The positive detections mentioned above were only found in sample 1ER01 that was collected by pouring lab grade deionized water over a stainless steel spoon. No SVOCs were detected in field blank sample. No SVOCs were detected in the QA/QC samples during the Additional Data Collection Effort.

There were no detections in the remaining organic analysis requested for the field blank and equipment rinsate samples (low level PAHs, pesticides/PCBs, op-pesticides, chlorinated herbicides, and dioxins/furans).

Of the inorganic (total) analysis requested during the July 2003 investigation, a total of seven constituents were positively detected, including barium, chromium, copper, lead, nickel, tin, and zinc, as presented in Table 3-24. The constituents mentioned above are most likely related to the lab grade deionized water that was used to collect the equipment rinsate and field blank samples. All but one of the metals was detected in the field blank sample.

Of the metals (total) analysis requested during the October 2004 investigation, all six of them were detected in at least on of the QA/QC samples as presented in Table 3-24. Four of the metals (copper, lead, nickel, and tin) were detected in the field blank at equal or similar concentrations in the equipment rinsate samples.

4.0 SCREENING-LEVEL ECOLOGICAL RISK ASSESSMENT AND STEP 3A OF THE BASELINE ECOLOGICAL RISK ASSESSMENT

This section presents a revision to a screening-level ecological risk assessment (ERA) and Step 3a of the baseline ERA for SWMUs 1 (Army Cremator Disposal Site) and 2 (Langly Drive Disposal Area), located at NSRR, Ceiba, Puerto Rico previously presented in the Draft Additional Data Collection Report and Screening-Level Ecological Risk Assessment and Step 3a of the Baseline Ecological Risk Assessment for SWMUs 1 and 2, Naval Station Roosevelt Roads, Ceiba, Puerto Rico (Baker, 2004a). This revised ERA integrates analytical data generated during an additional data collection effort conducted in October 2004 (Baker, 2004b) with analytical data used in the draft ERA. Additional sampling was conducted at each SWMU to address data gaps and uncertainties identified in the draft ERA. The revised ERA also incorporates revisions outlined in the Navy's Response (dated July 23, 2004) to USEPA Comments (dated April 9, 2004) on the draft Report.

The revised ERA was conducted in accordance with the Chief of Naval Operations (CNO) document entitled Navy Policy for Conducting Ecological Risk Assessments (CNO, 1999).

The Navy ERA process (see Figure 4-1) consists of eight steps organized into three tiers and represents a clarification and interpretation of the eight-step ERA process outlined in the USEPA ERA guidance for the Superfund program (USEPA, 1997). Tier 1 of the Navy ERA process represents the screening-level ERA:

- Screening-level problem formulation and ecological effects evaluation (Step 1).
- Screening-level exposure estimate and risk calculation (Step 2).

Under Navy policy, if the results of Steps 1 and 2 (Tier 1 screening-level ERA) indicate that, based on a set of conservative exposure assumptions, there are chemicals present in environmental media that may present a risk to receptor species/communities, the ERA process proceeds to the baseline ERA. According to Superfund guidance (USEPA, 1997), Step 3 represents the problem formulation phase of the baseline ERA. Under Navy policy, the baseline ERA is defined as Tier 2, and the first activity under Tier 2 is Step 3a. In Step 3a, the conservative exposure assumptions applied in Tier 1 are refined and risk estimates are recalculated using the same conceptual site model. The evaluation of risks in Step 3a may also include consideration of background data, chemical bioavailability, and the frequency of detection. If the re-evaluation of the conservative exposure assumptions does not support an acceptable risk determination, the site continues through the baseline ERA process (Step 3b baseline ERA problem formulation).

4.1 Environmental Setting

The sections that follow provide a brief description of SWMUs 1 and 2. The habitats occurring within and contiguous to each SWMU are also described, as well as the biota that may be present. The description of habitats and biota relies primarily on literature-based information for Puerto Rico and NSRR. This information is supplemented by observations recorded during a habitat characterization conducted within the vegetative units at and contiguous to SWMUs 1 and 2 from May 15 to May 19, 2000. The habitat characterization report, prepared by Geo-Marine, Inc (Plano, Texas), is included as Appendix F.

4.1.1 Site History

NSRR occupies over 8,600 acres on the East Coast of Puerto Rico, along Vieques Passage (see Figure 1-1), with Vieques Island lying approximately ten miles to the east. NSRR was commissioned in 1943 as a Naval Operations Base and re-designated a Naval Station in 1957. The current primary mission of NSRR is limited. NSRR is currently preparing for operational closure.

SWMU 1, located east of the Navy Lodge, encompasses an area of roughly 116 acres of land (see Figure 1-2). The SWMU is bounded to the north by Kearsage Road leading to the U.S. Customs Pier, Ensenada Honda to the east, estuarine wetlands to the south, and the Navy Lodge and Bowling Alley to the west. Based on previous reports, the Army Cremator Disposal Site operated from the early 1940s until the early 1960s and was the main station landfill during this period. The waste material was disposed of by piling, burning, and compacting (A.T. Kearney, Inc., 1988). According to the Naval Energy and Environmental Support Activity (NEESA), an estimated 100,000 tons of waste including scrap metal, inert ordnance, batteries, tires, appliances, cars, cables, dry cleaning solvent cans, paint cans, gas cylinders, construction debris, dead animals, and residential waste was disposed of at this SWMU (NEESA, 1984). No reliable information exists regarding the amounts of material present in the disposal area that could be hazardous; however, in 1984, an Initial Assessment Study (IAS) team estimated that as much as 1,000 tons of hazardous material could be present (NEESA, 1984).

SWMU 2 is located along Langley Drive approximately 1,000 feet northeast of the Navy Commissary (see Figure 1-2). The site extends from Langley Drive to an estuarine wetland system bordering the Ensenada Honda and has an estimated length of 1,300 feet in a northeast-southwest direction. This site operated as a landfill from approximately 1939 to 1959 and is documented as having been used for the disposal of both hazardous and non-hazardous wastes (NEESA, 1984). Debris noted during the IAS included partially buried metal and concrete objects, old fuel lines, flexible metal hoses, sample containers containing pellets, steel cables, hardened tar, rubble, and ten to fifteen 55-gallon drums that were corroded. The drum contents, generally consisting of a whitish solid with a green outer crust, were exposed (NEESA, 1984). The IAS team estimated the volume of disposed waste to be approximately 1,700 cubic yards, of which approximately 20,000 pounds could be hazardous.

4.1.2 Terrestrial and Marine Habitats

A description of terrestrial and marine habitats within and contiguous to SWMUs 1 and 2 is provided in the sections that follow. As discussed in Section 4.1, the description of vegetative habitat units is based primarily on a habitat characterization conducted by Geo-Marine, Inc. from May 15 to May 19, 2000.

4.1.2.1 Terrestrial Habitats

The upland habitat bounded by NSRR is classified as subtropical dry forest (Ewel and Witmore, 1973). Similar to other forested areas of Puerto Rico, this region was previously clear-cut in the early part of the century, primarily for pastureland (Geo-Marine, Inc., 1998). After acquisition by the Navy, a secondary growth of thick scrub, dominated by lead tree (*Leucaena spp.*), Christmas tree (*Randia aculeate*), sweet acacia (*Acacia famesiana*), and Australian corkwood (*Sesbania grandiflora*) grew in the previously grazed sections (Geo-Marine, Inc., 1998). Secondary growth communities (upland coastal forest communities and coastal scrub forest communities) exist today throughout the station's undeveloped upland. The upland vegetative communities at SWMU 1 are classified as coastal scrub forest and upland coastal forest

communities, while the upland vegetative communities at SWMU 2 are classified as upland coastal forest communities (see Figure 4-2).

The upland coastal forest community at SWMU 1 consists of a shrub layer (dominated by lead tree, almacigo [*Bursa simaruba*], and Christmas tree), a tree layer (oxhorn bucida [*Bucida buceras*], basket wiss [*Trichostigma octandrum*], and common guayaba [*Psidium guajava*]), and herbaceous areas dominated by *Panicum maximum* (no common name) (Geo-Marine, Inc., 2000). The coastal scrub forest community at SWMU 1 is limited to two stratums (shrub and herbaceous). Lead tree and *Panicum maximum* dominate the shrub and herbaceous stratums, respectively. A tree layer is not present within the coastal scrub forest community (Geo-Marine, Inc., 2000).

Identical to the coastal shrub forest community at SWMU 1, the upland coastal forest community at SWMU 2 is limited to a shrub layer and herbaceous layer. The shrub layer is dominated by lead tree, sweet acacia, and bottle wiss, while the herbaceous layer is dominated by *Sporobolis indicus* (no common name), *Panicum maximum*, cattle tongue (*Pluchea carolinenis*), and marsh mallow (*Waltheria indicia*).

Cobana negra (*Stahlia monosperma*), a federally threatened tree species, is known to occur between the boundary of black mangrove communities and coastal upland forest communities. This species is also known to occur in coastal forests of southeastern Puerto Rico (Little and Wadsworth, 1964). A single individual has been reported at NSRR. Although the location of the sighting was not documented, NSRR personnel believe the tree is located within the coastal forest community behind the former Navy Exchange store, northwest of SWMU 1 and Langley Drive. Cobana negra was not observed at SWMUs 1 and 2 during the May 15 to May 19, 2000 habitat characterization.

4.1.2.2 Marine Habitats

The marine environment surrounding NSRR includes mudflats, mangroves (black mangrove [*Avicenia germinans*] and red mangrove [*Rhizophora mangle*] communities), and seagrass beds (turtle grass [*Thalassia testudium*] and manatee grass [*Syringodium filliforme*]). The total area of mudflats, mangroves, and sea grass beds in the offshore environment is approximately 161 acres, 2,700 acres, and 1,900 acres, respectively (Geo-Marine, Inc., 1998). Coral reefs are also located in the offshore marine environment. Seagrass beds represent grazing areas for the green sea turtle (*Chelonia mydas*) and the West Indian manatee (*Trichechas manatus*). The green sea turtle is a federally threatened species, and the West Indian manatee is a federally endangered species. Both species have been reported from the marine environment surrounding NSRR.

The nearest open water habitat downgradient from SWMUs 1 and 2 is the Ensenada Honda. As evidenced by Figure 4-2, sea grass beds are prevalent throughout much of the Ensenada Honda, including the area immediately downgradient from SWMUs 1 and 2. A map showing the spatial relationship of each SWMU to the Ensenada Honda is provided as Figure 4-3. Included on this figure are wetland units identified by the Cowardin Wetland Classification System (Cowardin et al., 1979 [see Figure 4-4]). The wetlands depicted on Figure 4-3 were delineated by Geo-Marine, Inc. in December 1999 from 1993 color infrared and 1998 true color aerial photography. Twenty percent of the wetlands delineated by aerial photography were field checked to verify the accuracy of the delineations. Field verification was based on the 1987 Corps of Engineers wetland delineation manual (United States Army Corps of Engineers [USACE], 1987). As evidenced by Figures 4-2 and 4-3, there are no freshwater wetland units within or contiguous to SWMUs 1 and 2. However, both SWMUs infringe upon an estuarine wetland system bordering the Ensenada Honda.

The estuarine wetland system includes downgradient from each SWMU include both red and black mangrove communities, with red mangroves occurring immediately adjacent to the Ensenada Honda and black mangroves located between the red mangrove and coastal upland forest communities. The red mangrove community downgradient from each SWMU is sparsely vegetated (approximately 25 percent cover), with either large pools of water (SWMU 1) or mud flats (SWMU 2) present. Specific wetland units located within the estuarine wetland system downgradient from SWMUs 1 and 2 include the following Cowardin classifications: E2SS3 (Estuarine, Intertidal, Scrub-Scrub, Broad-Leaved Evergreen), E2US2 (Estuarine, Intertidal, Unconsolidated Shore, Sand), E2US3 (Estuarine, Intertidal, Unconsolidated Shore, Mud), and E2US4 (Estuarine, Intertidal, Unconsolidated Shore, Organic).

4.1.3 Biota

A description of the biota occurring within Puerto Rico and the landmass encompassed by NSRR is provided in the sections that follow. This description is supplemented by information contained within the habitat characterization report included as Appendix F.

4.1.3.1 Mammals

A total of 22 terrestrial mammal species are known historically from Puerto Rico; however, all mammals except bats (13 species) have been extirpated (United States Geological Society [USGS], 1999). None of the bats found on Puerto Rico are exclusive to the island. The West Indian manatee is known to occur in the marine environment surrounding NSRR. As depicted on Figure 4-2, sea grass beds are located throughout much of the shallow water habitat within the Ensenada Honda and the marine environment surrounding NSRR. Their locations represent potential feeding habitat for this marine mammal.

Several mammals have been introduced into Puerto Rico, including the black rat (*Rattus rattus*), Norway rat (*Rattus norvegicus*), and mongoose (*Herpestes javanicus*). These nonindigenous mammals have been implicated in the decline of native bird and reptile populations (USGS, 1999 and United States Fish and Wildlife Service [USFWS], 1996a).

4.1.3.2 Birds

A total of 239 bird species are native to Puerto Rico (Raffaele, 1989). This total includes breeding permanent residents and non-breeding migrants. In addition, many nonindigenous bird species have been introduced to Puerto Rico, including the shiny cowbird (*Molothrus bonariensis*) and several parrot species, such as the budgerigar (*Melopsittacus undulates*), orange-fronted parrot (*Aratinga canicularis*), and monk parrot (*Myiopsitta monaqchus*). Of the 239 species native to Puerto Rico, 12 are endemic to the island (Raffaele, 1989).

Numerous native and migratory bird species have been reported at NSRR (Geo-Marine, Inc., 1998). A list of bird species reported at NSRR or having the potential to occur is provided in Table 4-1. The list, compiled from literature-based information pre-dating 1990, includes the great blue heron (*Ardea herodias*), snowy egret (*Egretta thula*), little blue heron (*Florida caerulea*), black-crowned night heron (*Nycticorax nycticorax*), belted kingfisher (*Ceryle alcyon*), spotted sandpiper (*Actitis macularia*), greater yellowlegs (*Tringa melanoleauca*), black-bellied plover (*Squatarola squatarola*), clapper rail (*Rallus longirostris*), Royal tern (*Thalasseus maximus*), sandwich tern (*Thalasseus sandvicensis*), least tern (*Stema albifrons*), yellow warbler (*Dendroica petechia*), palm warbler (*Dendroica palmarum*), prairie warbler (*Dendroica discolar*), magnolia warbler (*Dendrocia magnolia*), mourning dove (*Zenaida macroura*), red-legged thrush

(*Mimocichla plumbea*), common nighthawk (*Chordeiles minor*), and red-tailed hawk (*Buteo jamaicensis*). Endemic species reported from NSRR include the Puerto Rican lizard cuckoo (*Saurothera vieilloti*), Puerto Rican flycatcher (*Myiarchus antillarum*), Puerto Rican woodpecker (*Malanerpes portoricensis*), Puerto Rican emerald (*Chlorostilbon maugaeus*), and yellow-shouldered blackbird (*Agelaius xanthomus*).

The yellow-shouldered blackbird is a federally endangered species. One of the principal reasons for the status of this species is attributed to parasitism by the nonindigenous shiny cowbird, which lays its eggs in blackbird nests and sometimes punctures the host's eggs (USFWS, 1983). Other factors contributing to the status of this species include nest predation by the introduced black rat, Norway rat, and mongoose, as well as habitat modification and destruction (USFWS 1996a). The entire land area of NSRR was declared critical habitat for the yellow-shouldered blackbird in 1976; however, a 1980 agreement with the USFWS exempted certain areas from this categorization (Geo-Marine, Inc., 1998). SWMUs 1 and 2 are located within the critical habitat designation. A study conducted by the Naval Facilities Engineering Command (NFEC, 1996) reported that the mangrove forests surrounding NSRR, including the mangrove communities downgradient from SWMUs 1 and 2, should be considered the most important nesting habitats for the yellow-shouldered blackbird. A survey conducted in July 2002 by the Puerto Rico Department of Natural Resources (PRDNR, 2002) reported fifteen yellow-shouldered blackbirds (including five juveniles) at NSRR. At the time of the survey, the birds were using the structures at the NSRR airport for resting cover. Although nesting pairs were not observed (the survey was not conducted during the breeding season), the airport structures contained several inactive nests. The inactive nests and juvenile birds indicate that a small breeding population is present at NSRR. Yellow-shouldered blackbirds were not observed within the coastal forest and mangrove communities at SWMUs 1 and 2 during the May 15 to May 19, 2000 habitat characterization (Geo-Marine, 2000). Other federally listed bird species that have been reported at NSRR or have the potential to occur are the brown pelican (*Pelecanus occidentalis occidentalis*), roseate tern (*Sterna dougallii dougallii*), and the piping plover (*Charadrius melodus*) (Geo-Marine, Inc., 1998). Given their habitat preferences (aquatic) and prey preferences (fish or aquatic invertebrates), these species have the potential to occur in the estuarine wetland system (piping plover) or open water habitat (brown pelican or roseate tern) downgradient from SWMUs 1 and 2.

Specific bird species observed within the vegetative communities at SWMUs 1 and 2 are listed in the habitat characterization report (see Appendix F). Species common to both SWMUs include the red-tailed hawk, yellow warbler, Puerto Rican woodpecker, loggerhead kingbird (*Tyannus caudifasciatus*), zenaida dove, and pearly-eyed thrasher (*Margarops fuscatus*). An active Wilson's plover (*Charadrius wilsonia*) nest was observed within the black mangrove community downgradient from SWMU 1.

4.1.3.3 Reptiles and Amphibians

A total of 23 amphibians and 47 reptiles are known from Puerto Rico and the adjacent waters (USGS, 1999). Fifteen of the amphibians and 29 of the reptiles are endemic, while four amphibian species and three reptilian species have been introduced (USGS, 1999). Puerto Rico's native amphibian species include 16 species of tiny frogs commonly called coquis. On the coastal lowlands, almost all coqui species are arboreal. The only amphibians listed under provisions of the Endangered Species Act of 1973 are the Puerto Rican ridge-headed toad (*Peltophryene lemur*) and the golden coqui (*Eleutherodactylus jasperi*). Both species are listed as threatened. Distribution of the golden coqui is restricted to areas of dense bromeliad growth. All specimens to date have been collected from a small semicircular area of a 6-mile radius south of Cayey (approximately 30 miles southwest of NSRR), generally at elevations above 700 meters

(USFWS, 1984). The Puerto Rican ridge-headed toad occurs at low elevations (below 200 meters) where there is exposed limestone or porous, well drained soil offering an abundance of fissures and cavities (USFWS, 1987). A single large population is known to exist from the southwest coast in Guánica Commonwealth Forest, and a small population is believed to survive on the north coast near Quebradillas, Arecibo, Barceloneta, Vega Baja, and Bayamón (USFWS, 1987). It has also been collected on the southeastern coastal plain near Coamo (USFWS, 1987). Given the habitat preferences and locations of known occurrences, these two species are not expected to occur at NSRR.

Puerto Rico's native reptilian species include 31 lizards, 8 snakes, 1 freshwater turtle, and 5 sea turtles (USGS, 1999). Of the five sea turtles, only the green sea turtle (*Chelonia mydas*), hawksbill sea turtle (*Eretmochelys imbricata*), and loggerhead sea turtle (*Dermochelys coriacea*) nest within Puerto Rico. These three sea turtles, as well as the leatherback sea turtle (*Caretta caretta*) and the Puerto Rican boa (*Epicrates inornatus*) represent the reptilian species listed under the provisions of the Endangered Species Act of 1973 (USGS, 1999). Given the presence of sea grass beds within the Ensenada Honda, this surface water body represents potential feeding habitat for the listed sea turtles. The Puerto Rican boa uses a variety of habitats but is most commonly found in karst forest habitats. Given the absence of karst forest habitat and the absence of any known occurrence of this species at or contiguous to SWMUs 1 and 2 (Geo-Marine, Inc., 1998), there is a low probability of occurrence for this species at each site.

Several lizard species were observed within the vegetative units at and contiguous to SWMUs 1 and 2 during the May 15 to May 19, 2000 habitat characterization, including the crested anole (*Anolis cristatellus*), *Anolis stratulus* (no common name), and *Anolis pulchellus* (no common name). Two amphibian species (i.e., frogs) were also observed within the upland coastal forest community at SWMU 2 (*Eleutherodactylus sp.* [no common name] and the white-lipped frog [*Leptodactylus albilabris*]).

4.1.3.4 Fish and Aquatic Invertebrates

A diverse fish and invertebrate community can be found in the marine environment surrounding NSRR. This can be attributed to the varied habitats that include marine and estuarine open water habitat, mud flats, sea grass beds, and mangrove forests. The fish community is represented by stingrays, herrings, groupers, needlefish, mullets, barracudas, jacks, snappers, grunts, snooks, lizardfishes, parrotfishes, gobies, filefishes, wrasses, damselfishes, and butterflyfish (Geo-Marine, Inc., 1998). The benthic invertebrate community includes sponges, corals, anemones, sea cucumbers, sea stars, urchins, and crabs. A list of known species residing within the estuarine wetland system and open water habitat (i.e., Ensenada Honda) downgradient from SWMUs 1 and 2 is not available. A large land crab (*Ucar sp.*) was observed within the black mangrove community downgradient from SWMU 2 during the May 15 to May 19, 2000 habitat characterization. Numerous fiddler crabs (*Uca sp.*) were also observed within the black and red mangrove communities downgradient from both SWMUs during the July 2003 field investigation.

4.2 Sources of Available Analytical Data

Sampling activities at SWMUs 1 and 2 have been conducted under six separate investigations:

- Confirmation Study in 1985/1986 (Round 1) and 1987 (Round 2): groundwater (SWMUs 1 and 2), surface soil (SWMU 2), subsurface soil (SWMU 2), surface water (SWMUs 1 and 2), and sediment (SWMUs 1 and 2);

- Supplemental Investigation in 1993: groundwater (SWMUs 1 and 2), surface soil (SWMU 2), and subsurface soil (SWMUs 1 and 2);

- Relative Risk Ranking in 1995: estuarine wetland sediment (SWMUs 1 and 2);

- RFI field investigation in 1996/1997: groundwater (SWMUs 1 and 2), surface soil (SWMUs 1 and 2), subsurface soil (SWMUs 1 and 2), and estuarine wetland sediment (SWMUs 1 and 2); and

- Additional data collection field investigation in 2003: estuarine wetland surface water and sediment (SWMUs 1 and 2 [Note: surface water was absent from the portion of the estuarine wetland downgradient from SWMU 2 sampled during the additional data collection field investigation]) and open water (Ensenada Honda) surface water and sediment (SWMUs 1 and 2).

- Additional data collection effort in 2004: surface soil (SWMUs 1 and 2), subsurface soil (SWMUs 1 and 2), estuarine wetland sediment (SWMUs 1 and 2), and open water sediment (SWMU1).

The Confirmation Study, Supplemental Investigation, Relative Risk Ranking, and RFI field investigations and associated analytical data were previously presented and discussed in the USEPA-approved Revised Draft RFI for OU 3/5 (Baker, 1999). A summary of the environmental media collected during these four investigations is presented in Tables 4-2 (SWMU 1) and 4-3 (SWMU 2). A description of the additional data collection field investigation and associated analytical data is presented in Sections 2.0 and 3.0, respectively.

The quality of the analytical data obtained during the Confirmation Study is questionable due to the unknown level of laboratory data quality objectives and the apparent lack of independent, third party data validation. Therefore, these data were deemed unacceptable for use in the screening-level ERA.

Analytical data for groundwater samples collected during the Supplemental and RFI field investigations also were excluded from evaluation for the following reasons:

- Groundwater does not represent an exposure point for ecological receptors.

- Although migration with groundwater to downgradient surface water and sediment represents a potential transport pathway at each SWMU, a sufficient number of surface water and sediment samples have been collected from the estuarine wetland system and open water habitat downgradient from SWMUs 1 and 2 to adequately address this transport pathway.

Analytical data for soil samples collected from the surface to a maximum depth of one-foot bgs during the Supplemental field investigation, 1996 and 1997 RFI field investigations, and the 2004 additional data collection field effort were quantitatively evaluated as surface soil in the screening-level ERA. This depth range is the most active biological zone (most soil heterotrophic activity occurs within the surface soil and soil invertebrates occur on the surface or within the oxidized root zone [Suter II, 1995]). As evidenced by Tables 4-2 and 4-3, subsurface soil samples were collected from various depth intervals at each SWMU during the Supplemental field investigation, 1996 and 1997 RFI field investigations, and 2004 additional data collection field effort. Analytical data for subsurface samples collected from the 0.5 to 1.5-, 0.5 to 1.67, and 1.0 to 2.0-foot depth interval also were quantitatively evaluated in the screening-level ERA.

However, analytical data for subsurface samples collected from deeper depth intervals were not evaluated since these depths are not likely to represent a significant exposure point for ecological receptors.

The estuarine wetland sediment samples collected during the relative risk ranking investigations were not quantitatively evaluated in the screening-level ERA. This decision is based on their age (analytical data reflecting current levels of exposure are available from the additional data collection field investigation conducted in July 2003). Furthermore, the locations of sediment samples collected during the additional data collection field investigation are in close proximity to the location of sediment samples collected during the Relative Risk Ranking. Sediment samples collected during the RFI field investigation at SWMUs 1 and 2 were taken from shallow, vegetated swales located within terrestrial habitat upgradient from the estuarine wetland system (Baker, 1999). Because they were collected from terrestrial habitat, not the estuarine wetland system, they are considered surface soil samples and were incorporated into the surface soil data set [Note: the sample identification numbers assigned to these samples were not changed].

Surface water was not collected from the estuarine wetland system and Ensenada Honda downgradient from SWMUs 1 and 2 during the Supplemental Investigation, Relative Risk Ranking, and RFI field investigations. Surface water was collected from the estuarine wetland system downgradient from SWMU 1 and the Ensenada Honda downgradient from SWMUs 1 and 2 during the additional data collection field investigation. Surface water was not collected from the estuarine wetland system downgradient from SWMU 2 due to the absence of standing water at designated sample locations. Surface water data collected during the additional data collection field investigation were quantitatively evaluated in the screening-level ERA.

A listing of the abiotic media qualitatively evaluated by the screening-level ERA (surface soil, subsurface soil, surface water, and sediment) is summarized in Table 4-4. Sample locations are depicted on Figures 4-5 (SWMU 1) and 4-6 (SWMU 2). Analytical data for each sample listed in Table 4-4 are included as Appendix G.

4.3 Screening-Level Problem Formulation

Problem formulation establishes the goals, scope, and focus of the ERA. The products of the screening-level problem formulation are (1) the preliminary conceptual model and (2) the assessment and measurement endpoints. The purpose of the preliminary conceptual model is to describe how ecological receptors may be exposed to chemicals originating from the site. The preliminary conceptual model is developed using information regarding major habitats and ecological receptors, media of concern, and potential contaminant sources in conjunction with an understanding of potential transport pathways, exposure pathways, and exposure routes. The fate, transport, and toxicological properties of the chemicals present at the site are also considered during this process. Assessment and measurement endpoints define the ecological attributes to be protected. They are selected to evaluate those receptors for which complete and potentially significant exposure pathways are likely to exist.

4.3.1 Preliminary Conceptual Model

Exposure, and thus potential for risk, can only occur if each of the following conditions are present (USEPA, 1998):

- A source of contamination must be present.

- Release and transport mechanisms must be available to move the contaminants from the source to an exposure point.

- An exposure point must exist where ecological receptors could contact affected media.

- An exposure route must exist whereby the contaminant can be taken up by ecological receptors.

Figures 4-7 and 4-8 present preliminary conceptual models for SWMUs 1 and 2, respectively. The conceptual models outline potential sources of contaminants, transport pathways, exposure media, potential exposure routes, and receptor groups. Specific components of the preliminary conceptual model (i.e., source areas, transport pathways, and exposure pathways and routes) are discussed in the sections that follow.

4.3.1.1 Source Areas

The disposal areas at SWMUs 1 and 2 represent potential source areas for the release of chemicals to abiotic media (i.e., surface and subsurface soil). Contaminated surface and subsurface soil also represent potential source areas for the release of chemicals to groundwater and/or downgradient surface soil, surface water, and sediment.

4.3.1.2 Transport Pathways

A transport pathway describes the mechanisms whereby chemicals may be transported from a source of contamination to ecologically relevant media. As depicted on Figures 4-7 and 4-8, the primary mechanisms for contaminant transport from potential source areas at SWMUs 1 and 2 are believed to include the following:

- Overland transport of chemicals with surface soil via surface runoff to downgradient surface soil, surface water, and sediment.

- Leaching of chemicals from surface soil and/or subsurface soil by infiltrating precipitation and transport to surface water and sediment with groundwater.

- Uptake by biota from surface soil, surface water, and/or sediment and trophic transfer to upper trophic level receptors.

4.3.1.3 Exposure Pathways and Routes

An exposure pathway links a source of contamination with one or more receptors via exposure to one or more media. Requirements for a complete exposure pathway were presented in Section 4.3.1. As depicted on Figures 4-7 and 4-8, potentially complete and significant exposure pathways exist at each SWMU.

An exposure route describes the specific mechanism(s) by which a receptor is exposed to a chemical present in an environmental medium. The most common exposure routes are dermal contact, direct uptake, ingestion, and inhalation. Terrestrial plants may be exposed to chemicals present in surface soil directly through their root surfaces during water and nutrient uptake. Unrooted, floating aquatic plants, rooted submerged aquatic plants, and algae may be exposed to chemicals directly from the water or (for rooted plants) from sediments. Terrestrial and aquatic invertebrates may be exposed to chemicals in surface soil, surface water, and sediment, through dermal adsorption and ingestion. Much of the toxicological data available for terrestrial and

aquatic invertebrates are based upon *in situ* studies that represent both pathways. Therefore, both pathways were considered together in this screening-level ERA. Invertebrates also represent a link between surface soil, surface water, and/or sediment and upper trophic level receptors through food web transfer. As such, they were included as prey items, where appropriate, for upper trophic level dietary exposures.

Birds and mammals may be exposed to chemicals through: (1) the inhalation of gaseous chemicals or chemicals adhered to particulate matter; (2) the incidental ingestion of contaminated abiotic media (e.g., soil or sediment) during feeding or cleaning activities; (3) the ingestion of contaminated water; (4) the ingestion of contaminated plant and/or animal tissues for chemicals that have entered food webs; and/or (5) dermal contact with contaminated abiotic media. These exposure routes, where applicable, are depicted on Figures 4-7 and 4-8. Their relative importance depends in part on the chemical being evaluated. For chemicals having the potential to bioaccumulate (e.g., PCBs and organochlorine pesticides), the greatest exposure to wildlife is likely to be from the ingestion of prey. For chemicals having a limited potential to bioaccumulate (e.g., aluminum), the exposure of wildlife to chemicals is likely to be greatest through the direct ingestion of abiotic media, such as surface soil.

Direct ingestion of drinking water is only considered if the salinity of a drinking water source is less than 15 parts per thousand (ppt), the approximate toxic threshold for wildlife receptors (Humphreys, 1988). As evidenced by Figures 4-2 and 4-3, there are no fresh surface water bodies within or contiguous to SWMUs 1 and 2. The only potential drinking water sources contiguous to SWMUs 1 and 2 are the estuarine wetland system and the Ensenada Honda. Salinity measurements were taken at each estuarine wetland and Ensenada Honda surface water sampling location during the July 2003 field investigation. The salinity of surface water within the estuarine wetland system downgradient from SWMU 1 ranged from 33.2 ppt to 40.1 ppt, while the salinity of surface water within the Ensenada Honda downgradient from SWMUs 1 and 2 ranged from 26.6 to 37.4 ppt. Given that all salinity values exceed the approximate toxic threshold for wildlife, the downgradient surface water bodies do not represent potential drinking water sources. Thus, ingestion of surface water is not a potential complete exposure pathway and was not considered in risk calculations for upper trophic level receptors.

Certain potential exposure pathways and/or routes identified on or excluded from Figures 4-7 and 4-8 were not evaluated in the screening-level ERA. Though potentially complete, these pathways were considered insignificant relative to other pathways due to low potential for exposure and low levels of relevant contaminants. For example, dermal exposures were not identified as significant relative to ingestion exposures for upper trophic level receptors and were not evaluated in the screening-level ERA. This approach is supported by evidence outlined in Suter II et al. (2000) and the USEPA (2000a), including the general fate properties of the majority of compounds detected in surface soil and sediment (e.g., low affinity for dermal uptake), the low potential exposure frequency and duration, and the protection offered by feathers, fur, and scales to avian, mammalian, and reptilian receptors. In addition, literature reviews indicate that dermal exposures to wildlife from classes of chemicals known or suspected to be of concern via dermal adsorption (e.g., VOCs, organophosphorous pesticides, and petroleum compounds) are often overestimated in laboratory studies (where feathers/fur are removed) and do not represent realistic exposure scenarios (USEPA, 2000a). Furthermore, though burrowing reptiles (which would be expected to experience the most significant exposure) inhabit the upland vegetative units at SWMUs 1 and 2, chemicals known or suspected to be of concern via dermal adsorption are not known to be associated with historical activities at the site and were either not detected (e.g., organophosphorous pesticides) or detected at a low frequency and concentration (e.g., VOCs). Moreover, in developing surface soil screening levels for twenty-four important compounds identified from National Priorities List (NPL) sites and Biological Technical Assistant Group

(BTAG) recommendations, USEPA calculated that the contribution of dermal exposures to the total dose received by terrestrial receptors to be 0.5 percent or less and therefore omitted the dermal pathway from their exposure estimates (USEPA, 2000a). Incidental ingestion of surface soil or sediment during feeding and preening activities was considered in risk estimates for upper trophic level receptors. Direct contact exposures were also considered for lower trophic level receptors (i.e., terrestrial and aquatic invertebrates).

Inhalation of gaseous chemicals and chemicals adhered to particulate matter (e.g., soil) were also excluded from evaluation in this screening-level ERA as the inhalation pathway is considered insignificant relative to ingestion pathways. As described above for dermal exposures, this approach is consistent with Suter II et al. (2000) and USEPA (1997 and 2000a), which recognize the relatively small contribution the inhalation pathway contributes to exposure estimates. For example, USEPA (2000a) estimates the expected contribution of exposure to dust particles and VOCs via inhalation to be 0.01 percent and 0.5 percent or less, respectively relative to ingestion. Site conditions further reduce the importance of this exposure route relative to ingestion. The vegetative groundcover at SWMUs 1 and 2 minimizes the suspension of dust and the potential for exposure via inhalation of chemicals adhered to soil particles. Furthermore, inhalation of gaseous chemicals that have volatilized from surface soil is likely to be insignificant given that VOCs were detected at a low frequency and concentration (see Appendix G).

Potentially complete and significant exposure pathways for terrestrial mammals (i.e., incidental ingestion of surface soil and ingestion of contaminated plant and/or animal tissues for chemicals that have entered food webs) were not selected for evaluation. The exclusion of mammals is appropriate because the potentially exposed mammalian receptors are limited to nonindigenous, nuisance species (see Section 4.1.3.1). However, because they represent a potential link between surface soil chemicals and terrestrial carnivores, they were included as food items in this screening-level ERA.

Though potentially complete exposure pathways have been identified for terrestrial reptiles (e.g., various lizard species), terrestrial amphibians (e.g., coquis), and aquatic reptiles (e.g., sea turtles) at SWMUs 1 and 2 (see Figures 4-7 and 4-8), there is a paucity of data concerning the toxicological effects of chemicals for reptiles and amphibians, rendering a quantitative evaluation problematic (USEPA, 2000a and 2003a). However, it can be qualitatively stated that reptiles and amphibians are not at risk if no risks are identified to other upper trophic level receptors utilizing the site that occupy a similar trophic level. This approach is consistent with USEPA Region III BTAG policy (USEPA, 2004a). Although this represents an uncertainty in the assessment, it is assumed that terrestrial reptiles and amphibians are not likely to be more sensitive to chemical exposures than the other receptor groups that are included in the screening-level ERA.

4.3.2 Endpoints and Risk Hypotheses

The conclusion of the screening-level problem formulation includes the selection of ecological endpoints, which are based on the preliminary conceptual model. Two types of endpoints, assessment endpoints and measurement endpoints, are defined as part of the ERA process as are risk hypotheses or risk questions (USEPA, 1997 and 1998). An assessment endpoint is an explicit expression of the environmental component or value that is to be protected. A measurement endpoint is a measurable ecological characteristic that is related to the component or value chosen as the assessment endpoint. The considerations for selecting assessment and measurement endpoints are summarized in USEPA (1992, and 1997) and discussed in detail in Suter II (1989, 1990, and 1993). Risk hypotheses are testable hypotheses about the relationship among the assessment endpoints and their predicted responses when exposed to contaminants.

Endpoints in the screening-level ERA define ecological attributes that are to be protected (assessment endpoints) and a measurable characteristic of those attributes (measurement endpoints) that can be used to gauge the degree of impact that has or may occur. Assessment endpoints most often relate to attributes of biological populations or communities, and are intended to focus the risk assessment on particular components of the ecosystem that could be adversely affected by chemicals attributable to the site (USEPA, 1997). Assessment endpoints contain an entity (e.g., spotted sandpiper) and an attribute of that entity (e.g., survival rate). Individual assessment endpoints usually encompass a group of species or populations (the receptor) with some common characteristic, such as specific exposure route or contaminant sensitivity, with the receptor then used to represent the assessment endpoint in the risk evaluation.

Assessment and measurement endpoints may involve ecological components from any level of biological organization, from individual organisms to the ecosystem itself (USEPA, 1992). Effects on individuals are important for some receptors, such as rare and endangered species; however, population- and community-level effects are typically more relevant to ecosystems. Population- and community-level effects are usually difficult to evaluate directly without long-term and extensive study. However, measurement endpoint evaluations at the individual level, such as an evaluation of the effects of chemical exposure on reproduction, can be used to predict effects on an assessment endpoint at the population or community level. In addition, use of criteria values designed to protect the vast majority (e.g., 95 percent) of the components of a community (e.g., National Ambient Water Quality Criteria [NAWQC] for the Protection of Aquatic Life) can be useful in evaluating potential community- and/or population-level effects.

Table 4-5 summarizes the assessment endpoints, risk hypotheses, and measurement endpoints selected for the screening-level ERA. Given the similarity of terrestrial habitats within and contiguous to each SWMU (see Appendix F and Section 4.1.2.1), identical endpoints and risk hypotheses were selected for the upland vegetative units. Identical endpoints and risk hypotheses also were selected for the open water habitat (i.e., Ensenada Honda) downgradient from each SWMU. Standing water is absent from that portion of the estuarine wetland system downgradient from SWMU 2. Although the sediment is saturated, insufficient standing water is present to support a fish community. As such, the survival, growth, and reproduction of avian piscivores within the estuarine wetland system was not selected as an assessment endpoint for SWMU 2. Suitable fish habitat is present within that portion of the wetland system downgradient from SWMU 1. For this reason, the survival, growth, and reproduction of avian piscivores within the estuarine wetland system was selected as an assessment endpoint for SWMU 1.

As evidenced by Table 4-5, the assessment endpoints selected were based on the survival, growth, and reproduction of terrestrial receptor groups (terrestrial plants and invertebrates), aquatic receptor groups (aquatic plants, benthic invertebrates, and fish), amphibians, reptiles, upper trophic level birds (herbivores, omnivores, and carnivores), and marine mammals (i.e., West Indian manatee). The population traits of interest for each of the assessment endpoints represent components of a healthy population. Failure or impairment of survival, growth, or reproduction will adversely affect the ability of the population to be healthy and viable and fill its appropriate role in an ecosystem.

4.3.2.1 Selection of Receptors

Because of the complexity of natural systems, it is generally not possible to directly assess the potential impacts to all ecological receptors present within an area. Therefore, specific receptor species (e.g., spotted sandpiper) or species groups (e.g., aquatic invertebrates) are often selected as surrogates to evaluate potential risks to larger components of the ecological community (e.g., aquatic invertebrate consumers) that are used to represent the assessment endpoints (e.g.,

survival, growth, and reproduction of aquatic invertebrate consumers). Selection criteria typically include those species that:

- Are known to occur, or are likely to occur, at the site;
- Have a particular ecological, economic, or aesthetic value;
- Are representative of taxonomic groups, life history traits, and/or trophic levels in the habitats present at the site for which complete exposure pathways are likely to exist;
- Can, because of toxicological sensitivity or potential exposure magnitude, be expected to represent potentially sensitive populations at the site; and
- Have sufficient ecotoxicological information available on which to base an evaluation.

Lower trophic level receptor species were evaluated based on those taxonomic groupings (e.g., terrestrial and aquatic plants and invertebrates) for which screening values have been developed. These groupings and screening values are used in most ERAs. As such, specific receptor species of lower trophic level terrestrial biota were not chosen because of the limited species-specific information available. These receptors were instead dealt with on a community level via a comparison to media-specific screening values.

The upper trophic level receptor species listed below were chosen for dietary exposure modeling based on the criteria listed above, the general guidelines presented in USEPA (1991a), the description of habitats and biota presented in Sections 4.1.2 and 4.1.3, respectively, and the assessment endpoints (see Table 4-5).

Terrestrial habitat (SWMUs 1 and 2):

- Mourning dove (*Zenaida macroura*) (avian herbivore)
- American robin (*Turdus migratorius*) (avian omnivore)
- Red-tailed hawk (*Buteo jamaicensis*) (avian carnivore)

Estuarine wetland system (SWMU 1):

- Belted Kingfisher (*Ceryle alcyon*) (avian piscivore)
- Great blue heron (*Ardea herodias*) (avian piscivore)
- Spotted sandpiper (*Actitis macroura*) (avian benthic invertebrate consumer)

Estuarine wetland system (SWMU 2):

- Spotted sandpiper (*Actitis macroura*) (avian benthic invertebrate consumer)

Open water habitat (SWMUs 1 and 2):

- Double-crested cormorant (*Phalacrocorax auritus*) (avian piscivore)

- West Indian manatee (*Trichechus manatus*) (mammalian herbivore)

With the exception of the American robin and double-crested cormorant, the upper trophic level receptors listed above are known to occur at NSRR (Raffaele, 1989). The American robin was selected as a surrogate species to represent birds reported from NSRR with similar feeding habits and dietary preferences (e.g., red-legged thrush). Although not previously reported from NSRR, the double-crested cormorant is known to occur in Puerto Rico (Raffaele, 1989). As discussed in Section 4.3.2, avian piscivores were not selected as ecological receptors for the estuarine wetland system downgradient from SWMU 2 based on the absence of suitable fish habitat.

As discussed previously in Section 4.3.1.3, a terrestrial mammal was not selected as an ecological receptor for the following reasons:

- With the exception of bats, all native terrestrial mammals have been extirpated from Puerto Rico. Life history information for Puerto Rico's native bat species is severely limited or lacking altogether.

- The terrestrial mammals represented by potentially complete exposure pathways are limited to nonindigenous, nuisance species (i.e., Norway rat, black rat, and mongoose) that have been implicated in the decline of native reptilian and bird populations.

While exposure pathways to terrestrial reptiles, terrestrial amphibians, and aquatic reptiles are likely to be complete, specific reptilian and amphibian species were not selected as receptors in this screening-level ERA since the life history and toxicological database concerning the effects of chemicals on reptiles and amphibians is severely limited. It is assumed that reptiles and amphibians potentially present at the site are not exposed to significantly higher concentrations of chemicals and are not more sensitive to chemicals than the other receptor species evaluated in the risk assessment. This assumption is a source of uncertainty in the screening-level ERA.

4.3.3 Fate and Transport Mechanisms

In the absence of measured values of chemicals within biotic media, the transport and partitioning of constituents into particular environmental compartments, and their ultimate fate in those compartments, can be predicted from key physical-chemical characteristics. The physical-chemical characteristics that are most relevant for exposure modeling in this assessment include water solubility, adsorption to solids, octanol-water partitioning, and degradability. These characteristics are defined below.

The water solubility of a compound influences it's partitioning to aqueous media. Highly water-soluble chemicals, such as most VOCs, have a tendency to remain dissolved in the water column rather than partitioning to sediment (Howard, 1991). Compounds with high water solubility also generally exhibit a lower tendency to bioconcentrate in aquatic organisms and a greater likelihood of biodegredation, at least over the short term (Howard, 1991).

Adsorption is a measure of a compound's affinity for binding to solids, such as soil or sediment particles. Adsorption is expressed in terms of partitioning, with either the adsorption coefficient (K_d), a unitless expression of the equilibrium concentration in the solid phase versus the water phase) or as organic carbon partition coefficient ($K_{oc,}$ K_d normalized to the organic carbon content of the solid phase; again unitless) (Howard, 1991). For a given organic chemical, the higher the K_{oc} or K_d, the greater the tendency for that chemical to adhere strongly to soil or sediment particles. K_{oc} values can be measured directly or can be estimated from either water solubility or

the octanol-water partition coefficient using one of several available regression equations (Howard, 1991).

Octanol-water partitioning indicates whether a compound is hydrophilic or hydrophobic. The Octanol-water partition coefficient (K_{ow}) expresses the relative partitioning of a compound between octanol (lipids) and water. A high affinity for lipids equates to a high K_{ow} and vice versa. As discussed above, K_{ow} has been shown to correlate well with adsorption to soil or sediment particles and the potential to bioaccumulate in the food chain (Howard, 1991). Typically expressed as log K_{ow}, a value of 3.0 or less generally indicates that the chemical will not bioconcentrate to a significant degree (Maki and Duthie, 1978). Log K_{ow} values and K_{oc} values for organic chemicals analyzed in environmental media collected from SWMUs 1 and 2 (Appendix IX VOCs, SVOCs, organochlorine pesticides, organophosphorous pesticides, chlorinated herbicides, and dioxins/furans) are presented in Table 4-6.

Degradability is an important factor in determining whether there will be significant loss of mass or change in the form of a chemical over time in the environment. The half-life of a compound is typically used to describe losses from either degradation (biological or abiotic) or from transfer from one compartment to another (e.g., volatilization from soil to air). The half-life is the time required for one-half of the mass of a compound to undergo the loss or degradation process.

4.4 Screening-Level Effects Evaluation

The purpose of the screening-level effects evaluation is the establishment of chemical exposure levels (screening values) that represent conservative thresholds for adverse ecological effects. One set of screening values is typically developed for each selected assessment endpoint. For this evaluation, two types of screening values were developed (media-specific screening values and ingestion-based screening values). Media-specific screening values were developed for surface soil, surface water, and sediment, while ingestion-based screening values were developed for food web (dietary) exposures.

4.4.1 Media-Specific Screening Values

The sections that follow describe the various criteria and toxicological benchmarks that were used as media-specific screening values (toxicological thresholds) for chemicals in surface soil, surface water, and sediment. The media-specific screening values, summarized in Tables 4-7 (surface soil), 4-8 (surface water), and 4-9 (sediment), represent conservative exposure thresholds above which adverse ecological effects may occur.

4.4.1.1 Surface Soil Screening Values

The literature-based toxicological benchmarks listed below, expressed as dry weight concentrations, were selected for use as surface soil screening values.

- Toxicological thresholds for earthworms and microorganisms (Efroymson et al., 1997a)
- Toxicological thresholds for plants (Efroymson et al., 1997b)

For a given chemical, when more than one screening value was available from the sources listed above, the lowest value was conservatively selected as the surface soil screening value. As evidenced by Table 4-7, the toxicological thresholds available from Efroymson et al. (1997a and 1997b) for chemicals analyzed in surface soil samples collected from SWMUs 1 and 2 are limited primarily to inorganics. For those chemicals lacking a toxicological threshold from Efroymson et

al. (1997a and 1997b), the following literature-based values, listed in their order of decreasing preference, were used as surface soil screening values:

- Toxicity reference values for plants and invertebrates listed in USEPA, 1999a.
- Soil standards developed by the Ministry of Housing, Spatial Planning and Environment (MHSPE, 1994) assuming a minimum default soil organic carbon content of 2.0 percent.
- Canadian soil quality guidelines (agricultural land use) developed by the Canadian Council of Ministers of the Environment (CCME, 2002).
- United States Fish and Wildlife Service (USFWS) screening values (Beyer, 1990) as listed in Friday (1998)

CCME screening values and screening values developed by Beyer (1990) were given the lowest preference since they are background-based values that do not represent effect concentrations.

4.4.1.2 Surface Water Screening Values

Chronic saltwater NAWQC (USEPA, 2002a) were selected for use as surface water screening values. USEPA NAWQC for cadmium, copper, chromium, lead, mercury, selenium, and zinc are expressed as dissolved concentrations. As a measure of conservatism in this screening-level ERA, they were converted to total recoverable concentrations using the appropriate conversion factors (USEPA, 2002a). For those chemicals lacking a saltwater NAWQC, surface water screening values were identified from the following information listed in their order of decreasing preference:

- Final Chronic Values (FCVs) for saltwater contained in Ecotox Thresholds (USEPA, 1996b)
- Chronic screening values for saltwater contained in Ecological Risk Assessment Bulletins – Supplement to Risk Assessment Guidelines (RAGS) (USEPA, 2001a)
- Minimum chronic toxicity test endpoints (No Observed Effect Concentration [NOEC] and Maximum Acceptable Toxicant Concentration [MATC] values) for saltwater species reported in the ECOTOX Database System (Aquatic Toxicity Information Retrieval [AQUIRE] database) (USEPA, 2003b)
- Chronic Lowest Observable Effect Levels (LOELs) for saltwater contained in National Oceanic and Atmospheric Administration (NOAA) Screening Quick Reference Tables (SQUIRTs) (Buchman, 1999)

The order of preference was selected based on their level of protection. For example, FCVs would be expected to offer a greater degree of protection than a single species NOEC, MATC, or LOEL since their derivation considers a larger toxicological database. In the absence of FCVs, USEPA Region IV chronic screening values, chronic test endpoints, and chronic LOELs, screening values were derived from the acute literature values listed below:

- Acute LOELs for saltwater contained in NOAA SQUIRTs (Buchman, 1999)
- Acute toxicity test endpoints (NOEC, Lowest Observed Effect Concentration [LOEC], median lethal concentration [LC_{50}], and median effective concentration [EC_{50}] values) for

saltwater species contained in the ECOTOX Database System (AQUIRE database) (USEPA, 2003b).

- LC_{50} values for saltwater species contained in Superfund Chemical Matrix (USEPA, 1996a)

Chronic-based screening values were extrapolated from acute NOEC, LOEC, LOEL, LC_{50}, and EC_{50} values as follows:

- An uncertainty factor of 10 was used to convert an acute NOEC, LOEC, or LOEL to a chronic-based screening value.

- An uncertainty factor of 100 was used to convert an EC_{50} or LC_{50} to a chronic-based screening value.

When acute toxicity data were used to extrapolate a chronic screening value, NOECs were given preference over LOECs/LOELs, LOECs/LOELs were given preference over LC_{50} and EC_{50} values, and EC_{50} values were given preference over LC_{50} values. When more than one value was available from the literature for a given test endpoint (e.g., NOEC), the minimum value was conservatively used to extrapolate a chronic screening value. In some cases, chronic and acute LOELs for chemical classes (e.g., PAHs) were available from Buchman (1999). A LOEL based on a chemical class was used to derive a chronic screening value only if that chemical lacked literature-based benchmarks and/or toxicity test endpoints.

For those chemicals lacking saltwater toxicological thresholds and literature values, surface water screening values were identified or developed from freshwater values using the sources and procedures discussed in the preceding paragraphs with one exception. This exception involved the consideration of freshwater Secondary Chronic Values (SCVs) developed by the USEPA (1996b) and Suter II (1996).

4.4.1.3 Sediment Screening Values

The literature-based toxicological benchmarks listed below, expressed as bulk sediment concentrations (dry weight), were used as sediment screening values.

- Effects-Range low (ER-L) marine and estuarine sediment quality guidelines (Long and Morgan, 1991 and Long et al., 1995)

- Threshold Effects Level (TEL) marine sediment quality guidelines (MacDonald, 1994)

- Apparent Effects Threshold (AET) marine sediment quality guidelines (Buchman, 1999)

A description of ER-L, TEL, and AET values and the methods used in their derivation are provided in the paragraphs that follow.

Effect Range-Low (ER-L) marine and estuarine sediment quality guidelines. Long and Morgan (1991) developed effects-based sediment quality guidelines using literature-based data from Equilibrium Partitioning (EqP) modeling, spiked-sediment toxicity tests, and matched sediment chemistry and biological effects measures. For a given chemical, the data were arranged in ascending order of concentration with each data entry assigned an "effects" or "no effects" descriptor, and the 10th percentile and 50th percentile concentrations of the ‘effects” data were

calculated. The 10th and 50th percentiles of the "effects" data represent the ER-L and Effects Range-Median (ER-M), respectively.

The ER-L and the ER-M delineate three concentration ranges for a given chemical. The concentration range below the ER-L value represents a minimal effects range (i.e., the concentration range in which effects would be rarely observed). Concentrations equal to or greater than the ER-L but less than the ER-M represent a possible effects range within which effects would occasionally occur, while concentrations greater than the ER-M represent a probable-effects range within which effects would frequently occur. The ER-L and ER-M values were recalculated by Long et al. (1995) after omitting a small amount of freshwater data included in the original calculations (Long and Morgan 1991) and incorporating more recent marine and estuarine data from the literature. Only ER-Ls were selected as sediment screening values in this screening-level ERA.

Threshold Effect Level (TEL) marine sediment quality guidelines. The updated and revised data set used by Long et al. (1995) also was used by MacDonald (1994) to calculate sediment quality assessment guidelines (TELs and Probable Effect Levels [PELs]) for Florida coastal waters. Unlike the methodology used by Long et al. (1991) to derive ER-L and ER-M values, the derivation of TELs and PELs took into consideration the "no effects" data set. Specifically, TELs were derived by calculating the geometric mean of the 15th percentile in the "effects" data set and the 50th percentile in the "no effects" data set, while PELs were derived by calculating the geometric mean of the 50th percentile in the "effects" data set and the 85th percentile in the "no effects" data set.

Identical to ER-Ls and ER-Ms, TELs and PELs delineate three concentration ranges for a given chemical. The TEL represent the upper limit of the range of sediment concentrations dominated by "no effects" data. Within this range, concentrations are not considered to represent significant hazards to sediment- associated biota. The PEL represents the lower limit of the range of sediment concentrations that are usually or always associated with adverse biological effects. The range of concentrations that could be associated with biological effects is delineated by the TEL and PEL. Within this range of concentrations, adverse biological effects are possible.

Apparent Effects Threshold (AET) marine sediment quality guidelines. The AET method, developed by Tetra Tech, Inc (1986), associates chemical concentrations in sediments with adverse biological effects (lethal and sub-lethal toxicity as measured using sediment toxicity tests or changes in benthic macroinvertebrate abundance and community structure as measured by *in situ* biological surveys). For a given chemical and measurement of biological effect (biological indicator), the AET value represents the sediment concentration above which statistically significant biological effects are always observed. The AET values shown in Table 4-7 represent the lowest AET value from a suite of seven biological indicators (amphipod mortality, oyster larval abnormality, Microtox luminescence, benthic macroinvertebrate abundance, bivalve larvae mortality/abnormality, Echinoderm larvae mortality/abnormality, and juvenile polychaete growth). It is noted that the AET values summarized in Table 7-6 are interim values subject to change (Buchman 1999).

Minimum chemical-specific AET values are used by the Washington Department of Ecology (1995) as sediment management standards for Puget Sound. Minimum AET values also are used by the U.S Army Corp of Engineers (USACE 1998) as "reason to believe" guidance for screening levels for the Dredged Material Management Program (DMMP). The DMMP screening levels are implemented for use in Puget Sound and Grays Harbor/Willapa Bay in the State of Washington. Current Washington State Department of Ecology sediment management standards

and USACE DMMP screening levels do not reflect the interim AET values reported by Buchman (1999).

For a given chemical, when more than one sediment quality guideline was available from the sources listed above, the minimum value was conservatively selected as the sediment screening value. For those organic chemicals lacking a literature-based, bulk-sediment toxicological threshold, equilibrium partitioning (EqP)-based benchmarks were used as sediment screening values. EqP-values were derived using the USEPA equilibrium partitioning (EqP) approach (USEPA, 1993a [see Appendix H]) or identified from the literature (Di Toro and McGrath, 2000). For a given chemical, when an EqP-based value was derived in accordance with USEPA (1993a) guidance and also was available from Di Toro and McGrath (2000), the minimum value was selected as the sediment screening value.

4.4.2 Ingestion-Based Screening Values

Ingestion-based screening values for dietary exposures were derived for each receptor species and chemical evaluated for food web exposures. Toxicological information from the literature for wildlife species most closely related to the receptor species was used if available. This information was supplemented by laboratory studies of non-wildlife species (e.g., laboratory mice) when necessary.

Chronic No Observed Adverse Effect Levels (NOAELs) based on growth or reproduction were preferentially used as ingestion-based screening values for upper trophic level receptors. NOAELs represent the highest dose of a chemical at which an effect being measured in a toxicity test does not occur. If several chronic toxicity studies were available from the literature, the most appropriate study was selected for each receptor species based on study design, study methodology, study duration, study endpoint and test species. When chronic NOAEL values were unavailable, estimates were derived or extrapolated from chronic Lowest Observed Adverse Effect Levels (LOAELs) or median lethal dose acute values (LD_{50}). LOAELs represent the lowest dose of a chemical at which an effect being measured in a toxicity test occurs, while an LD_{50} represents the dose of a chemical at which half of the organisms being tested die. An uncertainty factor of 10 was used to convert a reported chronic LOAEL to a chronic NOAEL, while an uncertainty factor of 100 was used to convert the acute LD_{50} to a chronic NOAEL (i.e., the LD_{50} was multiplied by 0.01 to obtain the chronic NOAEL).

Ingestion-based screening values for the bird species selected as ecological receptors (American robin, mourning dove, red-tailed hawk, spotted sandpiper, belted kingfisher, great blue heron, and double-crested cormorant), expressed as milligrams of the chemical per kilogram body weight of the receptor per day (mg/kg-BW/day), are summarized in Table 4-10. Ingestion-based screening values used in this ERA for the West Indian manatee are summarized in Table 4-11. The NOAEL and LOAEL values summarized in Table 4-11 were adjusted to reflect differences in body weights between the mammalian test species and the West Indian manatee. Using the NOAEL as an example, this was accomplished by using the following scaling equation (Sample et al., 1996):

$$NOAEL_r = NOAEL_t(BW_t/BW_r)^{1/4} \qquad \text{(Equation 4-1)}$$

where:

$NOAEL_r$	=	NOAEL of the receptor species (mg/kg-BW/day)
$NOAEL_t$	=	NOAEL of the test species (mg/kg-BW/day)
BW_r	=	Body weight of receptor species (kg)

BW_t = Body weight of test species (kg)

The scaled NOAELs and LOAELs are included in Table 4-11. Sample et al. (1996) consider a scaling factor of 1.0 most appropriate for interspecies extrapolation between birds. Therefore, the NOAEL and LOAEL values summarized in Table 4-10 were not adjusted to reflect differences in body weights between avian test species and avian receptor species.

Not all chemicals analyzed in abiotic media were evaluated for food web exposures. The organic chemicals evaluated for food web exposures were limited to those listed in Table 4-6 with the potential to bioaccumulate to a significant extent. Bioaccumulative organic chemicals are defined in the screening-level ERA as those with a maximum reported log octanol-water partition coefficient (log K_{ow}) greater than or equal to 3.0. Rational for using a log K_{ow} of 3.0 to define an organic chemical with the potential to bioaccumulate is included as Appendix I. For conservatism, all inorganic chemicals except cyanide also were evaluated for food web exposures. Cyanide was excluded from evaluation because it is readily metabolized and does not bioaccumulate (Eisler, 1991). The list of chemicals selected for evaluation of food web exposures contains many chemicals that are not identified as "important bioaccumulative compounds" by the USEPA (2000b). Their inclusion in the evaluation of terrestrial and aquatic food web exposures is consistent with the conservatism of this screening-level ERA.

4.5 Screening-Level Exposure Estimation

This section presents the analytical data, exposure assumptions, and the exposure models and input parameters that were used to estimate the potential exposure of ecological receptors to chemicals in surface soil, surface water, and sediment.

4.5.1 Selection Criteria for Analytical Data and Their Use in the Screening-Level ERA

The available analytical data (described in Section 4.2) were reviewed against a set of selection criteria to identify specific data that would be used to estimate potential ecological receptor exposures. The specific analytical data quantitatively evaluated in this screening-level ERA are included as Appendix G. The criteria used to select these analytical data are listed below.

- Data must have been validated by a qualified data validator using acceptable data validation methodology. Rejected (R) values were not used in the screening-level ERA. Unqualified data and data qualified as J, NJ, and CN were treated as detected, while data qualified as U or UJ were treated as non-detected.

- The available soil analytical data were divided into surface soil data (i.e., analytical data for soil samples collected from the 0 to 0.5- or 0 to 1.0-foot depth interval) and subsurface soil data (analytical data for soil samples collected from the 0.5 to 1.5-, 0.5 to 1.67-, and 1.0 to 2.0-foot depth interval), and evaluated independently from each other. The evaluation of available soil analytical data was limited to these depth ranges since most soil heterotrophic activity and soil invertebrates occur on the surface or within the oxidized root zone (Suter II, 1995).

- Surface soil was preferentially evaluated for terrestrial food web exposures as the upper depth is the most active biological zone (Suter II, 1995). However, for conservatism, evaluation of exposures to subsurface soil (soil collected 0.5 to 1.5 feet, 0.5 to 1.67, and 1.0 to 2.0 feet bgs) was performed if maximum concentrations occurred in the subsurface horizon.

- SWMUs 1 and 2 surface soil samples collected during the 1996 RFI field investigation and 2004 additional data collection field effort were analyzed for dioxin/furans. However, only the 2004 surface soil samples were analyzed for specific dioxin/furan congeners (1996 RFI surface soil data reported as total TCDD, PeCDD, HxCDD, TCDF, PeCDF, and HxCDF [see Appendix G.1]). Only the congener-specific analytical data were used in the exposure estimate and risk calculation for surface soil.

- For surface water, total (unfiltered) metals data were used in the media-specific screening evaluation. However, dissolved (filtered) metals data were used in the food web model for piscivorous birds to address surface water-to-fish bioaccumulation (see Section 4.5.2.2.1).

- The surface water and sediment data for samples collected from the estuarine wetland system and open water habitat (Ensenada Honda) downgradient from a given SWMU were evaluated independently from each other. Independent evaluation was necessary since unique species are expected to forage within each habitat type. For example, the spotted sandpiper would be expected to forage within the estuarine wetland system but not the Ensenada Honda. Conversely, the double-crested cormorant and West Indian manatee would be expected to forage within the Ensenada Honda, but not the estuarine wetland system.

- Although groundwater data were available for each SWMU, these data were not evaluated in the screening-level ERA since an adequate surface water and sediment dataset was available for the downgradient surface water bodies.

- Maximum reporting limits were conservatively used to estimate exposure for non-detected chemicals.

- In some instances, duplicate samples were collected in the field. The maximum concentration of each chemical (or the maximum non-detected value) in the original or duplicate sample was used as a conservative estimate of contaminant concentration at a particular sampling point. Results from duplicate samples were not evaluated individually.

4.5.2 Exposure Estimation

Maximum detected concentrations in surface soil, subsurface soil, surface water, and sediment were used to conservatively estimate potential chemical exposures for the ecological receptors selected to represent the assessment endpoints. For conservatism, maximum detection limits for chemicals that were analyzed for but not detected were also compared to media-specific screening values and (where appropriate) used for food web exposure modeling. This was done to ensure that reporting limits were similar to, or less than, chemical concentrations at which potential adverse effects to ecological receptors may occur. For samples with duplicate analyses, the higher of the two concentrations was used in the screening (when both values were detects or both values were non-detects). In cases where one result was a detection and the other a non-detect, the detected value was used in the assessment.

With the exception of 2,3,7,8-TCDD, media-specific and ingestion-based screening values are not available for dioxin/furan chemicals. The concentration of these chemicals were converted to 2,3,7,8-TCDD equivalents using the toxicity equivalence factors reported by the USEPA (2000b) and summarized in Tables 4-12 (SWMU 1) and 4-13 (SWMU 2). 2,3,7,8-TCDD equivalents

were summed and compared directly with the 2,3,7,8-TCDD media-specific screening values or used in the food web models.

4.5.2.1 Terrestrial and Aquatic Receptor Groups

Maximum measured chemical concentrations in surface soil, subsurface soil, surface water, and sediment were compared to the media-specific screening values (presented and discussed in Section 4.4.1) to conservatively evaluate the potential for adverse ecological effects to the lower trophic level receptor groups selected as assessment endpoints (terrestrial plants and invertebrates, aquatic plants and invertebrates, and fish). Exposure point concentrations for the terrestrial receptor groups (terrestrial plants and invertebrates) were maximum measured surface soil and subsurface soil concentrations. Maximum measured surface water and/or sediment concentrations were used as exposure point concentrations for the aquatic receptor groups (aquatic plants, invertebrates, and fish).

4.5.2.2 Upper Trophic Level Receptors

Exposures for upper trophic level receptor species via the food web were determined by estimating chemical-specific concentrations in each dietary component using uptake and food web models. Incidental ingestion of soil or sediment was also included when calculating the total level of exposure. Drinking water exposures were not considered when estimating the total level of exposure (see Section 4.3.1.3). As indicated previously, maximum measured soil and sediment concentrations were used in all calculations to provide a conservative assessment.

Tissue concentrations were modeled for terrestrial plants (food item for American robin and mourning dove), soil invertebrates (food item for American robin), small mammals (food item for red-tailed hawk), aquatic plants (food item for the West Indian manatee), aquatic invertebrates (food item for the spotted sandpiper, belted kingfisher, and great blue heron), and fish (food item for the belted kingfisher, great blue heron, and double-crested cormorant). Specific small mammals species were not selected as dietary items for the red-tailed hawk. Instead, a specific trophic level (omnivore) was used to represent the small mammals present in Puerto Rico that represent potential food items (e.g., Norway rat and black rat). Small mammal herbivores and insectivores were excluded as food items for the red-tailed hawk because they are not part of the Puerto Rican mammalian fauna (see Section 4.1.3.1).

4.5.2.2.1 Exposure Point Concentrations

The uptake of chemicals from the abiotic media into terrestrial and aquatic food items is based (where available) on conservative (e.g., maximum or 90th percentile) bioconcentration factors (BCFs) or bioaccumulation factors (BAFs) from the literature. A BCF indicates the degree to which a chemical may accumulate in organisms coincident with the concentration of the chemical in the surrounding media. They are calculated by dividing the concentration of a chemical in the tissue of organisms by the concentration in the surrounding media. BAF values consider both direct exposures to the surrounding media, as well as uptake from dietary exposures. As such, BAFs were given preference over BCFs when estimating prey item tissue concentrations. Default factors of 1.0 were used only when data are unavailable for chemicals in the literature. The methodology and models used to derive these estimates are described below.

Terrestrial Plants. Tissue concentrations in the aboveground vegetative portion of terrestrial plants were estimated by multiplying the maximum measured surface soil concentration for each chemical by chemical-specific soil-to-plant BCFs obtained from the literature. The BCF values used were based on root uptake from soil and on the ratio between dry-weight soil and dry-weight

plant tissue. Literature values based on the ratio between dry-weight soil and wet-weight plant tissue were converted to a dry-weight basis by dividing the wet-weight BCF by the estimated solids content for terrestrial plants (15 percent [0.15]; Sample et al., 1997a).

BCFs for terrestrial plants are those reported in Baes et al. (1984) or Bechtel Jacobs (1998a). For organic chemicals without literature based BCFs, soil-to-plant BCFs were estimated using the algorithm provided in Travis and Arms (1988):

$$Log\ B_v = 1.588 - (0.578)\ (Log\ K_{ow}) \qquad \text{(Equation 4-2)}$$

where:

$Log\ B_v$ = Log soil-to-plant BCF (unitless; dry weight basis)
$Log\ K_{ow}$ = Log octanol-water partitioning coefficient (unitless)

The Log K_{ow} values used in the calculations were obtained primarily from the USEPA (1995a and 1996a) and are listed in Table 4-6. The soil-to-plant BCFs used in the screening-level ERA are summarized in Table 4-14.

Earthworms. Tissue concentrations in soil invertebrates (earthworms) were estimated by multiplying the maximum measured surface soil concentration for each chemical by chemical-specific BCFs or BAFs obtained from the literature. BCFs are calculated by dividing the concentration of a chemical in the tissues of an organism by the concentration of that same chemical in the surrounding environmental medium (in this case, surface soil) without accounting for uptake via the diet. BAFs consider both direct exposure to soil and exposure via the diet. Since earthworms consume soil, BAFs are more appropriate values and were used in the food web models when available. BAFs based on depurated analyses (soil was purged from the gut of the earthworm prior to analysis) were given preference over undepurated analyses when selecting BAF values since direct ingestion of surface soil is accounted for separately in the food web model.

The BCF/BAF values used in this screening-level ERA (see Table 4-14) are based on the ratio between dry-weight soil and dry-weight earthworm tissue. Literature values based on the ratio between dry-weight soil and wet-weight earthworm tissue were converted to a dry-weight basis by dividing the wet-weight BCF/BAF by the estimated solids content for earthworms (16 percent [0.16]; USEPA, 1993b). For inorganic chemicals without available measured BCFs/BAFs, an earthworm BAF of 1.0 was assumed.

Small Mammals. Whole-body tissue concentrations in small mammals (omnivores) were estimated using one of two methodologies. For chemicals with literature-based soil-to-small mammal BAFs, the small mammal tissue concentration was obtained by multiplying the maximum measured surface soil concentration for each chemical by a chemical-specific soil-to-small mammal BAF. The BAF values used are based on the ratio between dry-weight soil and whole-body dry-weight tissue. Literature values based on the ratio between dry-weight soil and wet-weight tissue were converted to a dry-weight basis by dividing the wet-weight BAF by the estimated solids content for small mammals (32 percent [0.32]; USEPA, 1993b). The soil-to-small mammal BAFs used in the screening-level ERA (see Table 4-15) are those in Sample et al. (1998b) for omnivores (or for general small mammals if omnivore values were unavailable).

For those chemicals without soil-to-small mammal BAF values, an alternate approach was used to estimate whole-body tissue concentrations. Because most chemical exposure for small mammal species is via the diet, it was assumed that the concentration of each chemical in a small

mammal's tissues is equal to the chemical concentration in its diet, that is, a diet to whole-body BAF (wet-weight basis) of one was assumed. Resulting tissue concentrations (wet-weight) were converted to dry weight using an estimated solids content of 32 percent (see above).

The use of a diet to whole-body BAF of one is likely to result in a conservative estimate of chemical concentrations for chemicals that are not known to biomagnify in terrestrial food chains (e.g., aluminum). For chemicals that are known to biomagnify (e.g., PCBs), a diet to whole-body BAF value of one will likely result in a realistic estimate of tissue concentrations based on reported literature values. For example, a maximum BAF (wet weight) value of 1.0 was reported by Simmons and McKee (1992) for PCBs based on laboratory studies with white-footed mice. Menzie et al. (1992) reported BAF values (wet-weight) for DDT of 0.3 for voles and 0.2 for short-tailed shrews. Reported BAF (wet-weight) values for dioxin are only slightly above one (1.4) for the deer mouse (USEPA 1990).

Aquatic Plants. Tissue concentrations in the vegetative portion of aquatic plants (i.e., sea grass) were estimated using the same methodologies as described above for terrestrial plants except that maximum sediment (not surface soil) concentrations were used in the calculation.

Aquatic Invertebrates. Tissue concentrations in aquatic invertebrates were estimated by multiplying the maximum measured sediment concentration for each chemical by chemical-specific sediment-to-invertebrate BCFs or BAFs obtained from the literature. The BCF/BAF values are based on the ratio between dry-weight sediment and dry-weight invertebrate tissue. Because BAFs consider both direct exposure to sediment and exposure via the diet, BAFs are more appropriate values and were used in the food web models when available. BAFs based on depurated analyses (sediment was purged from the gut of the organism prior to analysis) were given preference over undepurated analyses when selecting BAF values since direct ingestion of sediment is accounted for separately in the food web model.

Literature values based on the ratio between dry-weight sediment and wet-weight invertebrate tissue were converted to a dry-weight basis by dividing the wet-weight BCF/BAF by the estimated solids content for aquatic invertebrates (21 percent [0.21]; USEPA 1993b). For chemicals without available measured literature BCF/BAF values, a BCF/BAF of 1.0 was assumed. The sediment-to-invertebrate BCFs/BAFs used in the screening-level ERA are summarized in Table 4-16.

Fish. The estimation of tissue concentrations in whole-body fish took into consideration bioaccumulation from surface water, as well as bioaccumulation from sediment. The contribution that sediment bioaccumulation has on whole-body fish tissue concentrations was estimated by multiplying the maximum measured sediment concentration for each chemical by chemical-specific sediment-to-fish BAFs obtained from the literature. The sediment-fish BAF values used were based on the ratio between dry-weight sediment and dry-weight fish tissue. Literature values based on the ratio between dry-weight sediment and wet-weight fish tissue were converted to a dry-weight basis by dividing the wet-weight BAF by the estimated solids content for fish (25 percent [0.25]; USEPA 1993b). For chemicals without literature based sediment-to-fish BAFs, a BAF of 1.0 was assumed. A summary of the sediment-to-fish BAFs used in the screening-level ERA is provided in Table 4-16.

The contribution that surface water bioaccumulation has on whole-body fish tissue concentrations was estimated by multiplying the maximum measured surface water concentration for each chemical by chemical-specific surface water-to-fish BAFs obtained from the literature. In the absence of literature-based BAFs, the contribution that surface water bioaccumulation has on whole-body fish tissue concentrations was estimated by the following equation (USEPA, 1995b):

$$C_{xf} = [(C_{sw})(BCF_{sw})(FCM)] \qquad \text{(Equation 4-3)}$$

where C_{xf} is the concentration of chemical x in whole-body fish (mg/kg), C_{sw} is the maximum surface water concentration (mg/L), BCF_{sw} is the measured surface water-to-fish BCF (L/kg). The surface water-to-fish BAF values obtained from the literature and the BCF values used in Equation 4-3 to estimate surface water-to-fish BAFs were based on dry weight fish tissue. Literature values based on wet-weight fish tissue were converted to a dry-weight basis by dividing the wet-weight BAF by 0.25 (see above).

For a given organic chemical, surface water-to-fish bioaccumulation was only considered if the chemical was detected in surface water and the chemical's log K_{ow} value is greater than or equal to 3.0. If an organic chemical with a log K_{ow} value greater than or equal to 3.0 was not detected in surface water, the contribution that surface water bioaccumulation has on the tissue concentration in whole-body fish was considered to be negligible. In this instance, only sediment bioaccumulation was considered in the estimation of whole-body fish tissue concentrations. Specific organic chemicals detected in surface water samples collected from the estuarine wetland or open water habitat downgradient from SWMUs 1 and/or 2 with log K_{ow} values greater than or equal to 3.0 were ethylbenzene, styrene, toluene, butylbenzylphthalate, di-n-octylphthalate, 2-methylnaphthalene, benzo(b)fluoranthene, naphthalene, and beta-BHC.

The surface water data used for metals in the estimation of surface water-to-fish bioaccumulation was based on the dissolved (filtered) fraction. Dissolved metals data were used since the dissolved fraction more closely approximates the bioavailable fraction of metals in the water column (USEPA, 1995b, 1999b, and 2002a). If a metal was not detected in the dissolved (filtered) fraction, the contribution that surface water bioaccumulation has on the whole-body fish tissue concentration of that metal was considered negligible. Specific metals detected in surface water samples (dissolved fraction) collected from the estuarine wetland system or open water habitat downgradient from SWMUs 1 and/or 2 were antimony, arsenic, barium, chromium, cobalt, copper, lead, nickel, selenium, tin, vanadium, and zinc.

Surface water-to-fish BAFs used in the screening-level ERA are summarized in Table 4-17. With the exception of selenium, surface water-to-fish BAFs were estimated using Equation 4-3. For selenium, a literature BAF was identified from the literature and used in this screening-level ERA. For metals, an FCM of 1.0 was used to convert a measured surface water-to-fish BCF to a surface water-to-fish BAF (USEPA, 1991b and 1995b and Sample et al., 1996). Although the USEPA (1995b) has derived FCMs for organic chemicals using a food chain model developed by Gobas (1993), an FCM of 1.0 was also used to convert measured surface water-to-fish BCFs to surface water-to-fish BAFs for PAHs. The 1993 Gobas model includes an input parameter to account for metabolism (metabolic rate constant); however, it was set to zero by the USEPA (1995c). As such, the FCMs developed by the USEPA do not take into consideration metabolism. Given that invertebrates and fish are capable of metabolizing PAHs (James, 1989, Varanasi et al., 1989, Buhler and Williams, 1989, and Foureman, 1989), use of an FCM would overstate bioaccumulation from dietary food items.

FCMs established by the USEPA (1995b) were used to estimate surface water-to-fish BAF values for ethylbenzene, styrene, toluene, butylbenzylphthalate, di-n-octylphthalate, and beta-BHC (see Table 4-17). For a given organic chemical, the FCM is based on the log K_{ow} of the chemical and the trophic level occupied by the prey. The USEPA (1995c) has reported that fish consumed by the belted kingfisher, great blue heron, and double-crested cormorant are trophic level 3 fish. As such, trophic level 3 FCMs were used in Equation 4-3 to estimate surface water-to-fish BAFs.

A final fish tissue concentration was derived by summing the individual contributions that surface water-to-fish bioaccumulation and sediment-to-fish bioaccumulation have on whole-body fish tissue concentrations:

$$C_{xf} = [(C_{sw})(BAF_{sw}) + (C_{sed})(BAF_{sed})] \qquad \text{(Equation 4-4)}$$

where C_{sed} is the maximum sediment concentration (mg/kg), BAF_{sed} is the sediment-to-fish BAF (unitless), BAF_{sw} is the surface water-to-fish BAF, and C_{xf}, and C_{sw}, are as previously described.

4.5.2.2.2 *Dietary Intakes*

Dietary intakes for each upper trophic level receptor species were calculated using the following formula (Equation 4-5) modified from USEPA (1993b).

$$DI_x = \frac{[[\sum_i [(FIR)(FC_{xi})(PDF_i)] + [(FIR)(SC_x)(PDS)]][AUF]}{BW}$$

where:

DI_x = Dietary intake for chemical x (mg chemical/kg body weight/day)
FIR = Food ingestion rate (kilograms per day [kg/day], dry-weight)
FC_{xi} = Maximum concentration of chemical x in food item i (mg/kg, dry weight)
PDF_i = Proportion of diet composed of food item i (mg/kg, dry weight)
SC_x = Maximum concentration of chemical x in surface soil/sediment (mg/kg, dry weight)
PDS = Proportion of diet composed of surface soil/sediment (dry weight basis)
BW = Body weight (kg, wet weight)
AUF = Area Use Factor (unitless)

Conservative receptor-specific exposure parameters (maximum food ingestion rates and minimum body weights) for the American robin, mourning dove, and red-tailed hawk, spotted sandpiper, great blue heron, belted kingfisher, double-crested cormorant, and West Indian manatee are provided in Table 4-18. The food items selected for each receptor species and the percent contribution to their total diet is provided in Table 4-19. As discussed previously in Section 4.3.1.3, receptor exposures via surface water ingestion were not included in the estimation of dietary intakes. As such, drinking water ingestion rates for the receptor species are not included in Table 4-18.

Table 4-18 contains exposure parameters and Table 4-19 contains a dietary composition for a small mammal omnivore. As discussed in Section 4.5.2.2, the diet of the red-tailed hawk (excluding surface soil) is assumed to be small mammal omnivores. This assumption is based on likely small mammal prey species present in Puerto Rico (rats). Identification of exposure parameters and food items was necessary when estimating small mammal whole body tissue concentrations for those chemicals that lack a literature-based soil-to-small mammal BAF (i.e., an exposure dose was necessary to estimate tissue concentrations). An assumed diet of 49 percent terrestrial vegetation, 49 percent terrestrial invertebrates, and 2 percent soil was selected as the diet for a small mammal omnivore.

For the screening-level ERA, an AUF of 1.0 was assumed (i.e., each receptor is assumed to spend 100 percent of its time on the site). As such, receptor-specific home ranges were not considered in the estimation of dietary intakes.

4.6 Screening-Level Risk Calculation

The screening-level risk calculation is the final step in a screening-level ERA. In this step, maximum chemical concentrations in abiotic media or maximum exposure doses for upper trophic level receptor species are compared with the corresponding screening values to derive screening risk estimates. The outcome of this step is a list of potential ecological COPCs for each media-pathway-receptor combination evaluated or a conclusion of negligible risk. The screening-level risk calculation for SWMUs 1 and 2 are presented in Sections 4.6.1.1 and 4.6.1.2, respectively.

4.6.1 Selection of Ecological Chemicals of Potential Concern

Ecological COPCs were selected using the hazard quotient (HQ) method. For a given chemical, an HQ was calculated by dividing the maximum chemical concentration in the medium being evaluated by the corresponding media-specific screening value or, in the case of upper trophic level receptors, by dividing the maximum exposure dose by the corresponding ingestion-based screening value.

The following conservative methodology was used to identify ecological COPCs for abiotic media:

- The maximum detected concentration in surface soil, surface water, and sediment were used to calculate media-specific HQs. For a given medium, chemicals with HQs greater than or equal to 1.0 based on maximum detected concentrations were identified as ecological COPCs.

- For non-detected chemicals, maximum reporting limits were used to calculate media-specific HQ values. Non-detected chemicals with HQs greater than or equal to 1.0 based on maximum reporting limits were identified as ecological COPCs.

- Detected and non-detected chemicals without media-specific screening values were identified as ecological COPCs.

To select preliminary ecological COPCs by evaluating food web exposures, maximum chemical concentrations in surface soil, surface water, and/or sediment were used to estimate dietary doses for each receptor. HQs were calculated with NOAELs, LOAELs, and Maximum Acceptable Toxicant Concentrations (MATCs). The MATC is derived by taking the geometric mean of the NOAEL and LOAEL. Calculations with NOAELs provide the most conservative risk estimate, while calculations with LOAELs provide the least conservative risk estimate. Calculations with MATCs provide realistic risk estimates since the MATC represents an estimation of the threshold concentration (i.e., the concentration above which a toxic effect on the test endpoint is produced). For the screening-level ERA, chemicals (detected and non-detected) with NOAEL-based HQs greater than or equal to 1.0 were identified as ecological COPCs. Identical to the media-specific screening evaluation, detected and non-detected chemicals without ingestion-based screening values were identified as ecological COPCs for upper trophic level receptor exposures.

HQs greater than or equal to 1.0 indicate the potential for risk since the chemical concentration or dose (exposure) exceeds the screening value (effect). However, screening values and exposure

doses are derived using intentionally conservative assumptions (maximum media concentrations, maximum ingestion rates, and minimum body weights) such that HQs greater than or equal to 1.0 do not necessarily indicate that risks are present or impacts are occurring. Rather, they identify chemical-pathway-receptor combinations requiring further evaluation. Following the same reasoning, HQs less than one indicate that risks are very unlikely, enabling a conclusion of no unacceptable risk to be reached with high confidence.

It is noted that the screening-level ERA considers independent effects of chemicals. However, the potential does exist for multiple chemicals in environmental media to interact. Much uncertainty is involved with the interpretation of chemical interactions due to the complexity of potential effects (e.g., synergistic, antagonistic, or additive), and due to varying toxicities of compounds in different species. For these reasons, cumulative effects were not addressed in this screening-level ERA. Chemical interactions can be addressed by site-specific studies conducted in Step 6 of the Navy ERA process (i.e., site investigation and data analysis [see Figure 4-1]).

4.6.1.1 SWMU 1 Screening-Level Risk Calculation

Screening-level risk calculations for SWMU 1 surface soil and subsurface soil, as well estuarine wetland and Ensenada Honda surface water and sediment downgradient from SWMU 1 are presented in Tables 4-20 through 4-25, while screening-level risk calculations for food web exposures (terrestrial and aquatic) are presented in Tables 26 through 29. Ecological COPCs were identified in Step 2 of the screening-level ERA using the procedure outlined in Section 4.6.1.

4.6.1.1.1 Surface Soil

Table 4-20 presents the results of the screening-level risk calculation for SWMU 1 surface soil. Detected concentrations greater than or equal to surface soil screening values are shown on Figure 4-9. Three VOCs (acetone, ethylbenzene, and tetrachloroethene) were each detected in one of eighteen surface soil samples collected at SWMU 1. Xylene was also detected in two of eighteen surface soil samples. Because maximum detected ethylbenzene, tetrachloroethane, and xylene concentrations (1 J ug/kg, 3 J ug/kg, and 5 J ug/kg, respectively) were less than surface soil screening values, they were not identified as ecological COPCs. Acetone was identified as an ecological COPC based on the lack of a surface soil screening value. Although not detected, vinyl chloride also was identified as an ecological COPC because the maximum reporting limit for this VOC exceeded the surface soil screening value (HQ = 1.36). Twenty-six additional non-detected VOCs were identified as ecological COPCs for SWMU 1 surface soil based on the lack of surface soil screening values.

Bis(2-ethylhexyl)phthalate and butylbenzylphthalate were detected in SWMU 1 surface soil. However, because maximum detected concentrations for both SVOCs were less than surface soil screening values, they were not identified as ecological COPCs. Six non-detected SVOCs (1,2,4,5-tetrachlorobenzene, 1,3,5-trinitrobenzene, 2,4-dimethylphenol, m,p-cresol, o-cresol, and pentachlorophenol) were identified as ecological COPCs for SWMU 1 surface soil because maximum reporting limits exceeded surface soil screening values. HQ values ranged from 1.45 for pentachlorophenol to 10.0 for 1,2,4,5-tetrachlorobenzene. An additional fifty-seven non-detected SVOCs were identified as ecological COPCs based on the lack of surface soil screening values.

Thirteen PAHs were detected in one or more of the SWMU 1 surface soil samples. Of these thirteen PAHs, benzo(b)fluoranthene (HQ = 1.25) and pyrene (HQ = 2.17) were identified as ecological COPCs because maximum detected concentrations exceeded surface soil screening

values. Maximum concentrations for the remaining detected PAHs, as well as maximum reporting limits for the non-detected PAHs were less than surface soil screening values. As such, they were not identified as ecological COPCs for SWMU 1 surface soil.

Three organochlorine pesticides (4,4'-DDT, 4,4-DDD, and 4,4'-DDE) were detected in SWMU 1 surface soil and identified as an ecological COPC because maximum detected concentrations exceeded surface soil screening values (HQs = 32.5 for 4,4'-DDD, 70.0 for 4,4'-DDE, and 108 for 4,4'-DDT). Although not detected, alpha-chlordane, gamma-chlordane, methoxychlor, and toxaphene were identified as ecological COPCs because maximum reporting limits exceeded surface soil screening values. The non-detected pesticide chlorobenzilate also was identified as an ecological COPC for SWMU 1 surface soil based on the lack of a surface soil screening value.

One PCB aroclor-1260 was detected in one of eighteen surface soil samples. However, because the maximum detected concentration was less than the surface soil screening value it was not identified as en ecological COPC.

Chlorinated herbicides, and organophosphorous pesticides were not detected in any of the SWMU 1 surface soil samples. However, three non-detected chlorinated herbicides (2,4,5-T, 2,4,5-TP, and 2,4-D) and nine non-detected organophosphorous pesticides were identified as ecological COPCs based on the lack of surface soil screening values. Because maximum reporting limits for all PCBs were less than surface soil screening values, they were not identified as ecological COPCs for SWMU 1 surface soil.

As evidenced by Table 4-12, seventeen dioxin/furan congeners were detected in SWMU 1 surface soil. Because the sum of the 2,3,7,8-TCDD equivalents (0.297 ug/kg), calculated using maximum detected concentrations and maximum reporting limits (for non-detected dioxin/furan chemicals), was less than the 2,3,7,8-TCDD surface soil screening value (500 ug/kg [see Table 4-7]), dioxin/furan chemicals were not identified as ecological COPCs for surface soil.

Sixteen metals were detected in SWMU 1 surface soil (see Table 4-20). Of these sixteen metals, maximum detected antimony, cadmium, chromium, cobalt, copper, lead, mercury, nickel, silver, tin, vanadium, and zinc concentrations exceeded surface soil screening values. HQ values ranged from 1.30 for silver to 123 for chromium. These twelve metals were identified as ecological COPCs for SWMU 1 surface soil.

4.6.1.1.2 Subsurface Soil

Table 4-21 presents the results of the screening-level risk calculation for SWMU 1 subsurface soil. Detected concentrations greater than or equal to surface soil screening values are shown on Figure 4-10. Three VOCs (acetone, carbon disulfide, and methylene chloride) were detected in one or more of the SWMU 1 subsurface soil samples. Because the maximum detected methylene chloride concentration was less than the surface soil screening value, this VOC was not identified as an ecological COPC. Acetone and carbon disulfide were identified as ecological COPCs due to the lack of surface soil screening values. Although not detected, vinyl chloride also was identified as an ecological COPC because the maximum reporting limit for this VOC exceeded the surface soil screening value (HQ = 1.18). Ten additional non-detected VOCs were identified as ecological COPCs for SWMU 1 subsurface soil based on the lack of surface soil screening values.

Bis(2-ethylhexyl)phthalate was detected in one of six subsurface soil samples collected at SWMU 1. However, because the single detected concentration (430 J ug/kg) was less than the surface soil screening value, this SVOC was not identified as an ecological COPC. Six non-detected

SVOCs (2,4,5-trichlorophenol. 2,4-dichlorophenol, 2,4-dimethylphenol, 2-chlorophenol, o-cresol, and pentachlorophenol) were identified as ecological COPCs for SWMU 1 subsurface soil because maximum reporting limits exceeded surface soil screening values. Nineteen additional non-detected SVOCs were identified as ecological COPCs based on the lack of surface soil screening values.

PAHs and PCBs were not detected in any of the SMWU 1 subsurface soil samples. However, fifteen non-detected PAHs were identified as ecological COPCs for SWMU 1 subsurface soil because maximum reporting limits exceeded surface soil screening values. HQ values based on maximum reporting limits was 1.33 for these fifteen PAHs. Because maximum reporting limits for all PCBs were less than surface soil screening values, they were not identified as ecological COPCs for SWMU 1 subsurface soil.

Six organochlorine pesticides (4,4'-DDT, 4,4'-DDD, 4,4'-DDE, endrin, heptachlor, and methoxychlor) were detected in SWMU 1 subsurface soil. Because maximum detected 4,4'DDD, endrin, heptachlor, and methoxychlor concentrations were less than surface soil screening values, they were not identified as ecological COPCs. 4,4'-DDT and 4,4-DDE were identified as ecological COPCs for SWMU 1 subsurface soil because maximum detected concentrations exceeded surface soil screening values (HQ = 8.75 and 1.30, respectively). Although not detected, toxaphene also was identified as an ecological COPC because the maximum reporting limit for this pesticide exceeded the surface soil screening value.

Eleven metals were detected in the SWMU 1 subsurface soil. Antimony, cadmium, chromium, cobalt, copper, vanadium, and zinc were identified as ecological COPCs because maximum detected concentrations exceeded surface soil screening values. HQ values ranged from 1.15 for cadmium to 120 for vanadium. Although not detected, mercury and silver also were identified as ecological COPCs for SWMU 1 subsurface soil because maximum reporting limits exceeded surface soil screening values.

4.6.1.1.3 Estuarine Wetland Surface Water

Table 4-22 presents the results of the screening-level risk calculation for estuarine wetland surface water downgradient from SWMU 1. Detection concentrations greater than or equal to surface water screening values are shown on Figure 4-11. Four VOCs (acetone, benzene, ethylbenzene, and toluene) were detected in one or more of the surface water samples. Because maximum detected concentrations, ranging from 0.13 J ug/L for ethylbenzene to 11 J ug/L for acetone, were less than surface water screening values, these four VOCs were not identified as ecological COPCs for estuarine wetland surface water. Six non-detected VOCs (chloroethane, chloroprene, ethyl methacrylate, iodomethane, methacrylonitrile, and trans, 1,4-dichloro-2-butene) were identified as ecological COPCs based on the lack of surface water screening values.

SVOCs (excluding PAHs), organochlorine pesticides, PCBs, organophosphorous pesticides, and chlorinated herbicides were not detected in estuarine wetland surface water. However, twenty-one SVOCs, twelve organochlorine pesticides, seven PCBs, and eight organophosphorus pesticides were identified as ecological COPCs for estuarine wetland surface water because maximum reporting limits exceeded surface water screening values. An additional eighteen non-detected SVOCs, one non-detected organochlorine pesticide (endrin aldehyde), and one non-detected organophosphorous pesticide (o,o,o-triethylphosphorothioate) were identified as ecological COPCs based on the lack of surface water screening values.

Three PAHs (2-methylnaphthalene, benzo(b)fluoranthene, and naphthalene) were detected in at least one estuarine wetland surface water sample. However, because maximum detected

concentrations for each PAH were less than surface water screening values (see Table 4-22), they were not identified as ecological COPCs for estuarine wetland surface water.

Dioxin/furan chemicals were not detected in estuarine wetland surface water (see Table 4-12). However, because the sum of the 2,3,7,8-TCDD equivalents (0.014 ug/L), calculated using maximum reporting limits, exceeded the 2,3,7,8-TCDD surface water screening value (0.00001 ug/L), dioxin/furan chemicals were identified as ecological COPCs.

Fourteen metals were detected in estuarine wetland surface water. Copper was identified as a ecological COPC because the maximum detected concentration (19 ug/L) exceeded the surface water screening value. Tin also was identified as an ecological COPC for estuarine wetland surface water based on the lack of a surface water screening value. Although not detected, cyanide was identified as an ecological COPC because the maximum reporting limit for this inorganic exceeded the surface water screening value.

4.6.1.1.4 Estuarine Wetland Sediment

Table 4-23 presents the results of the screening-level risk calculation for estuarine wetland sediment downgradient from SWMU 1. Detected concentrations greater than or equal to sediment screening values are shown on Figure 4-12. Seven VOCs (2-butanone, acetone, benzene, carbon disulfide, chlorobenzene, ethylbenzene, and toluene) were detected in two or more of the sediment samples collected from the estuarine wetland. Because maximum detected 2-butanone, benzene, carbon disulfide, chlorobenzene, ethylbenzene, and toluene concentrations were less than screening values, these six VOCs were not identified as ecological COPCs for sediment. Acetone was identified as an ecological COPC for estuarine wetland sediment because the maximum concentration for this VOC exceeded the sediment screening value (HQ = 56.8). Seventeen non-detected VOCs also were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values. Five additional non-detected VOCs (chloroprene, ethyl methacrylate, iodomethane, methacrylonitrile, and trans-1,4-dichloro-2-butene) were identified as ecological COPCs for estuarine wetland sediment based on the lack of sediment screening values.

SVOCs, PCBs, and organophosphorous pesticides were not detected in estuarine wetland sediment. However, sixty-seven non-detected SVOCs, seven non-detected PCBs, and eight non-detected organophosphorous pesticides were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values. Nineteen additional non-detected SVOCs and one non-detected organophosphorous pesticide (o,o,o-triethylphosphorothioate) were identified as ecological COPCs for estuarine wetland sediment based on the lack of sediment screening values.

Two PAHs (fluoranthene and pyrene) were detected in estuarine wetland sediment. The maximum detected fluoranthene concentration was less than the sediment screening value (HQ = 0.07), while the maximum detected pyrene concentration exceeded the sediment screening value (HQ = 1.44). Based on an HQ value greater than 1.0, pyrene was identified as an ecological COPC for estuarine wetland sediment. Five additional non-detected PAHs were identified as ecological COPCs for estuarine wetland sediment because maximum reporting limits exceeded sediment screening values.

Two organochlorine pesticides (4,4'-DDD and 4,4'-DDE) were detected in estuarine wetland sediment. Because maximum detections for both pesticides exceeded sediment screening values (HQ = 352 for 4,4'-DDD and HQ = 213 for 4,4'-DDE), they were identified as ecological COPCs. Although not detected, twelve organochlorine pesticides also were identified as

ecological COPCs because maximum reporting limits exceeded screening values. One additional non-detected organochlorine pesticide (endrin aldehyde) was identified as an ecological COPC for estuarine wetland sediment based on the lack of a sediment screening value.

Dioxin/furan chemicals were not detected in estuarine wetland sediment (see Table 4-12). However, because the sum of the 2,3,7,8-TCDD equivalents (7.0 ug/kg), calculated using maximum reporting limits, exceeded the 2,3,7,8-TCDD sediment screening value (0.0036 ug/kg), dioxin/furan chemicals were identified as ecological COPCs for estuarine wetland sediment.

Seventeen inorganics were detected in estuarine wetland sediment downgradient from SWMU 1. Maximum detected antimony, cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, silver, tin, and vanadium concentrations exceeded sediment screening values. As such, these twelve metals were identified as ecological COPCs for estuarine wetland sediment. Two detected inorganics (beryllium and cyanide) also were identified as ecological COPCs due to the lack of sediment screening values. One additional non-detected metal (thallium) was identified as an ecological COPC based on the lack of a sediment screening value.

4.6.1.1.5 Ensenada Honda Surface Water

Table 4-24 presents the results of the screening-level risk calculation for Ensenada Honda surface water downgradient from SWMU 1. Detected concentrations greater than or equal to surface water screening values are shown on Figure 4-13. Four VOCs (2-butanone, acetone, bromoform, and chloroethane) were detected in Ensenada Honda surface water. Because maximum detected 2-butanone, acetone, and bromoform concentrations (2.0 J ug/L, 8.8 J ug/L, and 0.24 J ug/L, respectively) were less than surface water screening values, they were not identified as ecological COPCs for Ensenada Honda surface water. Five non-detected VOCs (chloroprene, ethyl methacrylate, iodomethane, methacrylonitrile, and trans-1,4-dichloro-2-butene) also were identified as ecological COPCs based on the lack of surface water screening values.

Three SVOCs (di-n-butylphthalate, isophorone, and methapyrilene) were detected in Ensenada Honda surface water. Because the maximum detected isophorone concentration (14 ug/L) was less than the surface water screening value, this SVOC was not identified as an ecological COPC. The maximum detected di-n-butylphthalate concentration (5.9 J ug/L) exceeded the surface water screening value (HQ = 1.74 ug/L), while methapyrilene lacks a surface water screening value. As such, these two SVOCs were identified as ecological COPCs. Twenty non-detected SVOCs also were identified as ecological COPCs for Ensenada Honda surface water because maximum reporting limits exceeded surface water screening values. An additional seventeen non-detected SVOCs were identified as ecological COPCs based on the lack of surface water screening values.

One PAH (naphthalene) was detected in five of nine Ensenada Honda surface water samples downgradient from SWMU 1. Because the maximum detected concentration (0.048 J ug/L) was less than the surface water screening value, this PAH was not identified as an ecological COPC. Maximum reporting limits for the non-detected PAHs were less than surface water screening values. As such, none of the non-detected PAHs were not identified as ecological COPCs for Ensenada Honda surface water.

Organochlorine pesticides, PCBs, chlorinated herbicides, and organophosphorous pesticides were not detected in Ensenada Honda surface water. However, twelve organochlorine pesticides, seven PCBs, and eight organophosphorous pesticides were identified as ecological COPCs because maximum reporting limits exceeded surface water screening values. One additional non-detected organochlorine pesticide (endrin aldehyde) and one additional non-detected

organophosphorous pesticide (o,o,o-triethylphosphorothioate) were identified as ecological COPCs based on the lack of surface water screening values.

Dioxin/furan chemicals were not detected in Ensenada Honda surface water (see Table 4-12). However, because the sum of the 2,3,7,8-TCDD equivalents (0.014 ug/L), calculated using maximum reporting limits, exceeded the 2,3,7,8-TCDD surface water screening value (0.00001 ug/L), dioxin/furan chemicals were identified as ecological COPCs.

Twelve inorganics were detected in Ensenada Honda surface water. Copper was identified as an ecological COPC because the maximum detected concentration exceeded the surface water screening value (HQ = 3.78). Tin, detected in seven of nine surface water samples, also was identified as an ecological COPC for Ensenada Honda surface water based on the lack of a surface water screening value. Although not detected, cyanide and silver were identified as ecological COPCs because maximum reporting limits exceeded surface water screening values.

4.6.1.1.6 *Ensenada Honda Sediment*

Table 4-25 presents the results of the screening-level risk calculation for Ensenada Honda sediment downgradient from SWMU 1. Detected concentrations above surface water screening values are shown on Figure 4-14. Four VOCs (2-butanone, acetone, carbon disulfide, and ethylbenzene) were detected in Ensenada Honda sediment. Because maximum detected 2-butanone, carbon disulfide, and ethylbenzene concentrations were less than sediment screening values (HQ = 0.06, 0.03, and 0.4, respectively), they were not identified as ecological COPCs. Acetone was identified as an ecological COPC for Ensenada Honda sediment because the maximum detected concentration exceeded the sediment screening value (HQ = 53.3). Eleven non-detected VOCs also were identified as ecological COPCs for Ensenada Honda sediment because maximum reporting limits exceeded sediment screening values. Five additional non-detected VOCs were identified as ecological COPCs based on the lack of sediment screening values.

Seven SVOCs (2,4-dinitrotoluene, 2-nitroaniline, 4-chloropehenylphenyl ether, 4-nitroaniline, di-n-butylphthalate, di-n-octylphthalate, and phenol) were detected in Ensenada Honda sediment. Because maximum detected concentrations exceeded sediment screening values (HQ values ranged from 2.37 for 4-chlorophenylphenyl ether to 291 for 2,4-dinitrotoluene), these seven SVOCs were identified as ecological COPCs for Ensenada Honda sediment. Fifty-six non-detected SVOCs also were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values. An additional nineteen non-detected SVOCs were identified as ecological COPCs for Ensenada Honda sediment based on the lack of sediment screening values.

Eight PAHs were detected in Ensenada Honda sediment. As evidenced by Table 4-25, maximum detected concentrations were less than sediment screening values. As such, these eight PAHs were not identified as ecological COPCs. Five non-detected PAHs (2-methylnaphthalene, acenaphthene, acenaphthylene, dibenz(a,h)anthracene, and fluorene) were identified as ecological COPCs for Ensenada Honda sediment because maximum reporting limits exceeded sediment screening values.

Organochlorine pesticides, PCBs, and organophosphorous pesticides were not detected in Ensenada Honda sediment. However, fourteen non-detected organochlorine pesticides, seven non-detected PCBs, and eight non-detected organophosphorous pesticides were identified as ecological COPCs for Ensenada Honda sediment because maximum reporting limits exceeded sediment screening values. One non-detected organochlorine pesticide (endrin aldehyde) and one

non-detected organophosphorous pesticide (o,o,o-triethylphosphorothioate) also were identified as ecological COPCs based on the lack of sediment screening values.

Dioxin/furan chemicals were not detected in Ensenada Honda sediment (see Table 4-12). However, because the sum of the 2,3,7,8-TCDD equivalents (5.819 ug/kg), calculated using maximum reporting limits, exceeded the 2,3,7,8-TCDD sediment screening value (0.0036 ug/kg), dioxin/furan chemicals were identified as ecological COPCs.

Seventeen metals were detected in Ensenada Honda sediment. Arsenic, copper, selenium, tin, and vanadium were identified as ecological COPCs because maximum detected concentrations exceeded sediment screening values. HQ values ranged from 1.02 for vanadium to 2.12 for tin. The detected metals beryllium and thallium also were identified as ecological COPCs due to the lack of sediment screening values. Although not detected, cyanide was identified as an ecological COPC due to the lack of a sediment screening value.

4.6.1.1.7 Terrestrial and Aquatic Food Web Exposures

Results of the screening-level risk calculation for SWMU 1 terrestrial and aquatic food web exposures are presented in Tables 4-26 through 4-29. A discussion of these results is presented below.

Terrestrial Food Web Exposures: Surface Soil

Results of the risk calculation for terrestrial food web exposures to chemicals in surface soil are presented in Table 4-26. Based on the comparison of maximum exposure doses to NOAEL-based screening values, three detected organochlorine pesticides (4,4'-DDD, 4,4'-DDE, and 4,4'-DDT), one detected PCB (Aroclor-1260), and eight detected metals (cadmium, chromium, cobalt, copper, lead, mercury, tin, and zinc) had HQ values greater than or equal to 1.0 for one or more of the terrestrial avian receptors evaluated by this ERA. The highest HQ values were calculated for the American robin, including a HQ of 201 for mercury, 168 for cadmium, 129 for zinc, and 103 for lead. Dioxin/furan chemicals also were detected and identified as ecological COPCs because maximum exposure doses, based on the sum of 2,3,4,7-TCDD equivalents, exceeded NOAEL-based screening values for the American Robin and Red-Tailed Hawk. One detected VOC (ethylbenzene), one detected SVOC (butylbenzylphthalate), and one detected metal (beryllium) also were identified as ecological COPCs for terrestrial food web exposures to chemicals in surface soil based on the lack of ingestion-based screening values.

In addition to the detected chemicals identified in the preceding paragraph, two non-detected PCBs (Aroclor-1248 and Aroclor-1254) were identified as ecological COPCs because maximum exposure doses for the American robin exceeded NOAEL-based screening values. In addition to these non-detected PCBs, eight non-detected VOCs, twenty-seven non-detected SVOCs, three non-detected organochlorine pesticides, two non-detected chlorinated herbicides, and one non-detected organophosphorous pesticides also were identified as ecological COPCs for terrestrial food web exposures to chemicals in surface soil based on the lack of ingestion-based screening values.

Terrestrial Food Web Exposures: Subsurface Soil

Results of the screening-level risk calculations for terrestrial food web exposures to chemicals in subsurface soil are presented in Table 4-27. As discussed in Section 4.5.1, chemicals were evaluated for subsurface soil terrestrial food web exposures if maximum concentrations occurred in subsurface soil. Based on the comparison of maximum exposure doses to NOAEL-based

screening values, cobalt had HQ values greater than 1.0 for the American robin (HQ = 1.85). As such, this detected metal was identified as an ecological COPC for terrestrial food web exposures. Beryllium also was detected and identified as an ecological COPC for subsurface soil food web exposures based on the lack of an ingestion-based screening value. In addition to the detected chemicals identified above, two non-detected SVOCs (di-n-butyphthalate and hexachlorobenzene) and six non-detected PCBs were identified as ecological COPCs for subsurface soil terrestrial food web exposures because maximum exposure doses exceeded NOAEL-based screening values for the American robin. An additional thirteen non-detected SVOC were identified as ecological COPCs for subsurface soil terrestrial food web exposures based on the lack of ingestion-based screening values.

Aquatic Food Web Exposures: Estuarine Wetland Surface Water and Sediment

Results of the risk calculation for aquatic food web exposures to chemicals in estuarine wetland surface water and/or sediment are presented in Table 4-28. Based on the comparison of maximum exposure doses to NOAEL-based screening values, three detected organochlorine pesticides (4,4'-DDE, 4,4'-DDD, and 4,4'-DDT) and eight detected metals (chromium, cobalt, copper, lead, mercury, selenium, vanadium, and zinc) had HQ values greater than or equal to 1.0 for one or more of the aquatic receptors. These three organochlorine pesticides and eight metals were identified as ecological COPCs for aquatic food web exposures. Three detected VOCs (chlorobenzene, ethylbenzene, and toluene), one detected herbicide (2,4,5-T), and one detected metal (beryllium) also were identified as ecological COPCs for estuarine wetland aquatic food web exposures based on the lack of ingestion-based screening values.

In addition to the detected chemicals identified above, five non-detected SVOCs [bis(2-ethylhexyl)phthalate, di-n-butylphthalate, dinoseb, hexachlorobenzene, and hexachlorobutadiene), four non-detected organophosphorous pesticides, and seven non-detected PCBs were identified as ecological COPCs because maximum exposure doses exceeded NOAEL-based screening values for each of the avian receptors. Although not detected, dioxin/furan chemicals were identified as ecological COPCs because maximum exposure doses, based on the sum of 2,3,7,8-TCDD equivalents, exceeded NOAEL-based screening values for each of the avian receptors. An additional seven non-detected VOCs, twenty-nine non-detected SVOCs, two non-detected organochlorine pesticides (endrin ketone and isodrin), one non-detected chlorinated herbicides (2,4,5-TP), and one non-detected organophosphorous pesticide (sulfotep) were identified as ecological COPCs for estuarine wetland aquatic food web exposures based on the lack of ingestion-based screening values.

Aquatic Food Web Exposures: Ensenada Honda Surface Water and Sediment

Results of the risk calculation for aquatic food web exposures to bioaccumulative chemicals in Ensenada Honda surface water and/or sediment are presented in Table 4-29. Based on the comparison of maximum exposure doses to NOAEL-based screening values, arsenic, mercury, and di-n-butylphthalate had HQ values greater than or equal to 1.0 for either the West Indian manatee (arsenic) or the double-crested cormorant (mercury and di-n-butylphthalate). These three chemicals were identified as ecological COPCs for aquatic food web exposures. One VOC (ethylbenzene), one SVOC (4-chlorophenylphenyl ether), one chlorinated herbicide (2,4,5-T), and one metal (beryllium) were detected and identified as ecological COPCs based on the lack of ingestion-based screening values for one or more of the aquatic receptors. Although not detected, dioxin/furan chemicals were identified as ecological COPCs because maximum exposure doses, based on the sum of 2,3,7,8-TCDD equivalents, exceeded NOAEL-based screening values for the West Indian manatee and double-crested cormorant. Nine non-detected VOCs, twenty-seven non-detected SVOCs, two non-detected organochlorine pesticides, one non-detected chlorinated

herbicide, and three non-detected organophosphorous pesticides also were identified as ecological COPCs based on the lack of ingestion-based screening values for one or both of the receptors.

4.6.1.2 SWMU 2 Screening-Level Risk Calculation

Screening-level risk calculations for SWMU 2 surface soil and subsurface soil, as well estuarine wetland sediment and Ensenada Honda surface water and sediment downgradient from SWMU 2 are presented in Tables 4-30 through 4-34, while screening-level risk calculations for food web exposures (terrestrial and aquatic) are presented in Tables 4-35 through 4-38. Ecological COPCs were identified in Step 2 of the screening-level ERA using the procedure outlined in Section 4.6.1.

4.6.1.2.1 Surface Soil

Table 4-30 presents the results of the screening-level risk calculation for SWMU 2 surface soil. Detected concentrations greater than or equal to surface soil screening values are shown on Figure 4-15. Four VOCs (1,1,1-trichloroethane, acetone, methylene chloride, and toluene) were detected in SWMU 2 surface soil. Because maximum detected 1,1,1-trichloroethane, methylene chloride, and toluene concentrations (12 J ug/kg, 65 ug/kg, and 7 ug/kg, respectively) were less than surface soil screening values, they were not identified as ecological COPCs. Acetone was identified as an ecological COPC for SMWU 2 surface soil due to the lack of a surface soil screening value. Although not detected, vinyl chloride also was identified as an ecological COPC because the maximum reporting limit for this VOC exceeded the surface soil screening value (HQ = 1.36). An additional twenty-six non-detected VOCs were identified as ecological COPCs based on the lack of surface soil screening values.

Eleven PAHs were detected in SWMU 2 surface soil. Because maximum detected concentrations for each of the detected PAHs were less than surface soil screening values, they were not identified as ecological COPCs. Four non-detected PAHs (2-methylnaphthalene, acenaphthene, acenaphthylene, and naphthalene) were identified as ecological COPCs because maximum reporting limits exceeded screening values.

Seven organochlorine pesticides (4,4'-DDE, 4,4'-DDT, endosulfan II, endrin, endrin aldehyde, gamma-BHC, and heptachlor) were detected in at least one of the surface soil samples collected at SWMU 2. Because maximum detected concentrations were less than surface soil screening values, they were not identified as ecological COPCs for SWMU 2 surface soil. Four non-detected organochlorine pesticides (alpha-chlordane, gamma-chlordane, methoxychlor, and toxaphene) were identified as ecological COPCs because maximum reporting limits exceeded surface soil screening values. Although not detected, chlorobenzilate also was identified as an ecological COPC for SWMU 2 surface soil based on the lack of a surface soil screening value.

SVOCs, PCBs, chlorinated herbicides, and organophosphorous pesticides were not detected in SWMU 2 surface soil. However, six SVOCs were identified as ecological COPCs because maximum reporting limits exceeded surface soil screening values. An additional Fifty-six non-detected SVOCs, three non-detected chlorinated herbicides, and nine non-detected organophosphorous pesticides were identified as ecological COPCs for SWMU 2 surface soil based on the lack of surface soil screening values.

As evidenced by Table 4-13, twelve dioxin/furan congeners were detected in SWMU 1 surface soil. However, because the sum of the 2,3,7,8-TCDD equivalents (0.0094 ug/kg), calculated using maximum detections and maximum reporting limits, was less than the 2,3,7,8-TCDD

surface soil screening value (500 ug/kg), dioxin/furan chemicals were not identified as ecological COPCs for SWMU 2 surface soil.

Fifteen metals were detected in SWMU 2 surface soil. Antimony, arsenic, barium, chromium, cobalt, copper, lead, mercury, nickel, selenium, vanadium, and zinc were identified as ecological COPCs because maximum detected concentrations exceeded surface soil screening values.

4.6.1.2.2 *Subsurface Soil*

Table 4-31 presents the results of the screening-level risk calculation for SWMU 2 subsurface soil. Detected concentrations above surface soil screening values are shown on Figure 4-16. Three VOCs (acetone, carbon disulfide, and methylene chloride) were detected in one or more of the SWMU 2 subsurface soil samples. Because the maximum detected methylene chloride concentration was less than the surface soil screening value, this VOC was not identified as an ecological COPCs. Acetone and carbon disulfide were identified as ecological COPCs for SWMU 2 subsurface soil based on the lack of surface soil screening values. Although not detected, vinyl chloride also was identified as an ecological COPC because the maximum reporting limit for this VOC exceeded the surface soil screening value (HQ = 1.82). Ten additional non-detected VOCs were identified as ecological COPCs for SWMU 2 subsurface soil based on the lack of surface soil screening values.

Di-n-butylphthalate was detected in one of six SWMU 2 subsurface soil samples. However, because the single detected concentration was less than the surface soil screening value, di-n-butylphthalate was not identified as an ecological COPC. Six non-detected SVOCs (2,4,5-trichlorophenol, 2,4-dichlorophenol, 2,4-dimethylphenol, 2-chlorophenol, o-cresol, and pentachlorophenol) were identified as ecological COPCs for SWMU 2 subsurface soil because maximum reporting limits exceeded surface soil screening values. Nineteen additional non-detected SVOCs were identified as ecological COPCs based on the lack of surface soil screening values.

Twelve PAHs were detected in SWMU 2 subsurface soil. Maximum detected concentrations of benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, fluoranthene, phenanthrene, and pyrene exceeded surface soil screening values. As such, these six PAHs were identified as ecological COPCs. Although not detected, 2-methylnaphthalene, acenaphthylene, and naphthalene were identified as ecological COPCs for SWMU 2 subsurface soil because maximum reporting limits exceeded surface soil screening values.

Twelve organochlorine pesticides (4,4'-DDT, 4,4'-DDD, 4,4'-DDE, aldrin, alpha-BHC, beta-BHC, delta-BHC, dieldrin, endosulfan I, endosulfan sulfate, heptachlor, and heptachlor epoxide) were detected in SWMU 2 subsurface soil. Because maximum detected concentrations were less than surface soil screening values, they were not identified as ecological COPCs. Although not detected, toxaphene was identified as an ecological COPC for SWMU 2 subsurface soil because the maximum reporting limit for this organochlorine pesticide exceeded the surface soil screening value.

PCBs were not detected in SWMU 2 subsurface soil. Furthermore, maximum reporting limits were less than surface soil screening values. Based on the absence of detections and maximum reporting limits less than surface soil screening values, PCBs were not identified as ecological COPCs for SWMU 2 subsurface soil.

Fourteen metals were detected in SWMU 2 subsurface soil. Antimony, arsenic, barium, cadmium, chromium, cobalt, copper, lead, mercury, nickel, vanadium, and zinc were identified as

ecological COPCs because maximum detected concentrations exceeded surface soil screening values. HQ values ranges from 1.02 for barium to 193 for vanadium. Although not detected, silver also was identified as an ecological COPC for SWMU 2 subsurface soil because the maximum reporting limit for this metal exceeded the surface soil screening value.

4.6.1.2.3 Estuarine Wetland Sediment

Table 4-32 presents the results of the screening-level risk calculation for estuarine wetland sediment downgradient from SWMU 2. Detections greater than or equal to sediment screening values are shown on Figure 4-17. 2-Butanone, acetone, benzene, carbon disulfide, chlorobenzene, and methylene chloride were detected in estuarine wetland sediment. Because maximum detected 2-butanone, benzene, carbon disulfide, chlorobenzene, and methylene chloride concentrations were less than sediment screening values, they were not identified as ecological COPCs. Acetone was identified as an ecological COPC because the maximum detected concentration of this VOC (110 J ug/kg) exceeded the sediment screening value. Ten non-detected VOCs were also were identified as ecological COPCs for estuarine wetland sediment because maximum reporting limits exceeded sediment screening values. Five additional non-detected VOCs were identified as ecological COPCs based on the lack of sediment screening values.

2-Acetylaminofluorene, 5-nitro-o-toluidine, di-n-butylphthalate, methyl methanesulfonate, N-nitrosomethylethylamine, N-nitrosomorpholine, p-(dimethylamino)azobenzene, and pentachlorophenol were detected in at least one of the estuarine wetland sediment sample. Detected concentrations ranged from 54 J ug/kg for methyl methanesulfonate to 450 J ug/kg for di-n-butylphthalate. The maximum detected concentration of 2-acetylaminofluorene was less than the sediment screening value. As such, this SVOC was not identified as an ecological COPC. Because the detected concentrations of 5-nitro-o-toluidine, di-n-butylphthalate, and pentachlorophenol exceeded sediment screening values, they were identified as ecological COPCs for estuarine wetland sediment. Methyl methanesulfonate, N-Nitrosomorpholine, and p-(dimethylamino)azobenzene also were identified as ecological COPCs based on the lack of sediment screening values. Fifty-six non-detected SVOCs were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values. An additional fifteen non-detected SVOCs were identified as ecological COPCs for estuarine wetland sediment based on the lack of sediment screening values.

Benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, fluoranthene, and pyrene were detected in at least one estuarine wetland sediment sample. Because maximum detected concentrations for each PAH were less than sediment screening values, they were not identified as ecological COPCs. Five non-detected PAHs [2-methylnaphthalene, acenaphthene, acenaphthylene, benzo(a)pyrene, and dibenz(a,h)anthracene] were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values.

4,4'-DDE was the only organochlorine pesticide detected in estuarine wetland sediment collected downgradient from SWMU 2. Because the maximum detected concentration was less than the sediment screening value, 4,4'DDE was not identified as an ecological COPC. Twelve non-detected organochlorine pesticides were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values. One additional non-detected pesticide (endrin aldehyde) also was identified as ecological COPCs based on the lack of a sediment screening value.

PCBs, chlorinated herbicides, and organophosphorous pesticides were not detected in estuarine wetland sediment. However, seven PCBs and eight organophosphorous pesticides were

identified as ecological COPCs because maximum reporting limits exceeded sediment screening values. One additional non-detected organophosphorous pesticide (o,o,o-tetraethyldithiopyrophosphate) was identified as an ecological COPC for estuarine wetland sediment based on the lack of a sediment screening value.

Dioxin/furan chemicals were not detected in estuarine wetland sediment downgradient from SWMU 2 (see Table 4-13). However, because the sum of the 2,3,7,8-TCDD equivalents (4.48 ug/kg), calculated using maximum reporting limits, exceeded the 2,3,7,8-TCDD sediment screening value (0.0036 ug/kg), dioxin/furan chemicals were identified as ecological COPCs.

All Appendix IX metals were detected in at least three estuarine wetland sediment samples. Arsenic, cobalt, copper, lead, mercury, nickel, tin, and vanadium were identified as ecological COPCs because maximum detected concentrations exceeded sediment screening values. Beryllium and thallium also were identified as ecological COPCs based on the lack of sediment screening values. Although not detected, cyanide was identified as an ecological COPC based on the lack of a sediment screening value.

4.6.1.2.4 Ensenada Honda Surface Water

Table 4-33 presents the results of the screening-level risk calculation for Ensenada Honda surface water. Detections above surface water screening values are shown on Figure 4-18. 2-Butanone, acetone, benzene, styrene, and toluene were detected in Ensenada Honda surface water. Because maximum detected concentrations were less than surface water screening values, these five VOCs were not identified as ecological COPCs. Five non-detected VOCs (chloroethane, chloroprene, ethyl methacrylate, iodomethane, and trans-2,4-dichloro-2-butene) were identified as ecological COPCs for Ensenada Honda surface water based on the lack of surface water screening values.

Butylbenzylphthalate was the only SVOC detected in Ensenada Honda surface water. This SVOC was detected in a single surface water sample at 0.88 J ug/L. Because this single detected concentration was less than the surface water screening value, butylbenzylphthalate was not identified as an ecological COPC. Twenty-one non-detected SVOCs were identified as ecological COPCs because maximum reporting limits exceeded surface water screening values. Eighteen additional non-detected SVOCs also were identified as ecological COPCs based on the lack of surface water screening values.

PAHs, PCBs, chlorinated herbicides, and organophosphorous pesticides were not detected in Ensenada Honda surface water. However, one non-detected PAH [benzo(a)pyrene], seven non-detected PCBs, and eight non-detected organophosphorous pesticides were identified as ecological COPCs because maximum reporting limits exceeded surface water screening values. One additional non-detected organophosphorous pesticide (o,o,o-triethylphosphorothioate) was identified as an ecological COPC for Ensenada Honda surface water due to the lack of a surface water screening value.

Beta-BHC was the only organochlorine pesticide detected in Ensenada Honda surface water. This pesticide was detected in a single surface water sample at a concentration less than the surface water screening value. Therefore, beta-BHC was not identified as an ecological COPC. Twelve non-detected pesticides were identified as ecological COPCs because maximum reporting limits exceeded surface water screening values. One additional non-detected organochlorine pesticide (endrin aldehyde) also was identified as ecological COPCs based on the lack of surface water screening value.

Dioxin/furan chemicals were not detected in Ensenada Honda surface water (see Table 4-13). However, because the sum of the 2,3,7,8-TCDD equivalents (0.014 ug/L), calculated using maximum reporting limits, exceeded the 2,3,7,8-TCDD surface water screening value (0.00001 ug/L), dioxin/furan chemicals were identified as ecological COPCs.

Fourteen metals were detected in Ensenada Honda surface water downgradient from SWMU 2. Maximum detected concentrations for antimony, arsenic, barium, beryllium, chromium, cobalt, lead, nickel, selenium, silver, thallium, vanadium, and zinc were less than surface water screening values. As such, these twelve metals were not identified as ecological COPCs. Copper was identified as an ecological COPC because the maximum detected concentration (4.5 J ug/L) exceeded the surface water screening value. Tin, detected in nine of nine samples, also was identified as an ecological COPC based on the lack of a surface water screening value. Although not detected, cyanide was identified as an ecological COPC because the maximum reporting limit for this inorganic exceeded the surface water screening value.

4.6.1.2.5 Ensenada Honda Sediment

Table 4-34 presents the results of the screening-level risk calculation for Ensenada Honda sediment downgradient from SWMU 2. Detections above sediment screening values are shown on Figure 4-19. Four VOCs (2-butanone, acetone, carbon disulfide, and methylene chloride) were detected in Ensenada Honda sediment. Acetone was identified as an ecological COPC because the maximum detected concentration (91 µg/L) exceeded the sediment screening value. Maximum detected concentrations for 2-butanone, carbon disulfide, and methylene chloride were less than sediment screening values. As such these three VOCs were not identified as ecological COPCs. Ten non-detected VOCs were identified as ecological COPCs for Ensenada Honda sediment because maximum reporting limits exceeded sediment screening values. Five additional non-detected VOCs were identified as ecological COPCs based on the lack of sediment screening values.

Di-n-butylphthalate, n-nitrosodi-n-propylamine, and phenol were detected in Ensenada Honda sediment downgradient from SWMU 2. Because the maximum detected concentration of N-nitrosodi-n-propylamine was less than the sediment screening value, this SVOCs was not identified as an ecological COPC. The maximum detected concentration of di-n-butylphthalate and phenol exceeded the surface water screening value (HQs = 10.5 and 2.46, respectively). As such, these two SVOCs were identified as ecological COPCs. Fifty-seven non-detected SVOCs also were identified as ecological COPCs for Ensenada Honda sediment because maximum reporting limits exceeded sediment screening values. Eighteen additional non-detected SVOCs were identified as ecological COPCs based on the lack of sediment screening values.

Twelve PAHs were detected in Ensenada Honda sediment downgradient from SWMU 2. Of these twelve PAHs, benzo(a)pyrene, benzo(a)anthracene, chrysene, fluoranthene, and pyrene were identified as ecological COPCs because maximum detected concentrations exceeded sediment screening values. Although not detected, 2-methylnaphthalene, acenaphthene, dibenz(a,h)anthracene, and fluorene also were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values.

Methoxychlor was the only organochlorine pesticide detected in Ensenada Honda sediment downgradient from SWMU 2. Because the maximum detected concentration was less than the sediment screening value, methoxychlor was not identified as an ecological COPC. Thirteen non-detected organochlorine pesticides were identified as ecological COPCs for Ensenada Honda sediment because maximum reporting limits exceeded sediment screening values. One additional

non-detected pesticide (endrin aldehyde) was identified as an ecological COPC based on the lack of a sediment screening value.

PCBs, organophosphorous pesticides, and chlorinated herbicides were not detected in Ensenada Honda sediment downgradient from SWMU 2. However, seven PCBs and eight organophosphorous pesticides were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values. One additional organophosphorous pesticide (o,o,o-triethylphosphorothioate) was identified as an ecological COPC for Ensenada Honda sediment based on the lack of a sediment screening value.

Dioxin/furan chemicals were not detected in Ensenada Honda sediment (see Table 4-13). However, because the sum of the 2,3,7,8-TCDD equivalents (4.76 ug/kg), calculated using maximum reporting limits, exceeded the 2,3,7,8-TCDD sediment screening value (0.0036 ug/kg), dioxin/furan chemicals were identified as ecological COPCs.

Seventeen metals were detected in Ensenada Honda sediment downgradient from SWMU 2. Of these seventeen metals, arsenic, copper, lead, and tin were identified as ecological COPCs because maximum detected concentrations exceeded screening values. HQ values ranged from 1.02 for copper to 1.59 for lead. The detected metals beryllium and thallium were also identified as ecological COPCs based on the lack of a sediment screening value. Although not detected cyanide was identified as an ecological COPC based on the lack of a sediment screening value.

4.6.1.2.6 Terrestrial and Aquatic Food Web Exposures

Results of the screening-level risk calculation for SWMU 2 terrestrial and aquatic food web exposures are presented in Tables 4-35 through 4-38. A discussion of these results is presented below.

Terrestrial Food Web Exposures: Surface Soil

Results of the risk calculation for terrestrial food web exposures to chemicals in surface soil are presented in Table 4-35. Based on the comparison of maximum exposure doses to NOAEL-based screening values, ten detected metals (arsenic, barium, cadmium, chromium, cobalt, copper, lead, mercury, selenium, and zinc) had HQ values greater than or equal to 1.0 for one or more of the terrestrial avian receptors. The highest HQ values were calculated for the American robin, including a HQ of 507 for lead and a HQ of 4,499 for mercury. These ten metals were identified as ecological COPCs for terrestrial food web exposures to chemicals in surface soil. Dioxin/furan chemicals were detected and identified as ecological COPCs because maximum exposure doses, based on the sum of 2,3,7,8-TCDD equivalents, exceeded NOAEL-based screening values for the American Robin.. Toluene and beryllium were detected and identified as ecological COPCs based on the lack of ingestion-based screening values.

Two non-detected SVOCs (hexachlorobenzene and di-n-butylphthalate) and three non-detected PCBs (Aroclor-1248, Aroclor-1254, and Aroclor-1260) were identified as ecological COPCs because maximum exposure doses for the American robin exceeded NOAEL-based screening values. In addition to the non-detected SVOCs and PCBs identified above, nine non-detected VOCs, twenty-nine non-detected SVOCs, one non-detected organochlorine pesticide, two non-detected chlorinated herbicides, and one non-detected organophosphorous pesticides also were identified as ecological COPCs for terrestrial food web exposures due to the lack of ingestion-based screening values.

Terrestrial Food Web Exposures: Subsurface Soil

Results of the screening-level risk calculations for terrestrial food web exposures to chemicals in subsurface soil are presented in Table 4-36. As discussed in Section 4.5.1, chemicals were evaluated for subsurface soil terrestrial food web exposures if maximum detected or non-detected concentrations occurred in subsurface soil. Based on the comparison of maximum exposure doses to NOAEL-based screening values, six metals (arsenic, cadmium, chromium, copper, lead, and zinc) had HQ values greater than 1.0 for one or more of the terrestrial avian receptors. These six metals were identified as ecological COPCs for terrestrial food web exposures.

Although not detected, hexachlorobenzene was identified as an ecological COPC for subsurface soil terrestrial food web exposures because the maximum exposure dose (based on the maximum reporting limit) exceeded the NOAEL-based screening value for the American robin. An additional seven non-detected VOCs and fifteen non-detected SVOC were identified as ecological COPCs for terrestrial food web exposures based on the lack of ingestion-based screening values.

Aquatic Food Web Exposures: Estuarine Wetland Sediment

Results of the risk calculation for aquatic food web exposures to chemicals in estuarine wetland sediment are presented in Table 4-37. Based on the comparison of maximum exposure doses to NOAEL-based screening values, one detected SVOC (di-n-butylphthalate) and eight detected metals (arsenic, chromium, cobalt, copper, lead, mercury, vanadium, and zinc) had HQ values greater than or equal to 1.0 for the spotted sandpiper (i.e., maximum exposure doses exceeded NOAEL-based screening values). One detected VOC (chlorobenzene), two detected SVOCs [2-acetylaminofluorene and p-(dimethylamino)azobenzene], and one detected metal (beryllium) also were identified as ecological COPCs for estuarine wetland aquatic food web exposures due to the lack of ingestion-based screening values.

Two non-detected SVOCs (hexachlorobenzene and dinoseb), two non-detected organophosphorous pesticides (ethyl parathion and methyl parathion), seven non-detected PCBs, and dioxins/furans (based on the sum of 2,3,7,8-TCDD equivalents) were identified as ecological COPCs for aquatic food web exposures because maximum exposure doses (based on maximum reporting limits) exceeded NOAEL-based screening values for the spotted sandpiper. An additional nine non-detected VOCs, twenty-seven non-detected SVOCs, three non-detected organochlorine pesticides (isodrin and heptachlor epoxide), two non-detected chlorinated herbicides (2,4,5-T and 2,4,5-TP) and one non-detected organophosphorous pesticide (sulfotep) were identified as ecological COPCs for estuarine wetland aquatic food web exposures based on the lack of ingestion-based screening values.

Aquatic Food Web Exposures: Ensenada Honda Surface Water and Sediment

Results of the risk calculation for aquatic food web exposures to chemicals in Ensenada Honda surface water and sediment are presented in Table 4-38. Based on the comparison of maximum exposure doses to NOAEL-based screening values, three detected metals (arsenic, mercury, and selenium) had HQ values greater than or equal to 1.0 for either the West Indian manatee (arsenic, mercury, and selenium) or the double-crested cormorant (mercury). Three non-detected organophosphorous pesticides (disulfoton, ethyl parathion, and methyl parathion) and the non-detected dioxin/furan chemicals also were identified as ecological COPCs for Ensenada Honda aquatic food web exposures because maximum exposure doses (based on maximum reporting limits) exceeded NOAEL-based screening values for one or more of the open water aquatic receptors. In addition, two detected VOCs (styrene and toluene), one detected SVOC

(butylbenzylphthalate), and one detected metal (beryllium) also were identified as ecological COPCs for estuarine wetland aquatic food web exposures due to the lack of ingestion-based screening values for the double-crested cormorant.

In addition to the detected chemicals identified in the preceding paragraph, eight non-detected VOCs, twenty-eight non-detected SVOCs, two non-detected organochlorine pesticides, two non-detected chlorinated herbicides, and five non-detected organophosphorous pesticides were identified as ecological COPCs for Ensenada Honda aquatic food web exposures based on the lack of ingestion-based screening values for one or both of the aquatic receptors.

4.6.2 Uncertainties Associated With the Screening-Level Ecological Risk Assessment

The procedures used in this evaluation to assess risks to ecological receptors, as in all such assessments, are subject to uncertainties because of the limitations of the available data and the need to make certain assumptions and extrapolations based on incomplete information. The major uncertainties associated with the screening-level ERA for SWMU 1 and 2 and their effect on risk conclusions are presented and discussed below.

Analytical Data

- The analytical data used in the screening-level ERA for surface soil and subsurface soil were obtained from samples collected during the 1992 Supplemental Investigation (surface soil and subsurface soil) and 1996 RFI field investigation (surface soil). Given the age of these data, they do not represent current levels of potential exposure.

- A second source of uncertainty related to the analytical data applies to the spatial coverage of surface soil data at SWMUs 1 and 2. Surface soil samples were not collected from a large portion of each SWMU (see Figures 4-5 and 4-6). Although subsurface samples were collected from the 0.5 to 1.5-foot depth interval from those areas lacking surface soil data, the absence of data from the 0 to 0.5-foot depth interval at these locations is a source of uncertainty in the screening-level since it is not known if the surface soil data capture maximum concentrations.

- A third source of uncertainty related to the analytical data applies to "J" flagged data. Analytical data for many chemicals were were qualified as estimated, "J" because the results fall between the method detection limit (MDL) and method reporting limit (MRL). Although concentrations that fall between the MDL and MRL are considered detected, the confidence in the quantified values is low.

Reporting Limits

- Reporting limits for many chemicals exceeded surface soil, surface water, and sediment screening values. The specific media primary affected by elevated reporting limits were surface soil and sediment

Identification of Ecological COPCs

- Chemicals without available screening values were identified as ecological COPCs even if they were not detected. Non-detected chemicals with reporting limits greater than screening values were also identified as ecological COPCs in the screening-level ERA. This approach likely overstates the number of actual COPCs.

- A second source of uncertainty related to the selection of ecological COPCs applies to the use of NOAEL-based screening values in risk calculations for upper trophic level receptors. The use of NOAEL-based screening values is extremely conservative since they give no indication as to how much higher a concentration must be before adverse effects are observed.

Exposure Point Concentrations

- The maximum measured concentration provides a conservative estimate for immobile biota or those with a limited home range. The most realistic exposure estimates for mobile species with relatively large home ranges and for species populations (even those that are immobile or have limited home ranges) are those based on the mean chemical concentrations in each medium to which these receptors are exposed. This is reflected in the wildlife dietary exposure models contained in the Wildlife Exposure Factors Handbook (USEPA, 1993b), which specify the use of average media concentrations. Given the mobility of the upper trophic level receptor species used in the screening-level ERA, the use of maximum chemical concentrations (rather than mean concentrations) to estimate the exposure via food webs is very conservative. The use of mean concentrations to estimate exposure in Step 3a of the baseline ERA is more likely to provide a more accurate estimate of potential risks at SWMUs 1 and 2.

Media-Specific Screening Values

- Literature-based toxicological thresholds were not available for many of the chemicals evaluated in the screening-level ERA. Furthermore, many of the surface soil screening values used in the comparison to surface soil analytical data were background-based concentrations (see Table 4-7). Because the background-based screening values do not represent effect concentrations, their use in the screening-level ERA likely resulted in an overstatement of the actual number of ecological COPCs.

- A second source of uncertainty related to media-specific screening values applies to cyanide. For all media, only total cyanide data were available for evaluation in the screening-level ERA. This analysis incorporates cyanide in all its forms, including the free form (both hydrogen cyanide [HCN] and the cyanide ion [CN-], weak metal complexes (including those with copper, zinc, and nickel), and tightly bound metal complexes (including those with silver, gold, cobalt, and iron) (Souren, 2000). Cyanide speciation in the environment is a function of a variety of chemical, physical, and biological parameters and processes, including the cyanide source, the availability of metal ions, the presence of certain degrading microorganisms, light, temperature, pH, and redox potential (Kjeldsen, 1998 and Souren, 2000). Iron-cyanide complexes are both the most common and the most stable in the environment. This form is considered relatively inert and has a half-life of 100 to 1000 years (Ghosh et al., 1999 and Kjeldsen, 1998). Free cyanide, however is very rare in soils (Shifrin et al., 1996), and is often negligible when compared to complexed forms at contaminated sites (0.5 to 5 percent [Meeussen et al 1992]; 2 percent [Ghosh et al., 1999]). If present, free cyanide will primarily (70 to 100 percent) be in the form at common soil pHs (6 to 9 standard units), which rapidly volatilizes and diffuses through the soil (Kjeldsen, 1998).

  Cyanide toxicity is highly relative to its chemical form. For example, a safe dose of the iron complexed form for humans is 2 grams/day, while a one-time lethal dose of weakly bound thiocyanates can range from 50 to 80 mg/kg body weight/day, and free cyanide is

fatal in doses of 0.5 to 3.5 mg/kg body (Kjeldsen, 1998). Though the literature directly relating to cyanide speciation and toxicity to ecological receptors is limited, both the scientific and regulatory community recognize that it is free, biologically available form of cyanide in the environment that is of concern (Eisler, 1991, MADEP, 1998, Meeussen et al., 1992, and Sample et al., 1997b). Therefore, exposure estimates based on total cyanide likely overestimated the potential for risk.

- A third source of uncertainty related to media-specific screening values applies to surface soil screening values. When a toxicological threshold was available for both plants and invertebrates, the minimum value was selected as the screening value. For several chemicals, only a plant or earthworm toxicological threshold was available from the literature. It was assumed in the screening-level ERA that the screening value selected for these chemicals are protective of both receptor communities. If a given chemical does not have an available screening value for both terrestrial plants and invertebrates, this approach will result in an underestimation of potential risks if the screening value is not based on the most sensitive receptor community.

- A fourth source of uncertainty related to media-specific screening values applies to surface water screening values. USEPA NAWQC were used as surface water screening values for arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, and zinc. Although USEPA NAWQC for these nine metals are expressed in terms of the dissolved fraction in the water column, the surface water screening values used were expressed as the total recoverable concentrations. Because the filtered fraction more closely approximates the bioavailable fraction of these nine metals in the water column (USEPA, 1999b and 2002a), use of screening values expressed as total recoverable concentrations likely resulted in an overstatement of the actual number of ecological COPCs. This uncertainty does not apply to filter feeding organisms (e.g., clams and mussels), which may receive exposure from total metals in surface water.

- A fifth source of uncertainty related to media-specific screening values applies to sediment screening values. The literature-based, bulk-sediment toxicological thresholds (i.e., TELs, ER-Ls, and AETs) used as screening values in the screening-level ERA do not take into consideration site-specific conditions that can influence chemical bioavailability and toxicity. These conditions include total organic carbon (TOC) and acid volatile sulfide (AVS), which can influence the bioavailability of organic chemicals and metals, respectively. As exposure does not necessarily equate to risk, bulk-sediment screening values may overstate to benthic macroivertebrates.

<u>Ingestion-Based Screening Values</u>

- Data on the toxicity of many chemicals to the receptor species were sparse or lacking, requiring the extrapolation of data from other wildlife species or from laboratory studies with non-wildlife species. This is a typical limitation for ecological risk assessments because so few wildlife species have been tested directly for most chemicals. The uncertainties associated with toxicity extrapolation were minimized through the selection of the most appropriate test species for which suitable toxicity data were available. The factors that were considered in selecting a test species to represent a receptor species included taxonomic relatedness, trophic level, foraging method, and similarity of diet. Regardless, the use of NOAEL and LOAEL values derived from laboratory studies with non-wildlife species may have resulted in an overstatement or understatement of potential risks if the sensitivities of the receptor and test species differ appreciable

- A second source of uncertainty related to the derivation of ingestion screening values applies to metals. Most of the toxicological studies on which the ingestion-based screening values for metals were based used forms of the metal (such as salts) that have high water solubility and high bioavailability to receptors. Since the analytical samples on which site-specific exposure estimates were based measured total metal concentrations, regardless of form, and these highly bioavailable forms are expected to compose only a fraction of the total metal concentration, this is likely to result in an overestimation of potential risks for these chemicals.

- A third source of uncertainty related to the derivation of ingestion screening values concerns the use of uncertainty factors. For example, in some cases NOAELs were extrapolated from LOAELs using an uncertainty factor of ten. This approach is likely to be conservative since Dourson and Stara (1983) determined that 96 percent of the chemicals included in a data review had LOAEL/NOAEL ratios of five or less.

- A fourth source of uncertainty related to the derivation of ingestion screening values also concerns uncertainty factors. The NOAEL and LOAEL values summarized in Tables 4-10 and 4-11 were not adjusted to reflect interspecies differences between the test species and receptor species. Wentsel et al. (1996) recommend an extrapolation factor of two (2) for NOAELs/LOAELs derived from studies using test species that are not within the same genus as the receptor species and an extrapolation factor of four (4) when test species are not within the same family/order as the receptor species. The authors further recommend an extrapolation factor of two (2) for threatened or endangered species. For a given chemical, if NOAEL and LOAEL values used in this ERA were derived from studies with test species that are less sensitive than the receptor species, the lack of interspecies extrapolations resulted in an underestimation of potential risks.

- To evaluate the uncertainty associated with the lack of NOAEL and LOAEL adjustments using the extrapolation factors discussed above, risk estimates for the West Indian manatee were derived by adjusting NOAEL and LOAEL values using an extrapolation factor of eight (8). An extrapolation factor of eight reflects interspecies differences between the test species and the West Indian manatee (factor of four) and the endangered status of the West Indian manatee (factor of two). As evidenced by Table 4-38a, an extrapolation factor of eight would result in the identification of nine detected metals (antimony, arsenic, barium, cadmium, copper, mercury, selenium, vanadium, and zinc) as ecological COPCs for West Indian manatee food web exposures at SWMU 1. This compares to the identification of three detected metals (i.e., arsenic, mercury, and selenium) as ecological COPCs when NOAEL and LOAEL values were not adjusted to reflect interspecies differences and the Federal status of the West Indian manatee (see Table 4-29).

- An extrapolation factor of eight would result in the identification of ten detected metals (antimony, arsenic, barium, cadmium, copper, lead, mercury, selenium, vanadium, and zinc) as ecological COPCs for West Indian manatee food web exposures at SWMU 2 (see Table 4-38b). This compares to the identification of three detected metals (arsenic, mercury, and selenium) as ecological COPCs when NOAEL and LOAEL values are not adjusted to reflect interspecies differences and the Federal status of the West Indian manatee (see Table 4-38).

- A fifth source of uncertainty related to the derivation of ingestion-based screening values applies to mercury. The NOAEL-based mercury screening value used for birds (0.0064 mg/kg-BW/day) was based on an organometallic (methylated) form (methyl mercury

dicyandiamide). Avian screening values for inorganic forms of mercury are substantially higher (0.45 mg/kg-BW/day for mercuric chloride [see Table 4-10]). The USEPA (2001b) reports that less than 20 percent of the total mercury in the water column and 0.5 to 5.3 percent of the total mercury in soil is present as methyl mercury. The USEPA (2001b) further reports that sediment mercury levels follow the same trends as soil in regard to methyl mercury percentages. These data indicate that methyl mercury represents a fraction of the total mercury in surface water, sediment, and surface soil. As such, the use of an ingestion-based screening value based on a methylated form assumes that 100 percent of the detected mercury is present as methyl mercury, likely resulting in an overestimation of potential risk.

Ecological Receptors

- Although exposure pathways to terrestrial reptiles and amphibians and aquatic reptiles are likely to be complete, reptilian and amphibian species were not selected as ecological receptors because the life history and toxicological database concerning the effects on reptiles and amphibians is severely limited. It was assumed that any terrestrial reptiles and amphibians and aquatic reptiles present at SWMUs 1 and 2 are not exposed to significantly higher concentrations of chemicals and are not more sensitive to chemicals than the other receptor species evaluated in the risk assessment. If reptiles and amphibians are exposed to significantly higher concentrations of chemicals and/or are more sensitive to chemicals than the other receptor species evaluated by this ERA, this approach will result in an underestimation of potential risks to herpetofauna.

Exposure Routes

- Although inhalation and/or dermal adsorption represent potential exposure routes for upper trophic level receptors, they were not evaluated in the screening-level ERA because they were considered insignificant relative to ingestion exposures (see Section 4.3.1.3). While this is a reasonable assumption for the terrestrial birds selected as ecological receptors, the exclusion of inhalation and dermal adsorption represents a source of uncertainty that may have results in an underestimation of potential risks.

Food Web Exposure Modeling

- Chemical concentrations in terrestrial and aquatic food items (plants, earthworms, and small mammal omnivores) were modeled from measured media concentrations and were not directly measured. The use of generic, literature-derived exposure models and BAFs introduces some uncertainty into the resulting estimates and may have resulted in an overstatement or understatement of potential risks. The values selected and the methodology employed was intended to provide a reasonable estimate of potential food web exposure concentrations.

- A second source of uncertainty related to the food web models is the use of default assumptions for exposure parameters such as BCFs and BAFs. Although BCFs or BAFs for many chemicals were readily available from the literature and were used in the ERA, the use of a default factor of 1.0 to estimate the concentration of some chemicals in receptor prey items is a source of uncertainty. The assumption that the chemical body burden in the prey item is at the same concentration as in soil is conservative for chemicals that are not known not to accumulate to any significant degree. However, if a chemical does accumulate in receptor prey items, the use of a default factor of 1.0 may

have resulted in an underestimation of potential risks to the upper trophic level receptors evaluated by this ERA.

- A third source of uncertainty related to the food web models is the use of unrealistically conservative exposure parameters. The use of maximum ingestion rates and minimum body weights resulted in a conservative estimate of exposure. In addition, AUFs were assumed to equal one. This is a conservative assumption since a significant percentage of each upper trophic level receptor species time could be spent foraging off-site in areas not impacted by site-related chemicals or areas where chemical concentrations are expected to be significantly lower. For example, the Florida population of the West Indian manatee ranges over fairly large areas during the summer (covering up to 200 linear km of river or coastline). Unlike the Florida population, which aggregates within the confines of natural or artificial warm water refuges during winter periods (USFWS, 1996b), there is no evidence of periodicity in manatee behavior in Puerto Rico (USFWS, 1986). As such, it cannot be expected that West Indian manatees would exclusively forage within the Ensenada Honda.

Chemical Mixtures

- Information on the ecotoxicological effects of chemical interactions is generally lacking, which required (as is standard for ecological risk assessments) that the chemicals be evaluated on a compound-by-compound basis during the comparison to screening values. This could result in an underestimation of risk (if there are additive or synergistic effects among chemicals) or an overestimation of risks (if there are antagonistic effects among chemicals).

4.6.3 Screening-Level Ecological Risk Assessment Decision Point and Recommendations

The results of the screening-level ERA for each SWMU indicated that, based on a set of conservative exposure assumptions, there are one or more chemicals that may present risks to one or more receptors species/receptor groups evaluated (see Tables 4-39 and 4-40). Therefore, the ERA process at SWMUs 1 and 2 proceeded to Step 3a of the baseline ERA. This evaluation is presented in the sections that follow.

4.7 Step 3a of the Baseline Ecological Risk Assessment

The results of the screening-level risk calculation indicated that, based on a set of conservative assumptions, there are one or more chemicals that may present a risk to ecological receptor groups and/or specific species. As such, the ERA process at SWMUs 1 and 2 proceeded to the baseline risk assessment.

According to Superfund guidance (USEPA, 1997), Step 3 initiates the problem formulation phase of the baseline ERA. Under Navy guidance (CNO, 1999), the baseline ERA is defined as Tier 2, and the first activity under Tier 2 is Step 3a (see Figure 4-1). In Step 3a, the conservative assumptions employed in the screening-level ERA (Tier 1) are refined and risk estimates are recalculated using the same conceptual model. Step 3a may also include consideration of background data, the frequency at which chemicals were detected, and chemical bioavailability.

The specific assumptions, parameters, and methods that were modified for the recalculation of media-specific and food web HQ values are identified below, along with justification for each modification. These refinements and methods were used in Step 3a of the baseline ERA to weigh the evidence of potential risk for each ecological COPC identified for each media and receptor to

determine whether the ecological COPCs should be identified as potential ecological risk drivers and carried on to Step 3b of the baseline ERA.

- Refined risk estimates for surface soil, surface water, and terrestrial and aquatic food web exposures (excluding the West Indian manatee) were derived using average (arithmetic mean) chemical concentrations. For individual receptor species, average chemical concentrations provide a better estimate of the likely level of chemical exposure because each receptor would be expected to forage in several different areas of the site, and, in many cases, off-site. Average concentrations are also appropriate for evaluating impacts *to populations* of lower trophic level receptors (i.e., terrestrial invertebrates, terrestrial plants, aquatic plants, benthic invertebrates, and fish). Because some of these receptors are relatively immobile, *individuals* are likely to be impacted by locations of maximum concentrations. However, evaluation of the average exposure case is more indicative of the level of impact that might be expected at the *population* level. Based on the status of the West Indian manatee (Federally endangered species), refined risk estimates using mean chemical concentrations were not derived for this receptor.

- Literature-based BCFs and BAFs based on, or modeled from, central tendency estimates (e.g., mean, median, midpoint) were used in place of maximum or high-end (e.g., 90th percentile) estimates for many chemicals. An assumed BCF/BAF of 1.0 was still used for those chemicals lacking a literature-based BAF/BCF. The refined BCFs and BAFs for those chemicals carried into Step 3a of the baseline ERA are summarized in Tables 4-41 (soil-to-terrestrial plant BCFs and soil-to-terrestrial invertebrate BAFs), 4-42 (soil-to-small mammal BAFs), and 4-43 (sediment-to-aquatic invertebrate BAFs). The surface water-to-fish BAFs used in the screening-level ERA (see Table 4-17) also were used in Step 3a since many of the values shown are based on a single study. As discussed above, refined risk estimates were not derived for the West Indian manatee. For this reason, sediment-to-plant BCFs based on central tendency estimates were not identified from the literature.

- Central tendency estimates (e.g., mean, median, midpoint) for body weight and food ingestion rate (see Table 4-44) were used to develop exposure estimates for upper trophic level receptors rather than the minimum body weights and maximum food ingestion rates used in the screening-level ERA. The use of central tendency estimates is more relevant because they represent the characteristics of a greater proportion of the individuals in the population. The evaluation of food web exposures still assumed an AUF of 1.0. Because refined risk estimates were not derived for the West Indian manatee, Table 4-25 does not include central tendency estimates for West Indian manatee body weight and food ingestion rate.

- In addition to the NOAELs-based risk estimates used in the screening-level ERA, consideration was also given to food web exposure risk estimates based on LOAELs and MATCs. However, NOAEL-based risk estimates were used exclusively for the West Indian manatee.

- Consideration was given to available surface soil, surface water, and sediment background data provided in Appendix G (G.22 through G.33). This was accomplished by statistically comparing site concentrations to background concentrations in accordance with Navy guidance (NFESC, 2002a and 2002b). The process used to statistically evaluate the SWMUs 1 and 2 surface soil, subsurface soil, surface water, and sediment data is depicted on Figure 4-19a. As evidenced by the figure, statistical comparisons included descriptive summaries of each data set (maximum, minimum, and mean

concentrations), statistical tests on the mean/median of the distributions (i.e., student's t-test, Wilcoxin rank sum test, Gehan test, and/or Satterthwaite's t-test), and statistical tests on the right tail of the distributions (i.e., quantile test and/or slippage test). The significance level (the probability criteria for rejecting the null hypotheses that data sets were sampled from the same population) was set at 0.05 for all statistical tests (NFESC, 2002a and 2002b).

- The background surface soil data used in Step 3a of the baseline ERA were basewide background data and SWMU 9 background data. Basewide and SWMU 9 background surface soil sampling locations are depicted on Figures 4-20 and 4-21, respectively. The basewide background surface soil sampling locations and associated analytical data were previously presented and discussed in the Revised Draft RCRA Facility Investigation Report for Operable Unit 3/5, Naval Station Roosevelt Roads, Ceiba, Puerto Rico (Baker, 1999; approved by the USEPA September 9, 1999). The SWMU 9 background surface soil sampling locations and associated analytical data were previously presented and discussed in the Final Corrective Measures Study Investigation Report for SWMU 9, Naval Station Roosevelt Roads, Ceiba, Puerto Rico (Baker, 2003a; approved by the USEPA February 19, 2003). Both background surface soil data sets were combined into a unified data set for use in Step 3a of the baseline ERA. The Final CMS investigation report for SWMU 9 contained an evaluation that justified a unified basewide and SWMU 9 background surface soil data set.

- The estuarine wetland and open water sampling locations are depicted on Figures 2-1 and 4-22, respectively. Estuarine wetland surface water and sediment sampling locations and associated analytical data were previously presented and discussed in Baker (2003a) and the Draft Additional Data Collection Work Plan for SWMU 9, Naval Station Roosevelt Roads, Ceiba Puerto Rico (Baker, 2003b; approved by the USEPA June 3, 2003). The Final Additional Data Collection Investigation Report, Tow Way Fuel Farm, Naval Station Roosevelt Roads, Ceiba, Puerto Rico (Baker 2003c; approved by the USEPA June 10, 2003) and Baker (2003b) contained descriptions of the open water surface water and sediment sampling locations and associated analytical data.

- A background subsurface soil dataset was not available for statistical comparison to site subsurface soil data. However, the site subsurface soil data were statistically compared to background surface soil data. Although the depth intervals for site subsurface soil and background surface soil overlap, the comparison introduces a source of uncertainty in Step 3a of the baseline ERA.

- As exposure does not necessarily equate to risk, consideration was given to site-specific factors that can affect the bioavailability of chemicals in surface water and sediment to aquatic receptor groups. For surface water, consideration was given to the concentration of metals in the dissolved (unfiltered) fraction. For sediment, consideration was given to the affect TOC and acid volatile sulfide (AVS) have on the bioavailability of organic and inorganic chemicals, respectively.

- Chemicals that were not identified as ecological COPCs because maximum detected concentrations (or maximum reporting limits in the case of non-detected chemicals) were less than media-specific screening values were not evaluated in Step 3a of the baseline ERA since a conclusion of no unacceptable risk can be made with high confidence.

- Chemicals that were not detected but identified as ecological COPCs in the Step 2 screening-level risk calculation due to the lack of screening values were also not

evaluated in Step 3a of the baseline ERA. It is not possible to quantitatively address the potential for risk from chemicals which are not detected and which do not have established screening values with which to compare them to. Even considerations of the most conservative measurement (the maximum reporting limit) are not informative when no threshold value has been established. Because of these limitations, the approach follows that outlined in the NCP (40 CFR 300 Appendix A), which does not establish a release when the sample measurement is less than the contract required detection limit as determined by an EPA-certified laboratory. As all samples were analyzed by a certified laboratory, and were validated by an independent third party, the exclusion of non-detected chemicals is considered reasonable and appropriate. Although eliminated from further evaluation, they remain ecological COPCs but are not considered potential ecological risk drivers (they are likely not present at ecologically critical concentrations). It is additionally noted that any site-specific studies, which may be conducted during a baseline ecological risk assessment, will indirectly evaluate the impacts of non-detected chemicals.

4.7.1 SWMU 1 Refined Risk Calculation and Risk Evaluation

Refined media-specific screening evaluations for SWMU 1 abiotic media and refined food web exposure evaluations for SWMU 1 terrestrial and aquatic upper trophic level receptors are presented and discussed in the sections that follow. As discussed in Section 4.6.1, detected chemicals with maximum concentrations and/or maximum exposure doses greater than screening values, as well as detected chemicals lacking screening values were identified as ecological COPCs in Step 2 of the screening-level ERA. Non-detected chemicals with maximum reporting limits and/or maximum exposure doses greater than screening values, as well as non-detected chemicals lacking screening values also were identified as ecological COPCs in the Step 2 risk calculations. Only those detected and non-detected chemicals with maximum concentrations and/or maximum exposure doses greater than screening values, and those detected chemicals lacking screening values were addressed by the refined screening-level risk calculations. Although eliminated from further discussion in Step 3a, non-detected chemicals lacking screening values remain ecological COPCs, but are not considered potential ecological risk drivers.

4.7.1.1 Refined Risk Calculation and Risk Evaluation for Surface Soil

Table 4-20 presented the results of the Step 2 screening-level risk calculation for SWMU 1 surface soil. Benzo(b)fluoranthene, pyrene, 4,4'-DDT, 4,4'-DDD, 4,4'-DDE, antimony, cadmium, chromium, cobalt, copper, lead, mercury, nickel, silver, tin, vanadium, and zinc were identified as ecological COPCs in Step 2 of the screening-level ERA because maximum detected concentrations exceeded surface soil screening values. One VOC (acetone) was detected and identified as an ecological COPC based on the lack of a surface soil screening value. One non-detected VOC (vinyl chloride), six non-detected SVOCs, and four non-detected organochlorine pesticides (alpha-chlordane, gamma-chlordane, methoxychlor, and toxaphene) were identified as ecological COPCs because maximum reporting limits exceeded screening values. Figure 4-9 presented the concentration distribution across the site for each detected chemical identified as an ecological COPC with a maximum HQ grater than or equal to 1.0. Table 4-45 presents the results of the refined screening-level risk calculation for those detected and non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA with a maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation for surface soil is presented below.

As discussed above, one-non-detected VOC (vinyl chloride), six non-detected SVOCs (1,2,4,5-trichlorophenol, 1,3,5-trinitrobenzene, 2,4-dimethylphenol, m,p-cresol, o-cresol, and pentachlorophenol), and four non-detected organochlorine pesticides (alpha-chlordane, gamma-chlordane, methoxychlor, toxaphene) were identified as ecological COPCs because maximum reporting limits exceeded surface soil screening values. The mean concentration of vinyl chloride, pentchlorophenol, alpha-chlordane, gamma-chlordane, methoxychlor, and toxaphene were less than surface soil screening values (HQs = 0.55, 0.58, 0.34, 0.34, 0.34, and 0.68, respectively), while mean concentrations for 1,2,4,5-tetrachlorobenzene, 1,3,5-trichlorobenzene, 2,4-dimethylphenol, m,p-cresol, and o-cresol were greater than screening values (see Table 4-45). Because chemicals with mean HQs less than surface soil screening values have little potential to impact terrestrial invertebrate and plant populations, they are not identified as potential ecological risk drivers for SWMU 1 surface soil, and no additional evaluation is recommended.

Surface soil screening values used for the SVOCs with mean HQs greater than 1.0 were background-based Canadian soil quality guidelines (CCME, 2002) protective of agricultural land uses. Because these screening values are not based on toxicological data, there is uncertainty in their use as chemical exposure levels (background-based screening values do not represent threshold effect concentrations), likely resulting in an overestimation of risk estimates.

Furthermore, because 1,2,4,5-tetrachlorobenzene, 1,3,5-trinitrobenzene, 2,4-dimethylphenol, m,p-cresol, and o-cresol were all non-detect, they are likely not present at ecologically concentrations. For the reasons discussed above, 1,2,4,5-tetrachlorobenzene, 1,3,5-trichlorobenzene, 2,4-dimethylphenol, m,p-cresol, and o-cresol also are not identified as potential ecological risk drivers for SWMU 1 surface soil, and no additional evaluation is recommended.

As discussed in the preceding paragraph, acetone was detected and identified as an ecological COPC in Step 2 of the screening-level ERA based on the lack of a surface soil screening value. This VOC was detected in one of eighteen surface soil samples at a concentration of 15 J ug/kg. With the exception of the surface soil screening value for vinyl chloride (11 ug/kg), this detection is less than available surface soil screening values for other VOCs (see Table 4-7). This single detected concentration also is two orders of magnitude below a soil ecological screening value established by the USEPA Region 5 for RCRA Appendix IX hazardous constituents (2,500 ug/kg [USEPA, 2003c]). Based on a low magnitude and frequency of detection (detected in one of eighteen samples at 15 J ug/kg) and the comparison of the single detected concentration to surface soil screening values for other VOCs and the USEPA Region 5 screening value, acetone is not identified as a potential risk driver for SWMU 1 surface soil, and additional evaluation is not recommended.

Mean concentrations of benzo(b)fluoranthene (HQ = 0.25) and pyrene (HQ = 0.27) were less than surface soil screening values (see Table 4-45). As evidenced by Figure 4-9, detections above the surface soil screening value for both PAHs were spatially limited to a single sample location [benzo(b)fluoranthene and pyrene were detected in 1SS06 at 1,500 ug/kg and 2,600 ug/kg, respectively]. The surface soil screening value used in the Step 2 and Step 3a risk calculations for benzo(b)fluoranthene and pyrene (1,200 ug/kg) was a chronic NOAEL value from a study that investigated the chronic toxicity of benzo(a)pyrene and benzo(b)fluoranthene to wheat (USEPA, 1999a [the chronic NOAEL was used as a surrogate screening value for pyrene]). Based on mean HQs less than 1.0 and the low magnitude and frequency of detections above the plant-based toxicological threshold identified above, benzo(b)fluoranthene and pyrene concentrations in SWMU 1 surface soil are not likely impacting terrestrial plant communities

A search of the literature on the toxicological effects of pyrene to soil flora and fauna identified several studies. Sverdrup et al. (2003) investigated the effect of pyrene on the seed emergence and early life-stage growth (fresh weight and dry weight) of three terrestrial plants: red clover (*Trifolium pretense*), ryegrass (*Lolium perenne*), and mustard (*Sinapsis alba*). After 21 days of exposure, pyrene concentrations causing a 20 percent reduction in seedling wet weight ranged from 49,000 ug/kg (red clover) to >1,000,000 ug/kg (ryegrass), while concentrations causing a 20 percent reduction in dry weight ranged from 56,000 ug/kg (red clover) to >1,000,000 ug/kg (ryegrass). 21-day red clover, ryegrass, and mustard seed emergence LC_{50} values were >1,000,000 ug/kg for each species. Sverdrup et al. (2002a) investigated the toxic effect of pyrene on *Enchytraeus cryticus* (enchytraeid worm) survival and reproduction. The 21-day LC_{50} value for pyrene was reported at >2,300,000 ug/kg, while the 21-day NOEC value based on *Enchytraeus cryticus* reproduction was reported at 18,000 ug/kg. Invertebrate toxicity data also were identified from the literature for an earthworm and springtail species. Sverdrup et al. (2002b) investigated the acute and chronic toxicity of pyrene to the earthworm *Eisenia veneta*. The 28-day LC_{50} value based on earthworm survival was 155,000 ug/kg and the 28-day NOEC value based on *Eisenia veneta* growth was 29,000 ug/kg. Sverdrup et al. (2001) investigated the effect of pyrene on the survival and reproduction of the springtail *Folsomia fimetaria* (Order Collembola). The 21-day LC_{50} and NOEC value based on springtail reproduction was 53,000 ug/kg and 13,000 ug/kg, respectively. The last study identified from the literature (Sverdrup et al., 2002c) investigated the toxic effect of pyrene on soil–nitrifying bacteria (i.e., soil nitrification process). The 28-day NOEC value of 79,000 ug/kg was reported for effects on soil nitrification.

As evidenced by the information presented above, literature-based toxicity data were available for terrestrial plants, invertebrates, and microorganisms. The maximum detected pyrene concentration (2,600 ug/kg in 17-SS02) is an order of magnitude less than the minimum chronic effect concentration identified from the literature (21-day NOEC of 13,000 ug/kg based on springtail reproduction).

The discussion presented above has shown that pyrene is not present in SWMU 1 surface soil at concentrations greater than literature-based plant, invertebrate, and microbial toxicity values specific to this PAH. For this reason, pyrene is not identified as a potential ecological risk driver, and no additional evaluation is recommended. Although toxicity data were not available from the literature for benzo(b)fluoranthene, this PAH also is not identified as a potential ecological risk driver based on a mean HQ less than 1.0, the low magnitude and frequency of detections above a plant-based surface soil screening value derived using this PAH as the test material.

As evidenced by Table 4-45, mean concentrations for 4,4'-DDT, 4,4'-DDD, and 4,4'-DDE exceeded surface soil screening values (HQs = 4.00, 1.21, and 1.28, respectively). 4,4'-DDT was detected in six of twenty-eight surface soil samples at concentrations greater than the surface soil screening value (610 ug/kg at 1SB03-00, 1,300 ug/kg at 1SS06, 930 ug/kg at 1SD01, 810 ug/kg at 1SS10, 1,700 ug/kg at 1SS13, and 28,000 ug/kg at 1SS16 [see Figure 4-19]). 4,4'-DDD and 4,4'-DDE were each detected in one of twenty-eight surface soil sample at a concentration greater than the surface soil screening value (13,000 ug/kg and 43,000 ug/kg, respectively, in 1SS16). As evidenced by Step 2 risk estimates (i.e., HQ values), the frequency of detections above surface soil screening values was high for each pesticide, ranging from 32.5 for 4,4'-DDD to 108 for 4,4'-DDT. Based on the magnitude of maximum detected concentrations above the surface soil screening value and, in the case of 4,4'DDT, the frequency of detections above surface soil screening value, 4,4'-DDT, 4,4'-DDD, and 4,4'-DDE are identified as potential ecological risks drivers for SWMU 1 surface soil, and additional evaluation is recommended.
Mean cobalt (HQ = 0.96), nickel (HQ = 0.64), silver (HQ = 0.13), and tin (HQ = 0.40) concentrations were less than surface soil screening values. As evidenced by Figure 4-9, silver detections above the surface soil screening value were spatially limited to a single sample location (2.6 J mg/kg in 1SS07). Tin was detected in three of twenty-four surface soil samples at concentrations greater than the surface soil screening value (181 mg/kg in 1SD01, 100 mg/kg in 1SS10, and 88 mg/kg in 1SS13), nickel was detected in five of twenty-four surface soil samples at concentrations greater than surface soil screening values 38.1 mg/kg in 1SS07, 63.3 mg/kg in 1SD01, 31.6 mg/kg in 1SD02, 31 mg/kg in 1SS10, and 76 mg/kg in 1SS13), while cobalt was detected in seven of eighteen surface soil samples at concentrations greater than the surface soil screening value (22.7 mg/kg in 1MW03, 25.1 mg/kg in 1SB01, 28.2 mg/kg in 1SB02, 21.5 mg/kg in 1SB03, 30.9 mg/kg in 1SS04, 27.5 mg/kg in 1SS05, and 20.2 mg/kg in 1SS06).

Mean antimony (HQ = 1.02), cadmium (HQ = 1.28), chromium (HQ = 69.1), copper (HQ = 3.38), lead (HQ = 2.95), mercury (HQ = 1.40), vanadium (72.6), and zinc (HQ = 6.46) concentrations were greater than surface soil screening values. Antimony was detected in five of twenty-four surface soil samples at concentrations greater than the surface soil screening value (9.4 J mg/kg in 1SS07, 23.6 J mg/kg in 1SD01, 14.5 J mg/kg in 1SD02, 23 J mg/kg in 1SS10, and 20 J mg/kg in 1SS13, cadmium was detected in five of eighteen surface soil samples at concentrations greater than the surface soil screening value (83.8 mg/kg in 1SS07, 4.7 mg/kg in 1SD01, 5.1 mg/kg in 1SS10, 9.4 mg/kg in 1SS11, and 12 mg/kg in 1SS13), mercury was detected in six of twenty-four surface soil samples at concentrations greater than the surface soil screening value (0.11 mg/kg in 1SS08, 0.85 mg/kg in 1SD01, 0.2 mg/kg in 1SD02, 0.44 mg/kg in 1SS10, 0.15 mg/kg in 1SS11, and 0.59 mg/kg in 1SS13), lead was detected in six of twenty-four surface soil samples at concentrations greater than the surface soil screening value (79.4 mg/kg in 1SS06, 101 mg/kg in 1SS07, 659 J in 1SD01, 966 J mg/kg in 1SD02, 680 J mg/kg in 1SS10, and 660 J

mg/kg in 1SS13), and zinc was detected in ten of twenty-four surface soil samples at concentrations greater than the surface soil value (140J mg/kg in 1MW01-00, 61.6 mg/kg in 1SB03, 136J mg/kg in 1SS06, 223 mg/kg in 1SS07, 1,780 mg/kg in 1SD01, 1,100 mg/kg in 1SD02, 110 J mg/kg in 1SS09, 1,600 J mg/kg in 1SS10, 190 J mg/kg in 1SS11, and 2,000 J mg/kg in 1SS13). Copper was detected in fifteen of twenty-four surface soil samples at concentrations greater than the surface soil screening value. Copper concentrations above the screening value ranged from 50.8 mg/kg in 1SD03 to 1,020 mg/kg in 1SD01. Chromium and vanadium were detected in each surface soil sample at concentrations greater than the surface soil screening value.

The surface soil screening values used for antimony, cadmium, chromium, cobalt, copper, lead, mercury, nickel, vanadium, tin, and zinc in the Step 2 and Step 3a screening level risk calculations were either toxicological thresholds for invertebrates (Efroymson et al., 1997a) or toxicological thresholds for plants (Efroymson et al., 1997b). The USEPA has recently published Ecological Soil Screening Levels (SSLs) for antimony, cadmium, cobalt, and lead (USEPA, 2003d, 2003e, 2003f, and 2003g, respectively). Maximum and mean antimony and cobalt concentrations detected in SWMU 1 surface soil (23.6 J mg/kg and 5.12 mg/kg, respectively for antimony and 30.9 mg/kg and 19.2 mg/kg, respectively for cobalt) are less than their respective invertebrate-based ecological SSLs (78 mg/kg for antimony and 13 mg/kg for cobalt); however, plant-based ecological SSLs have not been established for these two metals. The maximum and mean cadmium concentration (83.8 mg/kg and 5.1 mg/kg) is less than the invertebrate-based ecological SSL established for this metal (140 mg/kg); however, the maximum cadmium concentration exceeds the plant-based ecological SSL of 32 mg/kg. Identical to cadmium, the maximum and mean lead concentration (966 J mg/kg and 147 mg/kg) is less than the invertebrate-based ecological SSL established for this metal (1,700 mg/kg), but exceeds the plant-based ecological SSL (110 mg/kg). Based on the comparison presented above, antimony, cadmium, cobalt, and lead can't be eliminated as potential ecological risk drivers. Additional evaluation of these four metals, as well as an evaluation of chromium, copper, mercury, nickel, silver, tin, vanadium, and zinc concentrations in SWMU 1 surface soil is presented in the paragraphs that follow.

To evaluate the significance of potential risks presented by antimony, cadmium, chromium, cobalt, copper, lead, mercury, nickel, silver, vanadium, tin, and zinc relative to background concentrations, the SWMU 1 surface soil data were statistically compared to background surface soil data (combined background dataset consisting of base background and SWMU 9 background surface soil data) in accordance with Navy guidance (NFESC, 2002a). Table 4-46 provides a summary and results of the statistical evaluation, while Figure 4-23 presents box plot diagrams illustrating the distribution of each data set. As evidenced by Table 4-46 and Figure 4-23, the distributions of chromium, cobalt, and vanadium concentrations in SWMU 1 surface soil are statistically equivalent to background levels, indicating that these metals are not likely to be site-related and not presenting a risk to terrestrial plants and invertebrates above background levels. The descriptive statistics presented in Table 4-46 support the distributional statistics. Maximum, mean, and 95 percent UCL concentrations for the cobalt and vanadium background data sets exceeded maximum, mean, and 95 percent UCL concentrations for the SWMU 1 data sets, while maximum, mean and 95 percent UCL concentrations for the SWMU 1 chromium data set were only slightly elevated above background chromium concentrations. Based on the descriptive and distributional statistics presented in Table 4-46, chromium, cobalt, and vanadium are not identified as potential risk drivers for SWMU 1 surface soil, and additional evaluation is not recommended.

A statistical evaluation of the silver analytical data could not be performed due to the absence of detected results in the background data set (see Table 4-46). Although a statistical comparison of

SWMU 1 surface soil and background data could not be performed, additional evaluation is not recommended for this metal. This recommendation is based on a mean HQ less than 1.0 (HQ = 0.13), the low spatial coverage of detected concentrations greater than the screening value (one of eighteen surface soil samples), and the low magnitude of detected concentrations greater than the surface soil screening value (the HQ based on the maximum detected concentration was only slightly elevated above 1.0 [HQ = 1.30]). It is noted that the screening value used in risk calculations for silver was a toxicological threshold for terrestrial plants (Efroymson et al., 1997b). An invertebrate-based toxicological threshold could not be identified from the literature for terrestrial invertebrates. As such, there is uncertainty in the exclusion of silver as a potential risk driver for terrestrial invertebrates. It is assumed in this ERA that terrestrial invertebrates are not more sensitive to silver than terrestrial plants.

The statistical evaluation of the nickel data sets was contradictory. The evaluation performed on the mean of the distributions (two sample t-test) concluded that nickel concentrations in SWMU 1 surface soil are elevated above background levels, while statistical tests performed on the right-tail of the distributions (qunatile test and slippage test) concluded that nickel concentrations in SWMU 1 surface soil are statistically equivalent to background concentrations. An examination of the descriptive statistics presented in Table 4-46 show that maximum, mean, and 95 percent UCL concentrations in SWMU 1 surface soil are slightly elevated above maximum, mean, and 96 percent UCL concentrations in background surface soil. Although detected in five of twenty-four surface soil samples at concentrations greater than the surface soil screening value of 30 mg/kg, the magnitude of detections above the screening value was low. This is evidenced by the maximum HQ value of 2.53. Based on a mean HQ less than 1.0 (HQ = 0.64) and the low magnitude of detected concentrations above the surface soil screening value, nickel is not identified as a potential ecological risk driver for SWMU 1 surface soil, and no additional evaluation is recommended.

The results of the distributional statistics performed on the copper and mercury data sets (mean or median of the distribution [Wilcoxin rank sum test and two sample t-test, respectively] and right tail of the distribution [quantile test and slippage test]) indicate that the concentration distributions for these two metals are statistically equivalent to background levels (see Table 4-46), while the statistical evaluations performed on the tin data sets (Gehan test, quantile test, and slippage test) concluded that SWMU 1 surface soil concentrations are elevated above background levels. The distributional statistics performed on the cadmium, lead, and zinc data sets were contradictory. For cadmium, the statistical method evaluating the mean of the distributions (Gehan test) concluded that SWMU 1 surface soil concentrations are elevated above background levels, while the evaluations performed on the right-tail of the distributions (quantile test and slippage test) concluded that SWMU 1 surface soil concentrations are statistically equivalent to background concentrations. Satterthwaite's t-test and slippage test concluded that the distribution of lead concentrations in SWMU 1 surface soil are statistically equivalent to background levels, while the quantile test concluded that the distribution of lead concentrations are elevated above background levels. Finally, the statistical method evaluating the median of the zinc distributions (Wilcoxin rank sum test), as well as the quantile test, concluded that the distribution of zinc concentrations in SWMU 1 surface soil are elevated above background levels, while the slippage test concluded that zinc concentrations in SWMU 1 surface soil are elevated above background levels.

The descriptive statistics presented in Table 4-46 show that mean, maximum, and 95 percent UCL concentrations for cadmium, copper, lead, mercury, tin, and zinc in SWMU 1 surface soil were elevated above background concentrations. The magnitude that maximum detected surface soil concentrations exceeded maximum background concentrations was high for several of these metals. For example, maximum detected cadmium, copper, lead, tin, and zinc concentrations in

SWMU 1 surface soil were 83.8 mg/kg, 1,020 mg/kg, 966 J mg/kg, 181 mg/kg, and 2,000 J mg/kg, respectively. This compares to maximum cadmium, copper, lead, tin, and zinc background concentrations of 0.92 J mg/kg, 250 mg/kg, 21 J mg/kg, 2.3 J mg/kg, and 106 mg/kg, respectively. The magnitude that maximum detected concentrations exceeded surface soil screening values also was high for cadmium, copper, lead, and zinc (maximum HQs = 21.0 for cadmium, 20.4 for copper, 19.3 for lead, and 40.0 for zinc, respectively [see Table 4-20]).

Although one or more of the statistics methods concluded that cadmium, copper, lead, mercury, and zinc concentrations in SWMU 1 surface soil are statistically equivalent to background levels, the magnitude that maximum detected concentrations exceeded maximum background concentrations and/or surface soil screening values indicates the presence of hot spots for each metal. An examination of the analytical data for each SWMU 1 surface soil sample showed that maximum detected copper and mercury, concentrations occurred in 1SD01 (1,020 mg/kg and 0.85 mg/kg, respectively). Elevated copper concentrations also were detected at 1SD02, 1SS10, and 1SS13 (608 mg/kg, 540 J mg/kg, and 740 J mg/kg, respectively), while elevated mercury concentrations also were detected at 1SS10 and 1SS13 (0.44 mg/kg and 0.59 mg/kg, respectively). The maximum detected cadmium concentration occurred at 1SS07 (83.8 mg/kg), while maximum detected lead and zinc concentrations occurred at 1SD02 and 1SS13, respectively (966 J mg/kg for lead and 2,000 J mg/kg for zinc). Elevated cadmium concentrations also were detected in the 1SD01, 1SS10, 1SS11, and 1SS13 surface soil samples, elevated lead concentrations also were detected in the 1SS07, 1SS10, and 1SS13 surface soil samples, while elevated zinc concentrations also were detected in 1SS07, 1SD01, 1SD02, and 1SS10 surface soil samples (see Appendix G.1). The surface soil analytical data indicate that elevated cadmium, copper, lead, mercury, and zinc concentrations are co-located at 1SS07, 1SD01, 1SD02, 1SS10, and/or 1SS13. As evidenced by Figure 4-9, 1SS07, 1SD01, 1SD02, 1SS10, and 1SS13 were collected from the same general area, along the southern boundary of SWMU 1. Several additional surface soil samples collected from this general location, including 1SS06, 1SS09, and 1SS11 also contained detected cadmium, copper, lead, mercury, and/or zinc concentrations greater than surface soil screening values. Based on mean HQs greater than 1.0, the frequency and magnitude of detections above surface soil screening values, and their co-location at multiple sample locations at concentrations greater than surface soil screening values, cadmium, copper, lead, mercury, and zinc are identified as potential ecological risk drivers for SWMU 1 surface soil, and additional evaluation is recommended.

The statistical evaluations performed on the tin data sets concluded that the distribution of tin concentrations in SWMU 1 surface soil is elevated above background levels. The descriptive statistics presented in 4-46 also show that maximum, mean, and 95 percent UCL concentrations for the SWMU 1 data set are elevated above background levels. An examination of the analytical data in Appendix G.1 indicates that the maximum detected tin concentration (181 mg/kg at 1SD01) is co-located with maximum copper and mercury concentrations. Elevated tin concentrations also were detected at sample locations containing elevated cadmium, copper, lead, mercury, and/or zinc concentrations (see Figure 4-9). Based on the magnitude of the maximum detection above the surface soil screening value and the co-location of elevated tin concentrations with elevated cadmium, copper, lead, mercury, and/or zinc concentrations, tin is identified as a potential ecological risk driver for SWMU 1 surface soil, and additional evaluation is recommended.

The statistical evaluation of the antimony data sets was impaired by the absence of detected results in background surface soil samples. Specifically, an evaluation of the right tail of the distributions (slippage test and quantile test) could not be performed. An examination of the SWMU 1 surface soil data indicates that concentrations greater than the surface soil screening value were detected in 1SS07 (9.4J mg/kg), 1SD01 (23.6 J mg/kg), 1SD02 (14.5 J mg/kg), 1SS10

(23 J mg/kg), and 1SS13 (20 J mg/kg). As discussed in the preceding paragraphs, these sample locations also represent hot spots or areas of elevated detections for cadmium, copper, lead, mercury, nickel, tin, and/or zinc. Based on the area encompassed by these five sample locations and the co-location of elevated concentrations with elevated cadmium, copper, lead, mercury, tin, and zinc concentrations, antimony is considered a potential ecological risk driver for SWMU 1 surface soil, and additional evaluation is recommended.

In summary, antimony, cadmium, copper, lead, mercury, tin, zinc, 4,4'-DDD, 4,4'-DDE, and 4,4'-DDT were identified as potential ecological risk drivers for surface soil at SWMU 1 and are recommended for additional evaluation in Step 3b of the ERA. Although acetone, benzo(b)fluoranthene, pyrene, chromium, cobalt, nickel, silver, and vanadium were detected and identified as ecological COPCs in Step 2 of the screening-level ERA, they are not considered potential risk drivers based on the discussion presented in the preceding paragraphs, and additional evaluation is not recommended. Additional evaluation also is not recommended for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

4.7.1.2 Refined Risk Calculation and Risk Evaluation for Subsurface Soil

Table 4-21 presented the results of the Step 2 screening-level risk calculation for SWMU 1 subsurface soil. Two pesticides (4,4'-DDT and 4,4'-DDE) and seven metals (antimony, cadmium, chromium, cobalt, copper, vanadium, and zinc) were identified as ecological COPCs in Step 2 of the screening-level ERA because maximum detected concentrations exceeded surface soil screening values. Two VOCs (acetone and carbon disulfide) were detected and identified as ecological COPCs due to the lack of a surface soil screening value. One non-detected VOC (vinyl chloride), six non-detected SVOCs, fifteen non-detected PAHs, one non-detected organochlorine pesticide (toxaphene), and two non-detected metals (mercury and silver) were identified as ecological COPCs because maximum reporting limits exceeded surface soil screening values. Figure 4-10 presented the concentration distribution across the site for each detected chemical identified as an ecological COPC with a maximum HQ greater than or equal to 1.0. Table 4-47 presents the results of the refined screening-level risk calculation for those detected and non-detected chemicals identified as ecological COPCs for subsurface soil in Step 2 of the screening-level ERA with a maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation for subsurface soil is presented below.

As discussed above, one non-detected VOC (vinyl chloride), six non-detected SVOCs (2,4,5-trichlorophenol, 2,4-dichlorophenol, 2,4-dimethylphenol, 2-chlorophenol, o-cresol, and pentachlorophenol), fifteen non-detected PAHs, one non-detected organochlorine pesticide (toxaphene), and two non-detected metals (mercury and silver) were identified as ecological COPCs because maximum reporting limits exceeded surface soil screening values. Mean vinyl chloride, 2,4,5-trichlorophenol, 2,4-dichlorophenol, 2-chlorophenol, mercury, and silver concentrations, as well as mean concentrations for each of the non-detected PAHs, were less than surface soil screening values, while mean 2,4-dimethylphenol, o-cresol, pentachlorophenol, and toxaphene concentrations exceeded screening values (see Table 4-47). Non-detected chemicals with mean HQs less than 1.0 have little potential to impact terrestrial invertebrate and plant populations. For this reason, they are not identified as potential ecological COPCs for SWMU 1 surface soil, and no additional evaluation is recommended. Additional evaluation also is not recommended for the non-detected chemicals with mean HQs greater than 1.0. Surface soil screening values used for 2,4-dimethylphenol and o-cresol in the Step 2 and Step 3a screening-level risk calculations were background-based Canadian soil quality guidelines (CCME, 2002) protective of agricultural land uses. The surface soil screening value used for toxaphene also was a background-based screening value. Because these screening values are not based on toxicological data, there is uncertainty in their use as chemical exposure levels (background-

based screening values do not represent threshold effect concentrations, likely resulting in an overestimation of risk estimates). Furthermore, because 2,4-dimethylphenol, o-cresol, and toxaphene (as well as pentachlorophenol) were not detected, they are likely not present at ecologically critical concentrations.

Acetone and carbon disulfide were detected and identified as ecological COPCs in Step 2 of the screening-level ERA due to the lack of surface soil screening values. Both VOCs were detected in four of six subsurface soil samples collected at SWMU 1. Detected concentrations ranged from 38 J ug/kg to 83 J ug/kg for acetone and 10 ug/kg to 18 ug/kg for carbon disulfide. Maximum detected acetone and carbon disulfide concentrations (83 J ug/kg and 18 ug/kg, respectively) are less than available surface soil screening values for other VOCs (excluding the surface soil screening value for vinyl chloride [see Table 4-7]). Maximum concentrations also are less than USEPA Region 5 soil screening values (2,500 ug/kg for acetone and 94.1 ug/kg, respectively). Based on the discussion presented above, it is not likely that these two VOCs are present at ecologically relevant concentrations, and additional evaluation is not recommended.

The mean concentration of 4,4'-DDT was greater than the surface soil screening value (HQ = 2.59). Detections above the surface soil screening value were spatially limited to two subsurface soil sample locations (3,500 CD ug/kg at 05SS130 and 960 J ug/kg at 1SS16). Given the magnitude of the maximum detection above the screening value, 4,4'-DDT is considered a potential risk drive for SWMU 1 subsurface soil, and additional evaluation is recommended in Step 3b of the baseline ERA.

The mean concentration of 4,4'DDE was less than the surface soil screening value (HQ = 0.28). This biodegradation product of 4,4'-DDT was detected in six of nine subsurface soil samples at concentrations ranging from 1.8 J ug/kg in 05SS126 to 520 J ug/kg in 05SS130. Detected concentrations at two locations exceeded the surface soil screening value (480 J ug/kg at 05SS103 and 520 J ug/kg at 1SS16; see Figure 4-10). Although the magnitude of detected concentrations above the surface soil screening value is low (maximum HQ value of 1.30; see Table 4-21), detections above the screening value are co-located at sample locations containing maximum 4,4'-DDT concentrations. As future biodegradation of 4,4'-DDT may result in an increase in the concentration of 4,4'-DDE at 05SS130 and 1SS16, 4,4'DDE is considered a potential ecological risk driver for SWMU 1 subsurface soil, and additional evaluation is recommended.

Mean concentrations of antimony (HQ = 0.996), cadmium (HQ = 0.42), and zinc (HQ = 0.92) were less than surface soil screening values. As evidenced by Figure 4-10, antimony and cadmium were each detected in a single subsurface soil sample (05SS135) at concentrations greater than surface soil screening values (5.9 J mg/kg and 4.6 mg/kg, respectively). Zinc was detected in two of six subsurface soil samples at concentrations greater than the surface soil screening value (84.9 mg/kg in 05SS126 and 63.8 mg/kg in 05SS130). Mean concentrations of chromium, cobalt, copper, and vanadium exceeded surface soil screening values (HQ = 45.0, 1.17, 1.35, and 79.0, respectively). As evidenced by Figure 4-10, cobalt and copper were each detected in four of seven subsurface soil samples at concentrations greater than surface soil screening values, while chromium and vanadium were detected in each of the subsurface soil samples at concentrations greater than surface soil screening values.

To evaluate the significance of potential risks presented by antimony, cadmium, chromium, cobalt, copper, vanadium, and zinc concentrations relative to background, the SWMU 1 subsurface soil data were statistically compared to background surface soil data in accordance with Navy guidance (NFESC 2002). As discussed in Section 4.7, a background subsurface soil data set was not available for the depth interval represented by the SWMU 1 subsurface soil samples. Table 4-48 presents a summary and results of the statistical evaluation. Box plot

diagrams illustrating the distribution of each data set is shown on Figure 4-24. As evidenced by Table 4-48 and Figure 4-24, the distribution of zinc concentrations in SWMU 1 subsurface soil is statistically equivalent to background, indicating that zinc is not likely to be site-related and not presenting risks to terrestrial plants and invertebrates at SWMU 1 above background levels. The descriptive statistics presented in Table 4-48 also show that the maximum zinc concentration occurred within the background data set. Based on the statistical evaluation presented in Table 4-48, a mean concentration less than the surface soil screening value (HQ = 0.92), and the low magnitude of detections above the surface soil screening value (see Figure 4-10), zinc is not considered a potential risk driver for subsurface soil at SWMU 1, and additional evaluation is not recommended.

A statistical evaluation of the antimony and cadmium data sets using the t-test or other statistical methods that evaluate central tendencies (i.e., Wilcoxin rank sum test) could not be performed due to the low frequency of detection in the background data sets. The statistical method evaluating the right tail of the distributions (i.e., slippage test) concluded that antimony concentrations in SWMU 1 subsurface soil are elevated above background surface soil concentrations, while cadmium concentrations in SWMU 1 subsurface soil data are statistically equivalent to background surface soil concentrations. However, a determination of equivalent or elevated concentrations cannot be made with high certainty due to the low power associated with this statistical method. Although the descriptive statistics presented in Table 4-48 indicate that antimony and cadmium concentrations in SWMU 1 subsurface soil are elevated above background, the low spatial coverage and magnitude of detections above screening values (see Figure 4-10) does not indicate that these two metals present unacceptable risk to terrestrial plant and invertebrate populations (both metals were detected in a single subsurface soil sample at a concentration greater than screening values; HQ values based on maximum detected concentrations were slightly elevated above 1.0 [HQ = 1.18 for antimony and 1.15 for cadmium). As such, these two metals are not considered potential risk drivers for subsurface soil at SWMU 1, and additional evaluation is not recommended.

Although mean concentrations of chromium (HQ = 45.0), cobalt (HQ = 1.17), copper (HQ = 1.35), and vanadium (HQ = 79.0) exceeded surface soil screening values, the statistical evaluations presented in Table 4-48, as well as the box plot diagrams illustrated on Figure 4-24 show that the distributions of chromium, cobalt, copper, and vanadium concentrations in SWMU 1 subsurface soil are statistically equivalent to background, indicating that they are not likely to be site-related and not presenting risks to terrestrial plants and invertebrates above background levels. Furthermore, the descriptive statistics presented in Table 4-48 show that maximum concentrations for these four metals occurred in the background data set. Based on the descriptive and distributional statistics presented in Table 4-48, chromium, cobalt, copper, and vanadium are not considered potential risk drivers for subsurface soil at SWMU 1, and additional evaluation is not recommended.

In summary, 4,4'-DDT and 4,4'-DDE were identified as potential ecological risk drivers for SWMU 1 subsurface soil and are recommended for additional evaluation in Step 3b of the ERA. Although acetone, carbon disulfide, antimony, cadmium, chromium, cobalt, copper, vanadium, and zinc were detected in SWMU 1 subsurface soil and identified as ecological COPCs in Step 2 of the screening-level ERA, they are not recommended for further evaluation based on the discussion presented in the preceding paragraphs. Further evaluation also is not recommended for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

4.7.1.3 Refined Risk Calculation and Risk Evaluation for Estuarine Wetland Surface Water

Table 4-22 presented the results of the Step 2 screening-level risk calculation for estuarine wetland surface water collected downgradient from SWMU 1. Two detected metals (copper and tin) were identified as ecological COPCs in Step 2 of the screening-level ERA. Copper was identified as an ecological COPC because the maximum detected concentration of this metal exceeded the surface water screening value. Tin was identified as an ecological COPC due to the lack of a surface water screening value. Twenty-one non-detected SVOCs, twelve non-detected organochlorine pesticides, seven non-detected PCBs, eight non-detected organophosphorous pesticides, dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration), and cyanide were identified as ecological COPCs because maximum reporting limits exceeded surface water screening values. Figure 4-11 presented the concentration distribution across the site for each detected chemical identified as an ecological COPC with a maximum HQ grater than or equal to 1.0. Table 4-49 presents the results of the refined screening-level risk calculation for those detected and non-detected chemicals identified as ecological COPCs for estuarine wetland surface water in Step 2 of the screening-level ERA with a maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation for estuarine wetland surface water is presented below.

As discussed above, twenty-one non-detected SVOCs, twelve non-detected organochlorine pesticides, seven non-detected PCBs, eight non-detected organophosphorous pesticides, dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration), and cyanide were identified as ecological COPCs because maximum reporting limits exceeded surface water screening values. Mean concentrations for six non-detected SVOCs (2,4-dinitrophenol, 2-nitroaniline, 3,3'-dichlorobenzidine, 4-chlorophenylphenyl ether, hexachlorobenzene, and hexachloroethane) were less than surface water screening values. Mean concentrations for the remaining non-detected chemicals identified as ecological COPCs in the Step 2 screening-level risk calculation exceeded surface water screening values (see Table 4-49). Of the fifty non-detected chemicals identified as ecological COPCs in the Step 2 screening-level risk calculation (non-detected chemicals with maximum reporting limits greater than surface water screening values, only 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, endrin, heptachlor, methoxychlor, and dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration) were detected in upgradient surface or subsurface soil. Based on the absence of detections in upgradient surface and subsurface soil, there is no indication that the remaining forty-three non-detected chemicals are associated with historical activities at the SWMU (SWMU 1 surface and subsurface soil analytical data provides no indication that a source is present or a release has occurred). For this reason, they are not identified as potential ecological risk drivers for estuarine wetland surface water, and no additional evaluation is recommended. Although detected in SWMU 1 surface and/or subsurface soil, additional evaluation also is not recommended for 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, endrin, heptachlor, methoxychlor, and dioxins/furans. Log K_{oc} values for 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, endrin, heptachlor, methoxychlor, and dioxins/furans (see table 4-6) indicate that these hydrophobic chemicals have a high affinity for adsorption to sediment particles. Therefore, it is unlikely that they are present within the dissolved fraction of the water column at ecologically important concentrations.

The mean concentration of copper exceeded the surface water screening value (HQ = 2.89). Copper was detected within the total recoverable fraction of each estuarine wetland surface water sample at a concentration greater than the screening value. The screening value used in this comparison was a USEPA CCC value expressed as a total (unfiltered) concentration. Because the filtered fraction of copper more closely approximates the bioavailable fraction of this metal in the water column (USEPA 1999b and USEPA 2002a), a dissolved CCC value for this metal (3.1 ug/l [USEPA 2002a]) was compared to dissolved copper data to further evaluate the significance

of potential risks presented by this metal. As evidenced by Table 4-50, the mean concentration of dissolved copper in estuarine wetland surface water downgradient from SMWU 1 was less than the dissolved CCC value (HQ = 0.74).

To further evaluate the significance of potential risks presented by total recoverable and dissolved copper relative to background concentrations, the estuarine wetland surface water data were statistically compared to background surface water data (see Table 4-51) in accordance with Navy guidance (NFESC 2002). The background surface water data used in the evaluation was previously discussed in Section 3.0. Box plots showing the distribution of each data set are illustrated on Figure 4-25. As evidenced by Table 4-51 and Figure 4-25, the distribution of total recoverable and dissolved copper concentrations in estuarine wetland surface water are statistically equivalent to background, indicating that copper is not likely to be site-related and not presenting a risk to aquatic receptors above background levels. Based on the comparison of the mean dissolved copper concentration to the dissolved CCC value (HQ = 0.74) and the statistical evaluation presented in Table 4-51, copper is not considered a potential risk driver for estuarine wetland surface water downgradient from SWMU 1, and additional evaluation is not recommended. Based on the descriptive and distributional statistics presented in Table 4-51, additional evaluation also is not recommended for total recoverable and dissolved tin. The statistical evaluation showed that the distribution of tin concentrations in estuarine wetland surface water downgradient from SWMU 1 is statistically equivalent to background, indicating that tin is unlikely to be site-related and not presenting a risk to aquatic receptors above background levels.

In summary, potential risk drivers were not identified for estuarine wetland surface water downgradient from SWMU 1. Although copper and tin were detected in estuarine wetland surface water and identified as ecological COPCs in Step 2 of the screening-level ERA, they are not recommended for further evaluation based on the discussion presented in the preceding paragraphs. Additional evaluation also is not recommended for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

4.7.1.4 Refined Risk Calculation and Risk Evaluation for Estuarine Wetland Sediment

Table 4-23 presented the results of the Step 2 screening-level risk calculation for estuarine wetland sediment collected downgradient from SWMU 1. One VOC (acetone), one PAH (pyrene), two organochlorine pesticides (4,4'-DDD and 4,4'-DDE), and twelve metals (antimony, cadmium, chromium, cobalt, copper, lead, mercury, nickel, selenium, silver, tin, and vanadium) were identified as ecological COPCs in Step 2 of the screening-level ERA because maximum detected concentration exceeded sediment screening values. Two detected inorganics (beryllium and cyanide) were also identified as ecological COPCs due to the lack of sediment screening values. Seventeen non-detected VOCs, sixty-seven non-detected SVOCs, five non-detected PAHs, twelve organochlorine pesticides, eight non-detected organophosphorous pesticides, seven non-detected PCBs, and dioxins/furans (based on 2,3,7,8-TCDD equivalent concentration) were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values. Figure 4-12 presented the concentration distribution across the site for each detected chemical identified as an ecological COPC with a maximum HQ grater than or equal to 1.0. Table 4-52 presents the results of the refined screening-level risk calculation for those detected and non-detected chemicals identified as ecological COPCs for estuarine wetland sediment in Step 2 of the screening-level ERA with a maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation for estuarine wetland sediment is presented below.

Seventeen non-detected VOCs, sixty-seven non-detected SVOCs, five non-detected PAHs, twelve non-detected organochlorine pesticides, eight non-detected organophosphorous pesticides,

seven non-detected PCBs, and dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration) were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values. Mean 1,1-dichloroethane, 1,2-dibromo-3-chloropropane, 1,2-dibromomethane, acetonitrile, bromomethane, tetrachloroethene, vinyl chloride, 1,2,4,5-tetrachlorobenzene, 2,3,4,6-tetrachlorophenol, 2-nitrophenol, N-nitrosdiethylamine, N-nitrosmethylethylamine, 2-methylnaphthalene, fluorene, pyrene, and toxaphene concentrations were less than sediment screening values, while mean concentrations for the remaining non-detected chemicals exceeded screening values. Of the 116 non-detected chemicals identified as ecological COPCs in the Step 2 screening-level ERA (non-detected chemicals with maximum reporting limits greater than sediment screening values), only tetrachloroethene, xylene, bis(2-ethylhexyl)phthalate, butylbenzylphthalate, 2-methylnaphthalene, acenaphthylene, dibenz(a,h)anthracene, 4,4'-DDT, endrin, heptachlor, methoxychlor, Aroclor-1260, and dioxins/furans were detected in upgradient surface or subsurface soil. Based on the absence of detections in SWMU 1 surface and subsurface soil, there is no indication that the remaining 103 non-detected chemicals are associated with historical site activities (no evidence of a source or release to surface and subsurface soil). For this reason, they are not identified as potential ecological risk drivers for estuarine wetland sediment, and no additional evaluation is recommended.

Although detected in SWMU 1 surface and/or subsurface soil, additional evaluation also is not recommended for tetrachloroethene, xylene, bis(2-ethylhexyl)phthalate, butylbenzylphthalate, 2-methylnaphthalene, acenaphthylene, dibenz(a,h)anthracene, 4,4'-DDT, endrin, heptachlor, methoxychlor, Aroclor-1260, and dioxins/furans. The absence of detections in sediment indicates that these chemicals are not migrating to the estuarine wetland system at ecologically important concentrations. The low frequency and magnitude of detections in upgradient surface soil and subsurface soil for many of these chemicals also precludes their migration to estuarine wetland sediment at ecologically important concentrations. For example, tetrachloroethene was detected in one of nineteen SWMU 1 surface soil samples at 3 J ug/kg, while xylene was detected in two of eighteen SWMU 1 surface soil samples at a maximum concentration of 5 J ug/kg (see Table 4-20). Furthermore, for the nonionic organic chemicals, the primary factor affecting bioavailability in sediment is TOC (USEPA, 1993a). Because bulk sediment screening values used for bis(2-ethylhexyl)phthalate, butylbenzylphthalate, 2-methylnaphthalene, acenaphthylene, dibenz(a,h)anthracene, 4,4'-DDT, endrin, heptachlor, methoxychlor, Aroclor-1260, and dioxins/furans in the Step 2 and Step 3a screening-level risk calculations do not account for the effect that TOC has on their bioavailability, HQs calculated based on reporting limits and bulk-sediment screening values are likely overestimates.

Cyanide was detected and identified as an ecological COPC in Step 2 of the screening-level ERA due the lack of a sediment screening value. As evidenced by Table 4-52, total cyanide was detected in one of nine sediment samples (01EWSSD02) at a concentration of 1.1 J mg/kg. A marine toxicological threshold was not available for use as a sediment screening value; however, Persaud et al. (1993) reported an open water disposal guideline of 0.1 mg/kg for freshwater systems. The single detection (1.1 J mg/kg at 01EWSSD02) exceeds this value. As described in Section 4.6.2, HQs calculated for cyanide are likely overestimates of the potential for risk, given that only total cyanide was measured and it is the free form that is biologically available and, thus, potentially toxic. Free cyanide takes the form of either hydrogen cyanide (HCN) or the cyanide anion (CN^-). As both of these forms are highly water soluble, detections in the water column would be expected if present. As cyanide was not detected in any form in surface water samples collected concurrently with sediment samples, it is likely that any cyanide present in sediment is in the complexed, inert form. As such, the singular total cyanide detection is not likely to present risk to populations of aquatic receptors in estuarine wetland sediment. Furthermore, cyanide was not detected in upgradient surface or subsurface soil samples (see

Appendix G.1 and G.2), indicating that the single cyanide detection in estuarine wetland sediment sample is not site-related. Based on the likely form of cyanide present at 01EWSSD02 and the absence of detections in upgradient surface and subsurface soil, which indicates that a source is not present and a release has not occurred at the SWMU, cyanide is not identified as a potential risk driver for estuarine wetland sediment downgradient from SWMU 1, and no further evaluation is recommended.

As evidenced by Table 4-52, the mean concentration of pyrene was less than the sediment screening value (HQ = 0.23). Pyrene was detected in two of nine sediment samples collected from the estuarine wetland downgradient from SWMU 1. Detections above the sediment screening value were spatially limited to a single sample location (220 ug/kg in 01EWSSD02 [see Figure 4-12]). Based on a mean concentration less than the sediment screening value and the low frequency and magnitude of detections above the sediment screening value, pyrene is not considered a potential risk driver for estuarine wetland sediment downgradient from SWMU 1, and additional evaluation is not recommended. This recommendation is further supported when the bioavailability of pyrene is taken into consideration. The sediment screening value used in risk calculations was a bulk-sediment toxicological threshold (i.e., TEL) developed by MacDonald (1994). This bulk-sediment value does not take into consideration site-specific factors that can influence bioavailability. For nonionic organic chemicals such as pyrene, the primary factor affecting bioavailability is TOC (USEPA 1993a). Di Toro and McGrath (2000) have developed a target lipid model that provides a method to evaluate the impact TOC has on the bioavailability of organic chemicals. Based on this model, Di Toro and McGrath (2000) report a sediment quality guideline (SQG) for pyrene of 11,800 ug/kg based on one percent organic carbon. As the maximum pyrene concentration was 220 ug/kg, it is unlikely that the single detection above a bulk-sediment screening value is ecologically relevant. Furthermore, given that the minimum TOC concentration measured in estuarine wetland sediment downgradient from SWMU 1 was 38,000 mg/kg (i.e., 3.8 percent), the Di Toro and McGrath (2000) target lipid model would predict even lower potential for bioavailability when site-specific TOC is considered (i.e., SQG of 44,809 ug/kg based on 3.8 percent TOC).

Mean concentrations of 4,4'-DDD and 4,4'-DDE were greater than sediment screening values (HQ = 50.0 and 43.9, respectively). 4,4'-DDD was detected in six of fourteen estuarine wetland sediment samples, while 4,4'DDE was detected in eight-of fourteen estuarine wetland sediment samples. As evidenced by Figure 4-12, all detections exceeded sediment screening values. To evaluate the significance of potential risks presented by 4,4'-DDD and 4,4'-DDE relative to background concentrations, the estuarine wetland sediment data were statistically compared to background estuarine wetland sediment data. The statistical comparison, presented in Table 4-53 and the box plot diagrams illustrating the distribution of the data sets (see Figure 4-26) show that the distribution of 4,4'-DDD and 4,4'-DDD concentrations in estuarine wetland sediment downgradient from SWMU 1 are statistically equivalent to background levels. This indicates that these two organochlorine pesticides are not likely to be site-related and do not present risks to aquatic receptors above background levels. The descriptive statistics presented in Table 5-53 also demonstrate that maximum, mean, and 95 percent UCL concentrations in the background sediment data set exceeded maximum, mean, and 95 percent UCL concentrations in the SWMU 1 sediment data set. A spatial examination of detected concentrations also shows that the nature and extent of 4,4'-DDD and 4,4'-DDE in estuarine wetland sediment has been adequately characterized. Specifically, maximum concentrations were not detected in the southernmost sediment samples (1EWS11, 1EWS13, and 1EWS14 [see Figure 4-12]). Based on the descriptive and distributional statistics presented in Table 4-12 and the spatial distribution of detected concentrations, 4,4'-DDD and 4,4'-DDT are not identified as potential ecological risk drivers for estuarine wetland sediment, and no additional evaluation is recommended.

Mean concentrations for antimony (HQ = 0.39), chromium (HQ = 0.77), cobalt (0.79), lead (0.40), mercury (0.82), nickel (HQ = 0.84), and selenium (0.70) were less than sediment screening values. Detections above sediment screening values for antimony, lead, and selenium were spatially limited to one of nine sediment samples, while chromium, cobalt, mercury, and nickel detections above sediment screening values were spatially limited to two of nine sediment (see Figure 4-12). As evidenced by HQ values calculated using maximum concentrations, the magnitude of detections above sediment screening values also was low (HQ = 1.35 for antimony, 1.57 for chromium, 1.70 for cobalt, 1.26 for lead, 2.0 for mercury, 1.82 for nickel, and 1.10 for selenium).

To evaluate the significance of potential risks presented by chromium, cobalt, lead, mercury, and selenium relative to background concentrations, the estuarine wetland sediment data were statistically compared to background concentrations (see Table 4-53). As evidenced by Table 4-53 and Figure 4-26, the distribution of cobalt, lead, mercury and, selenium concentrations in estuarine wetland sediment downgradient from SWMU 1 are statistically equivalent to background, indicating that these four metals are not site-related and not presenting risks to aquatic receptor groups above background levels. The descriptive statistics presented in Table 4-53 also show that maximum, mean, and 95 percent UCL background concentrations for cobalt and selenium exceeded maximum, mean, and 95 percent UCL concentrations in estuarine wetland sediment downgradient from SWMU 1, while maximum, mean, and 95 percent UCL concentrations for lead and mercury in estuarine wetland sediment were only slightly elevated above background concentrations. Based on mean concentrations less than sediment screening values, the low spatial coverage and magnitude of detections above sediment screening values, and the descriptive and distributional statistics presented in Table 4-53, cobalt, lead, mercury, and selenium are not considered potential risk drivers for estuarine wetland sediment, and additional evaluation is not recommended.

The descriptive and distributional statistics presented in Table 4-53 indicate that the distribution of chromium concentrations in estuarine wetland sediment downgradient from SWMU 1 is elevated above background levels. This metal was detected in two of nine estuarine wetland sediment samples at a concentration greater than the screening value (82 mg/kg in 01EWSSD05 and 54 mg/kg in 01EWSSD09). Furthermore, the descriptive and distributional statistics presented in Tables 4-46 and 4-48 for surface soil and subsurface soil, respectively showed that chromium is not present in upgradient soil at concentrations greater than background soil levels. Based on the low magnitude and frequency of detected chromium concentrations greater than the sediment screening value (detected in two of nine sediment samples at a concentration greater than the screening value; HQ value based on maximum detection was slightly elevated above 1.0 [1.57]) and the absence of evidence suggesting that a release of chromium has occurred at SWMU 1 (surface and subsurface soil concentrations are statistically equivalent to background surface soil and subsurface soil levels), chromium is not considered a potential risk driver for estuarine wetland sediment downgradient from SWMU 1, and additional evaluation is not recommended.

The distributional statistics presented in Table 4-53 for nickel (two sample t-test and slippage test) indicate that the concentration of this metal in estuarine wetland sediment downgradient from SWMU 1 is elevated above background sediment concentrations. However, as was previously discussed, the spatial coverage of nickel concentrations greater than the sediment screening value was limited to two samples. Furthermore, the magnitude of these two detections above the sediment screening value was low (HQ = 1.82). To further evaluate the significance of potential risks presented by nickel concentrations in estuarine wetland sediment downgradient from SWMU 1, alternative screening values were identified from the literature. The screening value used in the Step 2 and Step 3a risk calculations was a TEL value (16 mg/kg) reported by

MacDonald (1994). MacDonald (1994) also reported a Permissible Effect Level (PEL) of 42.8 mg/kg. Long et al. (1995) reported an ER-L and ER-M value of 20.9 mg/kg and 51.6 mg/kg, respectively for estuarine and marine sediments. Buchman (1999) reported an AET value of 110 mg/kg for marine sediments. MacDonald et al. (2000) reported a consensus-based Threshold Effect Concentration (TEC) and Probable Effect Concentration (PEC) value of 22.7 mg/kg and 48.6 mg/kg, respectively for freshwater ecosystems. Finally, Persuad et al. (1993) developed Provincial sediment quality guidelines (Low Effect Level [LEL] = 16 mg/kg and Severe Effect Level [SEL] = 75 mg/kg) for the protection and management of Ontario sediments. The maximum detected concentration of nickel falls within the range of the alternative screening values, while the mean nickel concentration (13.5 mg/kg) is less than these alternative values. Based on a mean concentration less than the sediment screening value for available literature-based toxicological thresholds and the low spatial coverage of concentrations above TEL, ER-L, TEC, and LEL values, nickel is not considered a potential risk driver for estuarine wetland sediment downgradient from SWMU 1, and additional evaluation is not recommended.

Mean concentrations for cadmium (HQ = 1.25), copper (HQ = 4.70), silver (HQ = 1.04), tin (HQ = 2.10), and vanadium (2.21) exceeded sediment screening values. The spatial coverage of cadmium and silver detections above sediment screening values was limited to one sample location (cadmium was detected in sediment sample 01EWSSD06 at 0.95 J mg/kg and silver was detected in 01EWSSD01 at 0.76 J ug/kg [see Figure 4-12]). To evaluate the significance of potential risks presented by cadmium and silver relative to background, the estuarine wetland sediment data were statistically compared to background concentrations. The statistical evaluation presented in Table 4-53 was inconclusive for both metals. A statistical evaluation of the mean/median of the distributions (e.g., t-test, Wilcoxin rank sum test, or Gehan Test) could not be performed because data assumptions for these tests were not met [see Figure 4-19a]). The slippage test did indicate that cadmium concentrations in estuarine wetland sediment downgradient from SWMU 1 are not elevated above background concentrations; however, a determination of equivalent or elevated concentrations cannot be made with high certainty due to the low power associated with this statistical method. Although the statistical evaluation was inconclusive, cadmium and silver are not considered potential risk drivers for estuarine wetland sediment downgradient from SWMU 1 and additional evaluation is not recommended. This recommendation is based on the low spatial coverage and magnitude of detections above sediment screening values. It is noted that maximum detected concentrations of each metal (0.95 J mg/kg for cadmium and 0.76 J mg/kg for silver) are less than ER-L values (1.2 mg/kg for cadmium and 1.0 mg/kg for silver) reported by Long et al. (1995). The maximum cadmium concentration also is less than the consensus-based TEC value (0.99 mg/kg) reported by MacDonald et al. (2000). This comparison also supports the recommendation for no further evaluation.

The statistical evaluation of the vanadium data presented in Table 4-53 (two sample t-test and slippage test), as well as the box plot diagram illustrating the distribution of the site and background datasets show that the distribution of vanadium concentrations in estuarine wetland sediment is statistically equivalent to background, indicating that vanadium is not likely to be site related and is not presenting a risk to aquatic receptors above background levels. The descriptive statistics presented in Table 4-53 also support this conclusion. Although the maximum vanadium concentration occurred in estuarine wetland sediment downgradient from SWMU 1 (230 mg/kg versus 220 mg/kg in background), the mean and 95 percent UCL concentrations in the background data set exceeded site mean and 95 percent UCL concentrations. For these reasons, vanadium is not considered a potential risk driver in estuarine wetland sediment downgradient from SWMU 1, and additional evaluation is not recommended.

The distributional statistics performed on the site and background data sets for tin were contradictory (see Table 4-53). The evaluation of the mean of distributions (Gehan Test) concluded that tin concentrations in estuarine wetland sediment downgradient from SWMU 1 are statistically equivalent to background concentrations, while the slippage test concluded that tin concentrations are elevated above background. The descriptive statistics presented in Table 4-53 show that mean and 95 percent UCLs for each data set are similar. The similarity of the SWMU and background data sets is also shown by the box plot diagram presented in Figure 4-26.

The sediment screening value used in the tin risk calculations was an AET value from Buchman (1999). The AET value, reported as >3.4 mg/kg, is based on tributyltin (TBT) toxicity to *Neanthes sp*. Use of this value as a sediment screening value is extremely conservative since it does not represent a threshold effect concentration (value reported as >3.4 mg/kg). Furthermore, the AET value is based on the most toxic form of tin (USEPA 2002b). An alternative value for tin, also based on TBT, was identified from the literature. Kristin et al. (1998) investigated the toxicity of TBT in sediment using spiked sediment toxicity tests with four benthic macroinvertebrate species (an oligochaete [*Tubifex tubifex*], a chironomid (*Chironomus riparius*], an amphipod [*Hyalella azteca*], and a mayfly [*Hexagenia sp.*]). *Hexagonia sp.* was the most sensitive benthic invertebrate tested. The test endpoint for this species was a median inhibition concentration (IC_{50}) based on growth. The reported IC_{50} value of 600 mg/kg (dry weight) resulted in a fifty percent reduction in the growth of the test organism when compared to a control. The maximum detected tin concentration in estuarine wetland sediment downgradient from SWMU 1 (26.8 J mg/kg in 01EWSSDO1; expressed as TBT) is an order magnitude below the IC_{50} value reported by Kristin et al. (1998). Based on this comparison, it is unlikely that tin is adversely impacting aquatic receptor populations within the estuarine wetland. A spatial examination of detected concentrations in estuarine wetland sediment also shows that the nature and extent of tin has been adequately characterized. Specifically, maximum concentrations were not detected in the southernmost sediment samples (1EWS11 and 1EWS12 [see Figure 4-12]). Based on the descriptive statistics presented in Table 4-53, a comparison of the maximum detected concentration to literature-based toxicity values, and the spatial distribution of detected concentrations, tin is not identified as a potential ecological risk driver for estuarine wetland sediment, and no additional evaluation is recommended.

Copper was detected in each estuarine wetland sediment sample at concentrations greater than the sediment screening value. As evidenced by Table 4-53, the statistical evaluation performed on the copper data sets indicates that the distribution of copper concentrations in estuarine wetland sediment downgradient from SWMU 1 is elevated above background levels. The descriptive statistics also demonstrate that maximum, mean and 95 percent UCL concentrations for the SWMU 1 data set exceed maximum mean, and 95 percent UCL concentrations for the background data set. Thirteen detections in estuarine wetland sediment exceeded the maximum background concentration.

Total sediment concentrations are usually not predictive of the bioavailability and toxicity of metals (Luoma, 1983). However, similar to nonionic chemicals, metal concentrations in sediment pore water have been correlated with toxicity (Adams et al., 1985, Swartz et al., 1985, and Kemp and Swartz, 1988). An important partitioning phase controlling the bioavailability and toxicity of cadmium, copper, lead, nickel, and zinc is AVS. AVS represents a reactive pool of solid-phase sulfide that is available to bind these metals, rendering them biologically unavailable and nontoxic to sediment-associated biota. Cadmium, copper, lead, nickel, and zinc, collectively termed simultaneously extracted metals (SEM), represent those metals that form a more stable complex with sulfide than does iron. The model states that if the SEM concentration is less than the concentration of AVS (i.e., when the SEM-to-AVS ratio is less than 1.0 or the SEM-to-AVS difference is less than 0), toxicity will not be observed. That is, if the SEM-to-AVS ratio is less

than 1.0 or the SEM-to-AVS difference is less than 0, sufficient AVS is available to bind all the SEM and sediment-associated biota will not be exposed to toxic concentrations of these metals in the sediment pore water.

Ankley et al., 1996 provide a summary of investigations that have studied the significance that sulfide partitioning has in controlling the acute and/or chronic toxicity of cadmium, copper, lead, nickel, and zinc in freshwater and marine sediments. Although specific literature-based information was not available on the significance of sulfide partitioning in controlling the chronic toxicity of copper in freshwater or marine sediments, Ankey (1996) states that, "The bioaccumulation of lead, zinc, and copper appears to be explained reasonably well by a model based on AVS binding." Ankey et al. (1996) also present four methods for deriving Sediment Quality Criteria (SQC) for cadmium, copper, lead, nickel, and zinc. One method includes the comparison of molar AVS concentrations to the summed molar concentration of the five metals simultaneously extracted with the AVS. Ankley et al. (1996) indicate that SQC derived using this approach should be considered no effect concentrations. This study shows acceptance by the scientific community regarding the applicability of the SEM-AVS model for predicting chronic effects of copper on benthic macroinvertebrates.

Estuarine wetland sediment samples collected during the October 2004 additional data collection field effort (i.e., 1EWS10 through 1EWS20) included analyses for AVS and SEM. A comparison of total SEM molar concentrations to AVS molar concentrations is provided in Table 4-53a (If an individual SEM was not detected in a given sample, the total SEM concentration for that sample was calculated assuming one-half the reporting limit for that metal). As evidenced by Table 4-53a, the molar concentration of AVS exceeded the molar concentration of SEM in each of the estuarine wetland sediment samples (SEM-to-AVS ratios ranged from 0.0277 [1EWS18] to 0.1727 [1EWS17]. The sediment sample with the maximum copper concentration (170 J mg/kg in 1EWS10) had an AVS-to-SEM ratio of 0.0299. The SEM-to-AVS ratios calculated for the estuarine wetland sediment samples indicate that benthic macroinvertebrates will not be exposed to toxic concentrations of SEM (including copper) in the sediment pore water. It is noted that Ankley et al. (1996) state that, "chemical equilibrium calculations suggest that the relative affinity of metals to AVS should be copper > lead > cadmium > zinc > nickel. Hence, the appearance of metals in pore water as AVS is exhausted should occur in an inverse order, e.g., zinc would replace nickel in a monosulfide complex and nickel would be liberated to the pore water, etc." Ankley et al. (1996) further state that, "This trend was, in fact, observed by Berry et al. (1996)." This information indicates that temporal fluctuations in estuarine wetland sediment AVS concentrations will not necessarily equate to the liberation of copper to sediment pore water at toxic concentrations. Based on the discussion presented above and the SEM-to-AVS ratios presented in Table 4-53a, copper is not identified as a potential ecological risk driver for estuarine wetland sediment downgradient from SWMU 1, and no additional evaluation is recommended.

As was previously discussed, beryllium was identified as an ecological COPC in Step 2 of the screening-level ERA based on the lack of a sediment screening value. The distributional statistics presented in Table 4-53 demonstrate that the concentration distribution of beryllium in estuarine wetland sediment is statistically equivalent to background levels. The descriptive statistics presented in Table 4-53 also show that the maximum estuarine wetland sediment concentration (0.31 mg/kg) is only slightly elevated above the maximum background concentration (0.25 J mg/kg), while the 95 percent UCL concentration in estuarine wetland sediment (0.27 mg/kg) is less than the background 95 percent UCL concentration in background sediment (0.58 mg/kg). Finally, a comparison of the beryllium surface water screening value to surface water screening values for other metals indicates that beryllium is less toxic to aquatic organisms than many of the Appendix IX metals, including cadmium, chromium, copper, lead, nickel, selenium, silver, and zinc. Given that the maximum beryllium concentration in estuarine wetland sediment

downgradient from SWMU 1 is less than sediment screening values for metals known to be more toxic to aquatic organisms (see Table 4-23), it is unlikely that beryllium is impacting benthic macroinvertebrate populations. Based on the descriptive and distributional statistics presented in Table 4-53 and the comparison of the maximum beryllium concentration to sediment screening values for other metals, this metal is not identified as a potential ecological risk driver for estuarine wetland sediment, and no additional evaluation is recommended.

A statistical evaluation of the antimony data was not possible due to the low number of data points in each data set and the lack of detected concentrations in the background data set. Antimony was detected in one of nine estuarine wetland sediment samples at a concentration greater than the sediment screeing value. The magnitude of this single detection above the screening value was low (maximum HQ = 1.35). The mean HQ for antimony is less than 1.0 (0.39), indicating acceptable risk to lower trophic level aquatic receptor populations. Based on a mean HQ less than 1.0 and the low frequency and magnitude of detections above the sediment screening value, antimony is not identified as a potential ecological risk driver for estuarine wetland sediment downgradient from SWMU 1, and no additional evaluation is recommended.

In summary, there are no potential ecological risk drivers identified for estuarine wetland sediment downgradient from SWMU 1. Although acetone, pyrene, 4,4'-DDD, 4,4'-DDE, total cyanide, antimony, cadmium, chromium, copper, cobalt, lead, mercury, nickel, selenium, silver, tin, and vanadium were detected in estuarine wetland sediment and identified as ecological COPCs in Step 2 of the screening-level, additional evaluation is not recommended based on the discussion presented in the preceding paragraphs. Additional evaluation also is not recommended for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

4.7.1.5 Refined Risk Calculation and Risk Evaluation for Ensenada Honda Surface Water

Table 4-24 presented the results of the Step 2 screening-level risk calculation for Ensenada Honda surface water downgradient from SWMU 1. Di-n-butylphthalate and copper were detected in Ensenada Honda surface water downgradient from SWMU 1 and were identified as ecological COPCs in Step 2 of the screening-level ERA because maximum detected concentrations exceeded surface water screening values. Chloroethane, methapyrilene, and tin were also detected and identified as ecological COPCs due to the lack of surface water screening values. Twenty non-detected SVOCs, twelve non-detected organochlorine pesticides, seven non-detected PCBs, eight non-detected organophosphorous pesticides, dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration), and two non-detected inorganics (silver and cyanide) were identified as ecological COPCs because maximum reporting limits exceeded surface water screening values. Figure 4-13 presented the concentration distribution across the site for each detected and non-detected chemical identified as an ecological COPC with a maximum HQ greater than or equal to 1.0. Table 4-54 presents the results of the refined screening-level risk calculation for those detected and non-detected chemicals identified as ecological COPCs for Ensenada Honda surface water in Step 2 of the screening-level ERA with a maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation for Ensenada Honda surface water is presented below.

Twenty non-detected SVOCs, twelve non-detected organochlorine pesticides, seven non-detected PCBs, eight non-detected organophosphorous pesticides, dioxins/furans (based on 2,3,7,8-TCDD equivalent concentration), and two non-detected inorganics (silver and cyanide) were identified as ecological COPCs because maximum reporting limits exceeded surface water screening values. Mean concentrations for 2,4-dinitrophenol, 2-nitroaniline, 3,3'-dichlorobenzidine, 4-chlorophenylphenyl ether, hexachlorobenzene, hexachloroethane were less than 1.0, while mean

concentrations for the remaining non-detected chemicals exceeded 1.0. Of the fifty non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA (non-detected chemicals with maximum reporting limits greater than surface water screening values), only 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, endrin, heptachlor, methoxychlor, Aroclor-1260, dioxins/furans, and silver were detected in upgradient surface soil and/or subsurface soil. Based on the absence of detections in SWMU 1 surface and subsurface soil, there is no indication that the remaining forty-one non-detected chemicals are associated with historical site activities (no evidence of a source or release to SWMU 1 surface and subsurface soil). For this reason, they are not identified as potential ecological risk drivers for estuarine wetland sediment, and no additional evaluation is recommended. Although detected in SWMU 1 surface and/or subsurface soil, additional evaluation also is not recommended for 4,4'-DDD, 4,4'-DDT, endrin, heptachlor, methoxychlor, Aroclor-1260, and silver. Log K_{ow} and Loc K_{oc} values for 4,4'-DDD, 4,4'-DDT, endrin, heptachlor, methoxychlor, and dioxins/furans (see table 4-6) indicate that these hydrophobic chemicals have a high affinity for adsorption to sediment particles. Therefore, it is unlikely that they are present within the dissolved fraction of the water column at ecologically important concentrations. Silver was detected in two of eighteen SWMU 1 surface soil samples at a maximum concentration of 2.6 J ug/kg. The low frequency and magnitude of detections in upgradient surface soil does not indicate that this metal is migrating to the Ensenada Honda at ecologically important concentrations.

Chloroethane and methapyrilene were identified as ecological COPCs in Step 2 of the screening-level ERA due to the lack of surface water screening values. Both chemicals were each detected in a single surface water sample (chloroethane was detected in 1OWSW03 at 7.2 ug/L and methapyrilene was detected in 1OWSW08 at 3.8 J ug/kg). These two organics were not detected in upgradient abiotic media (surface soil and subsurface soil and estuarine wetland surface water and sediment), indicating that their presence is not associated with a release from SWMU 1. Both detections also are less than the majority of available surface water screening values for other VOCs and SVOCs. Therefore, it is not likely that these two VOCs are present at ecologically relevant concentrations. As such, they are not identified as potential risk drivers, and additional evaluation is not recommended.

The mean concentration of di-n-butylphthalate in Ensenada Honda surface water downgradient from SWMU 1 exceeded the surface water screening value (HQ = 1.50). This SVOC was detected in one of nine surface water samples (5.9 J ug/kg in 01OWSW07). The surface water screening value used in risk calculations (3.4 ug/L) was a USEPA Region 4 surface water screening value (see Table 4-8). This single detection is less than a SCV (32.7 ug/L) reported by Suter II (1995) and a Tier II SCV reported by the USEPA (1996b). Based on the low magnitude of the detected concentration above the screening value used in risk calculations (3.4 ug/L [USEPA, 2001]), As well as the comparison of the single detection to SCVs developed by Sutter II (1995) and the USEPA (1996b), it is unlikely that this phthalate ester is impacting aquatic receptor communities within the Ensenada Honda. As such, di-n-butylphthalate is not identified as a potential risk driver for Ensenada Honda surface water downgradient from SWMU 1.

The mean concentration of copper in Ensenada Honda surface water collected downgradient from SWMU 1 was less than the surface water screening value (HQ = 0.78). Copper detections greater than the surface water screening value occurred in two of nine samples (3.9 J ug/L in 01OWSW03 and 14 ug/L in 01OWSW05). As discussed in Section 4.7.2.3, the screening value used in the Step 2 and Step 3a risk calculations for this metal was a USEPA CCC value expressed as a total (unfiltered) concentration. Because the filtered fraction of copper more closely approximates the bioavailable fraction of this metal in the water column (USEPA 1999b and USEPA 2002a), a dissolved CCC value for this metal (3.1 ug/l [USEPA 2002a]) was compared to dissolved copper data. As evidenced by Table 4-55, the mean concentration of dissolved copper

in estuarine wetland surface water downgradient from SMWU 1 also was less than the dissolved CCC value (mean HQ = 0.29). The maximum dissolved copper concentration (1.2 J ug/L) also is less than the dissolved CCC value (maximum HQ = 0.39).

To further evaluate the significance of potential risks presented by total recoverable and dissolved copper relative to background concentrations, the Ensenada Honda surface water data were statistically compared to background surface water data. Table 4-56 presents a summary and results of the statistical evaluation. Box plot diagrams illustrating the distribution of each data set is shown on Figure 4-27. As evidenced by Table 4-56 and Figure 4-27, the distributions of total recoverable and dissolved copper concentrations in Ensenada Honda surface water are statistically equivalent to background levels, indicating that copper is not likely to be site-related and not presenting a risk to aquatic receptors above background levels. The descriptive statistics presented in Table 4-56 also show that maximum, mean, and 95 percent UCL concentrations for the dissolved background data set exceeded maximum, mean, and 95 percent UCL concentrations for the dissolved Ensenada Honda data set. Based on the comparison of the mean total recoverable and dissolved copper concentrations to total recoverable and dissolved screening values and the statistical evaluation presented in Table 4-56, copper is not considered a potential risk driver for Ensenada Honda surface water downgradient from SWMU 1, and additional evaluation is not recommended.

Based on the statistical evaluation presented in Table 4-56, additional evaluation also is not recommended for tin. The distributional statistics demonstrate that the distribution of tin concentrations in Ensenada Honda surface water downgradient from SWMU 1 is statistically equivalent to background. This recommendation is supported by a comparison of the maximum tin concentration detected in Ensenada Honda surface water (1.9 J ug/L) to the USEPA Region 5 surface water screening value for total tin (180 ug/L [USEPA 2003c]). The maximum detected concentration is two orders of magnitude below the USEPA Region 5 value.

In summary, no chemicals were identified as a potential risk driver for Ensenada Honda surface water downgradient from SWMU 1. Although chloroethane, di-n-butylphthalate, methapyrilene, copper, and tin were detected and identified as ecological COPCs in Step 2 of the screening-level ERA, they are not recommended for further evaluation based on the discussion presented in the preceding paragraphs. Additional evaluation also is not recommended for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

4.7.1.6 Refined Risk Calculation and Risk Evaluation for Ensenada Honda Sediment

Table 4-25 presented the results of the Step 2 screening-level risk calculation for Ensenada Honda sediment downgradient from SWMU 1. Acetone, seven SVOCs (2,4-dinitrotoluene, 2-nitroaniline, 4-chloropehenylphenyl ether, 4-nitroaniline, di-n-butylphthalate, di-n-octylphthalate, and phenol) and five metals (arsenic, copper, selenium, tin, and vanadium) were detected in Ensenada Honda sediment downgradient from SWMU 1 and identified as ecological COPCs because maximum detected concentrations exceeded sediment screening values. Beryllium and thallium also were detected and identified as ecological COPCs due to the lack of sediment screening values. Eleven non-detected VOCs, fifty-six non-detected SVOCs, five non-detected PAHs, fourteen non-detected organochlorine pesticides, seven non-detected PCBs, eight non-detected organophosphorous pesticides, and dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration) were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values. Figure 4-14 presented the concentration distribution across the site for each detected chemical identified as an ecological COPC with a maximum HQ greater than or equal to 1.0. Table 4-57 presents the results of the refined screening-level risk calculation for those detected and non-detected chemicals identified as ecological COPCs for

Ensenada Honda sediment in Step 2 of the screening-level ERA with a maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation for Ensenada Honda sediment downgradient from SWMU 1 is presented below.

Eleven non-detected VOCs, fifty-six non-detected SVOCs, five non-detected PAHs, fourteen non-detected organochlorine pesticides, seven non-detected PCBs, eight non-detected organophosphorous pesticides, and dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration) were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values. Mean concentrations for acetonitrile, bromomethane, cis-1,3-dichloropropene, 1,3-dichlorobenzene, 2-acetylaminofluorene, 2-picoline, acetophenone, hexachlorophene, pronamide, 2-methylnaphthalene, fluorine, heptachlor epoxide, methoxychlor, and toxaphene were less than sediment screening values, while mean HQ for the remaining non-detected chemicals exceeded sediment screening values. Of the 102 non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA (non-detected chemicals with maximum HQ values greater than sediment screening values), only xylene, bis(2-ethylhexyl)phthalate, butylbenzylphthalate, 2-methylnaphthalene, acenaphthylene, dibenz(a,h)anthracene, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, endrin, heptachlor, methoxychlor, Aroclor-1260, and dioxins/furans were detected in upgradient surface or subsurface soil. Based on the absence of detections in SWMU 1 surface and subsurface soil, there is no indication that the remaining eighty-six non-detected chemicals are associated with historical site activities (no evidence of a source or release to surface and subsurface soil). For this reason, they are not identified as potential ecological risk drivers for estuarine wetland sediment, and no additional evaluation is recommended. Although detected in upgradient surface or subsurface soil, xylene, bis(2-ethylhexyl)phthalate, butylbenzylphthalate, 2-methylnaphthalene, acenaphthylene, dibenz(a,h)anthracene, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, endrin, heptachlor, methoxychlor, Aroclor-1260, and dioxins/furans also are not identified as potential ecological risk drivers. The low frequency and magnitude of detections in upgradient surface soil and subsurface soil for many of these chemicals precludes their migration to estuarine wetland sediment at ecologically important concentrations. For example, xylene was detected in two of eighteen SWMU 1 surface soil samples at a maximum concentration of 5 J ug/kg, butylbenzylphthalate was detected in one of eighteen surface soil samples at 75 J mg/kg, while endrin and methoxychlor were each detected in one of six sediment samples at 0.12 NJ ug/kg and 0.44NJ ug/kg, respectively (see Table 4-57). Furthermore, for nonionic organic chemicals, the primary factor affecting bioavailability in sediment is TOC (USEPA, 1993a). Because bulk-sediment screening values used for bis(2-ethylhexyl)phthalate, butylbenzylphthalate, 2-methylnaphthalene, acenaphthylene, dibenz(a,h)anthracene, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, endrin, heptachlor, Aroclor-1260, and dioxins/furans in the Step 2 and Step 3a risk calculations do not account for the effect that TOC has on their bioavailability, HQs calculated based on reporting limits and bulk-sediment screening values are likely overestimates.

As evidenced by Table 4-57, the mean concentration of acetone exceeded the sediment screening value (HQ = 27.5). Acetone was detected in nine of nine sediment samples at concentrations greater than the sediment screening value. The sediment screening value used for this VOC was derived using the USEPA equilibrium partitioning (EqP) approach (USEPA 1993a [see Appendix H]). As discussed in Appendix H, the USEPA EqP-approach derives a sediment benchmark by setting the dissolved chemical concentration in pore water equal to the surface water benchmark and calculates a corresponding particle-sorbed chemical concentration. This approach is appropriate for highly sorptive chemicals (e.g., PAHs), but it produces overly conservative sediment quality benchmarks for VOCs (Fuchsman, 2003). To evaluate the significance of the acetone detections in Ensenada Honda sediment, an alternative screening value was identified from the literature. Di Toro and McGrath (2000) reported a SQG of 2,265 ug/kg for acetone based on a target lipid model and one percent organic carbon. Given that the maximum detected

acetone (310 ug/L) is an order of magnitude below this value, it is unlikely that this common laboratory contaminant is present at ecologically significant concentrations. Furthermore, given that the minimum TOC concentration measured in Ensenada Honda sediment downgradient from SWMU 1 was 14,000 mg/kg (i.e., 1.4 percent), the Di Toro and McGrath (2000) target lipid model would predict even lower potential for bioavailability when site-specific TOC is considered (i.e., SQG of 3,171 ug/kg based on 1.4 percent TOC). Based on the comparison of the maximum acetone concentration to a SQG value developed by Di Toro and McGrath (2000), acetone is not considered a potential risk driver for Ensenada Honda sediment, and additional evaluation is not recommended

Mean concentrations for 2,4-dinitrotoluene (HQ = 49.5), 2-nitroaniline (HQ = 95.3), 4-chlorophenylphenyl ether (1.87), 4-nitroaniline (HQ = 68.6), di-n-butylphthalate (HQ = 9.16), di-n-octylphthalate (HQ = 11.34), and phenol (HQ = 5.87) exceeded sediment screening values. With the exception of di-n-butylphthalate, these SVOCs were detected in a single sediment sample (01OWSD05). Di-n-butylphthalate was detected in two sediment samples (01OWSD02 and 01OWSD06). There is uncertainty associated with the SVOC detections in sediment sample 01OWSD05, which was duplicated in the field during the July 2003 additional data collection field investigation. 2,4-dinitrotoluene, 2-nitroaniline, 4-chlorophenylphenyl ether, 4-nitrophenol, di-n-octylphthalate, and phenol were not detected in the original sample, only in the duplicate sample. In order to reduce the uncertainty associated with this pattern of detection, three sediment samples were collected within the immediate vicinity of 01OWSD05 during the October 2004 additional data collection field investigation and analyzed for 2,4-dinitrotoluene, 2-nitroaniline, 4-chlorophenylphenyl ether, 4-nitroaniline, di-n0butylphthalate, di-n-octylphthalate, and phenol. As evidenced by Appendix G.11, these seven SVOCs were not detected in the 1OWSD10, 1OWSD11, and 1OWSD12 sediment samples. These seven SVOCs also were not detected in SWMU 1 surface and subsurface soil (see Appendices G.1 and G.2). The absence of detections in upgradient surface and subsurface soil indicates that these SVOCs are not associated with historical site activities at SWMU 1 (no evidence of a source or release to surface and subsurface soil). Based on the absence of detections in the 1OWSD10, 1OWSD11, and 1OWSD12 sediment samples and the absence of detections in SWMU 1 surface and subsurface soil, 2,4-dinitrotoluene, 2-nitroaniline, 4-chlorophenylphenyl ether, 4-nitroaniline, di-n0butylphthalate, di-n-octylphthalate, and phenol are not identified as potential ecological risk drivers for Ensenada Honda sediment, and no additional evaluation is recommended.

Mean concentrations of arsenic (HQ = 0.94), selenium (HQ = 0.81) and vanadium (HQ = 0.61) were less than sediment screening values. As evidenced by Figure 4-14, arsenic was detected in three of nine samples at concentrations greater than the screening value (8.3 mg/kg in 01OWSD01, 8.5 mg/kg in 01OWSD08, and 8.7 mg/kg in 01OWSD03), selenium was detected in three of nine sediment samples at concentrations greater than or equal to the sediment screening value (1.0 J mg/kg in 01OWSD09, 1.1 J mg/kg in 01OWSD03, and 1.2 J mg/kg in 01OWSD04), and vanadium was detected in a single sediment sample at a concentration greater than the sediment screening value (58 mg/kg in 01OWSD04). Given the low spatial coverage and/or the low magnitude of detections greater than or equal to sediment screening values, it is not likely that arsenic, selenium, and vanadium are presenting unacceptable risks to aquatic receptor populations within the Ensenada Honda sediment downgradient from SWMU 1. To evaluate potential risks presented by selenium and vanadium concentrations relative to background concentrations, the estuarine wetland sediment data were statistically compared to background sediment data. Table 4-58 presents a summary and results of the statistical evaluation. Box plot diagrams illustrating the distribution of each data set is shown on Figure 4-28. As evidenced by Table 4-58 and Figure 4-28, the distribution of arsenic concentrations in Ensenada Honda sediment downgradient from SWMU 1 are elevated above background levels. The statistical methods used to evaluate the vanadium data sets were contradictory. Satterthwaite's t-test, which

evaluated the mean of the distributions, concluded that vanadium concentrations in Ensenada Honda sediment were elevated above background concentrations, whereas the slippage test, evaluating the right tail of the distributions, concluded that vanadium concentrations are not elevated above background concentrations. A statistical evaluation of the selenium data sets could not be performed due to the absence of detections in the background data set. The descriptive statistics for these three metals show that maximum, mean, and 95 percent UCL concentrations in the Ensenada Honda data set exceeded background concentrations.

Although the distributional and/or descriptive statistics presented in Table 4-58 show that arsenic, selenium, and vanadium concentrations in Ensenada Honda sediment downgradient from SWMU 1 are elevated above background levels, mean HQ values were less than 1.0 (see Table 4-57). The magnitude of detections greater than sediment screening values was low (HQ values based on maximum concentrations were 1.20 for arsenic, 1.20 for selenium, and 1.02 for vanadium). Alternative screening values were not available from the literature for selenium and vanadium; however, several alternative toxicological thresholds were identified for arsenic. The maximum arsenic concentration (8.7 mg/kg) is less than the consensus-based TEC value (9.79 mg/kg) reported by MacDonald et al. (2000) and slightly elevated above the ER-L value (8.2 mg/kg) reported by Long et al. (1995) and the LEL value (6 mg/kg) reported by Persaud et al. (1993). Based on mean HQs less than 1.0, the low magnitude of detections greater than screening values used in risk calculations, and, in the case of arsenic, the comparison of the maximum detected concentration to alternative screening values, these three metals are not identified as potential risk drivers, and additional evaluation is not recommended.

Mean concentrations of copper and tin exceeded sediment screening values (HQ = 1.04 and 1.51, respectively). The statistical evaluation presented in Table 4-58 indicates that the distribution of copper concentrations in Ensenada Honda sediment downgradient from SWMU 1 is elevated above background concentrations. Copper was detected in six of nine sediment samples at concentrations greater than the sediment screening value (see Figure 4-14); however, the magnitude of detections above the sediment screening value was low. This is evidenced by a maximum HQ of 1.39. The sediment screening value used for this metal in risk calculations was a TEL value developed by MacDonald (1994). To further evaluate the significance of detected concentrations greater than the TEL-based screening value, alternative screening values were identified from the literature. Long et al. 1995 reported ER-L and ER-M concentrations of 34 mg/kg and 270 mg/kg, respectively for estuarine and marine sediments, while Buchman (1999) reported an AET value of 390 mg/kg for marine sediments. MacDonald et al. (2000) developed consensus-based TEC (31.6 mg/kg) and PEC (149 mg/kg) values for freshwater ecosystems, while Persaud et al. (1993) developed Provincial sediment quality guidelines (LEL [16 mg/kg] and SEL [110]) for the management of sediment quality in Ontario. With the exception of the LEL value, maximum and mean copper concentrations measured in Ensenada Honda sediment downgradient from SWMU 1 are less than alternative values available from the literature. Based on the low magnitude of detections above TEL and LEL values and the absence of detections above ER-L and consensus-based TEC values, copper is not considered a potential risk driver for Ensenada Honda sediments, and additional evaluation is not recommended.

Identical to selenium, a statistical evaluation of the site and background tin and beryllium data sets could not be performed due to the lack of detections in background open water sediment samples (see Table 4-58). The statistical evaluation of the thallium data sets also was impaired by the low frequency of detection in the site and background data sets (an evaluation of the mean of the distributions could not be performed).

The sediment screening value used in the risk calculations for tin was an AET value from Buchman (1999). The AET value, reported as >3.4 mg/kg, is based on tributyltin (TBT) toxicity

to *Neanthes sp.* Use of this value as a sediment screening value is extremely conservative since it does not represent a threshold effect concentration (AET value reported as a concentration greater than 3.4 mg/kg). Furthermore, the AET value is based on the most toxic form of tin (USEPA 2002b). An alternative value for tin, also based on TBT, was identified from the literature. As discussed in Section 4.7.2.4, Kristin et al. (1998) investigated the toxicity of TBT in sediment using spiked sediment toxicity tests with four benthic macroinvertebrate species (an oligochaete [*Tubifex tubifex*], a chironomid (*Chironomus riparius*], an amphipod [*Hyalella azteca*], and a mayfly [*Hexagenia sp.*]). *Hexagonia sp.* was the most sensitive benthic invertebrate tested. The test endpoint for this species was an IC_{50} based on growth (600 mg/kg). The maximum detected concentration of tin in Ensenada Honda sediment downgradient from SWMU 1 (17.6 J mg/kg, expressed at TBT) is an order of magnitude below this value. Based on this comparison, tin is not considered a potential ecological risk driver for Ensenada Honda sediment, and additional evaluation is not recommended.

The slippage test concluded that thallium concentrations in Ensenada Honda sediment downgradient from SWMU 1 are statistically equivalent to background concentrations. However, as previously discussed, the power of this statistical procedure is low. An examination of the descriptive statistics presented in Table 4-58 for this metal indicates that the single detection in background open water sediment (0.91 J mg/kg) exceeds the maximum detected concentration in site sediment (0.16 J mg/kg). This indicates that potential risks presented by thallium are not elevated above background levels. For this reason, thallium is not considered a potential risk driver for Ensenada Honda sediment, and additional evaluation is not recommended. It is noted that evaluation of potential risks using toxicological benchmarks or toxicity data could not be performed since this information was not available from the literature.

Beryllium was detected at low concentrations in eight of nine sediment samples collected from the Ensenada Honda downgradient from SWMU 1. As was previously stated, the absence of detections in the background data set prevented a statistical evaluation. Because toxicological thresholds or toxicity data are not available from the literature, a direct evaluation of this metal's potential to impact aquatic life could not be performed. A comparison of the beryllium surface water screening value to surface water screening values for other metals indicates that beryllium is less toxic to aquatic organisms than many of the Appendix IX metals, including cadmium, chromium, copper, lead, nickel, selenium, silver, and zinc. Given that the maximum detected beryllium concentration in Ensenada Honda sediment (0.082 J mg/kg) is less than sediment screening values for metals known to be more toxic to aquatic organisms, it is unlikely that beryllium is impacting benthic macroinvertebrate populations. Based on a comparison of the maximum detected concentration to available sediment screening for other metals, as well as the low magnitude of detections (less than 0.1 mg/kg), beryllium is not considered a potential risk driver for Ensenada Honda sediments, and additional evaluation is not recommended.

In summary, there are no potential ecological risk drivers identified for Ensenada Honda sediment downgradient from SWMU 1. Although acetone, 2,4-dinitrotoluene, 2-nitroaniline, 4-chlorophenyphenyl-ether, 4-nitrophenol, di-n-butylphthalate, di-n-octylphthalate, phenol, arsenic, beryllium, copper, selenium, tin, thallium, and vanadium were detected and identified as ecological COPCs in Step 2 of the screening-level ERA, they are not considered potential risk driver for the reasons, presented above, and additional evaluation is not recommended. No additional evaluation also is recommended for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

4.7.1.7 Refined Risk Calculation and Risk Evaluation for Terrestrial and Aquatic Food Web Exposures

Tables 4-26 through 4-29 presented the results of the Step 2 screening-level risk calculation for terrestrial and aquatic food web exposures. The sections that follow present and discuss the refined screening-level risk calculation and risk evaluation for terrestrial and aquatic food web exposures.

4.7.1.7.1 Terrestrial Food Web Exposures

The refined risk calculation for terrestrial food web exposures to chemicals detected and non-detected in surface soil and subsurface soil with a maximum HQ greater than 1.0 is presented below.

Terrestrial Food Web Exposures: Surface Soil

Table 4-26 presented the results of the Step 2 screening-level risk calculation for terrestrial food web exposures to chemicals in surface soil. Based on the comparison of maximum exposure doses to NOAEL-based screening values, three detected organochlorine pesticides (4,4'-DDD, 4,4'-DDE, and 4,4'-DDT), one detected PCB (Aroclor-1260), and eight detected metals (cadmium, chromium, cobalt, copper, lead, mercury, tin, and zinc) had HQ values greater than or equal to 1.0 for one or more of the terrestrial avian receptors. Dioxin/furan chemicals were identified as ecological COPCs for terrestrial food web exposures because the maximum exposure dose, based on the sum of 2,3,7,8-TCDD equivalents, exceeded NOAEL based screening values for each of the avian receptors. One detected VOC (ethylbenzene), one detected SVOC (butylbenzylphthalate), and one detected metal (beryllium) also were identified as ecological COPCs for terrestrial food web exposures due to the lack of an ingestion-based screening value. Two non-detected PCBs (Aroclor-1248 and Aoclor-1254) were identified as ecological COPCs because maximum exposure doses for the American robin exceeded NOAEL-based screening values. Table 4-59 presents the results of the refined risk calculation for detected and non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA with a maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation is presented in the paragraphs that follow.

As discussed above, two non-detected PCBs (Aroclor-1248 and Aroclor-1254) were identified as ecological COPCs in Step 2 of the screening-level ERA because exposure doses based on maximum reporting limits for the American robin exceeded the NOAEL-based screening value for established for this receptor. Mean American robin exposure doses for both PCBs were less than NOAEL-based screening values (see Table 4-59). Based on mean exposure doses less than NOAEL-based screening values, it is unlikely that Aroclor-1248 and Aroclor-1254 are impacting avian omnivore populations. For this reason, these two non-detected PCBs are not identified as potential ecological risk drivers for SWMU 1 surface soil terrestrial food web exposures, and no additional evaluation is recommended.

Aroclor-1260 was detected and identified as an ecological COPC in Step 2 of the screening-level ERA because the maximum exposure dose for the American robin exceeded the NOAEL-based screening value (HQ = 2.76). This PCB was detected in one of eighteen surface soil samples (440 ug/kg in 1SD02. As evidenced by Table 4-59, the mean exposure dose is less than the NOAEL-based screening value. Based on mean HQ values less than NOAEL-, MATC-, and LOAEL-based screening values and the low magnitude and frequency of detection, it is unlikely that Aroclor-1260 is impacting avian omnivore populations. Therefore, Aroclor-1260 is not

identified as a potential ecological risk driver for SWMU 1 terrestrial food web exposures, and no additional evaluation is recommended.

Based on the comparison of mean exposure doses to NOAEL-based screening values, chromium, cobalt, copper, and tin had HQ values less than 1.0 for each of the terrestrial avian receptors. As evidenced by the statistical evaluation presented in Table 4-46, the concentration distributions of chromium and cobalt in SWMU 1 surface soil are statistically equivalent to background levels, indicating that these two metals are not site-related and not presenting risks to avian receptors populations above background levels. Based on mean HQ values less than one for each the avian receptors, as well as the descriptive and distributional statistics presented in Table 4-46, chromium and cobalt are not considered potential risk drivers for terrestrial food web exposures, and additional evaluation is not recommended.

Although the statistical evaluations presented in Table 4-46 also concluded that copper and tin concentrations in SWMU 1 surface soil are statistically equivalent to background concentrations, the descriptive statistics for the SWMU 1 and background surface soil data sets indicate that concentrations are elevated above background. To further evaluate the significance of copper and tin detections in SWMU 1 surface soil, a spatial examination of the data was performed to determine if individually detected concentrations have the potential to impact terrestrial avian populations that may be using SWMU 1 as foraging habitat. This was accomplished by calculating surface soil concentrations that would result in mean exposure doses greater the NOAEL-based screening values. If maximum detected concentrations are less than minimum concentrations that would result in mean exposure doses greater than NOAEL-based screening values, it can be concluded that these chemicals are not presenting risks to avian receptor populations. Surface soil concentrations equivalent to mean exposure doses greater than or equal to NOAEL-based screening values were derived and compared to individual samples results to determine the spatial distribution of ecologically significant concentrations. Surface soil concentrations resulting in mean exposure doses greater than or equal to NOAEL-based screening values were derived using the following formula (Equation 4-6):

$$SC_x = \frac{(0.99)(NOAEL_{ij})(BW_j)}{[[\sum_i (FIR_j)(BAF_{xi} / BAF_{xi})(PDF_{ij})] + [(FIR_j)(PDS_j)]][AUF]}$$

where:

SC_x	=	Mean concentration of chemical x in surface soil resulting in a mean exposure dose greater than or equal to the NOAEL-based screening value (mg/kg, dry weight)
$NOAEL_{ij}$	=	Ingestion-based screening value for chemical i applied to receptor j (mg chemical/kg body weight/day)
BCF_{xi}	=	*Mean/median surface soil-to-biota BCF for chemical x and food item i (dry weight basis)*
BAF_{xi}	=	*Mean/median surface soil-to-biota BAF for chemical x and food item i (dry weight basis)*
BW_j	=	Mean body weight for receptor j (kg, wet weight)
FIR_j	=	Mean food ingestion rate for receptor j (kg/day, dry-weight)
PDF_{ij}	=	Proportion of receptor j diet comprised of food item I (mg/kg, dry weight)
PDS_j	=	Proportion of receptor j diet composed of surface soil (dry weight basis)
AUF_j	=	Area Use Factor for receptor j (unitless)

Using Equation 4-6 above, a copper concentration in surface soil greater than 1,779 mg/kg would result in a modeled mean exposure dose for the most sensitive receptor (American robin) that is greater than or equal to the NOAEL-based screening value. Given that the maximum detected concentration of copper in SWMU 1 surface soil was 1,020 mg/kg, this metal would not be expected to impact avian receptor populations. A tin concentration greater than 122 mg/kg would result in a modeled mean exposure doses for the most sensitive receptor (American robin) that is greater than NOAEL-based screening values. Based on an examination of the SWMU 1 surface soil data, this threshold concentration was exceeded in a single surface soil sample (181 mg/kg in 1SD01). Given the low frequency and magnitude of detections above the concentration resulting in mean exposure doses greater than the NOAEL-based screening value for the most sensitive receptor, tin also would not be expected to impact avian receptor populations. Based on the evaluation presented above, copper and tin are not identified as potential risk drivers for avian terrestrial food web exposures, and additional evaluation is not recommended.

Mean cadmium, lead, mercury, and zinc exposure doses were greater than or, in the case of mercury, equal to NOAEL-based screening values for the American robin (HQs = 1.25, 1.85, 1.00, and 1.67, respectively). The mean lead exposure dose also exceeded the NOAEL-based screening value for the morning dove (HQ = 1.02). Mean cadmium, lead, mercury, and zinc exposure doses were less than MATC- and LOAEL-based screening values for each receptor. As discussed in Section 4.7.2.1, the descriptive statistics presented in Table 4-46 show that cadmium, mercury, lead, and zinc concentrations in SWMU 1 surface soil are elevated above background concentrations. To evaluate the significance of cadmium, lead, mercury, and zinc detections in SWMU 1 surface soil, surface soil concentrations equivalent to mean exposure doses greater than or equal to NOAEL-based screening values were derived and compared to individual samples results to determine the spatial distribution of ecologically significant concentrations. Using Equation 4-6, cadmium, lead, mercury, and zinc concentrations greater than 4.2 mg/kg, 60 mg/kg, 0.108 mg/kg, and 124 mg/kg, respectively would result in mean exposure doses greater than or equal to NOAEL-based screening values. Cadmium concentrations greater than 4.2 mg/kg were detected in 1SS07 (83.8 mg/kg), 1SD01 (4.7 mg/kg), 1SS10 (5.1 mg/kg), 1SS11 (9.4 mg/kg), and 1SS13 (12 mg/kg), lead concentrations greater than 60 mg/kg were detected in 1SS06 (79.4 mg/kg), 1SS07 (101 mg/kg), 1SD01 (659 J mg/kg), 1SD02 (966 J mg/kg), 1SS10 (680 J mg/kg), 1SS11 (94 J mg/kg), and 1SS13 (660 J mg/kg), mercury concentrations greater than 0.108 mg/kg were detected in 1SD01 (0.85 mg/kg), 1SD02 (0.2 mg/kg), 1SS08 (0.11 mg/kg), 1SS10 (0.44 mg/kg), 1SS11 (0.15 mg/kg), and 1SS13 (0.59 mg/kg), and zinc concentrations greater than 124 mg/kg were detected in 1MW01 (140 J mg/kg), 1SS06 (136 J mg/kg), 1SS07 (223 mg/kg), 1SD01 (1,780 mg/kg), 1SD02 (1,100 mg/kg), 1SS10 (1,600 J mg/kg), 1SS11 (190 J mg/kg), and 1SS13 (2,000 J mg/kg). Based on mean exposure doses greater than NOAEL-based screening values, the frequency and/or magnitude of detections greater than concentrations that would result in mean exposure doses greater than or equal to NOAEL-based screening values, and the their co-location at ecologically important concentrations, cadmium, lead, mercury, and zinc are identified as potential risk drivers for avian omnivore populations. Lead is also considered a potential risk driver for avian herbivores. Detected concentrations at 1SD01, 1SD02, 1SS10, and 1SS13 exceed the concentration that would result in a mean exposure dose greater than the NOAEL-based screening value for the morning dove (109 mg/kg). Although the frequency of detections above this ecologically significant concentration is low (four of twenty-three samples), the magnitude of these four detections is high.

Mean exposure doses for 4,4'-DDE, and 4,4'-DDT exceeded NOAEL-based screening values for the American robin (HQs = 11.42 and 1.14, respectively). The mean exposure dose for 4,4'-DDE also exceeded MATC- and LOAEL-based screening values established for this receptor. Mean exposures doses for these two organochlorine pesticides were less than NOAEL-based screening values for the mourning dove and red-tailed hawk. Mean exposure doses for 4,4'-DDD were less

than ingestion-based screening values for each receptor. A spatial examination of the 4,4'-DDD and 4,4'-DDT data sets indicates that one detected concentration for each pesticide (13,000 ug/kg and 43,000 J ug/kg, respectively in 1SS16) would result in mean modeled exposure doses greater than the NOAEL-based screening value (i.e., a 4,4'-DDD concentration greater than 540 ug/kg and a 4,4'-DDT concentration greater than 1,390 ug/kg would result in a modeled exposure doses greater than NOAEL-based screening values). Using an identical method of examination, eight detected 4,4'-DDE concentrations (28,000 ug/kg in 1SS16, 1,700 ug/kg in 1SS13, 1,300 ug/kg in 1SS06, 810 ug/kg in 1SS10, 610 ug/kg in 1SB03-00, 280 ug/kg in 1SS07, 930 ug/kg in 1SD01, and 370 ug/kg in 1SD02) would result in a modeled exposure dose greater than the NOAEL-based screening value (i.e., a 4,4'-DDE concentration greater than 106 mg/kg would result in a modeled exposure dose greater than the NOAEL-based screening value). Based on the frequency and/or magnitude of detections above concentrations that would result in a mean exposure doses greater than the NOAEL-based screening values, 4,4'-DDD, 4,4-DDE, and 4,4'-DDT are identified as a potential risk driver for SWMU 1 terrestrial food web exposures, and additional evaluation is recommended.

Dioxins/furans were identified as ecological COPCs in Step 2 of the screening-level ERA because maximum exposure doses exceeded NOAEL-based screening values for each of the avian receptors. The mean American robin exposure dose for dioxin/furan chemicals exceeded the NOAEL-based screening value (HQ = 2.22), but was less than MATC- and LOAEL-based screening values established for this receptor (HQs = 0.04 and 0.22, respectively). Mean exposure doses for the mourning dove and red-tailed hawk were less than NOAEL-based screening values (HQs = 0.04 and 0.20, respectively). Given that the mean dioxin/furan equivalent concentration did not exceed the MATC- and LOAEL-based screening value for the American robin and NOAEL-based screening values for the mourning dove and red-tailed hawk, it is unlikely that dioxins/furans are impacting avian omnivore, herbivore, and carnivore populations that may be foraging within the terrestrial habitat offered by SWMU 1. For this reason, dioxins/furans are not identified as potential ecological risk drivers for SWMU 1 terrestrial food web exposures, and no additional evaluation is recommended.

Ingestion-based screening values are not available for ethylbenzene, butylbenzylphthalate, and beryllium. Ethylbenzene and butylbenzylphthalate were each detected in a single surface soil sample (ethylbenzene was detected at 1 J ug/kg in 1SB01-00 and butylbenzylphthalate was detected in 1SS07 at 75 J ug/kg). Given the low frequency and magnitude of detections, ethylbenzene and butylbenzylphthalate are not likely to impact avian receptor populations. As such, they are not identified as potential risk drivers for terrestrial food web exposures, and additional evaluation is not recommended. Beryllium was detected in each surface soil sample. The statistical evaluation presented in Table 4-46 showed that beryllium concentrations at the site are elevated above background levels. However, this comparison to background provides no indication of potential impacts to avian receptor populations. As discussed in Section 4.4.2, all metals were evaluated for terrestrial food web exposures as a measure of conservatism. However, there is no indication that beryllium is an important bioaccumulative chemical (USEPA 2000a). For this reason, additional evaluation is not recommended for this metal.

In summary, cadmium, lead, mercury, zinc, 4,4'-DDD, 4,4'-DDE, and 4,4'-DDT were identified as potential risk drivers for avian omnivore populations. Lead was also identified as a potential risk driver for avian herbivore populations. Although ethylbenzene, butylbenzylphthalate, dioxins/furans, beryllium, chromium, and cobalt were detected and identified as ecological COPCs in Step 2 of the screening-level ERA for terrestrial food web exposures, they are not recommended for additional evaluation based on the discussion presented in the preceding paragraphs. Additional evaluation also is not recommended for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

Terrestrial Food Web Exposures: Subsurface Soil

Table 4-27 presented the results of the Step 2 screening-level risk calculation for terrestrial food web exposures to chemicals in subsurface soil. Based on the comparison of maximum exposure doses to NOAEL-based screening values, cobalt had an HQ value greater than 1.0 for the American robin. Beryllium also was detected and identified as an ecological COPC for terrestrial food web exposures due to the lack of an ingestion-based screening value. Two non-detected SVOCs (di-n-butylphthalate and hexachlorobenzene) and five non-detected PCBs were identified as ecological COPCs because exposure doses based on maximum reporting limits exceeded NOAEL-based screening values for the American robin. Table 4-60 presents the results of the refined risk calculation for detected and non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA with an HQs greater than 1.0. An evaluation of the refined screening-level risk calculation is presented in the paragraphs that follow.

Two non-detected SVOCs (di-n-butylphthalate and hexachlorobenzene) and five non-detected PCBs were identified as ecological COPCs in the Step 2 screening-level risk calculation because maximum exposure doses based on reporting limits exceeded NOAEL-based screening values for the American robin. As evidenced by Table 4-60, mean exposure doses for these non-detected SVOCs and PCBs are less than NOAEL-based screening values (HQs less than 1.0), indicating little potential to impact terrestrial avian populations. The non-detected SVOCs and PCBs also were not detected in SWMU 1 surface soil. The absence of detections in both surface and subsurface soil indicates that they are not likely to be related to historical activities at SWMU 1. Based on mean HQs less than 1.0 and/or no indication that that they are related to historical site activities, the non-detected chemicals identified as ecological COPCs in Step 2 of the ERA are not identified as potential ecological risk drivers for SWMU 1 terrestrial subsurface soil food web exposures.

Based on a comparison of mean exposure doses to NOAEL-based screening values, cobalt had a HQ less than 1.0 (American robin HQ = 0.35). A mean HQ less than 1.0 indicates that cobalt has little potential to impact terrestrial avian omnivore populations. As evidenced by the statistical evaluation presented in Table 4-48, the distribution of cobalt concentrations in SWMU 1 subsurface soil are statistically equivalent to background surface soil. Based on a mean HQ less than 1.0 and the statistical evaluation presented in Table 4-48, cobalt is not considered a potential risk driver for subsurface soil terrestrial food web exposures, and additional evaluation is not recommended.

Beryllium was identified as an ecological COPC for terrestrial food web exposures based on the lack of an ingestion-based screening value. Although concentrations detected in SWMU 1 subsurface soil were elevated above background levels (see Table 4-48), beryllium is not considered an important bioaccumulative chemical (USEPA, 2000a). As such, this metal is not likely to bioaccumulate in terrestrial avian prey items at ecologically important concentrations. For this reason, beryllium is not identified as a potential risk driver, and additional evaluation is not recommended.

In summary, there are no potential ecological risk drivers identified for SWMU 1 subsurface soil terrestrial food web exposures. Although beryllium and cobalt were detected and identified as ecological COPCs in the Step 2 screening-level risk calculation, they are not recommended for additional evaluation based on the discussion presented in the preceding paragraphs. Additional evaluation also is not recommended for the non-detected chemicals identified as ecological COPCs for subsurface soil terrestrial food web exposures in Step 2 of the screening-level ERA

4.7.1.7.2 Aquatic Food Web Exposures

The refined risk calculation for aquatic food web exposures to chemicals detected in estuarine wetland and Ensenada Honda surface water and sediment downgradient from SWMU 1 is presented below.

Aquatic Food Web Exposures: Estuarine Wetland Surface Water and Sediment

Table 4-28 presented the results of the Step 2 screening-level risk calculation for aquatic food web exposures to chemicals in estuarine wetland surface water and sediment downgradient from SWMU 1. Based on the comparison of maximum exposure doses to NOAEL-based screening values, three detected organochlorine pesticides (4,4'-DDD, 4,4'-DDE, and 4,4'-DDT) and eight detected metals (chromium, cobalt, copper, lead, mercury, selenium, vanadium, and zinc) had HQ values greater than or equal to 1.0 for one or more of the aquatic receptors. Three detected VOCs (chlorobenzene, ethylbenzene, and toluene), one detected chlorinated herbicide (2,4,5-T), and one detected metal (beryllium) also were identified as ecological COPCs for estuarine wetland aquatic food web exposures due to the lack of ingestion-based screening values. Five non-detected SVOCs [bis(2-ethylhexyl)phthalate, di-n-butylphthalate, dinoseb, hexachlorobenzene, and hexachlorobutadiene), four non-detected organophosphorous pesticides, and seven non-detected PCBs were identified as ecological COPCs because maximum exposure doses exceeded NOAEL-based screening values for each of the avian receptors. Although not detected, dioxin/furan chemicals were identified as ecological COPCs because maximum exposure doses, based on the sum of 2,3,7,8-TCDD equivalents, exceeded NOAEL-based screening values for each of the avian receptors. Table 4-61 presents the results of the refined risk calculation for detected and non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA with a maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation is presented in the paragraphs that follow.

As discussed above, five non-detected SVOCs, four non-detected organophosphorous pesticides, one-non-detected organochlorine pesticide (4,4'-DDT), dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration), and seven non-detected PCBs were identified as ecological COPCs because maximum exposure doses exceeded NOAEL-based screening values for each of the avian receptors. Mean bis(2-ethylhexyl)phthalate, dinoseb, hexachlorobutadiene, 4,4'-DDT, PCB, disulfoton, and phorate exposure doses were less than ingestion-based screening values for each of the avian receptors, while mean di-n-butylphthalate, hexachlorobenzene, ethyl parathion, and methyl parathion exposure doses exceeded ingestion-based screening values for each receptor. Of the eighteen non-detected chemicals identified as ecological COPCs for estuarine wetland food web exposures in the Step 2 screening-level risk calculation (i.e., non-detected chemicals with maximum exposure doses greater than ingestion-based screening values), only bis(2-ethylhexyl)phthalate, butylbenzylphthalate, 4,4'-DDT, Aroclor-1260, and dioxins/furans were detected in upgradient surface or subsurface soil. Based on the absence of detections in SWMU 1 surface and subsurface soil, there is no indication that the remaining thirteen non-detected chemicals are associated with historical site activities (no evidence of a source or release to surface and subsurface soil). For this reason, they are not identified as potential ecological risk drivers for estuarine wetland aquatic food web exposures, and no additional evaluation is recommended.

Although detected in upgradient surface or subsurface soil, bis(2-ethylhexyl)phthalate, butylbenzylphthalate, 4,4'-DDT, Aroclor-1260, and dioxins/furans also are not identified as potential ecological risk drivers for estuarine wetland aquatic food web exposures. Mean NOAEL-based HQ values for bis(2-ethylhexyl)phthalate and Aroclor-1260 are both less than 1.0, indicating little potential to impact avian receptor populations. These two organic chemicals also

were detected at a low frequency and magnitude of detection in upgradient surface and subsurface soil. Bis(2-ethylhexyl)phthalate was detected in two of eighteen surface soil samples at a maximum concentration of 200 J ug/kg and on of six subsurface soil samples at 430 J ug/kg. Aroclor-1260 was detected in one of eighteen SWMU 1 surface soil samples at 440 ug/kg. This PCB was not detected in SWMU 1 subsurface soil. The low frequency and magnitude of detections in upgradient media indicate little potential for migration to the estuarine wetland at ecologically important concentrations. Butylbenzylphthalate also has little potential to impact estuarine wetland avian receptor populations. This phthalate was only detected in one of eighteen surface soil samples at a concentration of 75 J ug/kg and was not detected in SWMU 1 subsurface soil. The low frequency and magnitude of detection does not suggest that butylbenzyphthalate is migrating to the estuarine wetland system at ecologically important concentrations. Dioxin/furan congeners were detected in each of the surface soil samples collected during the 2004 additional data collection field investigation, while 4,4'-DDT was detected in twenty-one of twenty-eight surface soil and seven of nine subsurface soil samples. However, they have little potential to migrate to the estuarine wetland system based on their physical/chemical properties (low water solubility and high K_{ow} and K_{oc} values) and the vegetative cover present within the SWMU 1 coastal forest community. Both of these factors would minimize potential migration with surface run-off in either a dissolved or solid state (i.e., adhered to soil particles). In summary, the non-detected chemicals are not identified as potential ecological risk drivers for estuarine wetland food web exposures, based on mean HQs less than 1.0, the absence of detections in upgradient media, the low frequency and magnitude of detections in upgradient media, and/or their physical/chemical properties.

Mean exposure dose for 4,4'-DDD and 4,4'-DDT were less than NOAEL-based screening values for each of the avian aquatic receptors (see Table 4-61). However, the mean exposure dose for 4,4'-DDE exceeded the NOAEL-based screening value for the spotted sandpiper, belted kingfisher, and great blue heron. Mean 4,4'-DDE exposure doses were less than MATC- and LOAEL-based screening values for each receptor. The statistical evaluation presented in Table 4-53 demonstrated that the distributions of 4,4'-DDD and 4,4'-DDE concentrations in estuarine wetland sediment downgradient from SWMU 1 are statistically equivalent to background concentrations. The descriptive statistics for the 4,4'-DDD and 4,4'-DDE data sets support this conclusion. For both pesticides, maximum, mean, and 95 percent UCL concentrations for the background data set exceeded maximum, mean, and 95 percent UCL concentrations for the estuarine wetland data set. Furthermore, as discussed in Section 4.7.1.4, a spatial examination of detected concentrations shows that the nature and extent of 4,4'-DDD and 4,4'-DDE detections in estuarine wetland sediment has been adequately characterized. Specifically, maximum concentrations were not detected in the southernmost sediment samples (1EWS11, 1EWS13, and 1EWS14 [see Figure 4-12]). Finally, the parent compound of 4,4'-DDD and 4,4'-DDT (i.e., 4,4'-DDT) was not detected in estuarine wetland sediment; therefore, the concentration of these two compounds would not be expected to increase due to the biodegradation of 4,4'-DDT. Based on the descriptive and distributional statistics presented in Table 4-53, the spatial distribution of detected concentrations, the absence of parent compound detections in estuarine wetland sediment, and in the case of 4,4'-DDD, a NOAEL-based HQ less than 1.0, these two pesticides are not identified as potential risk drivers for estuarine wetland sediment downgradient from SWMU 1, and no additional evaluation is recommended.

As evidenced by Table 4-61, mean exposure doses for cobalt, lead, selenium, and zinc were less than NOAEL-based screening values for each avian receptor (i.e., spotted sandpiper, belted kingfisher, and great blue heron). Furthermore, the results of the distributional and descriptive statistics presented in Table 4-53 showed that the distributions of cobalt, lead, selenium, and zinc concentrations in estuarine wetland sediment downgradient from SWMU 1 are statistically equivalent to concentrations in background sediment. Based on mean exposure doses less than

screening values and the statistical evaluation presented in Table 4-53, cobalt, lead, selenium, and zinc are not considered potential risk drivers for estuarine wetland food web exposures, and additional evaluation is not recommended.

Although mean HQ values for mercury and vanadium exceeded NOAEL-based screening values for the spotted sandpiper, belted kingfisher, and/or great blue heron, their concentration distributions in estuarine wetland sediment were found to be statistically equivalent to background levels. This indicates that these metals are not site-related and are not presenting risks to avian receptor populations above background levels. For this reason, mercury and vanadium are not considered potential risk drivers for estuarine wetland food web exposures, and additional evaluation is not recommended.

The mean exposure dose for chromium exceeded the NOAEL-based screening value for the spotted sandpiper (HQ = 1.82). The statistical evaluation presented in Table 4-53 demonstrated that the distribution of chromium concentrations in estuarine wetland sediment downgradient from SWMU 1 is elevated above background levels. A spatial examination of the chromium data set indicates that detected concentrations at each sampling point would result in a modeled exposure dose greater than the NOAEL-based screening value (i.e., a concentration greater than 21.9 mg/kg results in a modeled exposure dose greater than the NOAEL-based screening value for the spotted sandpiper). A spatial examination of the background data set also indicates that detected concentrations at two of six sampling locations would result in modeled exposure doses greater than the NOAEL-based screening value. Chromium concentrations in upgradient surface soil and subsurface soil also were found to be statistically equivalent to background levels (see Table 4-46 and 4-48, respectively), indicating that there has not been a release to surface soil or subsurface soil at SWMU 1.

The use of NOAEL-based screening values is extremely conservative since they give no indication as to how much higher a concentration must be before adverse effects are observed. Given that LOAEL- and MATC-based HQ values for chromium are less than 1.0 for the spotted sandpiper (see Table 4-61) and the conservatism associated with the use of NOAELs, it is unlikely that chromium is presenting unacceptable risks to avian invertebrate consumers foraging within the estuarine wetland downgradient from SWMU 1. The absence of evidence suggesting that a chromium release has occurred at SWMU 1 further supports the exclusion of this metal as a potential risk driver.

Mean exposure doses for copper were less than NOAEL-based screening values (see Table 4-61). A spatial examination of the estuarine wetland data set also indicates that detected concentrations at each sampling point, which ranged from 32 mg/kg at 01EWSSD07 to 140 mg/kg at 01EWSSD05, would result in a modeled exposure dose less than the NOAEL-based screening value (i.e., a concentration greater than 281 mg/kg would result in a modeled exposure dose greater than the NOAEL-based screening value for the most sensitive receptor [spotted sandpiper]). Based on mean HQ values less than 1.0 for each aquatic receptor, as well as the lack of detected sediment concentrations above a concentration resulting in a mean exposure dose greater than the NOAEL-based screening value for the most sensitive receptor, copper is not identified as a potential risk driver for aquatic food web exposures. As such, additional evaluation is not recommended.

Ingestion-based screening values are not available for chlorobenzene, ethylbenzene, toluene, 2,4,5-T, and beryllium. Chlorobenzene was detected in two sediment samples (3.4 J ug/kg in 01EWSSD01 and 5.8 J mg/kg in 01EWSSD06), ethylbenzene was detected in three sediment samples (2.5 J ug/kg in 01EWSSD01, 3 J ug/kg in 01EWSSD02, and 1.2 J ug/kg in 01EWSSD03), while toluene was detected in two sediment samples (3.1 J ug/kg in 01EWSSD01

and 3.7 J ug/kg in 01EWSSD06). Ethylbenzene and toluene were also detected in a single surface water sample (0.13 J ug/L and 0.2 J ug/L, respectively in 01EWSSW05). In the case of ethylbenzene and toluene, the low BCF values presented in Table 4-17 (15.1 and 37.2, respectively) indicate that these two VOCs would not be expected to bioaccumulate in aquatic prey items to a significant extent. Furthermore, chlorobenzene and toluene were not detected in upgradient surface or subsurface soil, while ethylbenzene was detected in a single upgradient surface soil sample (1 J ug/kg in 1SB01). These data indicate that chlorobenzene, ethylbenzene, and toluene are not likely to be related to historical activities at the SWMU (no evidence of a source or a release to surface soil). Given the low magnitude of detections, no indication that they are associated with a release from the SWMU, and/or the low potential to bioaccumulate in aquatic prey items, chlorobenzene, ethylbenzene, and toluene are not likely to impact avian receptor populations foraging within the estuarine wetland downgradient from SWMU 1. As such, they are not identified as potential risk drivers for estuarine wetland aquatic food web exposures, and additional evaluation is not recommended. 2,4,5-T was detected in one of nine estuarine wetland sediment samples (2.5 ug/kg in 1EWS07). This herbicide was not detected in SWMU 1 surface soil, indicating that 2,4,5-T is not associated with historical site activities at SWMU 1. Based on the low frequency and magnitude of detections and the absence of detections in SWMU 1 surface soil, 2,4,5-T is not identified as a potential ecological risk driver for estuarine wetland sediment, and no additional evaluation is recommended. Beryllium was detected in each sediment sample. Concentrations ranged from 0.088 J mg/kg in 01EWSSD07 to 0.31 mg/kg in 01EWSSD03. This metal was not detected in the total recoverable fraction of the estuarine wetland surface water samples. The statistical evaluation presented in Table 4-53 indicates that beryllium concentrations at the site are statistically equivalent to background concentrations. As such, this metal would not be expected to present risks to avian aquatic receptors above background levels. Based on the statistical evaluation presented in Table 4-53, beryllium also is not identified as a potential risk driver for estuarine wetland sediment downgradient from SWMU 1, and additional evaluation is not recommended.

In summary, there are no potential ecological risk drivers identified for aquatic food web exposures to chemicals in estuarine wetland surface water and/or sediment downgradient from SWMU 1. Although chlorobenzene, ethylbenzene, toluene, 4,4'-DDE, 4,4'-DDD, 2,4,5-TP, beryllium, chromium, cobalt, copper, lead, mercury, selenium, vanadium, and zinc were detected in estuarine wetland sediment and/or surface water and identified as ecological COPCs in Step 2 of the ERA, they are not recommended for further evaluation based on the discussion presented in the preceding paragraphs. Additional evaluation also is not recommended for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

Aquatic Food Web Exposures: Ensenada Honda Surface Water and Sediment

Table 4-29 presented the results of the Step 2 screening-level risk calculation for aquatic food web exposures to chemicals in Ensenada Honda surface water and sediment downgradient from SWMU 1. Based on the comparison of maximum exposure doses to NOAEL-based screening values, arsenic, mercury, selenium, and di-n-butylphthalate had HQ values greater than or equal to 1.0 for either the West Indian manatee (arsenic, mercury, and selenium) or the double-crested cormorant (mercury and di-n-butylphthalate). One VOC (ethylbenzene), one SVOC (4-chlorophenylphenyl ether), one chlorinated herbicide (2,4,5-T), and one metal (beryllium) also were detected and identified as ecological COPCs based on the lack of ingestion-based screening values for one or more of the aquatic receptors. Three non-detected organophosphorous pesticides (disulfoton, ethyl parathion, and methyl parathion) and dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration) were identified as ecological COPCs because maximum exposure doses for the West Indian manatee and/or double crested cormorant exceeded NOAEL-based screening values. Table 4-62 presents the results of the refined risk calculation for detected

and non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA with a maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation is presented in the paragraphs that follow. As discussed in Section 4.7, refined risk estimates were not derived for the West Indian manatee. As such, the risk characterization for this receptor did not include an evaluation of refined screening-level risk estimates.

Three non-detected organophosphorous pesticides (disulfoton, ethyl parathion, and methyl parathion) and dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration) were identified as ecological COPCs because maximum exposure doses exceeded NOAEL-based screening values for the West Indian manatee and/or double-crested cormorant. Disulfoton, ethyl parathion, and methyl parathion were not detected in SWMU 1 surface and subsurface soil. Based on the absence of detections in SWMU 1 surface and subsurface soil, there is no indication that these three organochlorine pesticides are associated with historical site activities (no evidence of a source or release to surface and subsurface soil). For this reason, they are not identified as potential ecological risk drivers for Ensenada Honda food web exposures, and no additional evaluation is recommended. Dioxin/furan congeners were detected in each of the surface soil samples collected during the 2004 additional data collection field investigation. However, they have little potential to migrate to the estuarine wetland system based on their physical/chemical properties (low water solubility and high K_{ow} and K_{oc} values) and the vegetative cover present within the SWMU 1 coastal forest community. Both of these factors would minimize potential migration with surface run-off in either a dissolved or solid state (i.e., adhered to soil particles).

Di-n-butylphthalate and mercury were detected and identified as an ecological COPC in Step 2 of the ERA because maximum exposure doses derived for the double-crested cormorant exceeded NOAEL-based screening values. As evidenced by Table 4-62, mean exposure doses for both chemicals are less than NOAEL-based screening values, indicating that di-n-butylphthalate and mercury are not likely to impact avian piscivore populations foraging within the Ensenada Honda. To further evaluate the significance of di-n-butylphthalate and mercury detections in Ensenada Honda sediment, Equation 4-7 was used to derive sediment concentrations for each chemical that would result in a mean exposure dose greater than or equal to the NOAEL-based screening value.

$$SC_x = \frac{(0.99)(NOAEL_{Ij})(BW_j)}{[[\sum_i (FIR_j)(BAF_{xi})(PDF_{ij})] + [(FIR_j)(PDS_j)]][AUF]}$$

where:

SC_x	=	Mean concentration of chemical x in sediment resulting in a mean exposure dose greater than or equal to the NOAEL-based screening value (mg/kg, dry weight)
$NOAEL_{ij}$	=	Ingestion-based screening value for chemical i applied to receptor j (mg chemical/kg body weight/day)
BAF_{xi}	=	*Mean/median surface soil-to-biota BAF for chemical x and food item i (dry weight basis)*
BW_j	=	Mean body weight for receptor j (kg, wet weight)
FIR_j	=	Mean food ingestion rate for receptor j (kg/day, dry-weight)
PDF_{ij}	=	Proportion of receptor j diet compised of food item I (mg/kg, dry weight)
PDS_j	=	Proportion of receptor j diet composed of surface soil (dry weight basis)
AUF_j	=	Area Use Factor for receptor j (unitless)

A di-n-butylphthalate concentration greater than 2,750 ug/kg and a mercury concentration greater than 0.049 mg/kg would result in mean exposure doses greater than NOAEL-based screening

values for the double-crested cormorant. Given that the maximum di-n-butylphthalate concentration in Ensenada Honda sediment was 610 J ug/kg (01OWSD02), no single detected concentration exceeds the threshold concentration of 2,750 ug/kg. As such, di-n-butylphthalate detections in Ensenada Honda sediment would not be expected to impact avian piscivore populations. Based on a mean HQ less than 1.0 and the lack of detected concentrations greater than the sediment concentration that would result in a mean exposure dose greater than the NOAEL-based screening value, di-n-butylphthalate is not identified as a potential ecological risk driver for Ensenada Honda food web exposures, and no additional evaluation is recommended. Mercury also is not identified as a potential ecological risk driver for Ensenada Honda food web exposures. Although mercury was detected in two Ensenada Honda sediment samples at concentrations above the threshold concentration that would result in a mean exposure dose greater than the NOAEL-based screening value (0.062 J mg/kg in 01OWSD03 and 0.066 mg/kg in 01OWSD05), the magnitude of the detections is low. A mean HQ less than 1.0 also indicates that mercury is not likely to impact avian piscivore populations foraging within the Ensenada Honda.

Arsenic, mercury, and selenium were identified as ecological COPCs for Ensenada Honda aquatic food web exposures in Step 2 of the screening-level ERA because maximum exposure doses for the West Indian manatee exceeded NOAEL-based screening values. Selenium was detected in each Ensenada Honda sediment sample at concentrations ranging from 0.53 J mg/kg in 1OWSD06 to 1.2 J mg/kg in 1OWSD04. An examination of SWMU 1 surface soil data (see Appendix G.1) indicates that selenium was detected in nine of eighteen surface soil samples at a maximum concentration of 0.8 J mg/kg (1SB01). Selenium was not detected in SWMU 1 subsurface soil (see Appendix G.2). The maximum detected selenium concentration in SWMU 1 surface soil (0.8 J mg/kg) is less than the maximum background surface soil concentration (1.2 J mg/kg in BGMW03 [see Appendix G.23]). These data indicate that selenium is not associated with historical activities at the SWMU (no evidence of a source or release of selenium to SWMU 1 surface or subsurface soil). Although selenium concentrations in Ensenada Honda sediment do not appear to be related to a release at SWMU 1, this metal is conservatively identified as a potential ecological risk driver for aquatic herbivore food web exposures based on a maximum HQ greater than 1.0 (NOAEL-based HQ = 3.58) and the Federal status of the West Indian manatee in Puerto Rico (i.e., endangered).

Mercury was detected in eight of nine Ensenada Honda sediment samples at concentrations ranging from 0.023 J mg/kg (1OWSD08) to 0.062 mg/kg (1OWSD03). All detected concentrations exceeded the maximum background concentration (0.0062 J mg/kg). Although mercury was identified as a potential ecological risk driver for SWMU 1 surface soil, the spatial distribution of mercury detections in Ensenada Honda sediment do not appear to be related to SWMU 1. First of all, the range of mercury detections in Ensenada Honda sediment was consistent from sample to sample, ranging from 0.023 J mg/kg to 0.062 mg/kg. This small range of detected concentrations does not indicate that mercury is migrating from SWMU 1 source areas to the Ensenada Honda. Secondly, a spatial examination of mercury detections in Ensenada Honda sediment does not appear to be linked to mercury detections in SWMU 1 surface soil. Specifically, the sediment sample containing the maximum mercury detection (1OWSD05) is not located downgradient from the SWMU 1 surface soil samples containing maximum detections (1SD01, 1SS10, and 1SS13 [see Figure 4-9]). Regardless, mercury was identified as a potential ecological risk driver for SWMU 1 surface soil. Based on this fact and the Federal status of the West Indian manatee, mercury is identified as a potential ecological risk driver for mammalian herbivore aquatic food web exposures, and additional evaluation is recommended.

Arsenic was detected in each of the Ensenada Honda sediment samples at concentrations ranging from 5.3 mg/kg (1OWSD04 and 1OWSD09) to 8.7 mg/kg (1OWSD03). All detected

concentrations exceeded the maximum arsenic concentration in background open water sediment (3.0 mg/kg). However, an examination of the SWMU 1 surface soil and subsurface soil data indicates that arsenic is not related to historical activities at SWMU 1. The maximum arsenic detection in SWMU 1 surface soil (5.5 J mg/kg [see Appendix G.1]) is only slightly elevated above the maximum background surface soil concentration (2.5 J mg/kg [see Appendix G.23]). Based on the lack of any indication that arsenic is related to historical activities at SWMU 1, this metal is not likely to be present in Enseanda Honda sediment due to a release at SWMU 1. Regardless, arsenic is conservatively identified as a potential ecological risk driver for aquatic herbivore food web exposures based on a maximum HQ greater than 1.0 (NOAEL-based HQ = 28.2) and the Federal status of the West Indian manatee in Puerto Rico (i.e., endangered).

In addition to arsenic, mercury, and selenium, three additional metals (cadmium, copper, and zinc) are conservatively identified as potential ecological risk drivers for aquatic herbivore food web exposures. These three metals are identified as potential ecological risk drivers based on their bioaccumulative potential (USEPA, 2000b) and maximum HQ values derived using toxicity reference values that incorporate an inter-species extrapolation factor (HQ = 5.66 for cadmium, 1.72 for copper, and 3.43 for zinc; see Table 4-38a).

Ethylbenzene, 2,4,5-T, beryllium, and 4-chlorophenylphenyl ether were detected in Ensenada Honda sediment and identified as ecological COPCs for aquatic food web exposures based on the lack of ingestion- based screening values for the double-crested cormorant. 4-Chlorophenylphenyl ether also lacks an ingestion-based screening value for the West Indian manatee. Ethylbenzene was detected in one of nine sediment samples (1.6 J ug/kg in 01OWSD09). Given the low frequency and magnitude of detected concentrations, it is unlikely that this VOC ethylbenzene is impacting avian piscivore populations foraging within the Ensenada Honda. The low surface water-to-fish BCF value for this VOC (15.1 [see Table 4-17]) also indicates that ethylbenzene will not bioconcentrate in aquatic food items. As such, ethylbenzene is not identified as a potential risk driver for Ensenada Honda aquatic food web exposures, and additional evaluation is not recommended. Identical to ethylbenzene, 2,4,5-T was detected in one of nine Ensenada Honda sediment samples (2.2 J ug/kg in 1OW01). Although detected in a single sediment sample, there is not indication that this chlorinated herbicide is associated with historical activities at the SWMU (2,4,5-T was not detected in SWMU 1 surface soil). Based on the low magnitude and frequency of detection and the absence of detections in upgradient surface soil, 2,4,5-TP is not identified as a potential ecological risk driver for Ensenada Honda aquatic food web exposures, and no additional evaluation is recommended.

Beryllium was detected in eight of nine sediment samples (ranging from 0.045 J mg/kg in 01OWSD09 to 0.082 J mg/kg in 01OWSD03). A statistical evaluation of the Ensenada Honda and background data could not be performed due to the absence of detections in the background data set (see Table 4-58). Because this metal is not identified as an important bioaccumulative chemical (USEPA 2000b), beryllium would not be expected to bioaccumulate in aquatic prey items. For this reason, as well as the low magnitude of detections, beryllium is not identified as a potential risk driver for Ensenada Honda aquatic food web exposures, and additional evaluation is not recommended.

4-Chlorophenylphenyl ether was detected in one of nine sediment samples (680 J ug/kg in 01OWSD05). As discussed in Section 4.7.1.6, there is uncertainty associated with this single detection (detected in a duplicate of sediment sample 01OWSD05 but not the original sample). In order to reduce the uncertainty associated with this pattern of detection, three sediment samples were collected within the immediate vicinity of 01OWSD05 during the October 2004 additional data collection field investigation and analyzed for 4-chlorophenylphenyl ether. As evidenced by Appendix G.11, this SVOC was not detected in the 1OWSD10, 1OWSD11, and

1OWSD12 sediment samples. This SVOC also was not detected in SWMU 1 surface and subsurface soil (see Appendices G.1 and G.2). The absence of detections in upgradient surface and subsurface soil indicates that 4-chlorophenylphenyl ether is not associated with historical site activities at SWMU 1 (no evidence of a source or release to surface and subsurface soil). Based on the absence of detections in the 1OWSD10, 1OWSD11, and 1OWSD12 sediment samples and the absence of detections in SWMU 1 surface and subsurface soil, 4-chlorophenylphenyl ether is not identified as a potential ecological risk driver for Ensenada Honda aquatic food web exposures, and no additional evaluation is recommended.

In summary, arsenic, mercury, and selenium were identified as potential ecological risk drivers for Ensenada Honda aquatic herbivore food web exposures. Cadmium, copper, and zinc also are identified as potential ecological risk drivers for West Indian manatee food web exposures based on their bioaccumulative potential and maximum HQ values derived using toxicity reference values that incorporate an interspecies extrapolation factor. Although beryllium, ethylbenzene, 4-chlorophenylphenyl ether, di-n-butylphthalate, and 2,4,5-T were detected and identified as ecological COPCs in Step 2 of the ERA, they are not considered potential risk drivers for Ensenada Honda aquatic food web exposures based on the discussion presented in the preceding paragraphs. Additional evaluation also is not recommended for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

4.7.1.8 Uncertainties Associated With Step 3a of the Baseline Ecological Risk Assessment for SWMU 1

Many of the uncertainties identified in Section 4.6.2 for the screening-level ERA (i.e., Steps 1 and 2 of the Navy ERA process) also apply to the refined screening-level risk calculation (i.e., Step 3a of the baseline ERA). Those uncertainties specific to the refined risk calculation for SWMU 1 are listed below.

Analytical Data

- Surface soil samples were not collected at several subsurface soil sample locations where elevated concentrations of chemicals identified as potential risk drivers were detected. Because terrestrial receptors would be expected to experience the greatest level of exposure to chemicals in surface soil, the lack of surface soil analytical data at these locations may have resulted in an understatement of potential risks.

- A background subsurface soil data set was not available for statistical comparison to SWMU 1 subsurface soil data. In the absence of background subsurface soil data, background surface soil data were used in the statistical comparison to SWMU 1 subsurface soil data. Because the background surface soil data and SWMU 1 subsurface soil were collected from different depth intervals (0 to 1 foot bgs and 0.5 to 1.5 or 1.0 to 2.0 feet bgs, respectively), the background surface soil data may not be comparable to the SWMU 1 subsurface soil data.

Identification of Potential Risk Drivers

- Non-detected chemicals lacking media-specific and/or ingestion-based screening values were not evaluated in the refined risk calculation, nor were they identified as potential risk drivers. This approach may have resulted in an understatement of the actual number of potential risk driver if any of the non-detected chemicals lacking screening values are present at ecologically significant concentrations. This uncertainty can be indirectly addressed in subsequent steps of the baseline ERA process through site-specific studies

(e.g., toxicity tests and bioaccumulation studies) designed to evaluate those detected chemicals identified as potential ecological risk drivers.

- A second source of uncertainty related to the identification of potential ecological risk drivers applies to the statistical analysis performed on SWMU 1 and background analytical data. Statistical comparisons of SWMU 1 surface soil, subsurface soil, estuarine wetland and open water surface water, and estuarine wetland and open water sediment analytical data to available background data were performed for all detected metals identified as ecological COPCs in the Step 2 screening-level risk calculation. Statistical comparisons included descriptive summaries of each data set (e.g., maximum and mean concentrations), tests of the mean/median of the distributions (two sample t-test, Satterthwaite's t-test, Wilcoxin rank sum test, or Gehan test), and tests of the right tail of the distribution (i.e., slippage test and/or quantile test).

  Full conformity with the standard assumptions underlying each test was not possible for certain chemicals, which introduced uncertainty in the determination of whether a chemical was site-related. This uncertainty was associated with slightly inadequate sample sizes (e.g., Gehan test and quantile test are typically performed on site and background sample sizes greater than ten), unequal sample sizes (maximum power and robustness occur when site and background sample sizes are equal), non-detected results (limits the type and power of statistical evaluations performed), and differences in reporting limits between the background and SWMU data sets. In addition, uncertainty was added when disagreements between statistical tests occurred for certain chemicals when more than one statistical evaluation was performed. When disagreements between tests occurred or when there was uncertainty associated with a particular test, a more detailed examination of the distribution of the data within and/or between data sets was performed. In these cases, comparisons to background were examined in conjunction with an evaluation of the potential for site release (i.e., preliminary conceptual model), the spatial pattern of concentration distributions, and the magnitude of detected concentrations above available screening values. As a conservative measure, chemicals determined to be statistically elevated over background concentrations were conventionally identified as being detected in site-related concentrations. This approach is particularly conservative, however, as the NCP Hazard Ranking System (40 CFR, Part 300, Appendix A) does not recognize a release when any chemical is detected in concentrations less than three times an appropriate background concentration.

- 2,4-dinitrotoluene, 2-nitroaniline, 4-chlorophenylphenyl ether, 4-nitoraniline, di-n-butylphthalate, di-n-octylphthalate, and phenol were were detected in a duplicate sediment sample collected from the Ensenada Honda during the 2003 additional data collection field investigation, but not the original sample (01OWSD05). To address the uncertainty associated with the analytical data, three Ensenada Honda sediment samples (1OWSD10, 1OWSD11, and 1OWSD12) were collected during the October 2004 additional data collection field effort and analyzed for seven SVOCs. 2,4-dinitrotoluene, 2-nitroaniline, 4-chlorophenylphenyl ether, 4-nitoraniline, di-n-butylphthalate, di-n-octylphthalate, and phenol were not detected in the 2004 sediment samples (see Appendix G.11). Based on the absence of detections in the 01OWSD05 sediment sample and the 2004 sediment samples, these seven SVOCs were not identified as potential ecological risk drivers for Ensenada Honda sediment. It is noted that these seven SVOCs were not detected in SWMU 1 surface and subsurface soil (see Appendices G.1 and G.2), indicating that these SVOCs are not associated with historical site activities at SWMU 1 (no evidence of a source or release to surface and subsurface soil).

- The lines of evidence used to identify chemicals as potential risk drivers in the risk characterization, included evaluations that statistically compared background data to SWMU 1 data. As discussed above, a subsurface soil data set was not available for a statistical comparison to SWMU 1 subsurface soil data. Due to the lack of background subsurface soil data, background surface soil data were used in the statistical comparison. Although the depth interval for background surface soil data and SWMU 1 subsurface soil data overlapped, the use of the background surface soil data may have resulted in an overstatement or understatement of potential risk drivers.

- The background surface water and sediment data used in the statistical comparison to Ensenada Honda surface water and sediment were collected from an embayment that does not receive anthropogenic inputs from industrial regions of the facility (e.g., storm water runoff). However, the Ensenada Honda receives inputs from a variety of sources that are associated with anthropogenic activities, including storm water runoff from industrial portions of the facility. As such, potential contamination associated with a release from the SWMU could not be differentiated from anthropogenic inputs. Because a statistical comparison to background concentrations was used as a line of evidence in the Step 3a risk characterization, the lack of background data from a surface water body receiving anthropogenic inputs similar to those potentially impacting surface water and sediment quality within the Ensenada Honda may have resulted in an overstatement of the actual number of potential risk drivers.

4.7.1.9 Step 3a Decision Point and Recommendations for SWMU 1

Table 4-63 presents a summary of the ecological COPCs identified in Step 2 of the screening-level ERA, as well as the potential risk drivers identified in Step 3a of the baseline ERA. Recommendations for each media and food web exposure pathway are presented in the sections that follow.

4.7.1.9.1 Surface Soil

Based on the refined media-specific risk calculation and risk evaluation presented in Section 4.7.1.1, antimony, cadmium, copper, lead, mercury, tin, zinc, 4,4'-DDD, 4,4'-DDE, and 4,4'-DDT were identified as potential risk drivers for SWMU 1 surface soil. Mean HQs for antimony, cadmium, copper, lead, mercury, zinc, and the three organochlorine pesticides exceeded surface soil screening values (see Table 4-45). Furthermore, as evidenced by the maximum HQ values presented in Table 4-20, the magnitude and/or frequency of detections above surface soil screening values was high for several of these chemicals, including cadmium (HQ = 21.0), copper (HQ = 20.4), lead (HQ = 19.3), 4,4'-DDD (HQ = 32.5), 4,4'-DDE (HQ = 70.0), and 4,4'-DDT (HQ = 108). The descriptive statistics presented in Table 4-44 also show that maximum, mean, and 95 percent UCL concentrations in the SWMU 1 data set exceeded maximum, mean, and 95 percent UCL background concentrations. Additional evaluation of antimony, cadmium, copper, lead, mercury, tin, zinc, 4,4'-DDD, 4,4'-DDE, and 4,4'-DDT is recommended in Step 3b of the baseline ERA.

4.7.1.9.2 Subsurface Soil

Based on the refined media-specific risk calculation and risk evaluation presented in Section 4.7.1.2, 4,4'-DDT and 4,4'-DDE were identified as potential ecological risk drivers for SWMU 1 subsurface soil. The mean 4,4'-DDT concentration exceeded the surface soil screening value (mean HQ = 1.46). Although detected in only two surface soil samples at a concentration greater than the screening value, the magnitude of the maximum detection (3,500 CD ug/kg in 05SS130)

was high (maximum HQ = 8.75). The mean 4,4'-DDE concentration was less than the surface soil screening value (HQ = 0.28); however, concentrations greater than the screening value were co-located at sample locations containing maximum 4,4'-DDT concentrations. Future biodegradation of 4,4'-DDT may result in an increase in the concentration of 4.4'-DDE at these locations. Additional evaluation is recommended for 4,4'-DDT and 4,4'-DDE in Step 3b of the baseline ERA.

4.7.1.9.3 Estuarine Wetland Surface Water

Based on the refined risk calculation and risk evaluation presented in Section 4.7.1.3, no potential ecological risk drivers were identified for estuarine wetland surface water downgradient from SWMU 1, and no further evaluation of estuarine wetland surface water is recommended.

4.7.1.9.4 Estuarine Wetland Sediment

Based on the refined risk calculation and risk evaluation presented in Section 4.7.1.4, no potential ecological risk drivers were identified for estuarine wetland sediment downgradient from SWMU 1, and no further evaluation of estuarine wetland sediment is recommended.

4.7.1.9.5 Ensenada Honda Surface Water

Based on the refined risk calculation and risk evaluation presented in Section 4.7.1.5, no potential ecological risk drivers were identified for Ensenada Honda surface water downgradient from SWMU 1, and no further evaluation of Ensenada Honda surface water is recommended.

4.7.1.9.6 Ensenada Honda Sediment

Based on the refined risk calculation and risk evaluation presented in Section 4.7.1.6, no potential ecological risk drivers were identified for Ensenada Honda sediment downgradient from SWMU 1, and no further evaluation of Ensenada Honda sediment is recommended.

4.7.1.9.7 Terrestrial and Aquatic Food Web Exposures

Based on the refined risk calculation and risk evaluation presented in Section 4.7.1.7.1, cadmium, lead, mercury, zinc, 4,4'-DDD, 4,4'-DDE, and 4,4'-DDT were identified as potential risk drivers for avian omnivore and/or avian herbivore terrestrial food web exposures to chemicals in surface soil. The magnitude of maximum exposure doses above NOAEL-based screening values for the American robin (avian omnivore) was high for several of these chemicals (see Table 4-26). Furthermore, as discussed in Section 4.7.1.9.1, descriptive statistics for the SWMU 1 and background surface soil data sets indicate that maximum, mean, and 95 percent UCL concentrations in the SWMU 1 data set were elevated above background concentrations. Finally, an evaluation of the spatial distribution of these seven chemicals indicated that the frequency and/or magnitude of detections above ecologically important concentrations (i.e., concentrations that would result in a mean American robin exposure dose greater than the NOAEL-based screening value) were high. Additional evaluation of cadmium, lead, mercury, zinc, 4,4'-DDD, 4,4'-DDE, and 4,4'-DDT is recommended in Step 3b of the baseline ERA.

There were no potential ecological risk drivers identified for subsurface soil food web exposures, nor were there any potential ecological risk drivers identified for estuarine wetland aquatic food web exposures. As such, no further evaluation is recommended for the receptor-pathway combinations evaluated by this ERA. Arsenic, cadmium, copper, mercury, selenium, and zinc were identified as potential ecological risk drivers for Ensenada Honda (West Indian manatee)

aquatic food web exposures. This recommendation is based on a number of factors, including detections in upgradient surface soil, maximum HQ values (see Tables 4-29 and 4-38a), the Federal status of the West Indian manatee in Puerto Rico, and/or the bioaccumulative potential of these six metals.

As discussed in Section 4.3.1.3, potentially complete exposure pathways have been identified for aquatic reptiles (i.e., sea turtles). However, based on the paucity of data concerning the toxicological effects of chemicals for reptiles, a quantitative evaluation could not be performed. Given the Federal status of sea turtles in Puerto Rico, additional evaluation is recommended in Step 3b of the baseline ERA. This evaluation will include an examination of their life history information to determine their potential for exposure to chemicals detected in Ensenada Honda sediment. Any toxicological data identified from the literature for aquatic reptiles also will be presented and discussed in Step 3b of the baseline ERA.

4.7.2 SWMU 2 Refined Risk Calculation and Risk Evaluation

Refined media-specific screening evaluations for SWMU 2 abiotic media and refined food web exposure evaluations for SWMU 2 terrestrial and aquatic upper trophic level receptors are presented and discussed in the sections that follow. As discussed in Section 4.6.1, detected chemicals with maximum concentrations or maximum exposure doses greater than screening values, as well as detected chemicals lacking surface soil screening values were identified as ecological COPCs in Step 2 of the screening-level ERA. Non-detected chemicals with maximum reporting limits or maximum exposure doses (based on maximum reporting limits) greater than screening values, as well as non-detected chemicals lacking screening values also were identified as ecological COPCs in the Step 2 screening-level risk calculations. Only those detected and non-detected chemicals with maximum concentrations and/or maximum exposure doses greater than screening values, and detected chemicals lacking screening values were addressed by the refined screening-level risk calculations. Although eliminated from further discussion in Step 3a, non-detected chemicals lacking screening values remain ecological COPCs, but are not considered potential ecological risk drivers.

4.7.2.1 Refined Risk Calculation and Risk Evaluation for Surface Soil

Table 4-30 presented the results of the screening-level risk calculation for SWMU 2 surface soil. Twelve metals (antimony, arsenic, barium, chromium, cobalt, copper, lead, mercury, nickel, selenium, vanadium, and zinc) were identified as ecological COPCs in Step 2 of the screening-level ERA because maximum detected concentrations exceeded surface soil screening values. The VOC acetone also was detected and identified as an ecological COPC due to the lack of an appropriate screening value. One non-detected VOC (vinyl chloride), six non-detected SVOCs (1,2,4,5-tetrachlorobenzene, 1,3,5-trinitrobenzene, 2,4-dimethylphenol, m,p-cresol, o-cresol, and pentachlorophenol), four non-detected PAHs (2-methylnaphthalene, acenaphthylene, anthracene, and naphthalene), and four non-detected organochlorine pesticides (alpha-chlordane, gamma-chlordane, methoxychlor, and toxaphene) were identified as ecological COPCs because maximum reporting limits exceeded surface soil screening values. Figure 4-15 presented the concentration distribution across the site for each detected chemical identified as an ecological COPC with a maximum HQ greater than or equal to 1.0. Table 4-64 presents the results of the refined screening-level risk calculation for those detected and non-detected chemicals identified as ecological COPCs for surface soil in Step 2 of the ERA with a maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation for surface soil is presented below.

One non-detected VOC (vinyl chloride), six non-detected SVOCs (1,2,4,5-tetrachlorobenzene, 1,3,5-trinitrobenzene, 2,4-dimethylphenol, m,p-cresol, o-cresol, and pentachlorophenol), four

non-detected PAHs (2-methylnaphthalene, acenaphthylene, anthracene, and naphthalene), and four non-detected organochlorine pesticides (alpha-chlordane, gamma-chlordane, methoxychlor, and toxaphene) were identified as ecological COPCs because maximum reporting limits exceeded surface soil screening values. Mean vinyl chloride, pentachlorophenol, 2-methylnaphthalene, acenaphthylene, anthracene, alpha-chlordane, gamma-chlordane, and methoxychlor were less than surface soil screening values, while mean concentrations for 1,2,4,5-tetrachlorobenzene, 1,3,5-trichlorobenzene, 2,4-dimethylphenol, m,p-cresol, o-cresol, and toxaphene were greater than screening values (see Table 4-64). Because non-detected chemicals with mean HQs less than surface soil screening values have little potential to impact terrestrial invertebrate and plant populations, they are not identified as potential ecological risk drivers for SWMU 2 surface soil, and no additional evaluation is recommended. Although mean concentrations for 1,2,4,5-tetrachlorobenzene, 1,3,5-trichlorobenzene, 2,4-dimethylphenol, m,p-cresol, o-cresol, and toxaphene exceed surface soil screening values, no additional evaluation also is recommended for these chemicals. Surface soil screening values used for the SVOCs with mean HQs greater than 1.0 were background-based Canadian soil quality guidelines (CCME, 2002) protective of agricultural land uses. The surface soil screening values used for toxaphene also is a background-based screening value (Friday, 1998). Because these screening values are not based on toxicological data, there is uncertainty in their use as chemical exposure levels (background-based screening values do not represent threshold effect concentrations), likely resulting in an overestimation of risk estimates. Furthermore, because 1,2,4,5-tetrachlorobenzene, 1,3,5-trinitrobenzene, 2,4-dimethylphenol, m,p-cresol, o-cresol, and toxaphene were all non-detect, they are likely not present at ecologically critical concentrations.

Acetone was detected in SWMU 2 surface soil and identified as an ecological COPC based on the lack of a surface soil screening value. This VOC was detected in only one of twelve surface soil samples at a concentration of 370 J ug/kg. The detected concentration is an order of magnitude lower than the 2,500 ug/kg ecological screening-level developed for RCRA Appendix IX hazardous constituents by USEPA Region V (USEPA, 2003c). Given this indication of an acceptable potential for risk, the minimal spatial coverage of detections, and the fact that USEPA identifies acetone as a common laboratory contaminant (USEPA, 1989), it is highly unlikely that acetone is either site-related or is present in SWMU 2 surface soil at ecologically relevant concentrations. As such, this VOC is not identified as a potential ecological risk driver for SWMU 2 surface soil, and no further investigation is recommended.

Antimony, arsenic, barium, chromium, cobalt, copper, lead, mercury, nickel, selenium, vanadium, and zinc were detected in SWMU 2 surface soil and identified as ecological COPCs in Step 2 of the ERA because maximum detected concentrations exceeded surface soil screening values. Mean concentrations of antimony, arsenic, barium, nickel, and selenium were less than screening values while mean concentrations of chromium, cobalt, copper, lead, mercury, vanadium, and zinc were greater than screening values (see Table 4-64). To evaluate the potential significance of risks presented by these twelve metals, the site surface soil data were compared statistically to available background surface soil concentrations in accordance with Navy guidance (NFESC, 2002). Table 4-65 provides a summary and results of the descriptive and distributional statistics comparing the SWMU 2 and background surface soil data sets. Figure 4-29 presents box plot diagrams illustrating the distribution of each data set in relation to the screening values used in the exposure estimate.

Table 4-65 demonstrates that the distribution of cobalt, selenium, and vanadium concentrations in SWMU 2 surface soil are statistically equivalent to the distribution of background surface soil concentrations, indicating that these three metals are not likely to be site-related and do not present a risk to terrestrial plant and invertebrates at the SWMU above background levels. The descriptive statistics presented in Table 4-65 support the distributional statistics. Maximum,

mean, and 95 percent UCL cobalt and selenium concentrations in background surface soil exceeded maximum, mean, and 95 percent UCL concentrations in SWMU 2 surface soil. Although the maximum vanadium concentration in SWMU 2 surface soil (281 J mg/kg) is slightly elevated above the maximum background concentration (270 mg/kg), mean and 95 percent UCL concentrations are less than background mean and 95 percent UCL concentrations. Based on the descriptive and distributional statistics presented in Table 4-65, cobalt, selenium, and vanadium are not considered potential risk drivers for terrestrial plants and invertebrate populations at SWMU 2, and additional evaluation is not recommended.

Due the absence of detections in the background surface soil data set, a statistical comparison of the antimony data sets could not be performed. An examination of Figure 4-15 indicates that potential hotspots may be present in the vicinity of locations 06SS145 (20.1 J mg/kg), 2SB03 (13.3 J mg/kg), and 2SS03 (24 J mg/kg). Antimony concentrations at these three locations are an order of magnitude greater than those detected at other SWMU 2 sample locations and the screening value protective of terrestrial plant communities (5 mg/kg, Efroymson et al 1997b). Detected concentrations are less than an invertebrate-based ecological SSL developed by the USEPA (78 mg/kg [USEPA, 2003d]). Although the mean antimony concentration is less than the surface soil screening value protective of terrestrial plant populations (HQ = 0.68), this metal is identified as a potential risk driver for SWMU 2 surface soil, and additional evaluation is recommended. This recommendation is based on the presence of potential hot spots at 06SS145, 2SB03, and 2SS03, where antimony concentrations were detected above the plant-based surface soil screening value.

The distributional statistics performed on the copper data sets (Wilcoxin rank sum test, quantile test, and slippage test) were contradictory. The Wilcoxin rank sum test and quantile test concluded that the distribution of copper concentrations in SWMU 2 surface soil is not elevated above background concentrations, while the slippage test concluded that copper concentrations in SWMU 2 surface soil are elevated above background levels. The descriptive statistics presented in Table 4-65 show that maximum, mean, and 95 percent UCL concentrations in the SWMU 2 data set exceeded maximum, mean, and 95 percent UCL background concentrations. An evaluation of the concentration distribution of copper across the site shows that this metal was detected in each surface soil sample at a concentration greater than the surface soil screening value (an invertebrate-based toxicological threshold of 50 mg/kg [Efroymson et al., 1997a]). Maximum copper concentrations occurred in 06SS145 (739 mg/kg), 2SB03 (374 J mg/kg), 2SB05 (919 J mg/kg), 2SD02 (399 mg/kg), 2SS03 (270 mg/kg), 2SSS05 (880 mg/kg), and 2SS09 (640 mg/kg). These seven detections also exceed a plant-based toxicological threshold available from the literature (100 mg/kg [Efroymson et al., 1997b]). All remaining detections above the surface soil screening value of 50 mg/kg were less than the maximum background concentration (250 mg/kg). Based on a mean HQ greater than 1.0 (5.08),the descriptive statistics presented in Table 4-65, and the frequency and magnitude of detections above the invertebrate-based surface soil screening value, copper is identified as a potential risk driver for SWMU 2 surface soil and additional evaluation is recommended.

The statistical evaluations performed on the arsenic, barium, chromium, lead, mercury, nickel, and zinc data sets indicate that these chemicals are present at the site at concentrations above background levels. Arsenic was detected in two of twelve surface soil samples at a concentration greater than the surface soil screening value (19.6 mg/kg in 06SS145 and 18.6 mg/kg in 2SB03), while nickel and barium were each detected in one of twelve surface soil samples at a concentration greater than the surface soil screening value (nickel was detected in 2SB03 at 34.6 mg/kg and barium was detected in 2SB05 at 1,060 mg/kg). The magnitude of the detections above the screening values was low. This is shown by a maximum arsenic HQ of 1.96 using a plant-based screening value of 10 mg/kg, a maximum barium HQ of 2.12 using an invertebrate-

based screening value of 500 mg/kg, and a maximum nickel HQ of 1.15 using a plant-based screening value of 30 mg/kg). Maximum detected arsenic, nickel, and barium concentrations did not exceed literature-based benchmarks protective of terrestrial invertebrates (arsenic: 60 mg/kg [Efroymson et al., 1997a], nickel: 200 mg/kg [Efroymson et. al., 1997a], and barium: 330 mg/kg [USEPA, 2003e]). Based on mean HQs less than 1.0 and the low magnitude of detected concentrations greater than minimum toxicological thresholds available from the literature, arsenic, barium, and nickel are not identified as potential risk drivers for SWMU 2 surface soil, and no additional evaluation is recommended.

Chromium was detected in each of the twelve surface soil samples at concentrations ranging from 15 to 59.5 mg/kg. Detected concentrations were two orders of magnitude greater than the screening value (0.4 mg/kg screening value protective of invertebrate populations (Efroymson et. al., 1997a), resulting in a calculated mean HQ of 92.23. Chromium concentrations also exceeded screening values protective of terrestrial plants (1.0 mg/kg [Efroymson et. al., 1997b]) at each location. Although the comparison to screening values indicates the potential for risk to terrestrial plants and invertebrates, it should be noted that all but the minimum measured chromium concentration in background surface soil also exceeded these screening values. Furthermore, the magnitude of chromium detections in SWMU 2 surface soil above the maximum background concentration (44.1 mg/kg) was low (55.3 J mg/kg in 2SB03 and 59.5 J mg/kg in 2SB05). Based on the low magnitude of detections above the maximum background concentration, chromium is not identified as a potential risk driver for SWMU 2 surface soil, and additional evaluation is not recommended.

Lead was detected in each surface soil sample, at concentrations ranging from 4 to 4,760 J mg/kg, with a mean HQ of 8.25 (based on a plant-based screening value of 50 mg/kg [Efroymson et. al., 1997b]). Concentrations exceeding the screening value were detected at fifteen locations, with maximum concentrations occurring at 06SS145 (4,760 J mg/kg), 2SS03 (1,400 mg/kg), 2SB03 (1,000 mg/kg), 2SB05 (512 mg/kg), and 2SD02 (390 J mg/kg). The detected lead concentrations at 06SS145, 2SB03, and 2SB05 exceeded an invertebrate-based screening value available from the literature (500 mg/kg [Efroymson et. al., 1997a]), while the detected concentration at 06SS145 (4,760 J mg/kg) exceeded an invertebrate-based ecological SSL (1,700 mg/kg [USEPA, 2003g]). Based on a mean HQ greater than 1.0 (8.25), the frequency and magnitude of detections above plant- and invertebrate-based toxicological thresholds, and the descriptive and distributional statistics presented in Table 4-65, lead is identified as a potential risk driver for SWMU 2 surface soil, and additional evaluation is recommended.

Mercury was detected in each of the twenty-four surface soil samples, at concentrations ranging from 0.04 to 19 mg/kg (mean HQ of 9.60). As evidenced by Figure 4-15, concentrations exceeding the screening value (a terrestrial invertebrate value of 0.10 mg/kg [Efroymson et. al., 1997a]) occurred at fourteen surface soil sampling locations, with a maximum concentration of 19 mg/kg at 2SS11. Detected mercury concentrations at six locations (06SS145, 2SB03, 2SB05, 2SS07, 2SS11, and 2SS12) also exceeded a toxicological threshold protective of terrestrial plants (0.3 mg/kg [Efroymson et. al., 1997b]). Based on a mean HQ greater than 1.0 (9.60), the frequency and magnitude of detections above the surface soil screening value, and the descriptive and distributional statistics presented in Table 4-65, mercury is identified as a potential ecological risk driver for SWMU 2 surface soil, and additional evaluation is recommended.

Identical to lead and mercury, zinc was detected in each surface soil samples, with detected concentrations ranging from 15.1 to 1,440 mg/kg. Concentrations exceeding the screening value (a plant-based toxicological threshold of 50 mg/kg [Efroymson et. al., 1997b]) occurred at twenty-three of twenty-four sample locations, resulting in a calculated mean HQ 8.24. Detected concentrations at twelve locations also exceeded an invertebrate-based toxicological threshold

available from the literature (200 mg/kg [Efroymson et al., 1997a]). Maximum concentrations occurred at 06SS145 (1,440 mg/kg), 2SB03 (845 mg/kg), 2SB05 (1,260 mg/kg), 2SD02 (841 mg/kg), 2SS05 (800 mg/kg), and 2SS10 (1,000 mg/kg). Based on a mean HQ greater than 1.0 (8.24), the frequency and magnitude of detections above the surface soil screening value, and the descriptive and distributional statistics presented in Table 4-65, zinc is identified as a potential risk driver for SWMU 2 surface soil and additional evaluation is recommended.

In summary, antimony, copper, lead, mercury, and zinc are identified as potential risk drivers for SWMU 2 surface soil, and additional evaluation is recommended. Although acetone, arsenic, barium, chromium, cobalt, nickel, selenium, and vanadium were detected in SWMU 2 surface soil and identified as ecological COPCs in Step 2 of the ERA, they are not recommended for further evaluation based on the discussion presented in the preceding paragraphs. Additional evaluation also is not recommended for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

4.7.2.2 Refined Risk Calculation and Risk Evaluation for Subsurface Soil

Table 4-31 presented the results of the screening-level risk calculation for SWMU 2 subsurface soil. Six PAHs (benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, fluoranthene, phenanthrene, and pyrene) and twelve metals (antimony, arsenic, barium, cadmium, chromium, cobalt, copper, lead, mercury, nickel, vanadium, and zinc) were identified as ecological COPCs in Step 2 of the screening-level ERA because maximum detected concentrations exceeded surface soil screening values. Two VOCs (acetone and carbon disulfide) were detected in SWMU 2 surface soil and identified as ecological COPCs based on the lack of surface soil screening values. One VOC (vinyl chloride), six SVOCs (2,4,5-trichlorophenol, 2,4-dichlorophenol, 2,4-dimethylphenol, 2-chlorophenol, o-cresol, and pentachlorophenol), three non-detected PAHs (2-methylnaphthalene, acenaphthylene, and naphthalene), one non-detected organochlorine pesticide (toxaphene), and one non-detected metal (silver) were identified as ecological COPCs because maximum reporting limits exceeded surface soil screening values. Figure 4-16 presented the concentration distribution across the site for each detected chemical identified as an ecological COPC with a maximum HQ greater than or equal to 1.0. Table 4-66 presents the results of the refined screening-level risk calculation for those detected and non-detected chemicals identified as ecological COPCs for subsurface soil in Step 2 of the ERA with a maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation for subsurface soil is presented below.

Twelve non-detected chemicals (one VOC, six SVOCs, 3 PAHs, one organochlorine pesticide, and one metal) were identified as ecological COPCs because maximum reporting limits exceeded surface soil screening values. Mean vinyl chloride, 2,4,5-trichlorophenol, 2-methylnaphthalene, acenaphthylene, naphthalene, and silver concentrations were less than surface soil screening values (see Table 4-66). Because non-detected chemicals with mean HQs less than surface soil screening values have little potential to impact terrestrial invertebrate and plant populations, they are not identified as potential ecological risk drivers for SWMU 2 subsurface soil, and no additional evaluation is recommended. Although mean concentrations for 2,4-dichlorophenol, 2,4-dimethylphenol, 2-chlorophenol, o-cresol, pentachlorophenol, and toxaphene exceeded surface soil screening value, they also are not identified as potential ecological risk drivers. Surface soil screening values used for 2,4-dimethylphenol and o-cresol are background-based Canadian soil quality guidelines (CCME, 2002) protective of agricultural land uses. The surface soil screening values used for toxaphene also is a background-based screening value (Friday, 1998). Because these screening values are not based on toxicological data, there is uncertainty in their use as chemical exposure levels (background-based screening values do not represent threshold effect concentrations), likely resulting in an overestimation of risk estimates.

Furthermore, because 2,4-dimethylphenol, o-cresol, and toxaphene, as well as 2,4-dichlorophenol, 2-chlorophenol and pentachlorophenol were all non-detected, they are likely not present at ecologically critical concentrations.

Acetone and carbon disulfide were detected in SWMU 2 subsurface soil and identified as ecological COPCs based on the lack of appropriate screening values. However, both were detected at maximum concentrations (acetone 560 J ug/kg, carbon disulfide 3 J ug/kg) an order of magnitude lower than screening levels developed for RCRA Appendix IX hazardous constituents by USEPA Region V (2,500 ug/kg for acetone and 94.1 ug/kg for carbon disulfide [USEPA 2003c]. Given this indication of an acceptable potential for risk, it is unlikely that either VOC is present in SWMU 2 subsurface soil at ecologically relevant concentrations. As such, acetone and carbon disulfide are not identified as risk drivers, and no further investigation is recommended.

The PAHs benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, fluoranthene, phenanthrene, and pyrene were detected in SWMU 2 subsurface soils at maximum concentrations exceeding screening values. Mean concentrations of benzo(b)fluoranthene, chrysene, and phenanthrene were less than screening values, the mean concentration of benzo(k)fluoranthene (HQ = 1.00) equaled the surface soil screening value, while mean concentrations of fluoranthene (HQ = 1.17), and pyrene (HQ = 1.01) exceeded screening values (see Table 4-66). The screening value used in the risk calculations for these six PAH compounds (1,200 ug/kg) is based on a chronic toxicity investigation of benzo(b)fluoranthene and benzo(a)pyrene to wheat and are therefore protective of terrestrial plant populations (Sims and Overcash, 1983). The low frequency of detections above the plant-based surface soil screening value (these six PAHs were each detected in one of six surface soil samples at a concentration greater than the screening value) indicate a low potential for risk to terrestrial plants populations. Van Staalen and Verweij (1991) examined the effects of the benzo(a)pyrene on the wood louse (*Porcellio scaber*). They calculated a chronic (28-day) NOAEL for growth of 25,000 ug/kg (25 mg/kg). Comparisons of maximum PAH concentrations to this surrogate value indicate acceptable risk for terrestrial invertebrate populations. PAH detections also are at least an order of magnitude lower than screening levels developed for RCRA Appendix IX hazardous constituents by USEPA Region V (USEPA 2003c). Given the low frequency and magnitude of detections greater than the plant-based screening value, as well as an indication of acceptable concentrations based on an invertebrate-based toxicological threshold and USEPA Region V surface soil screening values, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, fluoranthene, phenanthrene, and pyrene concentrations in SWMU 2 subsurface soil are not likely impacting lower trophic level plant and invertebrate populations.

A search of the literature on the toxicological effects of fluoranthene, phenanthrene, pyrene to soil flora and fauna identified several studies. Sverdrup et al. (2002a) reported the toxic effect of fluoranthene, phenanthrene, and pyrene on *Enchytraeus crypticus* (enchytraeid worm) reproduction following 21-days of exposure. Maximum detected fluoranthene, phenanthrene, and pyrene concentrations in SWMU 2 subsurface soil (3,200 ug/kg, 1,900 ug/kg, and 2,100 ug/kg, respectively) were an order of magnitude lower than the reported chronic NOEC values (38,000 ug/kg, 34,000 ug/kg, and 18,000 ug/kg for fluoranthene, phenanthrene, and pyrene, respectively). The toxic effect of fluoranthene, fluorene, phenanthrene, and pyrene on *Eisenia veneta* (earthworm) growth was reported by Sverdrup et al. (2002a). Identical to the study using *Enchytraeus crypticus*, maximum detected concentrations for these three PAHs are less than the 28-day NOEL values (98,000 ug/kg, 31,000 ug/kg, and 29,000 ug/kg, respectively for fluoranthene, phenanthrene and pyrene, respectively). One additional study investigating the toxic effect of fluoranthene, phenanthrene, and pyrene on a soil invertebrate species was identified from the literature. Sverdrup et al. (2001) reported 21-day NOEC values of 47,000 ug/kg (fluoranthene), 21,000 ug/kg (phenanthrene), and 13,000 ug/kg (pyrene) for *Folsomia*

fimetaria (springtail) reproduction. Maximum detected concentrations for these three PAHs are less than the springtail NOEC values.

In addition to the invertebrate toxicity values discussed above, plant and microorganisms toxicity values were identified from the literature. Sverdrup et al. (2003) investigated the effect of fluoranthene, phenanthrene, and pyrene on red clover, ryegrass, and mustard seedling growth. After 21-days of exposure, fluoranthene EC_{20} values ranged from 140,000 ug/kg to 650,000 ug/kg on a fresh weight basis and 150,000 ug/kg to >1,000,000 ug/kg on a dry weight basis. 21-day EC_{20} values for phenanthrene ranged from 37 mg/kg to 300 mg/kg on a fresh weight basis and 46 mg/kg to 340,000 ug/kg on a dry weight basis, while pyrene EC_{20} values ranged from 49,000 ug/kg to >1,000,000 ug/kg on a fresh weight basis and 56,000 ug/kg to >1,000,000 ug/kg on a dry weight basis. In addition to effects on seeding growth, Sverdrup et al. (2003) reported toxic effects on red clover, ryegrass, and mustard seed emergence for each PAH compound. 21-day fluoranthene, phenanthrene, and pyrene LC_{50} values were greater than 1,000,000 ug/kg for each species. Identical to the invertebrate data, maximum detected fluoranthene, fluorene, phenanthrene, and pyrene concentrations were less than EC_{20} values based on seedling growth and 21-day LC_{50} values based on seed emergence for each species. The toxic effects of fluoranthene, phenanthrene, and pyrene on soil nitrification processes, total number of protozoa, and total number of herterotrophic flagellates was reported by Sverdrup et al. (2002c). NOEC values for soil nitrification ranged from 24,000 ug/kg for fluoranthene to 79,000 ug/kg for pyrene. An EC_5 value of 2,400,000 ug/kg for total number of protozoa was reported for phenanthrene, while EC_5 values of 250,000 ug/kg for phenanthrene and 2,000,000 ug/kg for fluoranthene were reported for total number of heterotrophic flagellates. EC_5 values based on reduction in total number of protozoa could not be estimated for phenanthrene and pyrene, while an EC_5 values based on the reduction in total number of heterotrophic flagellates could not be estimated for pyrene. Maximum detected fluoranthene, phenanthrene, and pyrene concentrations were less than available soil microbial NOEC and EC_5 values.

Based on the comparison of maximum detected concentrations to literature-based plant, invertebrate and microbial toxicity values specific to these PAH compounds, fluoranthene, phenanthrene, and pyrene are not identified as potential ecological risk drivers for SWMU 2 subsurface soil, and no additional evaluation is recommended. Benzo(b)fluoranthene, benzo(k)fluoranthene, and chrysene also are not identified as potential ecological risk drivers based on mean HQ values less than or equal to 1.0 [pyrene and benzo(k)fluoranthene], the low magnitude and frequency of detections above the plant-based toxicological threshold used in the Step 2 risk calculations, and/or the absence of detections above plant, invertebrate and microbial toxicity values identified from the literature for other PAH compounds. Finally, benzo(b)fluoranthene is not identified as a potential ecological risk driver based on the low magnitude and frequency of detections above the plant-based toxicological threshold used in the Step 2 risk calculations.

Twelve metals (antimony, arsenic, barium, cadmium, chromium, cobalt, copper, lead, mercury, nickel, vanadium, and zinc) were identified as ecological COPCs in Step 2 of the screening-level ERA because maximum detected concentrations exceeded surface soil screening values. Mean concentrations of arsenic, barium, cadmium, cobalt, nickel, and silver were less than surface soil screening values (see Table 4-66), while mean concentrations of antimony (HQ = 1.47), chromium (HQ = 79.00), copper (HQ = 13.68), lead (HQ = 13.61), mercury (HQ = 1.50), vanadium (HQ = 70.79), and zinc (HQ = 13.17) exceeded screening values.

To evaluate the potential significance of risks presented by these metals, SWMU 2 subsurface soil data were compared statistically to available background concentrations in accordance with Navy guidance (NFESC 2002). Table 4-67 provides a summary and results of the descriptive and

distributional statistics, while Figure 4-30 presents box plot diagrams illustrating the distribution of each data set in relation to the screening values used in the exposure estimate. As evidenced by Table 4-67 and Figure 4-30, the distribution of cobalt concentrations in SWMU 2 surface soil are statistically equivalent to background surface soil concentrations, indicating that this metal is not likely to be site-related and does not present a risk to terrestrial wildlife at the site above background levels. The descriptive statistics also show that maximum, mean and 95 percent UCL concentrations in the background data set exceeded maximum, mean, and 95 percent UCL concentrations in the SWMU 2 data set. Based on a mean HQ less than 1.0 (0.99) and the descriptive and distributional statistics presented in Table 4-67, cobalt is not identified as a potential risk driver for SWMU 2 subsurface soil, and additional evaluation is not recommended.

The distributional statistics presented in Table 4-67 also indicate that the distributions of barium, chromium, vanadium, and nickel concentrations in SWMU 2 subsurface soil are statistically equivalent to background surface soil concentrations. However, the descriptive statistics for barium, chromium and nickel show that maximum, mean, and upper 95 percent UCL concentrations occurred in the SWMU 2 data set. Maximum and 95 percent UCL concentrations for vanadium also exceeded background maximum and 95 percent UCL concentrations. To further evaluate the significance of chromium, nickel, and vanadium in SWMU 2 subsurface soil, the frequency and magnitude of detections above surface soil and/or background concentrations was considered. As evidenced by Figure 4-16, chromium was detected in each subsurface soil sample at a concentration greater than the surface soil screening value; however, only one detected concentration (59.8 mg/kg in 06SS141) exceeded the maximum background surface soil concentrations (44.1 mg/kg). Barium and nickel were each detected in a single subsurface soil samples at a concentration greater than surface soil screening values (barium was detected in 06SS146 at 509 mg/kg and nickel was detected in 06SS141 at 49.9 mg/kg). The maximum HQ values for barium and nickel show that these concentrations were only slightly elevated above screening values (1.02 and 1.66, respectively). Vanadium was detected in all but one subsurface soil sample at a concentration greater than the screening value. However, this metal also was detected in each background surface soil sample at concentrations greater than the screening value. Only one detected vanadium concentration in SWMU 2 subsurface soil exceeded the maximum background concentration (386 mg/kg in 06SS150). The mean vanadium background concentration also exceeded the mean vanadium concentration in SWMU 2 subsurface soil. Based on mean HQs less than 1.0 (barium and nickel), the distributional statistics presented in Table 4-67, and the low frequency and magnitude of detections greater than surface soil screening values and/or maximum background concentrations, barium, chromium, nickel, and vanadium are not identified as potential risk drivers for SWMU 2 subsurface soil, and additional evaluation is not recommended.

The statistical evaluation of the cadmium data sets was limited to the right tail of the distribution (i.e., slippage test). The slippage test concluded that cadmium concentrations in SWMU 2 subsurface soil are elevated above background surface soil concentrations. Given the low power of this statistical method, a conclusion regarding the statistical equivalence of cadmium concentrations cannot be made with high confidence. To further evaluate the significance of cadmium detections in SWMU 2 subsurface soil, an examination of the concentration distributions across the site relative to surface soil screening values was performed. Cadmium was detected in four of the seven subsurface soil locations sampled at concentrations ranging from 3.9 to 8.7 mg/kg, with a calculated mean HQ of 0.94. Detected concentrations at each location were elevated over the range of background concentrations and exceeded the terrestrial plant screening value of 4.0 mg/kg (Efroymson et al 1997b) at locations 06SS141 (8.7 mg/kg), 06SS146 (5.0 mg/kg), and 06SS150 (6.3 mg/kg). However, concentrations at each location were lower than invertebrate-based toxicological thresholds available from the literature (20 mg/kg [(Efroymson et al 1997a] and 140 mg/kg [USEPA, 2003e), indicating acceptable risk to terrestrial

invertebrate populations. The maximum detected cadmium concentration also is less than a plant-based ecological SSL (32 mg/kg [USEPA, 2003e]). As these comparisons to toxicological thresholds protective of lower trophic level receptors indicate an acceptable potential for risk to invertebrates at the individual level (maximum HQ < 1.0) and to terrestrial vegetation at individual or population level (maximum HQ < 1.0 based on the ecological SSL and a mean HQ less than 1.0 based on the screening value used in the Step 3a risk calculation), it is unlikely that cadmium is impacting invertebrate and plant populations at SWMU 2. As such, cadmium is not identified as a potential risk driver for SWMU 2 subsurface soil, and additional evaluation is not recommended.

The distributional statistics performed on the arsenic data sets were contradictory. The evaluation of the median of the distributions (Wilcoxin rank sum test) concluded that concentrations are statistically equivalent, whereas the evaluation of the right tail of the distribution (slippage test) concluded that arsenic concentrations in the SWMU 2 data set are elevated above background surface soil samples. The descriptive statistics support the conclusions of the slippage test (maximum, mean, and 95 percent UCL concentrations in the SWMU 2 data set exceeded maximum, mean, and 95 percent UCL background concentrations). As evidenced by Figure 4-16, arsenic was detected in two subsurface soil samples (06SS141 and 06SS146) at concentrations (17.5 mg/kg and 18.7 mg/kg, respectively) greater than the surface soil screening value (plant-based toxicological threshold of 10 mg/kg [Efroymson et al., 1997b]). All detections were less than an invertebrate-based toxicological threshold available from the literature (60 mg/kg [Efroymson et al., 1997a]). The low magnitude of detections above the plant-based screening value is shown by the maximum HQ value, which was slightly elevated above 1.0 (1.87). Based on a mean HQ less than 1.0 (0.69) and the low magnitude of detections greater than the most conservative screening value, arsenic is not identified as a potential risk driver for SWMU 2 subsurface soil, and additional evaluation is not recommended.

The statistical evaluations performed on the lead and zinc data sets indicate that these two metals are present in SWMU 2 subsurface soil at concentrations elevated above background surface soil concentrations, while the statistical evaluations performed on the copper data sets concluded that SWMU 2 subsurface soil concentrations for this metal are statistically equivalent to background surface soil concentrations. An evaluation of the box plot diagrams presented in Figure 4-30 and the spatial distribution of concentrations across the site (see Figure 4-16) indicates potential hotspots of these three metals at locations 06SS01 and 06SS103. Detected copper concentrations at 06SS141 and 06SS146 were 5,850 mg/kg and 774 mg/kg, respectively, detected lead concentrations at 06SS141 and 06SS146 were 1,210 J mg/kg and 5,850 J mg/kg, respectively, and detected zinc concentrations at 06SS141 and 06SS146 were 3,350 mg/kg and 2,010 mg/kg, respectively. The surface soil screening value used in the risk calculations for copper was an invertebrate-based toxicological threshold (50 mg/kg {Efroymson et al., 1997a]). The detected concentrations at 06SS101 and 06SS103 also exceeded a plant-based toxicological threshold available from the literature (100 mg/kg [Efroymson et al., 1997b]). The surface soil screening values used in the risk calculations for lead and zinc were plant-based toxicological thresholds (50 mg/kg [Efroymson et al., 1997b]). Detected lead and zinc concentrations at these two subsurface soil sample locations also exceeded invertebrate-based toxicological thresholds available from the literature (500 mg/kg for lead and 200 mg/kg for zinc [Efroymson et al., 1997a]). The detected lead concentration at 06SS146 also exceeded an invertebrate-based ecological SSL (1,700 [USEPA, 2003g]) Based on mean HQs greater than 1.0 (20.35 for copper, 20.81 for lead, and 16.87 for zinc), the descriptive statistics presented in Table 4-67, and the apparent co-location of elevated concentrations (see Figure 4-16), copper, lead, and zinc are identified as potential risk drivers for subsurface soil, and additional evaluation is recommended.

Statistical evaluations performed on the SWMU 2 subsurface soil and background surface soil data sets were contradictory. The evaluation of the mean of the distributions (i.e., Gehan test) concluded that mercury concentrations in SWMU 2 subsurface soil are elevated above background levels, while the evaluation performed on the right tail of the distribution concluded that the SWMU 2 mercury concentrations are statistically equivalent to background. The statistical evaluation of the antimony data could not be performed due to the lack of detections in the background antimony data set. Mercury was detected in four subsurface soil sample (0.68 mg/kg in 06SS146, 0.16 J mg/kg in 2SB03, 0.14 J mg/kg in 2SB05, and 0.26 J mg/kg in 2SB07). Detected concentrations at 06SS146, 2SB03, and 2SB07 exceeded the surface soil screening value (an invertebrate-based toxicological threshold of 0.1 mg/kg [Efroymson et al., 1997a]). This detected concentration at 06SS146 also exceeded a plant-based toxicological threshold available from the literature (0.3 mg/kg [Efroymson et al., 1997b]). Antimony was detected in four of seven surface soil samples. The detected concentration of antimony at 06SS141 (17.9 J mg/kg) and 06SS146 (19.8 J mg/kg) exceeded the surface soil screening value (a plant-based toxicological threshold of 10 mg/kg [Efroymson et al., 1997b]). Given that detected concentrations are co-located with elevated copper, lead, and zinc concentrations and mean HQs exceeded 1.0 (1.47 for antimony and 1.56 mg/kg for mercury), these two metals also are identified as potential ecological risk drivers for SWMU 2 subsurface soil, and additional evaluation is recommended.

In summary, antimony, copper, lead, mercury, and zinc are identified as potential ecological risk drivers for SWMU 2 subsurface soil, and additional evaluation is recommended. As discussed above, elevated concentrations of these five metals are co-located at subsurface soil sample locations 06SS141 and/or 06SS146. Although acetone, carbon disulfide, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, fluoranthene, phenanthrene, pyrene, arsenic, barium, cadmium, chromium, cobalt, nickel, and vanadium were detected in subsurface soil and identified as ecological COPCs in Step 2 of the ERA, they are not recommended for further evaluation based on the discussion presented in the preceding paragraphs. Additional evaluation also is not recommended for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

4.7.2.3 Refined Risk Calculation for Estuarine Wetland Sediment

Table 4-32 presented the results of the screening-level risk calculation for estuarine wetland sediment downgradient from SWMU 2. One VOC (acetone), three SVOCs (5-nitro-o-toluidine, di-n-butylphthalate, and pentachlorophenol), and eight metal (arsenic, cobalt, copper, lead, mercury, nickel, tin, and vanadium) were detected and identified as ecological COPCs in Step 2 of the screening-level ERA because maximum detected concentrations exceeded sediment screening values. Methyl methanesulfonate, N-nitrosomorpholine, and p-(dimethylamino)azobenzene, beryllium, and thallium were also identified as ecological COPCs due to the lack of sediment screening values. Ten non-detected VOCs, fifty-six non-detected SVOCs, five non-detected PAHs, twelve non-detected organochlorine pesticides, seven non-detected PCBs, eight non-detected organophosphorous pesticides, and the non-detected dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration) were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values. Figure 4-17 presented the concentration distribution across the site for each detected chemical identified as an ecological COPC with a maximum HQ greater than 1.0. Table 4-68 presents the results of the refined screening-level risk calculation for those detected and non-detected chemicals identified as ecological COPCs for sediment in Step 2 of the ERA with an HQ greater than 1.0. An evaluation of the refined screening-level risk calculation for sediment is presented below.

As discussed above, ten non-detected VOCs, fifty-six non-detected SVOCs, five non-detected PAHs, twelve non-detected organochlorine pesticides, seven non-detected PCBs, eight non-detected organophosphorous pesticides, and the non-detected dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration) were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values. Mean concentrations for six VOCs (2-hexanone, 4-methyl-2-pentanone, cis-1,3-dichloroprepene, ethylbenzene, propionitrile, and trans-1,3-dichloropropene), five SVOCs [1,3-dichlorobenzene, 1,4-dioxane, acetophenone, and bis(2-chloroethoxy)methane], one PAH (2-methylnaphthalene), and two organochlorine pesticides (heptachlor epoxide and methoxychlor) were less than sediment screening values, while mean concentrations for the remaining non-detected chemicals exceeded sediment screening values. Of the ninety-nine non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA (non-detected chemicals with maximum HQ values greater than sediment screening values), only benzo(a)pyrene, dibenz(a,h)anthracene, 4,4'-DDD, 4,4'-DDT, dieldrin, endosulfan I, endosulfan II, endrin, endrin aldehyde, gamma-BHC, heptachlor, heptachlor epoxide, and dioxins/furans were detected in SWMU 2 surface or subsurface soil. Based on the absence of detections in SWMU 2 surface and subsurface soil, there is no indication that the remaining eighty-six non-detected chemicals are associated with historical site activities (no evidence of a source or release to SWMU 2 surface and subsurface soil). For this reason, they are not identified as potential ecological risk drivers for estuarine wetland sediment, and no additional evaluation is recommended.

Although detected in upgradient surface or subsurface soil, benzo(a)pyrene, dibenz(a,h)anthracene, 4,4'-DDD, 4,4'-DDT, dieldrin, endosulfan I, endosulfan II, endrin, endrin aldehyde, gamma-BHC, heptachlor, heptachlor epoxide, and dioxins/furans also are not identified as potential ecological risk drivers. The low frequency and magnitude of detections in upgradient surface soil and subsurface soil for many of these chemicals precludes their migration to estuarine wetland sediment at ecologically important concentrations. For example, endrin, endrin aldehyde, gamma-BHC, and heptachlor were each detected in one of twelve SWMU 2 surface soil samples at 0.15 NJ ug/kg, 0.71 J ug/kg, 1.2 NJ ug/kg, and 0.17 J ug/kg, respectively. Furthermore, for nonionic organic chemicals, the primary factor affecting bioavailability in sediment is TOC (USEPA, 1993a). Because bulk-sediment screening values used for benzo(a)pyrene, dibenz(a,h)anthracene, 4,4'-DDD, 4,4'-DDT, dieldrin, endrin, gamma-BHC, heptachlor, and dioxins/furans do not account for the effect that TOC has on their bioavailability, HQs calculated based on reporting limits and bulk-sediment screening values are likely to be overestimates of potential risk. In addition, the EqP-based screening values used for endosulfan I, endosulfan II, endosulfan sulfate, endrin aldehyde, and heptachlor epoxide are based on a default TOC content (one percent [10,000 mg/kg]). Given that the minimum TOC concentration measured in estuarine wetland sediment downgradient from SWMU 2 was 26,000 ug/kg, HQs calculated based on a default TOC concentration also are likely to be overestimates of potential risk.

As evidenced by Table 4-68, the mean concentration of acetone exceeded the sediment screening value (HQ = 5.27). Acetone was detected in seven of nine sediment samples. Six of the seven detected concentrations were greater than the sediment screening value. The sediment screening value used in the risk calculations was an EqP-based value derived using the USEPA EqP approach (USEPA, 1993a). As discussed in Section 4.7.1.6, EqP values derived for VOCs using this approach are overly conservative (Fuchsman, 2003). To further evaluate the significance of acetone detections in estuarine wetland sediment, detected concentrations were compared to an alternative screening value identified from the literature. Di Toro and McGrath (2000) reported a SQG of 2,265 ug/kg for acetone based on a target lipid model and one percent organic carbon. Given that the maximum acetone detection (110 J ug/kg) is an order of magnitude below this value, it is unlikely that this common laboratory contaminant is present at ecologically significant

concentrations. Furthermore, given that the minimum TOC concentration measured in estuarine wetland sediment downgradient from SWMU 2 was 26,000 mg/kg (i.e., 2.6 percent), the Di Toro and McGrath (2000) target lipid model would predict even lower potential for bioavailability when site-specific TOC is considered (i.e., SQG of 5,889 ug/kg based on 2.6 percent TOC). Based on the comparison of the maximum acetone concentration to a SQG value developed by Di Toro and McGrath (2000), acetone is not considered a potential risk driver for estuarine wetland sediment, and additional evaluation is not recommended.

The SVOCs methyl methanesufonate, n-nitrosomorpholine, and p-(dimethylamino)azobenzene were detected and identified as ecological COPCs based on the lack of sediment screening values (EqP-based screening values could not be calculated due to the lack of surface water screening values). These chemicals were each detected in one of nine sediment samples (02EWSSD09), but were not detected in surface soil or subsurface soil samples collected at SWMU 2, suggesting that there presence is not related to a release from potential source areas at the SWMU. Detected concentrations of each chemical (54 J ug/kg for methyl methanesufonate, 240 J ug/kg for n-nitrosomorpholine, and 86 J ug/kg for p-[dimethylamino]azobenzene) were less than method reporting limits, adding uncertainty to the presence of these chemicals in sediment. Detected concentrations also were within the range of available screening values for other SVOCs (1.0 ug/kg for 1,4-phenylenediamine to 965,742 ug/kg for 7,12-dimethylbenz[a]anthracene [see Table 4-9)]. Due to the lack of detections in upgradient site media, and the relatively low level of detected concentrations, these chemicals are not identified as potential risk drivers for estuarine wetland sediment downgradient from SWMU 2 and no further evaluation is recommended.

Beryllium and thallium were identified as ecological COPCs in sediment based on the lack of sediment screening values. Beryllium was detected in each sediment sample, while thallium was detected in three of nine sediment samples. To evaluate the significance of potential risks presented by beryllium and thallium relative to background concentrations, the estuarine wetland sediment data were statistically compared to background sediment data in accordance with Navy guidance (NFESC, 2002). Table 4-69 provides a summary and results of the descriptive and distributional statistics comparing SWMU 2 estuarine wetland and background sediment concentrations. Figure 4-31 presents box plot diagrams illustrating the distribution of each data set. As evidenced by Table 4-69 and Figure 4-31, the distribution of beryllium concentrations in estuarine wetland sediment downgradient from SWMU 2 is statistically equivalent to background levels. The descriptive statistics presented in Table 4-69 support the conclusions of the distributional statistics. Although the maximum beryllium concentration for the estuarine wetland sediment data set was greater than the maximum background concentration, mean and 95 percent UCL concentrations were less than background mean and 95 percent UCL concentrations. Based on the descriptive and distributional statistics presented in Table 4-69, beryllium is not identified as a potential risk driver for estuarine wetland sediment downgradient from SWMU 2, and additional evaluation is not recommended.

Thallium was not detected in the background data set. As such, a statistical comparison of site and background concentrations could not be conducted. Furthermore, toxicological thresholds or toxicity data are not available from the literature for this metal. Therefore, a direct evaluation of potential impacts to aquatic life cannot be performed. The maximum detected concentration in estuarine wetland sediment (0.11 J mg/kg) is lower than conservative sediment screening values for other metals. While these screening values are not indicative of thallium toxicity, they provide an indication of the potential for this metal to present risks to benthic invertebrates. A comparison of the thallium surface water screening value to surface water screening values for other metals also indicates that thallium is less toxic to aquatic organisms than many of the Appendix IX metals, including cadmium, copper, lead, mercury, nickel, and silver. Based on the comparison of the maximum detected concentration to available sediment screening values for

other metals known to be more toxic to aquatic organisms (based on surface water screening values), as well as the low magnitude of detections, thallium is not considered a potential risk driver for estuarine wetland sediment downgradient from SWMU 2, and additional evaluation is not recommended.

The SVOCs 5-nitro-o-toluidine, di-n-butylphthalate, and pentachlorophenol were detected and identified as ecological COPCs in Step 2 of the screening-level ERA because maximum detected concentrations exceeded sediment screening values. As evidenced by Table 4-68, mean HQs for 5-nitro-o-toluidine, di-n-butylphthalate, and pentachlorophenol exceeded 1.0 (2.44, 6.22, and 14.71, respectively). 5-Nitro-o-toluidine and pentachlorophenol were each detected in one of eight and one of nine sediment samples, respectively (5-nitro-o-toluidine was detected in 02EWSSD09 at 330 J ug/kg and pentachlorophenol was detected in 02EWSSD05 at 250 J ug/kg), while di-n-butylphthalate was detected in two of nine samples (450 J ug/kg in 02EWSSD02 and 340 J ug/kg in 02EWSSD05). The screening value used for 5-nitro-o-toluidine in risk calculations was an EqP-based sediment screening value calculated in accordance with USEPA procedures using a default TOC of 1.0 percent (USEPA 1993a). Using the TOC concentration for sediment sample 02EWSSDO9, which contained the single detection of 5-nitro-o-toluidine, a site-specific EqP-based screening value can be derived using the procedures presented in Appendix I. Based on a sample-specific TOC concentration of 27,000 ug/kg (2.7 percent), the site-specific screening value for 5-nitro-o-toluidine is 354 mg/kg. Given that the detected concentration of 5-nitro-o-toluidine in 02EWSSD09 (330 J ug/kg) was less than this site-specific EqP-based value, it is unlikely that this SVOC is present at an ecologically relevant concentration.

An identical evaluation for pentachlorophenol and di-n-butylphthalate also indicates that detected concentrations of these two SVOCs are not ecologically relevant. The sediment screening values used in risk calculations were bulk-sediment toxicological thresholds (AET values). These bulk-sediment values do not take into consideration site-specific factors (i.e., TOC) that can influence bioavailability. Based on a sample-specific TOC concentration of 33,000 mg/kg (3.3 percent) in sediment sample 02EWSSD05 (sediment sample containing the single detection of this SVOC), the site-specific EqP-based screening value for pentachlorophenol is 26,296 ug/kg. The single detected concentration of pentachlorophenol (250 J ug/kg) is two orders of magnitude below this site-specific value. For di-n-butylphthalate, a site-specific TOC was calculated using the minimum TOC concentration for the sample containing the maximum detection (2.7 percent in 02EWSDD05). The resultant value, 3,124 ug/kg, is an order of magnitude higher than the maximum detected concentration. Based on the comparison of maximum detected concentrations to site-specific, EqP-based screening values, as well as the absence of detections in upgradient surface soil and subsurface soil, 5-nitro-o-toluidine, pentachlorophenol, and di-n-butylphthalate are not identified as potential risk drivers for estuarine wetland sediment downgradient from SWMU 2, and additional evaluation is not recommended.

Eight metals (arsenic, cobalt, copper, lead, mercury, nickel, tin, and vanadium) were detected and identified as ecological COPCs in Step 2 of the screening-level ERA because maximum detected concentrations exceeded sediment screening values. Each metal was detected in all nine estuarine wetland sediment samples. Mean concentrations of arsenic, cobalt, lead, nickel, and vanadium were less than screening values, while mean concentrations of copper, mercury, and tin exceeded screening values (see Table 4-68). Sediment concentrations of each metal were statistically compared to background sediment concentrations to evaluate the potential significance of risks in accordance with Navy guidance (NFESC, 2002).

As evidenced by Table 4-69 and Figure 4-31, the distributions of cobalt and vanadium are statistically equivalent to background sediment concentrations. This indicates that these metals

are not site-related and do not present a risk to aquatic receptors above background levels. The descriptive statistics presented in Table 4-69 support the conclusions of the distributional statistics. For both metals, maximum, mean, and 95 percent UCL concentrations in the background data set exceeded maximum, mean, and 95 percent UCL concentrations in the estuarine wetland sediment data set. Based on the descriptive and distributional statistics presented in Table 4-69, cobalt and vanadium are not identified as potential risk drivers and additional evaluation is not recommended.

The statistical evaluations presented in Table 4-69 also indicate that the distributions of arsenic, mercury, and nickel concentrations are statistically equivalent to background levels. The descriptive statistics show that maximum, mean, and 95 percent UCL concentrations in the estuarine wetland data set were slightly elevated above background concentrations. An examination of the concentration distribution of detected concentrations across the estuarine wetland downgradient from SWMU 2 shows that nickel was detected in a single sediment sample while arsenic was detected in two sediment samples at concentrations greater than sediment screening values ([nickel was detected in 02EWSSD01 at 17 mg/kg, while arsenic was detected in 02EWSSD01 at 14 mg/kg and 02EWSSD09 at 7.4 mg/kg) [see Figure 4-17]). These detections were only slightly elevated above their respective screening values (7.2 mg/kg for arsenic and 15.9 mg/kg for nickel [MacDonald, 1994]). Based on mean HQs less than 1.0 (HQ = 0.67 mg/kg, 0.54, and 0.50 for arsenic, lead, and nickel, respectively), the distributional statistics presented in Table 4-69, and the low frequency and magnitude of detected concentrations greater than sediment screening values, arsenic and nickel are not identified as potential risk drivers for estuarine wetland sediment downgradient from SWMU 2, and additional evaluation is not recommended.

Tin was detected in four of nine estuarine wetland sediment samples at a concentration greater than the sediment screening value (02EWSSD01, 02EWSSD02, 02EWSSD03, and 02EWSSD06 at 6 J mg/kg, 3.5 J mg/kg, 4.2 J mg/kg, and 4.5 J mg/kg, respectively). Detected concentrations were only slightly elevated above the sediment screening value (3.4 mg/kg [Buchman 1999]). The descriptive statistics presented in Table 4-69 also show that maximum, mean, and 95 percent UCL concentrations for the SWMU 2 data set were only slightly elevated or below background concentrations. Finally, the maximum detected concentration (14.64 J mg/kg [expressed as TBT]) is two orders of magnitude below an alternative screening value identified from the literature (an IC_{50} of 600 mg/kg based on *Hexagonia sp.* growth reported by Kristin et al., 1998 [see Section 4.7.1.4]). Based on a mean HQ less than 1.0 (for mercury), a comparison of the maximum detected concentration to an alternative screening value for tin, as well as the descriptive and distributional statistics presented in Table 4-69 and the low magnitude of detected concentrations above screening values used in risk calculations, mercury and tin are not identified as potential risk drivers for estuarine wetland sediment downgradient from SWMU 2, and additional evaluation is not recommended.

The statistical evaluations presented in Table 4-69 for copper and lead were contradictory. The evaluation of the mean of the distributions for both metals concluded statistical equivalence between the estuarine wetland and background data sets, while the evaluations performed on the right-tail of the distributions (slippage test and quantile test) concluded that estuarine wetland concentrations are elevated above background concentrations. The descriptive statistics show that maximum, mean, and upper 95 percent UCL concentrations in the SWMU 2 estuarine wetland data sets exceeded maximum, mean, and 95 percent UCL background sediment concentrations for both metals. A spatial examination of the copper and lead data also show that copper was detected in sixteen of nineteen sediment samples and lead was detected in five of seventeen sediment samples at concentrations greater than sediment screening values.

To further evaluate the significance of copper and lead detections in estuarine wetland sediment downgradient from SWMU 2, total SEM molar concentrations at each location sampled during the October 2004 additional data collection field effort were compared to AVS molar concentrations (see Table 4-69a). With the exception of sediment sample 2EWS12, the molar concentration of AVS exceeded the molar concentration of SEM in nine of ten sediment samples (i.e., SEM-to-AVS rations were less than 1.0). The SEM-to-AVS ratio at 2EWS11 exceeded 1.0 (SEM-to-AVS ratio of 126). The high SEM-to-AVS ratio at this sample location can be attributed to the lack of detectable AVS (AVS not detected at a reporting limit of 0.1 umole/gram). As discussed in the preceding paragraph, the 2EWSD12 sediment sample also contained the maximum detected copper and lead concentration (280 J mg/kg and 170 mg/kg, respectively). The absence of any detectable AVS at 2EWSD12 indicates that benthic macroinvertebrates at this sample location may be exposed to toxic concentrations of SEM metals in sediment pore water. An examination of the analytical data for the individual SEM metals (see Appendix G.16) show that copper, lead, and zinc are responsible for the SEM-to-AVS ratio at 2EWSD12. The molar concentration of each metal (2.0458 umole/gram, 0.6757 umole/gram, and 3.5174 umole/gram, respectively) exceeded the AVS reporting limit (0.1 umole/gram). Based on the magnitude and frequency of detected concentrations above bulk-sediment screening values, the descriptive statistics presented in Table 4-69, and the SEM-to-AVS ratio for sediment sample 2EWSD12, copper and lead are identified as potential ecological risk drivers for estuarine wetland sediment downgradient from SWMU 2, and additional evaluation is recommended. Although the Step 2 screening-level risk calculation indicated that bulk-sediment zinc concentrations in estuarine wetland sediment are less than the bulk-sediment screening value (see Table 4-32), zinc also is identified as a potential ecological risk driver for estuarine wetland sediment. This decision is based on the molar concentration of this metal at 2EWSD12. The SEM data for zinc also suggest that the nature and extent of zinc concentrations in estuarine wetland sediment may not have been adequately characterized by the 2003 additional data collection field investigation.

In summary, copper, lead, and zinc are identified as a potential risk drivers for the estuarine wetland benthic macroinvertebrate community downgradient from SWMU 2, and additional evaluation is recommended. Although acetone, 5-nitro-o-toluidine, di-n-butylphthalate, pentachlorophenol, methyl methanesufonate, p-(dimethylamino)azobenzene, n-nitrosomorpholine, arsenic, beryllium, cobalt, mercury, nickel, thallium, tin, and vanadium were identified as ecological COPCs in Step 2 of the ERA, they are not recommended for further evaluation based on the discussion presented in the preceding paragraphs. Additional evaluation also is not recommended for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

4.7.2.4 Refined Risk Calculation and Risk Evaluation for Ensenada Honda Surface Water

Table 4-33 presented the results of the Step 2 screening-level risk calculation for Ensenada Honda surface water collected downgradient from SWMU 2. Copper was detected and identified as an ecological COPC in Step 2 of the screening-level ERA because the maximum detected concentration exceeded the surface water screening value. Tin also was detected and identified as an ecological COPC based on the lack of a surface water screening value. Twenty-one non-detected SVOCs, one non-detected PAH [benzo(a)pyrene], seven non-detected PCBs, twelve non-detected organochlorine pesticides, eight non-detected organophosphorous pesticides, dioxins/furans (based of the 2,3,7,8-TCDD equivalent concentration), and one non-detected inorganic (cyanide) were identified as ecological COPCs based on the lack of surface water screening values. Figure 4-18 presented the concentration distribution across the site for each detected chemical identified as an ecological COPC with a maximum HQ greater than 1.0. Table 4-70 presents the results of the refined screening-level risk calculation for those detected and non-

detected chemicals identified as ecological COPCs for surface water in Step 2 of the ERA with a maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation for surface water in the open water habitat is presented below.

As discussed above, twenty-one non-detected SVOCs, seven non-detected PCBs, twelve non-detected organochlorine pesticides, eight non-detected organophosphorous pesticides, dioxins/furans (based of the 2,3,7,8-TCDD equivalent concentration), and one non-detected inorganic (cyanide) were identified as ecological COPCs based on the lack of surface water screening values. Mean concentrations for seven of the non-detected SVOCs and the non-detected PAH were less than surface water screening values, while mean concentrations for the remaining non-detected chemicals exceeded surface water screening values (see Table 4-70). Of the fifty-one non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA (non-detected chemicals with maximum HQ values greater than sediment screening values), only di-n-butylphthalate, benzo(a)pyrene, and dioxins/furans were detected in SWMU 2 surface and/or subsurface soil. Based on the absence of detections in SWMU 2 surface and subsurface soil, there is no indication that the remaining forty-eight non-detected chemicals are associated with historical site activities (no evidence of a source or release to SWMU 2 surface and subsurface soil). For this reason, they are not identified as potential ecological risk drivers for estuarine wetland sediment, and no additional evaluation is recommended. Although detected in SWMU 2 surface and/or subsurface soil, additional evaluation also is not recommended for di-n-butylphthalate, benzo(a)pyrene, and dioxins/furans. Log K_{ow} and Log K_{oc} values for di-n-butylphthalate, benzo(a)pyrene, and dioxins/furans (see table 4-6) indicate that these hydrophobic chemicals have a high affinity for adsorption to sediment particles. Therefore, it is unlikely that they are present within the dissolved fraction of the water column at ecologically important concentrations.

The mean concentration of total recoverable copper in Ensenada Honda surface water collected downgradient from SWMU 2 was less than the surface water screening value (HQ = 0.62). Copper detections greater than the surface water screening value occurred in a single surface water sample (4.5 J ug/L in 02OWSW01). The concentration of dissolved copper in Ensenada Honda surface water ranged from 0.52 J ug/L in 02OWSW09 to 1.6 ug/L in 02OWSW02. The maximum dissolved copper concentration was less than a dissolved CCC value (3.1 ug/L) reported by the USEPA (2002a). Given that the dissolved concentration of copper more closely approximates the bioavailable fraction of this metal in the water column (USEPA, 1999b and 2002a), the absence of detections above the dissolved CCC value indicates that copper is not impacting aquatic receptor groups within the Ensenada Honda.

To further evaluate the significance of potential risks presented by total recoverable copper relative to background, the Ensenada Honda surface water data were statistically compared to background surface water data. Table 4-71 presents a summary and results of the statistical evaluation. A box plot diagram illustrating the distribution of each data set is shown on Figure 4-32. As evidenced by Table 4-71 and Figure 4-32, the distribution of total recoverable copper concentrations in Ensenada Honda surface water downgradient from SWMU 2 is statistically equivalent to background levels. Based on a mean HQ less than 1.0 for total recoverable copper, the distributional statistics presented in Table 4-71, and the absence of dissolved copper detections above the USEPA dissolved CCC value, copper is not identified as a potential risk driver for Ensenada Honda surface water, and additional evaluation is not recommended

Based on the statistical evaluation presented in Table 4-71, additional evaluation also is not recommended for tin. Although distributional statistics could not be performed on the tin data due to the large number of non-detected results in the background data sets, the evaluation of the right-tail of the distribution shows that tine concentrations in Ensenada Honda surface water

downgradient from SWMU 2 are statistically equivalent to background. Furthermore, the descriptive statistics for total recoverable tin show that maximum, mean, and 95 percent UCL background concentrations exceed Ensenada Honda surface water concentrations.

In summary, there are no potential risk drivers identified for Ensenada Honda surface water downgradient from SWMU 2. Although copper and tin were identified as ecological COPCs in Step 2 of the ERA, they are not recommended for further evaluation based on the discussion presented in the preceding paragraphs. Additional evaluation also is not recommended for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

4.7.2.5 Refined Risk Calculation and Risk Evaluation for Ensenada Honda Sediment

Table 4-34 presented the results of the Step 2 screening-level risk calculation for sediment of the open water habitat associated with SWMU 2. One VOC (acetone), two SVOCs (di-n-butylphthalate and phenol), five PAHs [benzo(a)pyrene, benzo(a)anthracene, chrysene, fluoranthene, and pyrene], and four metals (arsenic, copper, lead, and tin) were detected and identified as ecological COPCs in Step 2 of the screening-level ERA because maximum detected concentrations exceeded sediment screening values. Beryllium and thallium were detected and identified as ecological COPCs based on the lack of sediment screening values. Ten non-detected VOCs, fifty-seven non-detected SVOCs, five non-detected PAHs, thirteen non-detected organochlorine pesticides, seven non-detected PCBs, eight non-detected organophosphorous pesticides, non-detected dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration), and one non-detected inorganic (cyanide) were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values. Figure 4-19 presented the concentration distribution across the site for each detected chemical identified as an ecological COPC with a maximum HQ greater than 1.0. Table 4-72 presents the results of the refined screening-level risk calculation for those detected and non-detected chemicals identified as ecological COPCs for sediment in Step 2 of the ERA with a maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation for sediment is presented below.

As discussed above, ten non-detected VOCs, fifty-seven non-detected SVOCs, five non-detected PAHs, thirteen non-detected organochlorine pesticides, seven non-detected PCBs, eight non-detected organophosphorous pesticides, non-detected dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration), and one non-detected inorganic (cyanide) were identified as ecological COPCs because maximum reporting limits exceeded sediment screening values. Mean concentrations for five of the non-detected VOCs, four of the non-detected SVOCs, two of the non-detected PAHs, and two of the non-detected organochlorine pesticides were less than sediment screening values, while mean concentrations for the remaining non-detected chemicals exceeded sediment screening values (see Table 4-72). Of the 101 non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA (non-detected chemicals with maximum reporting limits greater than sediment screening values), only di-n-butylphthalate, benzo(a)pyrene, dibenz(a,h)anthracene, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, dieldrin, endrin, endosulfan I, endosulfan II, gamma-BHC, heptachlor, heptachlor epoxide, and dioxins/furans were detected in SWMU 1 surface and/or subsurface soil. Based on the absence of detections in SWMU 2 surface and subsurface soil, there is no indication that the remaining eighty-seven non-detected chemicals are associated with historical site activities (no evidence of a source or release to SWMU 2 surface and subsurface soil). For this reason, they are not identified as potential ecological risk drivers for estuarine wetland sediment, and no additional evaluation is recommended.

di-n-butylphthalate, benzo(a)pyrene, dibenz(a,h)anthracene, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, dieldrin, endrin, endosulfan I, endosulfan II, gamma-BHC, heptachlor, heptachlor epoxide, and

dioxins/furans are not identified as potential ecological risk drivers. The low frequency and magnitude of detections in upgradient surface soil and subsurface soil for many of these chemicals (see Tables 4-30 and 4-31) precludes their migration to the Ensenada Honda at ecologically important concentrations. Furthermore, for nonionic organic chemicals, the primary factor affecting bioavailability in sediment is TOC (USEPA, 1993a). Because bulk-sediment screening values used for di-n-butylphthalate, benzo(a)pyrene, dibenz(a,h)anthracene, 4,4'-DDD, 4,4'-DDE, 4,4'-DDT, dieldrin, endrin, gamma-BHC, heptachlor, and dioxins/furans do not account for the effect that TOC has on their bioavailability, HQs calculated based on reporting limits and bulk-sediment screening values are likely to be overestimates of potential risk. In addition, the EqP-based screening values used for endosulfan I, endosulfan II, and heptachlor epoxide are based on a default TOC content (one percent [10,000 mg/kg]). Given that the minimum TOC concentration measured in Ensenada Honda sediment downgradient from SWMU 2 was 18,000 ug/kg, HQs calculated based on a default TOC concentration also are likely to be overestimates of potential risk.

Beryllium and thallium were identified as ecological COPCs in sediment based on the lack of sediment screening values. Beryllium was detected in eight of nine sediment samples, with detected concentrations ranging from 0.033 J mg/kg in 02OWSD06 to 0.082 J mg/kg in 02OWSD03. Thallium was detected in two of nine sediment samples (0.043 J mg/kg in 02OWSD04 and 0.12 J mg/kg in 02OWSD08). As evidenced by Table 4-73, the statistical evaluations performed on the Ensenada Honda and background beryllium and thallium data sets was impaired by the low number and/or absence of detections in background sediment. The statistical evaluation of the thallium data sets was limited to the right-tail of the distribution (slippage test), while the statistical evaluation of the beryllium data sets could not be performed at all. The limited statistics performed on the thallium data sets show that the distribution of thallium concentrations in Ensenada Honda sediment is statistically equivalent to background levels. The descriptive statistics for this metal also show that the maximum detected thallium concentration occurred in the background data set. Based on statistical evaluation presented in Table 4-73 and the low magnitude of detections (maximum detected concentration was only 0.12 J mg/kg) thallium is not considered a potential risk driver for Ensenada Honda sediment downgradient from SWMU 2, and additional evaluation is not recommended.

Although a statistical evaluation of the beryllium data sets could not be performed due to the absence of detections in background sediment, this metal also is not identified as a potential risk driver for Ensenada Honda sediments downgradient from SWMU 2. This metal was not detected in upgradient media (surface soil and estuarine wetland sediment) at concentrations elevated above background (see Tables 4-65 and 4-69, respectively). This metal also was not detected in upgradient surface soil and subsurface soil at ecologically significant concentrations (see Tables 4-30 and 4-31, respectively). In addition, beryllium is not identified as an important bioaccumulative chemical by the USEPA (2000a).

Acetone was identified as an ecological COPC in sediment because the maximum detected concentration of this VOC (91 J ug/kg) exceeded the sediment screening value (5.8 ug/kg). This common laboratory contaminant was detected in eight of nine sediment samples, with a mean HQ of 8.62. As discussed elsewhere in this document, the sediment screening value used in sediment risk calculations for acetone was an EqP-based value derived using the USEPA EqP approach (USEPA 1993a). VOC EqP-based values derived in accordance with this approach are overly conservative (see Section 4.7.1.6). To further evaluate the significance of acetone detections in Ensenada Honda sediment downgradient from SWMU 2, detected concentrations were compared to an alternative screening value identified from the literature. Di Toro and McGrath (2000) reported a SQG of 2,265 ug/kg for acetone based on a target lipid model and one percent organic carbon. Given that the maximum acetone detection (91 J ug/kg) is two orders of magnitude

below this value, it is unlikely that this VOC is present at ecologically significant concentrations. Furthermore, given that the minimum TOC concentration measured in estuarine wetland sediment downgradient from SWMU 2 was 18,000 mg/kg (i.e., 1.8 percent), the Di Toro and McGrath (2000) target lipid model would predict even lower potential for bioavailability when site-specific TOC is considered (i.e., SQG of 4,077 ug/kg based on 1.8 percent TOC). Based on the comparison of the maximum acetone concentration to a SQG value developed by Di Toro and McGrath (2000), acetone is not considered a potential risk driver for Ensenada Honda sediment downgradient from SWMU 2, and additional evaluation is not recommended.

The SVOCs di-n-butylphthalate and phenol were detected and identified as ecological COPCs in Step 2 of the screening-level ERA because maximum detected concentrations exceeded sediment screening values. Di-n-butylphthalate was detected in four of nine sediment samples (mean HQ = 8.01). The sediment screening value used in the Step 2 and 3a risk calculations for this SVOC (58 mg/kg [Buchman 1999] was a bulk sediment screening value that does not take into account site-specific factors (i.e., TOC) that may influence bioavailability. Using a site-specific TOC concentration of 18,000 mg/kg (minimum TOC concentration measured in Ensenada Honda sediment) and the procedures presented in Appendix I, a site-specific screening value for di-n-butylphthalate can be derived. This value (2,083 mg/kg) is an order of magnitude higher than the maximum detected concentration (610 J ug/kg in 02OWSD03). Based on this analysis, di-n-butylphthalate is not identified as a potential risk driver for Ensenada Honda sediment downgradient from SWMU 2, and additional evaluation is not recommended.

Phenol was detected in one of nine sediment samples (320 J ug/kg in 02OWSD04), with a mean HQ of 2.46. Identical to di-n-butylphthalate, the screening value used for phenol in risk calculations was a bulk sediment screening value (130 ug/kg [Buchman 1999]). The derivation of an EqP-based screening value would not be appropriate, since the resultant value would be overly conservative (the K_{oc} value for phenol [28.5 L/kg] does not indicate that this organic is a highly sorptive chemical). An examination of the analytical data for upgradient media showed that this SVOC was not detected in surface soil and subsurface soil collected from the SWMU, nor was it detected in estuarine wetland sediment. The absence of detections in upgradient media indicates that phenol is not site-related. As such, this SVOC is not identified as a potential risk driver, and additional evaluation is not recommended.

Benzo(a)pyrene, benzo(a)anthracene, chrysene, fluoranthene, and pyrene were identified as ecological COPCs in Step 2 of the screening-level ERA because maximum detected concentrations exceeded sediment screening values. Mean HQs for these PAHs were each less than 1.0 (see Table 4-72), indicating acceptable risk to ecological receptor populations. Furthermore, each PAH was detected at a concentration greater than sediment screening values in a single sample (02OWSD05). The total PAH concentration in this sample (1,716 mg/kg [one-half the detection limit was used for non-detected PAHs]) is less than a total PAH toxicological threshold of 4,022 mg/kg reported by Long et al. (1995). To further evaluate the significance of PAH detections in Ensenada Honda sediment downgradient from SWMU 2, detected concentrations were compared to SQGs derived by Di Toro and McGrath (2000) using a target lipid model and 1.0 percent TOC. SQGs for benzo(a)pyrene, benzo(a)anthracene, chrysene, fluoranthene, and pyrene (16,324 mg/kg, 14,222 mg/kg, 14,268 mg/kg, 11,974 mg/kg, and 11,792 mg/kg, respectively) are two orders of magnitude above maximum detected concentrations (190 J ug/kg 250 J ug/kg, 180 J ug/kg, 290 J mg/kg, and 220 J mg/kg, respectively). Based on the discussion presented above, benzo(a)pyrene, benzo(a)anthracene, chrysene, fluoranthene, and pyrene are not identified as potential risk drivers for Ensenada Honda sediment downgradient from SWMU 2, and additional evaluation is not recommended.

Mean concentrations of arsenic and tin exceeded sediment screening values (HQs = 1.11 and 1.15, respectively). Both metals were detected in six of nine Ensenada Honda sediment samples at concentrations greater than sediment screening values. To evaluate the significance of arsenic and tin concentrations in Ensenada Honda sediment relative to background concentrations, the data sets were statistically compared in accordance with Navy guidance (NFESC 2002). As evidenced by Table 4-73, the distributional statistics (Satterthwaite's t-test and slippage test) concluded that the distribution of arsenic concentrations in Ensenada Honda sediment downgradient from SWMU 2 are elevated above background concentrations. Although the arsenic concentrations in Ensenada Honda sediment are statistically elevated relative to background, arsenic was not identified as an ecological risk driver based on the following lines of evidence: (1) As evidenced by a maximum HQ of 1.52, the magnitude of detections above the sediment screening value is low; (2) The range of arsenic detections in upgradient surface soil (1.1 mg/kg to 19.6 mg/kg) and subsurface soil (3.3 mg/kg to 18.7 mg/kg), and estuarine wetland sediment (2 mg/kg to 14 mg/kg) is very consistent with the range of detections in Ensenada Honda sediment (3.5 mg/kg to 11 mg/kg). The consistency of arsenic detections in SWMU 2 surface soil, subsurface soil, estuarine wetland sediment, and Ensenada Honda sediment is not indicative of a release at SWMU 2. Based on the low magnitude of detected concentrations above the sediment screening value and no indication that a release has occurred at SWMU 2, arsenic is not identified as a potential ecological risk driver for Ensenada Honda sediment, and no additional evaluation is recommended.

A statistical evaluation of the tin data sets could not be performed due to the absence of detections in the background data set. However, the maximum detected tin concentration (12.69 J mg/kg in 02OWSD02 [expressed as TBT]) is an order of magnitude below an alternative screening value identified from the literature (an IC_{50} of 600 mg/kg based on *Hexagonia sp.* growth reported by Kristin et al., 1998 [see Section 4.7.1.4]). Based on the comparison of the maximum tin concentration to an effect-based concentration, tin also is not identified as potential risk drivers for Ensenada Honda sediment, and additional evaluation is not recommended.

Mean concentrations of copper and lead were less than sediment screening values (HQs = 0.72 and 0.27, respectively). Both metals were detected in a single sediment sample at concentrations greater than the sediment screening values (lead was detected in 02OWSD04 at 48 mg/kg and copper was detected in 02OWSD03 at 19 mg/kg). As evidenced by their maximum HQ values (copper = 1.02 and lead = 1.59), both detections were only slightly elevated above screening values. Although the statistical evaluations presented in Table 4-73 indicate that the distribution of copper and lead concentrations in Ensenada Honda sediment are elevated above background levels, these two metals are not identified as potential risk drivers. This determination is based on mean HQ values (less than 1.0 for each metal) and the low frequency and magnitude of detections above sediment screening values.

In summary, there are no potential risk drivers identified for Ensenada Honda sediment downgradient from SWMU 2. Although acetone, di-n-butylphthalate, phenol, benzo(a)pyrene, benzo(a)anthracene, chrysene, fluoranthene, pyrene, arsenic, beryllium, copper, lead, thallium, and tin were identified as ecological COPCs in Step 2 of the ERA, they are not recommended for further evaluation based on the discussion presented in the preceding paragraphs.

4.7.2.6 Refined Risk Calculation and Risk Evaluation for Terrestrial and Aquatic Food Web Exposures

Tables 4-35 through 4-38 presented the results of the Step 2 screening-level risk calculation for SWMU 2 terrestrial and aquatic food web exposures. The sections that follow present and

discuss the refined screening-level risk calculation and risk evaluation for terrestrial and aquatic food web exposures.

4.7.2.6.1 Terrestrial Food Web Exposures

The refined risk calculation for terrestrial food web exposures to chemicals detected in SWMU 2 surface soil and subsurface soil is presented below.

Terrestrial Food Web Exposures: Surface Soil

Table 4-35 presented the results of the Step 2 screening-level risk calculation for terrestrial food web exposures to chemicals in surface soil. Based on the comparison of maximum exposure doses to NOAEL-based screening values, ten detected metals (arsenic, barium, cadmium, chromium, cobalt, copper, lead, mercury, selenium, and zinc) had NOAEL-based HQ values greater than or equal to 1.0 for one or more of the terrestrial avian receptors. These ten metals were identified as ecological COPCs for terrestrial food web exposures. Dioxin/furan chemicals also detected and identified as ecological COPCs because the maximum exposure dose, based on the sum of 2,3,7,8-TCDD equivalents, exceeded NOAEL-based screening values for each of the avian receptors. In addition, one detected VOC (toluene) and one detected metal (beryllium) were identified as ecological COPCs for terrestrial food web exposures due to the lack of ingestion-based screening value. In addition to the detected chemicals identified above, two non-detected SVOCs (hexachlorobenzene and di-n-butylphthalate) and three non-detected PCBs (Aroclor-1248, Aroclor-1254, and Aroclor-1260) were identified as ecological COPCs because maximum American robin exposure doses (based on maximum reporting limits) exceeded NOAEL-based screening values. Table 4-74 presents the results of the refined risk calculation for detected and non-detected chemicals with a maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation is presented in the paragraphs that follow.

As discussed above, two non-detected SVOCs (hexachlorobenzene and di-n-butylphthalate) and three non-detected PCBs (Aroclor-1248, Aroclor-1254, and Aroclor-1260) were identified as ecological COPCs in Step 2 of the screening-level ERA because exposure doses based on maximum reporting limits for the American robin exceeded NOAEL-based screening values established for this receptor. Mean American robin exposure doses for the SVOCs and PCBs were less than the NOAEL-based screening value (see Table 4-74). Based on mean exposure doses less than NOAEL-based screening values, it is unlikely that hexachlorobenzene, di-n-butylphthalate, Aroclor-1248, Aroclor-1254, and Aroclor-1260 are impacting avian omnivore populations. For this reason, they are not identified as potential ecological risk drivers for SWMU 2 surface soil terrestrial food web exposures, and no additional evaluation is recommended.

Mean cobalt and selenium exposure doses were less than NOAEL-based screening values for each of the terrestrial avian receptors. The statistical evaluation presented in Table 4-65 also demonstrated that the distribution of cobalt and selenium concentrations in SWMU 2 surface soil are statistically equivalent to background levels, indicating that these metals are not likely to be site-related and do not present risks to the avian receptors at the SWMU above background levels. Based on mean exposure doses less than NOAEL-based screening values and the statistical evaluations presented in Table 4-65, cobalt and selenium are not identified as potential risk drivers for terrestrial food web exposures, and additional evaluation is not recommend.

Mean arsenic, barium, cadmium, chromium, and copper exposure doses were less than NOAEL-based screening values. However, mean lead, mercury, and zinc exposure doses for the American robin and mourning dove exceeded NOAEL-based screening values (see Table 4-74). The

statistical evaluations presented in Table 4-65 demonstrated that the distributions of arsenic, barium, and chromium concentrations in SWMU 2 surface soil were elevated above background concentrations. The statistical evaluations performed on the copper data sets were contradictory. The Wilcoxin rank sum test and quantile test concluded that the two data sets were statistically equivalent, while the slippage test concluded that copper concentrations in SWMU 2 surface soil were elevated above background concentrations. However, an examination of the descriptive statistics presented in Table 4-65 show that maximum, mean, and 95 percent UCL concentrations in the SWMU 2 data set exceeded maximum, mean, and 95 percent UCL background concentrations. The statistical evaluations performed on the cadmium data sets were impaired by the low number of detections in background surface soil. As evidenced by Table 4-65, only the slippage test could be performed, which concluded that the SWMU 2 data set was statistically equivalent to the background data set. However, identical to copper, the distributional statistics show that SWMU 2 cadmium concentrations are elevated above background concentrations.

To further evaluate the significance of arsenic, barium, cadmium, chromium, copper, lead, mercury, and zinc detections in SWMU 2 surface soil, Equation 4-6 was used to derive surface soil concentrations for each metal that would result in a mean exposure doses greater than or equal to the NOAEL-based screening value for the most sensitive receptor (American robin [arsenic, cadmium, chromium, copper, lead, mercury, and zinc] or morning dove [barium]). The results of this evaluation are presented in the paragraphs that follow.

Arsenic, cadmium, and copper surface soil concentration greater than 148 mg/kg, 4.2 mg/kg, and 1,775 mg/kg, respectively would result in mean American robin exposure doses greater than or equal to NOAEL-based screening values. Given that the maximum detected arsenic, cadmium, and copper concentrations were 19.6 mg/kg in 06SS145, 3.9 mg/kg in 06SS145, and 919 mg/kg in 06SS145), respectively, no single detected concentration exceeded these threshold concentrations. Based on this evaluation and the mean HQ values presented in Table 4-74, arsenic, cadmium, and copper would not be expected to impact avian omnivore populations at SWMU 2. A chromium concentration greater than 52 mg/kg would result in a mean American robin exposure dose greater than or equal to the NOAEL-based screening value. Chromium was detected in two surface soil samples at a concentration greater than this threshold value (55.3J mg/kg in 2SB03 and 59.5 J in 2SB05). Based on the low magnitude of detections greater than the threshold value, chromium detections in surface soil also would not be expected to impact avian omnivore populations using the terrestrial habitats at SWMU 2. A barium concentration greater than 895 mg/kg would result in a mean exposure dose for the mourning dove greater than or equal to the NOAEL-based screening value. Barium was detected in a single surface soil at a concentration greater than 895 mg/kg (1,060 mg/kg in 2SB05). Given the low frequency and magnitude of detections greater than the threshold value, barium detections in surface soil would not be expected to impact avian herbivore populations at SWMU 2. Based on mean HQs less than 1.0 for each of the avian receptors and the discussion presented above, arsenic, barium, cadmium, chromium, and copper are not identified as potential risk drivers for terrestrial food web exposures at SWMU 2, and additional evaluation is not recommended.

Lead, mercury, and zinc surface soil concentrations greater than 60 mg/kg, 0.108 mg/kg, and 124 mg/kg, respectively would result in mean American robin exposure doses greater than or equal to NOAEL-based screening values. Lead was detected in fourteen surface soil samples at concentrations greater than the threshold value of 60 mg/kg, including 06SS145 (4,760 J mg/kg), 2SS03 (1,400 mg/kg), 2SB03 (1,000 mg/kg), 2SB05 (512 mg/kg) and 390 J (2SD02). Zinc was detected in twelve surface soil samples at concentrations greater than the threshold value of 124 mg/kg, including 06SS145 (1,440 mg/kg), 2SB05 (1,260 mg/kg), 2SS10 (1,000 mg/kg), 2SB03 (845 mg/kg), 2SD02 (841 mg/kg), 2SS05 (800 mg/kg), and 2SS03 (720 mg/kg). Mercury was detected in fourteen surface soil concentrations greater than the threshold value of 0.108 mg/kg,

including 2SS11 (19 J mg/kg), 2SS07 (59 J mg/kg), 2SS12 (0.57 J mg/kg), and 06SS145 (0.45 mg/kg). Based on mean HQs greater than 1.0 and the frequency and magnitude of detections greater than concentrations that would result in mean American robin exposure doses greater than NOAEL-based screening value, lead, mercury, and zinc are identified as potential risk drivers for SWMU 2 surface soil and additional evaluation is recommended.

The mean exposure dose for dioxin/furan chemicals was less than the NOAEL-based screening value for the American robin (HQ = 0.11). As evidenced by Table 4-13, the most toxic congeners (2,3,7,8-TCDD and 1,2,3,7,8-PCDD) were not detected in any of the SWMU 2 surface soil samples. Based on a mean HQ less than 1.0, dioxin/furan chemicals are not identified as a potential ecological risk driver for avian omnivore food web exposures, and no additional evaluation is recommended.

Ingestion-based screening values are not available for toluene and beryllium. Toluene was detected in one of twelve surfaces soil samples (7 ug/kg in 06SS145), while beryllium was detected in ten of twelve surface soil samples (detected concentrations ranged from 0.07 mg/kg in 2SD03 to 0.54 mg/kg in 2MW02. Neither chemical has been identified as an important bioaccumulative chemical (USEPA, 2000a). The statistical evaluation presented in Table 4-65 also demonstrated that beryllium concentrations in SWMU 2 surface soil are statistically equivalent to background concentrations. Based on the low frequency and/or magnitude of detections and, in the case of beryllium, the statistical evaluation presented in Table 4-65, toluene and beryllium are not identified as potential risk drivers for SWMU 2 terrestrial food web exposures, and additional evaluation is not recommended.

In summary, lead, mercury, and zinc were identified as potential risk drivers for surface soil terrestrial food web exposures. These three metals were also identified as potential risk drivers for SWMU 2 surface soil (see Section 4.7.2.1). Although toluene, dioxins/furans, arsenic, barium, beryllium, cadmium, chromium, cobalt, copper, and selenium were detected and identified as ecological COPCs for terrestrial food web exposures in Step 2 of the screening-level ERA, they are not recommended for further evaluation based on the discussion presented above. No additional evaluation also is not recommended for the non-detected chemicals identified as ecological COPCs for surface soil terrestrial food web exposures in Step 2 of the screening-level ERA.

Terrestrial Food Web Exposures: Subsurface Soil

Table 4-36 presented the results of the screening-level risk calculations for terrestrial food web exposures to chemicals in subsurface soil. Based on the comparison of maximum exposure doses to NOAEL-based screening values, five detected metals (cadmium, chromium, copper, lead, and zinc) had HQ values greater than 1.0 for one or more of the terrestrial avian receptors. Beryllium was also detected and identified as an ecological COPC based on the lack of ingestion-based screening values. These seven metals were identified as ecological COPCs for terrestrial food web exposures. Although not detected, hexachlorobenzene was identified as an ecological COPC for subsurface soil terrestrial food web exposures because the maximum American robin exposure dose (based on the maximum reporting limit) exceeded the NOAEL-based screening value. Table 4-75 presents the results of the refined risk calculation for detected and non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA. An evaluation of the refined screening-level risk calculation is presented in the paragraphs that follow.

As discussed above, one non-detected SVOC (hexachlorobenzene) was identified as an ecological COPC in Step 2 of the screening-level ERA because the maximum American robin exposure dose based on the maximum reporting limit exceeded the NOAEL-based screening value. The mean

American robin exposure doses for the SVOCs and PCBs were less than the NOAEL-based screening value (see Table 4-74). Based on a mean exposure dose less than the NOAEL-based screening value, it is unlikely that hexachlorobenzene is impacting avian omnivore populations. For this reason, this SVOC is not identified as a potential ecological risk driver for subsurface soil terrestrial food web exposures, and no additional evaluation is recommended.

Mean cadmium, chromium, and copper exposure doses were less than NOAEL-based screening values for each of the terrestrial avian receptors. The discussion presented in Section 4.7.2.2 indicated that the concentrations of these three metals are elevated above background concentrations. To further evaluate the significance of cadmium, chromium, and copper detections in SWMU 2 subsurface soil, Equation 4-6 was used to derive surface soil concentrations for each metal that would result in a mean exposure dose greater than or equal to the NOAEL-based screening value for the most sensitive receptor (American robin). The results of this evaluation are presented in the paragraphs that follow.

As presented in the discussion for SWMU 2 surface soil, cadmium, chromium, and copper soil concentration greater than 4.2 mg/kg, 52 mg/kg, and 1,775 mg/kg, respectively would result in mean American robin exposure doses greater than or equal to NOAEL-based screening values. Chromium was detected in one subsurface soil sample at a concentration greater than the threshold value of 52 mg/kg (59.8 mg/kg in 06SS141). Given the low magnitude of this single detected concentration above the threshold value, it unlikely that chromium concentrations in SWMU 2 subsurface soil would impact avian omnivore populations at SWMU 2. Based on mean HQ values less than 1.0 for each avian receptor and the evaluation presented above, chromium is not identified as a potential risk driver for SWMU 2 subsurface soil, and additional evaluation is not recommended. Cadmium was detected in three subsurface soil samples at concentrations greater than the threshold value of 4.2 mg/kg (8.7 mg/kg in 06SS141, 5 mg/kg in 06SS146, and 6.3 mg/kg in 06SS150). However, based on a mean HQ less than 1.0 and the low magnitude of detections greater than the 4.2 mg/kg threshold value, cadmium is not considered a potential risk driver for SWMU 2 terrestrial food web exposures. Copper was detected in a single subsurface soil sample at a concentration greater than the threshold value of 1,775 mg/kg (5,850 mg/kg in 06SS141). Although mean HQ values for this metal were less than 1.0 for each of the avian receptors, copper is identified as a potential risk driver for SWMU 2 subsurface soil based on the magnitude of this single detection above the threshold concentration resulting in a mean exposure dose greater than the NOAEL-based screening value for the American robin.

Mean lead and zinc HQ values exceeded 1.0 for the American robin (HQs = 11.19 and 5.23, respectively) and the morning dove (HQs = 6.19 and 2.12, respectively). The lead LOAEL-based HQ value for the American robin also exceeded 1.0 (1.12). The descriptive statistics presented in Table 4-67 showed that maximum, mean, and 95 percent UCL concentrations in the SWMU 2 subsurface soil data set exceeded background surface soil concentrations. To further evaluate the significance of lead and zinc concentrations in SWMU 2 subsurface soil, Equation 4-6 was used to derive surface soil concentrations for each metal that would result in a mean exposure doses greater than or equal to the NOAEL-based screening value for the most sensitive receptor (American robin). As presented in the discussion for SWMU 2 surface soil, lead and zinc concentrations greater than 60 mg/kg and 124 mg/kg would result in mean exposure doses greater than the NOAEL-based screening value for the American robin. Lead was detected in seven of twelve subsurface soil samples at a concentration greater than the threshold value of 60 mg/kg, including 5,850 J mg/kg in 06SS146, 1,210 J mg/kg in 06SS141, 470 mg/kg in 2SS03, and 190 mg/kg in 2SS05). Zinc was detected in nine of twelve subsurface soil samples at a concentration greater than the threshold value of 124 mg/kg, including 3,350 mg/kg in 06SS141, 2,010 mg/kg in 06SS146, 780 mg/kg in 2SS03. Based on mean NOAEL-based HQs greater than 1.0 for the American robin and mourning dove, as well as the magnitude and frequency of detected

concentrations greater than threshold values that would result in mean exposure doses greater than NOAEL-based screening values, lead and zinc are identified as potential risk drivers for subsurface soil terrestrial food web exposures, and additional evaluation is recommended.

Beryllium was identified as an ecological COPC in Step 2 of the screening-level ERA based on the lack of an ingestion-based screening value. This metal was detected in a single subsurface soil sample (1.4 mg/kg in 06SS155D). Beryllium has not been identified as an important bioaccumulative chemical. The low magnitude and frequency of detections also indicates that beryllium is not likely impacting terrestrial avian populations at SWMU 2. As such, beryllium is not identified as a potential risk driver for subsurface soil terrestrial food web exposures, and no further evaluation is recommended.

In summary, copper, lead, and zinc are identified as potential risk drivers for subsurface soil terrestrial food web exposures. These three metals also were identified as ecological COPCs for subsurface soil (see Section 4.7.2.2). Although cadmium, chromium, mercury, and beryllium were detected in SWMU 2 subsurface soil and identified as ecological COPCs in Step 2 of the screening-level ERA, they are not recommended for further evaluation based on the discussion presented in the preceding paragraph. No additional evaluation also is recommended for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

4.7.2.6.2 Aquatic Food Web Exposures

The refined risk calculation for aquatic food web to chemicals detected estuarine wetland and Ensenada Honda surface water and/pr sediment downgradient from SWMU 2 is presented below.

Aquatic Food Web Exposures: Estuarine Wetland Sediment

Table 4-37 presented the results of the screening-level risk calculations for aquatic food web exposures to chemicals in estuarine wetland sediment downgradient from SWMU 2. Based on the comparison of maximum exposure doses to NOAEL-based screening values, one detected SVOC (di-n-butylphthalate) and eight detected metals (arsenic, chromium, cobalt, copper, lead, mercury, vanadium, and zinc) were identified as ecological COPCs in Step 2 of the screening-level ERA because maximum exposure doses exceeded NOAEL-based screening values for the spotted sandpiper. One detected VOC (chlorobenzene), two detected SVOCs [2-acetylaminofluorene and p-(dimethylamino)azobenzene], and one detected metal (beryllium) also were identified as ecological COPCs for estuarine wetland aquatic food web exposures due to the lack of ingestion-based screening values. Two non-detected SVOCs (hexachlorobenzene and dinoseb), seven non-detected PCBs, two non-detected organophosphorous pesticides (ethyl parathion and methyl parathion), and dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration) were identified as ecological COPCs for aquatic food web exposures because maximum exposure doses (based on maximum reporting limits) exceeded NOAEL-based screening values for one or more of the aquatic avian receptors. Table 4-76 presents the results of the refined risk calculation for detected and non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA with a maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation is presented in the paragraphs that follow.

As discussed above, two non-detected SVOCs (hexachlorobenzene and dinoseb), two non-detected organochlorine pesticides (ethyl parathion and methyl parathion), seven non-detected PCBs, and dioxins/furans (based on the 2,3,7,8-TCDD equivalent concentration) were identified as ecological COPCs for aquatic food web exposures because maximum exposure doses (based on maximum reporting limits) exceeded NOAEL-based screening values for one or more of the

aquatic avian receptors. Mean exposure doses for hexachlorobenzene, dinoseb, methyl parathion, and the seven non-detected PCBs were less than ingestion-based screening values. These chemicals also were not detected in SWMU 2 surface and subsurface soil (see Appendix G.12 and G.13). Therefore, there is no indication that they are site-related (no evidence of a source or release to surface or subsurface soil). Although the mean ethyl parathion exposure dose is greater than the ingestion-based screening value, this organophosphorous pesticide also was not detected in SWMU 2 surface soil and subsurface soil. Dioxins/furans were detected in SWMU 2 surface soil. However, they have little potential to migrate to the estuarine wetland system based on their physical/chemical properties (low water solubility and high K_{ow} and K_{oc} values) and the vegetative cover present within the SWMU 1 coastal forest community. Both of these factors would minimize potential migration with surface run-off in either a dissolved or solid state (i.e., adhered to soil particles). In summary, hexachlorobenzene, dinoseb, ethyl parathion, methyl parathion, the seven non-detected PCBs, and dioxins/furans are not identified as potential ecological risk drivers for estuarine wetland food web exposures, based on mean HQs less than 1.0, the absence of detections in upgradient media, and/or their physical/chemical properties.

Di-n-butylphthalate was identified as an ecological COPC for estuarine wetland aquatic food web exposures because the maximum exposure dose for this SVOC exceeded the NOAEL-based screening value for the spotted sandpiper. Di-n-butylphthalate was detected in two of nine sediment samples (450 J ug/kg in 02EWSSD02 and 340 J ug/kg in 02EWSSD05). As evidenced by Table 4-76, the mean HQ value for this SVOC was less than the NOAEL-based screening value (HQ = 0.58). Neither of the detected concentrations exceeds the sediment concentration that would result in a mean exposure dose greater than or equal to the NOAEL-based screening value (610 ug/kg calculated using Equation 4-7). Based on a mean HQ value less than 1.0 and the comparison of detected concentrations to the threshold value of 610 ug/kg, di-n-butylphthalate is not identified as a potential risk driver for estuarine wetland aquatic food web exposures.

Mean exposure doses for arsenic, chromium, cobalt, copper, vanadium, and zinc were less than ingestion-based screening values (HQs = 0.19, 0.94, 0.83, 0.32, 0.88, and 0.58, respectively, while mean exposure doses for lead and mercury exceeded NOAEL-based screening values (5.39 and 2.12, respectively). The mean mercury exposure dose for the spotted sandpiper also exceeded the MATC-based screening value (HQ = 1.70). With the exception of lead, the statistical evaluations presented in Table 4-69 showed that the concentration distributions of these eight metals in estuarine wetland sediment downgradient from SWMU 2 are statistically equivalent to background concentrations, indicating that they are not presenting risks to avian invertebrate consumer populations above background levels. The statistical evaluations performed on the lead data set were contradictory (see Table 4-69). The descriptive statistics presented in Table 4-69 support the results of the distributional statistics for arsenic, chromium, cobalt, and vanadium. Maximum, mean, and 95 percent UCL concentrations are less than or slightly elevated above maximum, mean and 95 percent UCL background concentrations. Based on mean HQs less than NOAEL-based screening values and the statistical evaluations (distributional and descriptive statistics) presented in Table 4-69, arsenic, chromium, cobalt, and vanadium are not identified as potential risk drivers for estuarine wetland aquatic food web exposures, and additional evaluation is not recommended.

An examination of the descriptive statistics presented in Table 4-69 show that copper, lead, mercury, and zinc concentrations (maximum, mean, and 95 percent UCL concentrations) in estuarine wetland sediment downgradient from SWMU 2 are elevated above background levels. To further evaluate the significance of detections in estuarine wetland sediment, detected concentrations for these four metals were compared to minimum concentrations that would result in a mean HQ greater than 1.0 (281 mg/kg, 13.7 mg/kg, 0.035 mg/kg, and 84 mg/kg for copper,

lead, mercury, and zinc, respectively [threshold values derived using equation 4-7]). The maximum copper concentration (280 J mg/kg) is less than the threshold concentration for this metal. Based on a mean HQ less than 1.0 and the absence of detections above the concentration that would result in a mean exposure dose greater than the NOAEL-based screening value, copper is not identified as a potential ecological risk driver for estuarine wetland aquatic food web exposures, and no additional evaluation is recommended. Zinc was detected in two of nine sediment samples at concentrations greater than the threshold value of 84 mg/kg (110 mg/kg in 02EWSSD01 and 02EWSSD02). Given the low magnitude of detections above the threshold concentration and a mean HQ less than 1.0, it is unlikely that zinc is impacting aquatic invertebrate consumers foraging with the estuarine wetland unit downgradient from SWMU 2. For this reason, zinc is not identified as a potential ecological risk driver for avian invertebrate consumers, and no further evaluation is recommended. Lead and mercury were detected in seven of seventeen and twelve of seventeen estuarine wetland sediment samples at concentrations greater than threshold concentrations that would result in a mean exposure dose greater than the NOAEL-based screening value. Of the twelve mercury detections greater than the threshold concentration, seven exceeded the maximum background concentration (0.14 mg/kg), including the detection in 2EWSD11 (1.3 mg/kg). All lead detections greater than the threshold value also exceeded the maximum background concentration. Based on mean HQ values greater than NOAEL-based screening values and the frequency and/or magnitude of detections above background concentrations and concentrations that would result in mean exposure doses greater than NOAEL-based screening values, lead and mercury are identified as potential ecological risk drivers for estuarine wetland sediment downgradient from SWMU 2, and additional evaluation is recommended.

Chlorobenzene, 2-acetylaminofluorene, p-(dimethylamino)azobenzene, and beryllium were identified as ecological COPCs for estuarine wetland aquatic food web exposures due to the lack of ingestion-based screening values. Chlorobenzene, 2-acetylaminofluorene, and p-(dimethylamino)azobenzene were each detected in a single sediment sample. Chlorobenzene was detected in 02EWSSD03 at 3.9 J ug/kg, 2-acetylaminofluorene was detected in 02EWSSD09 at 400 J ug/kg, and p-(dimethylamino)azobenzene was detected in 02EWSSD09 at 86 J ug/kg. These SVOCs are not identified as important bioaccumulative chemicals by the USEPA. Based on the low frequency of detection and the unlikely ability to bioaccumulate in aquatic prey items, these four SVOCs are not identified as potential risk drivers for estuarine wetland aquatic food web exposures, and additional evaluation is not recommended.

Beryllium was detected in each of the estuarine wetland sediment samples. Concentrations ranged from 0.074 J in 02EWSSD05 to 0.33 mg/kg in 02EWSSD01. The descriptive and distributional statistics presented in Table 4-69 showed that the distribution of beryllium concentrations in estuarine wetland sediment downgradient from SWMU 2 is statistically equivalent to background concentrations. This indicates that beryllium is not presenting a risk to avian invertebrate consumer populations above background levels. Based on the descriptive and distributional statistics presented in Table 4-69, beryllium is not identified as a potential risk driver for estuarine wetland sediment downgradient from SWMU 2, and additional evaluation is not recommended.

In summary, lead and mercury were identified as potential ecological risk drivers for estuarine wetland food web exposures downgradient from SWMU 2. Although chlorobenzene, 2-acetylaminofluorene, di-n-butylphthalate, p-(dimethylamino)azobenzene, arsenic, beryllium, chromium, cobalt, copper, vanadium, and zinc were identified as ecological COPCs in Step 2 of the screening-level ERA, they are not recommended for further evaluation based on the discussion presented in the preceding paragraphs. Additional evaluation also is not recommended

for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

Aquatic Food Web Exposures: Ensenada Honda Surface Water and Sediment

Table 4-38 presented the results of the screening-level risk calculation for Ensenada Honda aquatic food web exposures. Based on the comparison of maximum exposure doses to NOAEL-based screening values, two detected metals (arsenic and mercury) had HQ values greater than or equal to 1.0 for either the West Indian manatee (arsenic) or the double-crested cormorant (mercury). Two detected VOCs (styrene and toluene), one detected SVOC (butylbenzylphthalate), and one detected metal (beryllium) also were identified as ecological COPCs for estuarine wetland aquatic food web exposures due to the lack of ingestion-based screening values for the double-crested cormorant. In addition to the detected chemicals identified above, three non-detected organophosphorous pesticides (disulfoton, ethyl parathion, and methyl parathion) and the non-detected dioxin/furan chemicals were identified as ecological COPCs because maximum exposure doses (based on maximum reporting limits) exceeded NOAEL-based screening values for one or more of the open water receptors. Table 4-77 presents the results of the refined risk calculation for detected and non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA with an maximum HQ greater than 1.0. An evaluation of the refined screening-level risk calculation is presented in the paragraphs that follow. As discussed in Section 4.7, refined risk estimates were not derived for the West Indian manatee. As such, the risk characterization for this receptor did not include an evaluation of refined screening-level risk estimates.

Three non-detected organophosphorous pesticides (disulfoton, ethyl parathion, and methyl parathion) and the non-detected dioxin/furan chemicals were identified as ecological COPCs because maximum exposure doses (based on maximum reporting limits) exceeded NOAEL-based screening values for one or more of the open water receptors evaluated in this ERA. An examination of the analytical data presented in Appendix G.20 shows that disulfoton, ethyl parathion, and methyl parathion were not detected in SWMU 2 surface or subsurface soil. The absence of detections in upgradient surface soil indicates that these three organophosphorous pesticides are not related to historical activities at the SWMU (no evidence of a source or a release to SWMU 2 surface and subsurface soil). For this reason, disulfoton, ethyl parathion, and methyl parathion are not identified as potential ecological risk drivers for AEAnsenada Honda aquatic food web exposures, and no additiona evaluation is recommended. Dioxin/furan congeners were detected in SWMU 2 surface soil. However, they have little potential to migrate to the estuarine wetland system based on their physical/chemical properties (low water solubility and high K_{ow} and K_{oc} values) and the vegetative cover present within the SWMU 1 coastal forest community. Both of these factors would minimize potential migration with surface run-off in either a dissolved or solid state (i.e., adhered to soil particles). Based on their physical/chemical properties, dioxins/furans are no identified as potential ecological risk drivers for Ensenada Honda aquatic food web exposures

Styrene, toluene, butylbenzylphthalate, and beryllium were identified as ecological COPCs for aquatic food web exposures in Step 2 of the screening-level ERA based on the lack of ingestion-based screening values for the double-crested cormorant. Toluene was detected in two surface water samples (0.19 J ug/L in 02OWSW01 and 0.22 J ug/kg in 02OWSW07), while styrene was detected in a single surface water sample (0.24 J ug/L in 02OWSW01). Butylbenzylphthalate was detected in a single surface water sample (0.88 J ug/L in 02OWSW05). These three organics were not detected in Ensenada Honda sediment. An examination of the analytical data presented in Appendix G.13 and Appendix G.14 shows that butylbenzylphthalate and styrene were not detected in SWMU 2 surface and subsurface soil, while toluene was detected in a single surface

soil sample (7 ug/kg in 06SS145). These data do not indicate that they are related to site activities at the SWMU (no evidence of a source or release to SWMU 2 surface and subsurface soil). Beryllium was not detected within the dissolved fraction of surface water collected from the Ensenada Honda. This metal was detected in eight of nine sediment samples, at concentrations ranging from 0.033 J ug/kg in 02OWSD06 to 0.082 J ug/kg in 02OWSD03). Although conservatively evaluated in this ERA for food web exposures, these four chemicals are not identified as important bioaccumulative chemicals by the USEPA (2000b). Given the low frequency and/or magnitude of detections in Ensenada Honda sediment, the absence of detections in SWMU 2 surface and subsurface soil (in the case of styrene and butybenzylphthalate), and their lack of identification as an important bioaccumulative chemical (USEPA, 2000b), styrene, toluene, butylbenzylphthalate, and beryllium are not identified as potential risk drivers for avian piscivores and mammalian herbivores foraging within the Ensenada Honda, and additional evaluation is not recommended.

The mean mercury exposure dose was less than the NOAEL-based screening value for the double crested cormorant (mean HQ = 0.48). The maximum detected sediment concentration (0.034 mg/kg in 02OWSD03) also is less than the concentration that would result in a mean exposure dose greater than the NOAEL-based screening value (0.049 mg/kg) As such, it is unlikely that detected concentrations of mercury would impact avian piscivore populations foraging withint he Ensenada Honda. Based on a mean exposure dose less than the NOAEL-based screening value and the lack of detections greater than the threshold concentration that would result in mean exposure dose greater than the NOAEL-based screening value, mercury is not identified as a potential risk drivers for avian piscivore aquatic food web exposures, and no additional evaluation is recommended..

Maximum arsenic, mercury, and selenium exposure doses for the West Indian manatee exceeded NOAEL-based screening values. Selenium was detected in nine of nine Ensenada Honda sediment samples at concentrations ranging from 0.27 J mg/kg (02OWSD09) to 0.51 J mg/kg (02OWSD05). As evidenced by Appendix G.33, this metal was not detected in background open water sediment. Maximum surface and subsurface soil selenium concentrations (1.1 J mg/kg and 0.72 J mg/kg [see Appendix G.13 and Appendix G.14, respectively) are less than the maximum background surface soil sample (1.2 J mg/kg). These data indicate that selenium concentrations in SWMU 2 surface and subsurface soil are not related to site activities (no evidence of a source or release). Although there does not appear to be a clear relationship between SWMU 2 surface soil/subsurface soil concentrations and Ensenada Honda sediment concentrations, selenium is conservatively identified as a potential ecological risk driver for mammalian herbivore food web exposures based on the Federal status of the West Indian manatee in Puerto Rico (i.e., endangered)..

Mercury was detected in eight of nine Ensenada Honda sediment samples at concentrations ranging from 0.0094 J mg/kg (02OWSD06) to 0.034 mg/kg (02OWSD03). This range is similar to the range of mercury detections in Ensenada Honda sediment downgradient from SWMU 1 (0.023 J mg/kg to 0.062 J mg/kg). Mercury detections in Ensenada Honda sediment downgradient from SWMUs 1 and 2 are also similar to mercury detections in Enseanda Honda sediment collected downgradient from SWMUs 7/8 (0.0092 J mg/kg to 0.075 mg/kg [Baker, 2003d]. These data do not indicate that mercury detections in Enseanda Honda sediment downgradient from SWMU 2 are related to activities at the site. Arsenic was detected in nine of nine sediment samples at concentrations ranging from 3.5 mg/kg (02OWSD09) to 11 mg/kg (020WSD07). All detected concentrations exceeded the maximum background concentration (3.0 mg/kg). An examination of the SWMU 2 surface and subsurface soil (see Appendix G.13 and Appendix G.14, respectively) and estuarine wetland sediment data (see Appendix G.16) does not indicate that Ensenada Honda sediment concentrations are site-related (results across all

media were consistent). Although their does not appear to be a clear relationship between SWMU 2 surface/subsurface soil concentrations and Ensenada Honda sediment concentrations, arsenic and mercury are identified as potential ecological risk drivers for mammalian herbivore aquatic food web exposures, and additional evaluation is recommended. This recommendation is based on the Federal status of the West Indian manatee.

In addition to arsenic, mercury, and selenium, four additional metals (cadmium, copper, lead, and zinc) are conservatively identified as potential ecological risk drivers for West Indian manatee food web exposures. These four metals are identified as potential ecological risk drivers based on their bioaccumulative potential (USEPA, 2000b) and maximum HQ values derived using toxicity reference values that incorporate an interspecies extrapolation factor (HQ = 3.13 for cadmium, 1.26 for copper, 4.56 for lead, and 3.11 for zinc; see Table 4-38b).

In summary arsenic, mercury, and selenium are identified as potential ecological risk drivers for Ensenada Honda mammalian herbivore food web exposures. Cadmium, copper, lead, and zinc also are identified as potential ecological risk drivers based on their bioaccumulative potential and maximum HQ values derived using toxicity reference values that incorporate an interspecies extrapolation factor. Although, styrene, butylbenzylphthalate, toluene, and beryllium were detected and identified as ecological COPCs in Step 2 of the screening-level ERA, they are not recommended for further evaluation based on the discussion presented in the preceding paragraphs. Additional evaluation also is not recommended for the non-detected chemicals identified as ecological COPCs in Step 2 of the screening-level ERA.

4.7.2.7 Uncertainties Associated With Step 3a of the Baseline Ecological Risk Assessment for SWMU 2

Many of the uncertainties identified in Section 4.6.2 for the screening-level ERA (i.e., Steps 1 and 2 of the Navy ERA process) also apply to the refined screening-level risk calculation (i.e., Step 3a of the baseline ERA) for SWMU 2. Those uncertainties specific to the refined risk calculation for SWMU 2 are listed below.

Analytical Data

- Surface soil samples were not collected at several subsurface soil sample locations where elevated concentrations of chemicals identified as potential risk drivers were detected. Because terrestrial receptors would be expected to experience the greatest level of exposure to chemicals in surface soil, the lack of surface soil analytical data at these locations may have resulted in an understatement of potential risks. This uncertainty was reduced by an additional data collection field effort conducted in October 2004, which included the collection of surface soil samples in the immediate vicinity of historical subsurface soil sample locations containing elevated concentrations of ecological COPCs.

- A background subsurface soil data set was not available for statistical comparison to SWMU 2 subsurface soil data. In the absence of background subsurface soil data, background surface soil data were used in the statistical comparison to SWMU 2 subsurface soil data. Because the background surface soil data and SWMU 2 subsurface soil were collected from different depth intervals (0 to 1 foot bgs, and 0.5 to 1.5 or 1 to 2-feet bgs, respectively), the background surface soil data may not be comparable to the SWMU 2 subsurface soil data. This may have resulted in an overestimation or underestimation of potential risks.

Identification of Potential Risk Drivers

- Non-detected chemicals lacking media-specific and/or ingestion-based screening values were not evaluated in the refined risk calculation, nor were they identified as potential risk drivers. This approach may have resulted in an understatement of the actual number of potential risk drivers if any of the non-detected chemicals lacking screening values are present at ecologically significant concentrations. This uncertainty can be indirectly addressed in subsequent steps of the baseline ERA process through site-specific studies (e.g., toxicity tests and bioaccumulation studies) designed to evaluate detected chemicals identified as potential ecological risk drivers.

- A second source of uncertainty related to the identification of potential ecological risk drivers applies to the statistical analysis performed on SWMU 2 and background analytical data. Statistical comparisons of SWMU 2 surface soil, subsurface soil, open water surface water, and estuarine wetland and open water sediment analytical data to available background data were performed for all detected metals identified as ecological COPCs in the Step 2 screening-level risk calculation. Statistical comparisons included descriptive summaries of each data set (e.g., maximum and mean concentrations), tests of the mean/median of the distributions (two sample t-test, Satterthwaite's t-test, Wilcoxin rank sum test, or Gehan test), and tests of the right tail of the distribution (i.e., slippage test and/or quantile test).

  Full conformity with the standard assumptions underlying each test was not possible for certain chemicals, which introduced uncertainty in the determination of whether a chemical was site-related. This uncertainty was associated with slightly inadequate sample sizes (e.g., Gehan test and quantile test are typically performed on site and background sample sizes greater than ten), unequal sample sizes (maximum power and robustness occur when site and background sample sizes are equal), non-detected results (limits the type and power of statistical evaluations performed), and differences in reporting limits between the background and SWMU data sets. In addition, uncertainty was added when disagreements between statistical tests occurred for certain chemicals when more than one statistical evaluation was performed. When disagreements between tests occurred or when there was uncertainty associated with a particular test, a more detailed examination of the distribution of the data within and/or between data sets was performed. In these cases, comparisons to background were examined in conjunction with an evaluation of the potential for site release (i.e., preliminary conceptual model), the spatial pattern of concentration distributions, and the magnitude of detected concentrations above available screening values. As a conservative measure, chemicals determined to be statistically elevated over background concentrations were conventionally identified as being detected in site-related concentrations. This approach is particularly conservative, however, as the NCP Hazard Ranking System (40 CFR, Part 300, Appendix A) does not recognize a release when any chemical is detected in concentrations less than three times an appropriate background concentration.

- The lines of evidence used to identify chemicals as potential risk drivers in the risk characterization, included evaluations that statistically compared background data to SWMU 2 data. As discussed above, a subsurface soil data set was not available for a statistical comparison to SWMU 2 subsurface soil data. Due to the lack of background subsurface soil data, background surface soil data were used in the statistical comparison. The use of the background surface soil data may have resulted in an overstatement or understatement of potential ecological risk drivers.

- The background surface water and sediment data used in the statistical comparison to Ensenada Honda surface water and sediment data were collected from an embayment that does not receive anthropogenic inputs from industrial regions of the facility (e.g., storm water runoff). However, the Ensenada Honda receives inputs from a variety of sources that are associated with anthropogenic activities, including storm water runoff from industrial portions of the facility. As such, potential contamination associated with a release from the SWMU could not be differentiated from anthropogenic inputs. Because a statistical comparison to background concentrations was used as a line of evidence in the Step 3a risk characterization, the lack of background data from a surface water body receiving anthropogenic inputs similar to those potentially impacting surface water and sediment quality within the Ensenada Honda may have resulted in an overstatement of the actual number of potential ecological risk drivers.

4.7.2.8 Step 3a Decision Point and Recommendations for SWMU 2

Table 4-78 presents a summary of the ecological COPCs identified in Step 2 of the screening-level ERA, as well as the potential risk drivers identified in Step 3a of the baseline ERA. Recommendations for each media and food web exposure pathway are presented in the sections that follow.

4.7.2.8.1 Surface Soil

Based on the refined media-specific risk calculation and risk evaluation presented in Section 4.7.2.1, antimony, copper, lead, mercury, and zinc were identified as potential risk drivers for SWMU 2 surface soil. Mean HQs for copper (HQ = 5.08), lead (HQ = 8.25), mercury (HQ = 9.60), and zinc (HQ = 8.24) exceeded surface soil screening values. As evidenced by the maximum HQs presented in Table 4-30, the magnitude of maximum detections above surface soil screening values also was high (maximum HQs for copper, lead, mercury, and zinc were 18.4, 95.2, 190, and 28.8, respectively). Furthermore, the descriptive statistics presented in Table 4-65 also showed that maximum, mean, and 95 percent UCL concentrations in SWMU 2 surface soil exceeded maximum, mean, and 95 percent UCL background concentrations for these five metals. Finally, elevated concentrations of each metal were co-located at several sampling locations. Additional evaluation of antimony, copper, lead, mercury, and zinc is recommended in Step 3b of the baseline ERA.

4.7.2.8.2 Subsurface Soil

Based on the refined media-specific risk calculation and risk evaluation presented in Section 4.7.2.2, antimony, copper, lead, mercury, and zinc were identified as a potential risk driver for SWMU 2 subsurface soil. Mean copper, lead, mercury, and zinc concentrations exceeded surface soil screening values (mean HQs = 13.7, 13.6, 1.5, and 13.2, respectively). As evidenced by maximum HQ values, the magnitude of maximum detected concentrations above surface soil screening values was high for copper, lead, and zinc (maximum HQs = 117, 117, and 67, respectively). Identical to the surface soil data, maximum, mean, and 95 percent UCL concentrations in SWMU 2 surface soil exceeded maximum, mean, and 95 percent UCL background concentrations (see Table 4-67). Finally, elevated concentrations of each metal are co-located. Additional evaluation of antimony, copper, lead, mercury, and zinc is recommended in Step 3b of the baseline ERA.

4.7.2.8.3 Estuarine Wetland Sediment

Based on the refined risk calculation and risk evaluation presented in Section 4.7.2.3, copper, lead, and zinc were identified as potential risk drivers for estuarine wetland sediment downgradient from SWMU 2. This recommendation is based on the SEM-to-AVS ratio calculated for sample 2EWSD12 (126), indicating that benthic macroinvertebrates at this sample location may be exposed to toxic concentrations of SEM metals (i.e., copper, lead, and zinc) in sediment pore water. Additional evaluation of copper, lead, and zinc is recommended in Step 3b of the baseline ERA.

4.7.2.8.4 Ensenada Honda Surface Water

Based on the refined risk calculation and risk evaluation presented in Section 4.7.2.4, no potential ecological risk drivers were identified for Ensenada Honda surface water downgradient from SWMU 2, and no further evaluation is recommended.

4.7.2.8.5 Ensenada Honda Sediment

Based on the refined risk calculation and risk evaluation presented in Section 4.7.2.5, no potential ecological risk drivers were identified for Ensenada Honda sediment downgradeint from SWMU 2, and no further evaluation is recommended.

4.7.2.8.6 Terrestrial and Aquatic Food Web Exposures

Based on the refined risk calculation and risk evaluation presented in Section 4.7.2.6.1, lead, mercury, and zinc were identified as potential risk drivers for avian omnivore terrestrial food web exposures to chemicals in surface soil. The magnitude of maximum exposure doses above NOAEL-based screening values for the American robin (avian omnivore) was high for each metal (maximum HQ = 507, 4,499, and 92.9 for lead, mercury, and zinc, respectively). Furthermore, as discussed in Section 4.7.1.9.1, the descriptive statistics presented in Table 4-65 show that SWMU 2 surface soil concentrations are elevated above background levels. Finally, an evaluation of the spatial distribution of these chemicals indicated that the frequency and magnitude of detections above concentrations resulting in mean American robin exposure doses greater than NOAEL-based screening values was high. Additional evaluation of lead, mercury, and zinc are recommended in Step 3b of the baseline ERA.

Based on the refined risk calculation and risk evaluation presented in 4.7.2.6.1, copper, lead, and zinc were identified as a potential risk driver for subsurface soil terrestrial food web exposures. As evidenced by the maximum HQ values presented in Table 4-36, maximum American robin exposure doses for each metal were high (HQ = 15.3 for copper, 623 for lead, and 216 for zinc). Mean American robin exposure doses for lead and zinc also exceeded NOAEL-based screening values (mean HQs = 11.19 and 5.23, respectively). The descriptive statistics presented in Table 4-67 also showed that SWMU 1 subsurface soil concentrations are elevated above background conditions. Finally, a spatial evaluation of these metals inidicated that the frequency and/or magnitude of detections above concentrations resulting in mean American robin exposure doses greater than NOAEL-based screeining values was high. Additional evaluation of copper, lead, and zinc is recommended in Step 3b of the baseline ERA.

Lead and mercury were identified as potential risk drivers identified for aquatic food web exposures to chemicals detected in estuarine wetland sediment (see Section 4.7.2.6.2). Mean spotted sandpiper exposure doses for lead and mercury exceeded NOAEL-based screening values (mean HQs = 2.12 and 5.39, respectively). The magnitude of maximum exposure doses above

NOAEL-based screening values also was high (maximum HQ = 21.3 for lead and 103 for mercury). Finally, the frequency of detections greater than concentrations resulting in mean exposure doses greater than NOAEL-baseed screening values was high. Additional evaluation of lead and mercury is recommended in Step 3b of the baseline ERA.

Based on the refined risk calculation and risk evaluation presented in Section 4.7.2.6.2, arsenic, cadmium, copper, lead, mercury, selenium, and zinc were identified as potential ecological risk drivers for aquatic food web exposures. This recommendation is based on detections in upgradient surface and subsurface soil, maximum HQ values (see Tables 4-38 and 4-38b), the bioaccumulative potential of these seven metals, and/or the Federal status of the West Indian manatee in Puerto Rico (i.e., endangered). As discussed in Section 4.3.1.3, potentially complete exposure pathways have been identified for aquatic reptiles (i.e., sea turtles). However, based on the paucity of data concerning the toxicological effects of chemicals for reptiles, a quantitative evaluation could not be performed. Given the Federal status of sea turtles in Puerto Rico, additional evaluation is recommended in Step 3b of the baseline ERA. This evaluation will include an examination of their life history information to determine their potential for exposure to chemicals detected in Ensenada Honda sediment. Any toxicological data identified from the literature for aquatic reptiles also will be presented and discussed in Step 3b of the baseline ERA.

4.8 References

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5.0 CONCLUSIONS AND RECOMMENDATIONS

The objective of this report was to present a screening-level ERA and Step 3A of the baseline ERA. The results of the ERA were used to determine if the SWMUs should either proceed to Step 3b of the baseline ERA (baseline ERA problem formulation) or continue in the CMS planning stage. Based on the results of the ERA presented in Section 4.0, it was recommended that each SWMU proceed to Step 3b of the baseline ERA.

The sections that follow provide a summary of the potential risk drivers identified for abiotic media at each SWMU in Step 3a of the baseline ERA. Specific recommendations regarding each potential risk driver are also presented and discussed.

5.1 SWMU 1

Table 4-63 presents a summary of the ecological COPCs identified in Step 2 of the screening-level ERA, as well as the potential risk drivers identified in Step 3a of the baseline ERA for SWMU 1. Recommendations for each media and food web exposure pathway are presented in the subsections that follow.

5.1.1 Surface Soil

Based on the refined media-specific risk calculation and risk evaluation presented in Section 4.7.1.1, antimony, cadmium, copper, lead, mercury, tin, zinc, 4,4'-DDD, 4,4'-DDE, and 4,4'-DDT were identified as potential risk drivers for SWMU 1 surface soil. Mean HQs for antimony, cadmium, copper, lead, mercury, zinc, and the three organochlorine pesticides exceeded surface soil screening values (see Table 4-45). Furthermore, as evidenced by the maximum HQ values presented in Table 4-20, the magnitude and/or frequency of detections above surface soil screening values was high for several of these chemicals, including cadmium (HQ = 21.0), copper (HQ = 20.4), lead (HQ = 19.3), 4,4'-DDD (HQ = 32.5), 4,4'-DDD (HQ = 70.0), and 4,4'-DDT (HQ = 108). The descriptive statistics presented in Table 4-44 also show that maximum, mean, and 95 percent UCL concentrations in the SWMU 1 data set exceeded maximum, mean, and 95 percent UCL background concentrations. Additional evaluation of antimony, cadmium, copper, lead, mercury, tin, zinc, 4,4'-DDD, 4,4'-DDE, and 4,4'-DDT is recommended in Step 3b of the baseline ERA.

5.1.2 Subsurface Soil

Based on the refined media-specific risk calculation and risk evaluation presented in Section 4.7.1.2, 4,4'-DDT and 4,4'-DDE were identified as potential ecological risk drivers for SWMU 1 subsurface soil. The mean 4,4'-DDT concentration exceeded the surface soil screening value (mean HQ = 1.46). Although detected in only two surface soil samples at a concentration greater than the screening value, the magnitude of the maximum detection (3,500 CD ug/kg in 05SS130) was high (maximum HQ = 8.75). The mean HQ for 4,4'-DDE was less than the surface soil screening value; however, detections above the surface soil screening value are co-located at sample locations containing maximum 4,4'-DDT concentrations. Because 4,4'-DDE is a biodegradation product of 4,4'DDT, concentrations have the potential to increase over time. Additional evaluation is recommended for 4,4'-DDT and 4,4'-DDE in Step 3b of the baseline ERA.

5.1.3 Estuarine Wetland Surface Water

Based on the refined risk calculation and risk evaluation presented in Section 4.7.1.3, no potential ecological risk drivers were identified for estuarine wetland surface water downgradient from SWMU 1, and no further evaluation of estuarine wetland surface water is recommended.

5.1.4 Estuarine Wetland Sediment

Based on the refined risk calculation and risk evaluation presented in Section 4.7.1.4, no potential ecological risk drivers were identified for estuarine wetland sediment downgradient from SWMU 1, and no further evaluation of estuarine wetland sediment is recommended.

5.1.5 Ensenada Honda Surface Water

Based on the refined risk calculation and risk evaluation presented in Section 4.7.1.5, no potential ecological risk drivers were identified for Ensenada Honda surface water downgradient from SWMU 1, and no further evaluation of Ensenada Honda surface water is recommended.

5.1.6 Ensenada Honda Sediment

Based on the refined risk calculation and risk evaluation presented in Section 4.7.1.6, no potential ecological risk drivers were identified for Ensenada Honda sediment downgradient from SWMU 1, and no further evaluation of Ensenada Honda sediment is recommended.

5.1.7 Terrestrial and Aquatic Food Web Exposures

Based on the refined risk calculation and risk evaluation presented in Section 4.7.1.7.1, cadmium, lead, mercury, zinc, 4,4'-DDD, 4,4'-DDE, and 4,4'-DDT were identified as potential risk drivers for avian omnivore and/or avian herbivore terrestrial food web exposures to chemicals in surface soil. The magnitude of maximum exposure doses above NOAEL-based screening values for the American robin (avian omnivore) was high for several of these chemicals (see Table 4-26). Furthermore, as discussed in Section 4.7.1.9.1, descriptive statistics for the SWMU 1 and background surface soil data sets indicate that maximum, mean, and 95 percent UCL concentrations in the SWMU 1 data set were elevated above background concentrations. Finally, an evaluation of the spatial distribution of these seven chemicals indicated that the frequency and/or magnitude of detections above ecologically important concentrations (i.e., concentrations that would result in a mean American robin exposure dose greater than the NOAEL-based screening value) were high. Additional evaluation of cadmium, lead, mercury, zinc, 4,4'-DDD, 4,4'-DDE, and 4,4'-DDT is recommended in Step 3b of the baseline ERA.

There were no potential ecological risk drivers identified for subsurface soil food web exposures, nor were there any potential ecological risk drivers identified for estuarine wetland aquatic food web exposures. As such, no further evaluation is recommended for the receptor-pathway combinations evaluated by this ERA. Arsenic, cadmium, copper, mercury, selenium, and zinc were identified as a potential ecological risk driver for Ensenada Honda aquatic food web exposures. This recommendation is based on their bioaccumulative potential, detections in upgradient surface/subsurface soil, maximum HQ values (see Tables 4-29 and 4-38a), and the Federal status of the West Indian manatee in Puerto Rico (endangered).

As discussed in Section 4.3.1.3, potentially complete exposure pathways have been identified for aquatic reptiles (i.e., sea turtles). However, based on the paucity of data concerning the toxicological effects of chemicals for reptiles, a quantitative evaluation could not be performed.

Given the Federal status of sea turtles in Puerto Rico, additional evaluation is recommended in Step 3b of the baseline ERA. This evaluation will include an examination of their life history information to determine their potential for exposure to chemicals detected in Ensenada Honda sediment. Any toxicological data identified from the literature for aquatic reptiles also will be presented and discussed in Step 3b of the baseline ERA.

5.2 SWMU 2

Table 4-78 presents a summary of the ecological COPCs identified in Step 2 of the screening-level ERA, as well as the potential risk drivers identified in Step 3a of the baseline ERA. Recommendations for each media and food web exposure pathway are presented in the subsections that follow.

5.2.1 Surface Soil

Based on the refined media-specific risk calculation and risk evaluation presented in Section 4.7.2.1, antimony, copper, lead, mercury, and zinc were identified as potential risk drivers for SWMU 2 surface soil. Mean HQs for copper (HQ = 5.08), lead (HQ = 8.25), mercury (HQ = 9.60), and zinc (HQ = 8.24) exceeded surface soil screening values. As evidenced by the maximum HQs presented in Table 4-30, the magnitude of maximum detections above surface soil screening values also was high (maximum HQs for copper, lead, mercury, and zinc were 18.4, 95.2, 190, and 28.8, respectively). Furthermore, the descriptive statistics presented in Table 4-65 also showed that maximum, mean, and 95 percent UCL concentrations in SWMU 2 surface soil exceeded maximum, mean, and 95 percent UCL background concentrations for these five metals. Finally, elevated concentrations of each metal were co-located at several sampling locations. Additional evaluation of antimony, copper, lead, mercury, and zinc is recommended in Step 3b of the baseline ERA.

5.2.2 Subsurface Soil

Based on the refined media-specific risk calculation and risk evaluation presented in Section 4.7.2.2, antimony, copper, lead, mercury, and zinc were identified as a potential risk driver for SWMU 2 subsurface soil. Mean copper, lead, mercury, and zinc concentrations exceeded surface soil screening values (mean HQs = 13.7, 13.6, 1.5, and 13.2, respectively). As evidenced by maximum HQ values, the magnitude of maximum detected concentrations above surface soil screening values was high for copper, lead, and zinc (maximum HQs = 117, 117, and 67, respectively). Identical to the surface soil data, maximum, mean, and 95 percent UCL concentrations in SWMU 2 surface soil exceeded maximum, mean, and 95 percent UCL background concentrations (see Table 4-67). Finally, elevated concentrations of each metal are co-located. Additional evaluation of antimony, copper, lead, mercury, and zinc is recommended in Step 3b of the baseline ERA.

5.2.3 Estuarine Wetland Sediment

Based on the refined risk calculation and risk evaluation presented in Section 4.7.2.3, copper, lead, and zinc were identified as potential risk drivers for estuarine wetland sediment downgradient from SWMU 2. This recommendation is based on the SEM-to-AVS ratio calculated for sample 2EWSD12 (126), indicating that benthic macroinvertebrates at this sample location may be exposed to toxic concentrations of SEM metals (i.e., copper, lead, and zinc) in sediment pore water. Additional evaluation of copper, lead, and zinc is recommended in Step 3b of the baseline ERA.

5.2.4 Ensenada Honda Surface Water

Based on the refined risk calculation and risk evaluation presented in Section 4.7.2.4, no potential ecological risk drivers were identified for Ensenada Honda surface water downgradient from SWMU 2, and no further evaluation is recommended.

5.2.5 Ensenada Honda Sediment

Based on the refined risk calculation and risk evaluation presented in Section 4.7.2.5, no potential ecological risk drivers were identified for Ensenada Honda sediment downgradient from SWMU 2, and no further evaluation is recommended.

5.2.6 Terrestrial and Aquatic Food Web Exposures

Based on the refined risk calculation and risk evaluation presented in Section 4.7.2.6.1, lead, mercury, and zinc were identified as potential risk drivers for avian omnivore terrestrial food web exposures to chemicals in surface soil. The magnitude of maximum exposure doses above NOAEL-based screening values for the American robin (avian omnivore) was high for each metal (maximum HQ = 507, 4,499, and 92.9 for lead, mercury, and zinc, respectively). Furthermore, as discussed in Section 4.7.1.9.1, the descriptive statistics presented in Table 4-65 show that SWMU 2 surface soil concentrations are elevated above background levels. Finally, an evaluation of the spatial distribution of these chemicals indicated that the frequency and magnitude of detections above concentrations resulting in mean American robin exposure doses greater than NOAEL-based screening values was high. Additional evaluation of lead, mercury, and zinc are recommended in Step 3b of the baseline ERA.

Based on the refined risk calculation and risk evaluation presented in 4.7.2.6.1, copper, lead, and zinc were identified as a potential risk driver for subsurface soil terrestrial food web exposures. As evidenced by the maximum HQ values presented in Table 4-36, maximum American robin exposure doses for each metal were high (HQ = 15.3 for copper, 623 for lead, and 216 for zinc). Mean American robin exposure doses for lead and zinc also exceeded NOAEL-based screening values (mean HQs = 11.19 and 5.23, respectively). The descriptive statistics presented in Table 4-67 also showed that SWMU 1 subsurface soil concentrations are elevated above background conditions. Finally, a spatial evaluation of these metals indicated that the frequency and/or magnitude of detections above concentrations resulting in mean American robin exposure doses greater than NOAEL-based screening values was high. Additional evaluation of copper, lead, and zinc is recommended in Step 3b of the baseline ERA.

Lead and mercury were identified as potential risk drivers identified for aquatic food web exposures to chemicals detected in estuarine wetland sediment (see Section 4.7.2.6.2). Mean spotted sandpiper exposure doses for lead and mercury exceeded NOAEL-based screening values (mean HQs = 2.12 and 5.39, respectively). The magnitude of maximum exposure doses above NOAEL-based screening values also was high (maximum HQ = 21.3 for lead and 103 for mercury). Finally, the frequency of detections greater than concentrations resulting in mean exposure doses greater than NOAEL-based screening values was high. Additional evaluation of lead and mercury is recommended in Step 3b of the baseline ERA.

Based on the refined risk calculation and risk evaluation presented in Section 4.7.2.6.2, arsenic, cadmium, copper, lead, mercury, selenium, and zinc were identified as potential ecological risk drivers for aquatic food web exposures. This recommendation is based on detections in upgradient surface/subsurface soil, maximum HQ values (see Tables 4-38 and 4-38b), the bioaccumulative potential of these seven metals, and/or the Federal status of the West Indian

manatee in Puerto Rico (i.e., endangered). As discussed in Section 4.3.1.3, potentially complete exposure pathways have been identified for aquatic reptiles (i.e., sea turtles). However, based on the paucity of data concerning the toxicological effects of chemicals for reptiles, a quantitative evaluation could not be performed. Given the Federal status of sea turtles in Puerto Rico, additional evaluation is recommended in Step 3b of the baseline ERA. This evaluation will include an examination of their life history information to determine their potential for exposure to chemicals detected in Ensenada Honda sediment. Any toxicological data identified from the literature for aquatic reptiles also will be presented and discussed in Step 3b of the baseline ERA.

6.0 REFERENCES

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