BASELINE ECOLOGICAL RISK ASSESSMENT
SAN JACINTO RIVER WASTE PITS SUPERFUND SITE

Prepared for

McGinnes Industrial Maintenance Corporation
International Paper Company
U.S. Environmental Protection Agency, Region 6

Prepared by

Integral Consulting Inc.
411 1st Avenue S, Suite 550
Seattle, Washington 98104

August 2012
Revised May 2013

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LIST OF ACRONYMS AND ABBREVIATIONS

Abbreviation	Definition
95UCL	95 percent upper confidence limit on the mean
AhR	aryl hydrocarbon receptor
AWQC	ambient water quality criteria for the protection of aquatic life
BEHP	bis(2-ethylhexyl)phthalate
BMF	biomagnification factor
bw	body weight
CCC	criterion continuous concentration
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act of 1980
CMC	criterion maximum concentration
COI	chemical of interest
COPC	chemical of potential concern
$COPC_E$	chemical of potential ecological concern
CSM	conceptual site model
CT	central tendency
CTR	critical tissue residue
DMP	Data Management Plan
DQO	Data Quality Objective
dw	dry weight
EcoSSL	ecological soil screening level
EPC	exposure point concentration
ER-L	effects range-low
ER-M	effects range-median
EROD	ethoxyresorufin-*O*-deethylase
FCA	fish collection area
FSR	Field Sampling Report
HQ	hazard quotient
HQ_L	HQ calculated using a lowest-observed-adverse-effects level
HQ_N	HQ calculated using a no-observed-adverse-effects level

I-10	Interstate Highway 10
Integral	Integral Consulting Inc.
IPC	International Paper Company
LOAEL	lowest-observed-adverse-effect level
lw	lipid weight
MIMC	McGinnes Industrial Maintenance Corporation
MWW	Mann Whitney Wilcoxon
NOAEC	no-observed-adverse-effects concentration
NOAEL	no-observed-adverse-effects level
OCDD	octachlorinated dibenzo-*p*-dioxin
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-*p*-dioxin
PCDF	polychlorinated dibenzofuran
ppt	parts per thousand
PSCR	Preliminary Site Characterization Report
QA	quality assurance
RACR	Removal Action Completion Report
RBA	relative bioavailability adjustment
REV	reference envelope value
RI/FS	Remedial Investigation and Feasibility Study
RM	reasonable maximum
RMin	reasonable minimum
SAP	Sampling and Analysis Plan
Site	San Jacinto River Waste Pits site in Harris County, Texas
SJRWP	San Jacinto River Waste Pits
SLERA	Screening Level Ecological Risk Assessment
SQG	sediment quality guideline
SSD	species sensitivity distribution
SWAC	surface area-weighted average concentration
TCDD	tetrachlorinated dibenzo-*p*-dioxin
TCDF	tetrachlorinated dibenzofuran
TCEQ	Texas Commission on Environmental Quality

TCRA	time-critical removal action
TEF	toxic equivalency factor
TEQ	toxicity equivalent
TEQ_{DF}	TEQ concentrations calculated using only dioxins and furans
TEQ_{DFP}	TEQ concentrations calculated using dioxins and furans and dioxin-like PCBs
TEQ_{P}	TEQ concentrations calculated using only dioxin-like PCBs are referred to as TEQ_{P}
TMDL	total maximum daily load
TOC	total organic carbon
TRV	toxicity reference value
UAO	Unilateral Administrative Order
UCL	upper confidence limit on the mean
USEPA	U.S. Environmental Protection Agency
ww	wet weight

1 INTRODUCTION

This baseline ecological risk assessment (BERA) was prepared on behalf of International Paper Company (IPC) and McGinnes Industrial Maintenance Corporation (MIMC; collectively referred to as the Respondents) in fulfillment of the 2009 Unilateral Administrative Order (2009 UAO), Docket No. 06-03-10, issued by the U.S. Environmental Protection Agency (USEPA) to IPC and MIMC on November 20, 2009 (USEPA 2009), for the San Jacinto River Waste Pits (SJRWP) site in Harris County, Texas (the Site). The 2009 UAO directs the Respondents to perform a Remedial Investigation and Feasibility Study (RI/FS) for the Site, and indicates that the RI include a BERA. This document fulfills the UAO requirement for the BERA, building on the conceptual site models (CSMs) described in the Preliminary Site Characterization Report (PSCR) for the impoundments north of Interstate Highway 10 (I-10) and surrounding aquatic environments (Figure 1-1).

A Screening Level Ecological Risk Assessment (SLERA) for the overall Site was presented as Appendix B to the RI/FS Work Plan (Anchor QEA and Integral 2010). That SLERA did not address the south impoundment, because it was written prior to USEPA's requirement that the south impoundment undergo investigation. In March 2011, soil samples were collected from the south impoundment area and analyzed for chemicals of interest (COIs). The resulting data have been used to perform a SLERA for the south impoundment, which is included as Appendix E to this document. USEPA has requested that additional studies be conducted with respect to the south impoundment area. This document presents a SLERA for the south impoundment in Appendix E to provide the screening-level problem formulation and the selection of receptors and assessment endpoints. Appendix E also includes analysis of the soil data collected in 2011 and identification of chemicals of potential ecological concern (COPCEs) for ecological receptors that may use that area. Following USEPA approval of this draft south impoundment SLERA and completion of the investigation of that part of the Site, a BERA for the south impoundment will be prepared. It will be presented in the Remedial Investigation Report.

1.1 Purpose

The Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA) guidance requires that remedies at contaminated sites be protective of human health and the environment (USEPA 1988). The baseline risk assessments evaluate the potential threats to human health and the environment in the absence of any remedial action, help determine whether remedial action is needed, and serve as the basis for the evaluation of the effectiveness of any subsequent remedial action. Ecological risk assessment addresses the likelihood that adverse effects on the environment, and to specific ecological receptors, may occur or are occurring as a result of exposure to one or more stressors (USEPA 1997).

The purpose of this BERA is to determine the nature and magnitude of risks to ecological receptors that result from any releases of hazardous substances from the impoundments north of I-10 at the Site. Results of the baseline risk assessments support risk managers by providing a point of reference for evaluation of the no-action alternative in the feasibility study, and for quantification of risk reduction that can be achieved by each remedial alternative considered in the feasibility study.

1.2 Document Organization

The approaches and methodologies presented in this BERA are consistent with USEPA guidance for conducting ecological risk assessments (USEPA 1997, 1998), and with Data Quality Objectives (DQOs) and related statements and information presented by the sediment, tissue, and soil sampling and analysis plans (SAPs) (Integral and Anchor QEA 2010; Integral 2010a, 2011a), and the RI/FS Work Plan (Anchor QEA and Integral 2010a). The document is organized according to specifications in the *Guidelines for Ecological Risk Assessment* (USEPA 1998), and includes the following:

- Section 2. Background Information
- Section 3. Problem Formulation
- Section 4. Exposure Assessment
- Section 5. Effects Characterization
- Section 6. Risk Characterization
- Section 7. Uncertainty Analysis
- Section 8. Summary of Ecological Risks and Risk Conclusions.

This document also includes six Appendices:

- Appendix A Receptor Profiles
- Appendix B Ecotoxicity Profiles
- Appendix C Exposure Point Concentrations Used for Exposure Assessment in the BERA
- Appendix D Estimation of Dioxin and Furan Concentrations in Terrestrial Invertebrate Tissue for the Exposure Model
- Appendix E Screening-Level Ecological Risk Assessment, South Impoundment
- Appendix F EPA Comments Relating to the Draft Baseline Ecological Risk Assessment (BERA) Dated March 15, 2012, and Responses, and Draft-Final BERA Dated August 2012, and Responses.

2 BACKGROUND INFORMATION

This BERA is presented to USEPA following completion of several studies and documents and provides a key component of the analyses required for the RI Report. Relevant background information on the Site setting and CSMs, and information supporting determination of the baseline dataset have been described previously. This section briefly reviews information relevant to the BERA that has been presented in earlier, approved documents. The problem formulation is presented subsequently in this context.

2.1 Site Setting and General Conceptual Site Models

The Site setting was described in the RI/FS Work Plan (Anchor QEA and Integral 2010) and later updated in the PSCR (Integral and Anchor QEA 2012). The PSCR provides a detailed description of the topography, hydrology, hydrogeology, and hydrodynamic environment at the Site. The draft Chemical Fate and Transport Modeling Report (Anchor QEA 2012) provides additional detail on the hydrodynamics and sediment physical environment, as well the fate of 2,3,7,8-tetrachlorinated dibenzo-*p*-dioxin (TCDD), 2,3,7,8-tetrachlorinated dibenzofuran (TCDF), and octachlorinated dibenzo-*p*-dioxin (OCDD).

Also described in the PSCR are two CSMs that provide the basis for the ecological exposure and risk analyses. These most recent iterations of the CSMs form the conceptual framework of chemical transport and exposure pathways that could lead to exposure of ecological receptors. Existing CSMs describe the environment of the northern and southern impoundments in the following general context:

- **The area north of I-10 and surrounding aquatic environment.** This area consists of a set of impoundments approximately 14 acres in size, built in the mid-1960s for disposal of paper mill wastes, and the surrounding areas containing sediments and soils potentially contaminated with chemicals originating in the waste materials that had been disposed of in the impoundments. The set of impoundments is located on a partially submerged 20-acre parcel on the western bank of the San Jacinto River, immediately north of the I-10 bridge (Figure 2-1). Dredging activities by third parties have occurred in the vicinity of the perimeter berm at the northwest corner of these impoundments; samples of sediment in nearby waters north and west of these impoundments indicate that dioxins and furans are present in nearby sediments.

Other sources of dioxins and furans are present upstream and on the Site, including chemical manufacturing facility outfalls, wastewater treatment plant outfalls, stormwater runoff and outfalls, and atmospheric deposition (Integral and Anchor QEA 2012). The Baytown West Wastewater Treatment Plan outfall occurs directly north of the I-10 bridge on the river's eastern shore. University of Houston and Parsons (2006) presents information on dioxins and furans in effluent from this wastewater treatment plant. Also on the eastern shore and to the north is a stormwater outfall draining a very large area, and atmospheric sources of dioxins and furans are present on the Site as well. Section 4.2.1 of the PSCR provides additional detail. The CSM that provides a summary of the chemical sources and the release and transport pathways is depicted in Figure 1-1.

- **The impoundment south of I-10.** Another impoundment may be present south of I–10, on the peninsula of land south of the 20-acre parcel. Portions of the peninsula are believed to have been used in the 1960s as a disposal area for paper mill waste similar to that disposed of in the impoundments north of I-10. Currently available information about the area south of I-10 indicates that wastes other than those originating from the Champion Papers Inc. paper mill were also deposited in the impoundment (Integral and Anchor QEA 2012), but the origins of the other waste and debris in that area are unknown. The CSM for the south impoundment primarily addresses the terrestrial environment (Figure 2-2); USEPA has requested additional studies to address data gaps and identify materials present in the impoundments south of I-10.

Finally, since this Site was added to the National Priorities List in 2008, a time-critical removal action (TCRA) has been implemented. Construction of the TCRA, which involved installation of a cap over the area within the original perimeter of the impoundments north of I-10, was completed in July 2011. The TCRA is relevant to the BERA because it has substantially changed ecological conditions and exposure pathways at the Site (Figure 2-3), reducing the potential for exposure of ecological receptors to the contaminated waste and sediment present on the Site. Details describing implementation of the TCRA can be found in the draft Removal Action Completion Report (RACR) (Anchor QEA 2011a).

Problem formulation (Section 3) integrates available information to describe specific pathways and exposure routes of interest to ecological receptors in the area north of I-10 and aquatic environment, building from the CSM of that area described above. The CSM describing ecological exposures, and from which the analysis steps in the BERA are determined, are detailed in Section 3.8. The south impoundment SLERA, including a screening level problem formulation and a CSM for ecological receptors, is presented in Appendix E.

2.2 Baseline Risk Assessment Datasets and Data Treatment Rules

Determination of an appropriate baseline dataset, which will be used to describe the current site conditions, is a key step of the RI/FS process. Once the appropriate data are identified, calculations are performed using a specified set of data treatment rules.

2.2.1 Baseline Dataset

The RI/FS Work Plan describes the rationale for selection of data to be used in the baseline risk assessments: data to be used in the baseline risk assessments should be of known quality, which includes only Category 1 data (as described in Section 3 of the RI/FS Work Plan) and should reflect the current, pre-remediation condition, which does not include conditions present in 2005 or previously (Integral 2011c). The Exposure Assessment Memorandum (Integral 2012) describes the process for incorporating additional data for polychlorinated biphenyl (PCB) congeners in catfish fillet and sediment collected on Site by the Texas Commission on Environmental Quality's (TCEQ) Total Maximum Daily Load (TMDL) program in 2008 and 2009 (University of Houston and Parsons 2009; Koenig 2010, Pers. Comm.). Appendix A to the Exposure Assessment Memorandum (Integral 2012) documents Integral Consulting Inc.'s (Integral's) independent validation of TCEQ's PCB congener data for tissue and sediment on the Site and for tissues in background areas according to procedures described by the RI/FS Work Plan. This validation effort resulted in a change to the classification of these PCB data from Category 2 to Category 1.

Both Site and background data are used in the risk assessment. Analysis of background information allows for consideration of other sources of risk at the Site, which is relevant to both risk assessment and evaluation of remedial alternatives. Background conditions provide the basis for understanding the incremental risks due to the site (Section 3.8.4.5). Such context informs risk management by ensuring that remedial actions that may be taken at the Site will actually result in reduction of exposure and risk originating from Site-related sources.

The baseline dataset for the Site consists of:

- Sediment, tissue, and soil data collected for the RI/FS (all new data collected by respondents since December 2009), including soil from the south impoundment collected in 2012[1]
- Sediment and surface water data collected by URS (2010) for TCEQ in 2009.
- PCB congener data for fish tissue and sediments resulting from sampling conducted by TCEQ in 2008 and 2009.

The background dataset developed for the RI consists of:

- Upstream surface (0- to 6-inch) sediment samples for 21 subtidal and 10 intertidal locations
- Background soil samples (0- to 6-inch and 6- to 12-inch) from 10 locations in the I-10 Beltway 8 East Green Space and from 10 locations in Burnet Park
- Clam and killifish from two locations upstream and hardhead catfish and blue crab from locations in Cedar Bayou.[2]

Background tissue and soil data were collected prior to publication of the PSCR (Integral and Anchor QEA 2012) and are described in that document; additional sediment samples were collected upstream in 2011 and will be described in the RI Report.

Although the ecological risk assessment uses different data types and uses data differently than the human health risk assessment, the baseline dataset described above is comprehensive for both of the baseline risk assessments to be conducted for the RI.

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[1] Sampling is documented in Addendum 3 to the Soil SAP for additional soil sampling south of I-10 (Integral 2011d); Addendum 2 to the Sediment SAP (Integral 2011b) and Addendum 1 to the Groundwater SAP (Anchor QEA 2011b).

[2] Background tissue data have also been collected for edible crab and catfish south of the Fred Hartman bridge; these data are for use in the human health risk assessment only.

2.2.2 Data Treatment Rules

RI/FS data are managed according to the project Data Management Plan (DMP), which is provided as Appendix A to the RI/FS Work Plan. Section 6.5 of the DMP also describes data averaging rules such as the averaging of results for replicates and treatment of qualified data. Data accessed for analyses in this report were prepared according to those rules. For performance of various analyses in this report, general data treatment rules are as follows:

- Nondetects were estimated at one-half the detection limit for use in all calculations, unless otherwise specified.
- TCDD toxicity equivalent (TEQ) concentrations were calculated using the toxicity equivalency factors (TEFs) most appropriate for the receptor being analyzed. These are discussed further in Section 3.2.
- TEQ concentrations in samples for which one or more dioxin and furan congeners were not detected were calculated using nondetects equal to one-half the detection limit. TEQ concentrations for PCB congeners for which one or more PCB congeners was not detected were calculated using nondetects equal to one-half the detection limit. If one or more congener concentration was estimated in calculation of a TEQ concentration, the TEQ is reported as estimated (J-qualified) in the database. If all congeners were not detected in a sample, the TEQ is reported as not detected (U–qualified).
- Any nondetects for a given analyte and medium that were higher than the maximum detected concentration for the same analyte and medium were considered "high-biasing non-detects," and were removed prior to use of the dataset in the BERA, following USEPA (1989) guidance.

In the calculation of exposure point concentrations (EPCs), and in statistical evaluations of the datasets (e.g., characterization of data distributions), specific rules were applied for estimating values for the censored data. Procedures for substituting values for censored data varied, depending on the sample size and the detection frequency, as follows:

- For each dataset used in calculation of an EPC or in evaluating the data distribution, the detection frequency was calculated as the percentage of values flagged with a "U" qualifier (not detected).

- Nondetects in datasets with sample sizes equal to or greater than 10 and detection frequencies equal to or greater than 50 percent were set to one-half the detection limit and included in all calculations.
- Datasets with sample sizes equal to or greater than 10 and detection frequencies between 20 and 50 percent were addressed using statistical substitution methods. The substitution method used depends on the distribution of the dataset; for normally or lognormally distributed data, upper confidence limits on the mean (UCLs) were estimated using robust regression on order statistics (Helsel 2005); for datasets with unknown data distributions (those that could not be defined as normal or lognormal), a nonparametric Kaplan Meier approach for imputing nondetects was used (Helsel 2005; Singh et al. 2006).
- Nondetects in datasets with sample sizes less than 10, regardless of detection frequency, or in datasets with detection frequencies less than 20 percent, regardless of sample size, were not subject to statistically derived substitutions, because the pool from which information about the data distribution can be drawn is insufficient for robust substitution methods. These datasets were treated with nondetects substituted at one-half the detection limit.

Finally, the data to describe PCBs in the media sampled on Site is variable. In sediment, dioxin-like PCBs were measured in a subset of the samples collected to describe nature and extent of contamination. Within the northern impoundments, samples were collected and analyzed for Aroclors, and elevated detection limits resulted from matrix interferences in several samples from the western cell. In soils, either PCB congener or Aroclor data are available, and in some samples, PCBs were not analyzed. Finally, in tissue, all 209 PCB congeners were measured in all samples. Data treatment rules for calculation of aggregate variables for PCBs (total PCBs and TEQ_P) were consistent with those laid out above. For total PCBs as a sum of Aroclors, this approach is likely to overestimate total PCB concentrations because of the inflated detection limits in some sediment samples.

3 PROBLEM FORMULATION

The problem formulation for the BERA provides a synthesis of ecological conditions, information on fate and transport and relevant toxicological information available at the start of the risk assessment process to finalize the assessment endpoints and risk questions to be addressed by the BERA (USEPA 1998). Information contained in or generated by earlier SJRWP documents prepared under the RI/FS is assembled in this section to provide a complete problem formulation, which results in definition of the approaches and methods used to perform the BERA. Specifically, this section draws on the following previously approved documents:

- The RI/FS Work Plan (Anchor QEA and Integral 2010), which describes the methods and approach to be used to perform the BERA, and provides the SLERA for the area north of I-10 and aquatic environment as an appendix. The SLERA identifies the species potentially present at the Site, including threatened and endangered species, specifies ecological receptors to serve as surrogates or representatives for the species potentially present, and provides a preliminary summary of ecotoxicological information for dioxins and furans.
- The Sediment SAP and the COPC [Chemical of Potential Concern] Technical Memorandum (Integral 2011c), both of which address selection of COPCES. Components of the Sediment SAP addressing selection of COPCs were excerpted and included as Appendix C of the RI/FS Work Plan.
- Technical Memorandum on Bioaccumulation Modeling (Integral 2010b), which addresses patterns of dioxin and furan bioaccumulation in invertebrates, fish, and birds using scientific literature and analyses of site and regional data. This technical memorandum includes results of statistical analyses that may be used to predict tissue concentrations of dioxins and furans from sediment or water concentrations, and an analysis of dioxin and furan bioaccumulation patterns in the area surrounding the Site.
- The PSCR (Integral and Anchor QEA 2012) for the most recent iteration of the CSMs. The latest CSMs were developed from the initial CSMs developed in the RI/FS Work Plan and Soil SAP Addendum 1. Although CSMs are introduced above, the problem formulation describes the final evaluation of exposure pathways relevant to the ecological risk assessment.

This BERA incorporates information from the entire area within USEPA's preliminary Site perimeter, and information from all of these documents as well as other publications. Dioxins and furans were identified in Appendix C of the RI/FS Work Plan as indicator chemicals for the purposes of the remedial investigation. This designation acknowledges that the relatively high toxicity of dioxins and furans in combination with the relatively elevated concentrations make dioxins and furans the focus of risk evaluation and risk management. For this reason, this BERA often incorporates more detail and depth for the evaluation of risks due to dioxins and furans, while using a generally conservative approach to address risks from the other COPC$_E$s.

3.1 Chemicals of Potential Ecological Concern

COPC$_E$s for the area north of I-10 and surrounding aquatic environments (Table 3-1) were determined using the RI dataset according to methods specified in the COPC Technical Memorandum, and are subdivided into those of potential concern to benthic invertebrates and those of concern to fish and wildlife. Chemicals in sediment with a detection frequency of at least 5 percent (following the sediment study) that were either a) present in at least one sample at a concentration greater than sediment screening concentrations protective of benthic invertebrate communities, or b) have no screening value protective of benthic invertebrate communities and c) were not correlated with dioxins and furans, are considered COPC$_E$s for benthic macroinvertebrate communities. If a chemical was detected in greater than 5 percent of sediment samples in the RI dataset, and is thought to be bioaccumulative (TCEQ 2006), it is considered to be a COPC$_E$ to be evaluated for fish and wildlife.

3.2 Overview of Ecological Effects

All of the COPC$_E$s have some potential to adversely affect the survival, growth, and/or reproduction of one or more ecological receptors if exposures are sufficiently elevated. Information about the types of effects associated with each COPC$_E$ in various species, and the information used to interpret exposure estimates for ecological receptors in the BERA are provided in Appendix B.

As the indicator chemical group, dioxins and furans are expected to be the most important ecological risk driver at the Site. In this context, the following section explains how the toxicity of dioxins, furans, and "dioxin-like" PCBs are assessed in this BERA, and provides an overview of the potential biological and ecological effects of all of the dioxin-like compounds in broad categories of ecological receptors. The approach used in this document is consistent with USEPA's (2008) Framework for Application of the Toxicity Equivalence Methodology for Polychlorinated Dioxins, Furans, and Biphenyls in Ecological Risk Assessment. Much of this information has been presented in prior submittals (Appendix B to the RI/FS Work Plan; Exposure Assessment Memorandum). More detailed discussion of the toxicity reference values and benchmarks used to interpret exposure estimates for dioxins and furans as well as the other COPCES is provided in Appendix B of this document.

3.2.1 Evaluating Exposure and Toxicity of "Dioxin-Like" Compounds

For each of 17 dioxin and furan congeners with chlorine substitutions in the 2,3,7,8-positions of the molecule, toxicity to fish, birds, and mammals is widely believed to occur through a common biochemical mechanism, one that is initiated by the binding of the congener to the aryl hydrocarbon receptor (AhR). Of the 17 AhR-active congeners, 2,3,7,8-TCDD exhibits the greatest potential for binding with AhR in many assays. The common toxicological mechanism among the 17 congeners, and the availability of a single potency index (2,3,7,8-TCDD potency) provides the basis for calculating the cumulative exposure to all AhR-active congeners for the purposes of evaluating toxicity and establishing thresholds of toxicological effects. The magnitude of toxicity of each of these 17 dioxin and furan congeners can be related to the toxicity of 2,3,7,8-TCDD using a congener-specific TEF. The concentration of each congener is converted to equivalent concentrations of 2,3,7,8-TCDD by multiplication with its TEF, and all the TEQs for individual congeners (the product of each congener and its TEF) are added to compute the total toxic equivalency of the mixture to 2,3,7,8-TCDD. The resulting total TEQ concentration provides the metric of exposure to "dioxin-like" compounds. Separate sets of TEFs have been derived for mammals, birds, and fish, and are provided in Table 3-2.

The toxic equivalency approach was first developed in 1977 for screening risks from dioxins and furans in combustion sources and incinerator emissions (Eadon et al. 1986; Erickson

1997). It was first used as an “interim” screening tool to evaluate the toxicity of mixtures of dioxins and furans. In 1989, USEPA stated that the TEQ approach “remains ‘interim’ in character and should be replaced as soon as practicable with a bioassay method” (USEPA 1989). The toxicological basis and rationale for the use of the TEF approach is described in Van den Berg et al. (1998; 2006). Guidance for the use of this methodology in ecological risk assessment is provided by USEPA (2008). This guidance explicitly recognizes that, due to interspecies variability in biochemistry and sensitivity to dioxin toxicity, the TEFs in Table 3-2 may be substituted with species-specific TEF schemes in ecological risk assessment, if sufficient rationale is provided. Although this document does not propose alternative TEF schemes, USEPA (2008) guidance highlights the uncertainty of using the TEF methodology across a wide range of species.

The application of the TEQ approach to PCB congeners was introduced in 1991. For some species and some types of toxicological endpoints, 12 of the 209 PCB congeners are considered to have dioxin-like toxicity, and as a result, these PCB congeners are considered to be additive with TEQs calculated from dioxins and furans (Safe 1990). TEFs for these 12 PCB congeners were assigned on the basis of a variety of endpoints demonstrated by *in vitro* assays and *in vivo* animal studies, most of which are noncancer endpoints (Van den Berg et al. 1998). Concentrations of the dioxin-like PCB congeners within a PCB mixture are first converted to TEQ concentrations using various TEFs. Once the TEQs have been calculated for each dioxin-like congener, they can be added to TEQs for dioxin and furan congeners to determine a total TEQ concentration.

3.2.2 TEQ Nomenclature

Toxicity equivalents are calculated and presented in several different ways for ecological risk assessment. To simplify presentation of these concepts, the term “TEQ” is qualified using subscripts to indicate the congeners included in its calculation, and the TEF scheme applied.

- TEQ concentrations calculated using only dioxins and furans are referred to as TEQ_{DF}
- TEQ concentrations calculated using only dioxin-like PCBs are referred to as TEQ_{P}
- TEQ concentrations calculated using dioxins and furans and dioxin-like PCBs are referred to as TEQ_{DFP}.

To specify the TEF scheme used, an additional subscript is applied, including "F" for the use of TEFs for fish; "M" for the use of TEFs for mammals; and "B" for the use of TEFs for birds. For example, using this notation scheme, the following indicates a TEQ calculated for dioxins, furans, and dioxin-like PCBs using TEFs for fish: $TEQ_{DFP,F}$. If the term "TEQ" is used with no subscript notation, it is to reference a general concept and not a specific concentration. TEFs for fish and birds are taken from Van den Berg et al. (1998); TEFs for mammals are taken from Van den Berg et al. (2006).

3.2.3 Mechanisms of Toxicity

In vertebrates, interactions of 2,3,7,8-TCDD and the dioxin-like compounds with AhR leads to alterations in gene expression and signal transduction that are believed to be the biochemical determinants of toxic effects (Birnbaum 1994). AhR is a member of a family of transcription factors that includes aryl hydrocarbon receptor nuclear translocators (e.g., ARNT, ARNT2, and ARNT3) and others. These proteins are involved in the sensation of and adaptation to changing environmental and developmental conditions. Once activated, AhR combines with ARNT and moves into the cell nucleus, where the complex can then bind specific DNA sequences, leading to altered gene expression. A role of the ligand-AhR complex in non-nuclear signal transduction has also been proposed. The functional consequences of AhR activation in fish and wildlife are diverse and involve numerous target organs, including the liver, thyroid, heart, immune system, and reproductive system (Fox 2001; Carney et al. 2006). Although AhR homologues have been identified in various invertebrate species, invertebrate AhR homologues lack specific, high-affinity binding for TCDD and other prototypical AhR ligands (Hahn et al. 1992; Butler et al. 2001).

There is also potential for non-AhR-mediated dioxin and furan toxicity in both vertebrates and invertebrates, but at much higher doses (USEPA 2008). Non-AhR-mediated dioxin and furan toxicity to vertebrates is not addressed because AhR-mediated toxicity is expected to occur at lower exposures. For invertebrates, non-AhR-mediated toxicity is addressed where data are available.

3.2.4 Overview of Toxicity to Ecological Receptors

From an ecological risk perspective, adverse effects of dioxins and furans on reproductive success, growth, and survival are relevant to evaluating the potential for population-level effects in any receptor. A range of reproductive and developmental effects such as reduced fertility, early-stage embryotoxicity, early life-stage mortality, and reduced growth and development of offspring are also relevant reproductive effects, because these effects can conceivably affect the growth or viability of a population. Studies have shown substantial inter-species and inter-taxa differences in susceptibility to the adverse effects of dioxins and furans, which presents a challenge for interpreting risks to species that have not been tested.

Below is a summary of information on the toxicity of dioxin-like compounds to broad categories of ecological receptors. This information was presented in greater detail in Appendix B of the RI/FS Work Plan (Anchor QEA and Integral 2010). Detailed information on the potential effects of $COPC_{ES}$ needed to support this risk assessment is presented in Appendix B.

3.2.4.1 Benthic Macroinvertebrates

The literature on the toxicity of dioxins and furans to aquatic invertebrates is less extensive than for fish, birds, and mammals, and the majority of studies were published more than a decade ago (USEPA 2008; Anchor QEA and Integral 2010). Most of these historical studies have found that aquatic invertebrates are relatively insensitive to dioxin toxicity. Studies summarized in Appendix B include tests on crustaceans, molluscs, insects, oligochaetes, and polychaetes from freshwater, estuarine, and marine environments. Exposure routes tested among these studies include ingestion and direct exposure to contaminated water and contaminated sediment, and several studies note whole body concentrations in animals exhibiting no adverse responses. While the full spectra of possible exposures and invertebrate taxa are not represented by the data, the evidence generally indicates that invertebrates can tolerate relatively high exposures to TCDD, and in some cases, to TCDF as well. Other 2,3,7,8-substituted congeners are not as well studied.

Recent studies have found that bivalve molluscs exhibit reproductive and developmental effects in response to exposure to 2,3,7,8-TCDD (Cooper and Wintermyer 2009) at exposures lower than no-effects levels for other tested species. The mechanism by which 2,3,7,8-TCDD affects bivalve molluscs has yet to be identified with certainty, but researchers agree that it is independent of AhR homologues (Cooper and Wintermyer 2009; Wintermyer and Cooper 2007; Butler et al. 2004). It is possible that other kinds of invertebrates may exhibit reproductive and developmental effects following exposure to 2,3,7,8-TCDD, or other dioxins and furans; most historical studies have evaluated only survival or growth in adult organisms, following exposures only to 2,3,7,8-TCDD. The results of these studies with bivalves will be used to interpret Site-specific invertebrate tissue data.

3.2.4.2 Fish

Substantial literature indicates that dioxin toxicity is mediated through AhR in fish, with some species having more than one receptor type of varying functionality. The period of greatest sensitivity to dioxins of many fish species is the embryo and post-hatch stages, with toxicity manifesting at the lower exposure concentrations as edema, with circulatory and metabolic changes leading to secondary effects. Cartilaginous malformations and growth effects also occur but may be less sensitive endpoints than edema and cardiac effects. Reproductive effects have been shown to occur in the range of thousands of nanograms per kilogram of tissue. Sublethal effects on juveniles and adults are less well studied; however, the literature reviewed suggests that these later life stages are not as sensitive to dioxin toxicity as are early life stages. The literature suggests that population resistance to dioxin toxicity can also occur over time in some fish, as shown for a killifish population living in the vicinity of a Superfund site with high dioxin levels (Nacci et al. 2002).

Within species, many dose response curves are steep, reflecting a relatively narrow range within which toxicity can manifest. However, there is a high level of variability in sensitivity to dioxins among fish species, with effects associated with concentrations ranging over two orders of magnitude. This variability argues for the development of a species sensitivity distribution (SSD) for evaluating effect levels relevant for risk assessment. Expression of dioxin and furan exposures as concentrations in egg tissue (a tissue residue-based effect level) is an appropriate basis on which to express exposure. Effects in early life stage fish appear to be relatively independent of the route of exposure (Steevens et al. 2005) such that studies using a variety of methods of egg exposure (e.g., water, egg injection) are appropriate for effects evaluation. Steevens et al. (2005) derived two SSDs providing excellent representation of the early life stage toxicity of 2,3,7,8-TCDD to fish.

Population-level risks may be mediated by interspecies and interpopulation differences in sensitivity to dioxins, as well as differential sensitivity to different types of dioxins and furans. Sensitivity to dioxins is not necessarily static within species; some fish populations with long-term exposure to contaminated conditions have apparently developed a resistance to toxicity of dioxins and PCBs. At a Superfund site in New Bedford Harbor, Massachusetts, with high concentrations of dioxins, PCBs, and other hazardous chemicals, the resident *Fundulus heteroclitus* (killifish) population has been shown to exhibit heritable resistance to PCBs in terms of reduced mortality relative to populations from uncontaminated reference sites (Nacci et al. 2002).

3.2.4.3 Reptiles

Generally, the literature describing potential effects of environmental toxicants to reptiles is poor, and available data are dominated by studies on turtles, with a paucity of information on lizards and snakes (Sparling et al. 2000b). Portelli and Bishop (2000) performed a review of the literature for organic chemicals other than pesticides. They found no reports of reptiles dying as a result of PCB, dioxin, or furan exposure, despite fairly elevated concentrations in tissues of specimens captured in the wild. Bishop et al. (1991) reported developmental abnormalities (e.g., abnormal eyes, claws and bills) and behavioral abnormalities in turtles exposed to dioxins, furans, PCBs, and organochlorine pesticides, but dose-response relationships have not been reported. This and other studies cited by Portelli and Bishop (2000) suggest correlations between concentrations of PCBs, polychlorinated dibenzo-*p*-dioxins (PCDDs), and polychlorinated dibenzofuran (PCDFs) and abnormalities in developing embryos, but these data are confounded by the presence of pesticides and other chemicals in the environment and tissues of organisms studied.

One recent laboratory study (Hecker et al. 2006) induced ethoxyresorufin-*O*-deethylase (EROD) activity in hepatocytes from the African brown snake (*Lamprophis fuliginosus*) by *in vitro* exposure to TCDD and PCB126, but dose-dependent EROD activity was not induced by two other dioxin-like PCB congeners. Portelli and Bishop (2000) note that there is no correlation between dioxin, furan, and PCBs in eggs and incidence of abnormalities when TEQ was used to characterize exposure, regardless of the TEF scheme used. More information is needed to understand the extent of both potential AhR-mediated toxicity and other toxicity of dioxins and dioxin-like compounds in reptiles.

3.2.4.4 Birds

Exposure to dioxins and furans is associated with adverse effects on bird reproduction and development. Changes in the heart have also been observed, although in some cases the impact of these effects on ultimate reproductive success or survival is unclear. Early life stages, including the embryo and recently hatched chick, appear to be the most sensitive to dioxin toxicity, and an overview of the literature appears to confirm USEPA's (2003b, 2008) position that egg exposure is an appropriate basis for predicting effects. The literature indicates there are substantial differences in susceptibility and sensitivity among species, even within the limits of reproductive toxicity. Among tested bird species, sensitivity to early life stage toxicity spans several orders of magnitude. As for fish (above), the similarity in the ranges of the sublethal and lethal effect concentrations reflects the steep dose-response associated with dioxin-like toxicity.

3.2.4.5 Mammals

Exposure to dioxins and furans is associated with adverse effects on mammalian reproduction and development, more so for the rat than for the mink. Early life stages, including the fetus and newly born pup/kit, appear to be the most sensitive to dioxin toxicity. Similar to birds, there is substantial inter-species variability in sensitivity to dioxins and furans.

Concentrations of dioxins and furans are commonly measured in liver or adipose tissue, because lipophilic compounds such as TCDD may accumulate in these tissues. Moreover, due to toxicokinetic differences between species, administered dose or content of compounds in foods is not as reliable an indicator of exposure to dioxins and furans as organ concentrations. Whole-body or tissue burden is the preferred metric of exposure in laboratory studies and may facilitate inter-species and inter-study comparisons. However, reliable literature on mink expresses exposure as ingested dose (e.g., Zwiernik et al. 2009), providing a useful, non-invasive means of evaluating exposure-response in a risk assessment context.

3.3 Fate and Transport of Chemicals of Potential Ecological Concern

Fate and transport processes relevant to the BERA include uptake and bioaccumulation, biomagnification, degradation and weathering of compounds, and the sequestration of chemicals in environmental matrices such as soils or sediments. The term "bioaccumulation" describes the extent to which an organism retains substances following uptake through any exposure route, resulting in the organism having a higher concentration in its tissues than in the surrounding environment (USGS 2007). "Biomagnification" occurs if concentrations are increasingly greater in higher trophic levels (USGS Toxic Substances Hydrology Program).

For the BERA, Site-specific data describing concentrations of COPCEs in tissues of some organisms are used to estimate the ingestion rate of COPCEs by birds and mammals, and in some cases, are directly compared to toxicity reference values (TRVs) to evaluate the potential for effects. With the exception of the Technical Memorandum on Bioaccumulation Modeling, which reviews patterns of dioxin and furan bioaccumulation in aquatic macroinvertebrates, fish and birds, bioaccumulation has been addressed by following TCEQ guidance; specifically TCEQ's list of chemicals considered bioaccumulative (TCEQ 2006). This list of bioaccumulative chemicals was specifically consulted in selection of COPCs for fish, reptiles, birds, and mammals, because these receptor groups are likely to be significantly exposed to bioaccumulative contaminants through ingestion of prey. Chemicals that are COPCEs (in addition to dioxins and furans) and listed by TCEQ as bioaccumulative from sediments include PCBs, cadmium, copper, mercury, nickel, and zinc. Site-specific data for fish, clam, and crab tissues provide empirical evidence of the bioaccumulation potential of each chemical, and are used as appropriate to evaluate species-specific exposures in the BERA.

The Technical Memorandum on Bioaccumulation Modeling (Integral 2010b) uses site specific data, regional data, and the literature to describe controls on bioaccumulation of dioxins and furans, and resulting bioaccumulation patterns. The technical memorandum finds several lines of evidence indicating that 1) rates of uptake of dioxin and furan congeners by both vertebrates and invertebrates for which data are available are variable (e.g., Tietge et al. 1998), and are controlled to a large extent by the size of the molecule, with the smaller, lower-chlorinated congeners taken up more readily across gill and gut membranes than the larger, more chlorinated congeners (Opperhuizen and Sijm 1990); 2) dioxins and furans can be metabolized and excreted, and this also occurs at different rates for different congeners (Hu and Bunce 1999; Nichols et al. 1998); 3) metabolism results in generation of soluble moieties which can be excreted, and does not occur by dehalogenation, except in bacteria (Hu and Bunce 1999); 4) elimination rates of tetrachlorinated congeners are lower than the more chlorinated congeners (e.g., Niimi 1996); and 5) dioxins and furans do not biomagnify, unlike PCBs which do biomagnify (Naito et al. 2003; Wan et al. 2005; Broman et al. 1992; and Jarman et al. 1997).

The Technical Memorandum on Bioaccumulation Modeling (Integral 2010b) also reports Site-specific statistical regression models that can be used to predict tissue concentrations for some congeners with a measurable degree of uncertainty. Both the conceptual model of bioaccumulation reported by the technical memorandum, and the regression models reported are used in the baseline risk assessments. Analysis of Site-specific tissue data in the PSCR (Integral and Anchor QEA 2012) supported the conceptual model of bioaccumulation developed by Integral (2010b).

3.4 Ecosystems Potentially at Risk

The Site is located in a low gradient, tidal estuary near the confluence of the San Jacinto River and the Houston Ship Channel. The surrounding area includes a mix of land uses, including two constructed reservoirs: Lynchburg Reservoir to the southeast and Lost Lake on the island in the center of the San Jacinto River west of Lynchburg Reservoir (Figure 3-1). Upland, riparian, and aquatic habitats are present.

3.4.1 Upland Habitats

Upland natural habitat adjacent to the San Jacinto River in the Site vicinity is generally low-lying, with little topographic variation, and consisting primarily of clay and sand that supports loblolly pine-sweetgum, loblolly pine-shortleaf pine, water oak-elm, pecan-elm, and willow oak-blackgum forest communities along the river's banks (TSHA 2009). Upland natural habitat occurs along narrow sections of land on either side of the river, as well as on several small islands, to the north and south of I-10 and east of the impoundments. Most of these islands are vegetated with a mixture of shrubs and trees, with fringing shallow waters. These habitats could support mammals, such as marsh rice rats and deer, that could migrate to the islands close to mainland areas, as well as passerines that could use the vegetated uplands for nesting and foraging, and shoreline birds such as sandpipers and herons that could wade and forage in the shallow areas adjacent to the islands.

Uplands on the western edge of the site north of I-10 are generally less densely developed than across the river along the Site's eastern border, which is developed with a mix of residential and commercial land uses (Figure 3-2). The I-10 freeway fragments the natural areas to the north and south of the highway, reducing the connectivity of these habitats. On the peninsula to the south of I-10, most of the upland habitat is zoned for commercial or industrial use, with the exception of a narrow segment of land on the western edge of the Site south of I-10 (Figure 3-2). The upland vegetation present on the southern peninsula is primarily low-lying grasses, with a few shrubs and trees adjacent to the shoreline.

3.4.2 Upland Wildlife

There is no site-specific data describing wildlife uses of the upland portions of the Site. Based on local wildlife lists and the types of habitat and land uses present at the Site, it is reasonable to expect a suite of generalist terrestrial species that are not highly specialized in their habitat requirements and are adapted to moderate levels of disturbance. The reptiles and amphibians that could occur in the vicinity of the Site include snakes, alligators, and turtles (Table 3-3). Avian taxa using upland habitats may include sparrows and other generalist passerines, starlings, pigeons and doves, corvids, and killdeer. Mammals expected in a semi-urban environment like the Site include small mammals (rodents), skunks, raccoons, coyotes, and opossums.

3.4.3 Aquatic and Riparian Habitats

Habitats on the northern portion of the Site include shallow and deep estuarine waters, and shoreline areas occupied by estuarine riparian vegetation. Because the Site is within an estuary, the salinity of the San Jacinto River in the vicinity of the Site can be low at times

(1 to 5 parts per thousand [ppt]; Clark et al. 1999); it was 2 to 12 ppt in a recent study (University of Houston and Parsons 2009). The in-water portion of the Site is unvegetated, with a deep (20- to 30-foot) central channel, and shallow (3 feet or less) sides (NOAA 1995; Clark et al. 1999). Except in the impoundments north of I-10, sediments are sandy and characterized by low organic matter content; most surface sediment samples collected within the northern impoundment ranged between 1 and 5 percent total organic carbon (TOC), and TOC in samples collected from within USEPA's preliminary Site perimeter but outside the impoundments was lower, with most samples between 0.5 and 2 percent TOC (Integral and Anchor QEA 2012) and having a high sand content. In surface sediment samples collected on the Site, the fraction consisting of sand ranged from 4 to 98 percent, with an average of about 50 percent sands.

3.4.3.1 Fish and Invertebrates

The tidal portions of the San Jacinto River and upper Galveston Bay provide rearing, spawning, and adult habitat for a variety of marine and estuarine fish (Table 3-4) and invertebrate species (Table 3-5). Species known to occur in the vicinity of the Site include clams and oysters, blue crab (*Callinectes sapidus*), black drum (*Pagonius cromis*), southern flounder (*Paralichthys lethostigma*), hardhead (*Ariopsis afelis*) and blue catfish (*Ictalurus furcatus*), spotted sea trout (*Cynoscion nebulosis*), and grass shrimp (*Paleomonetes pugio*) (Gardiner et al. 2008; Usenko et al. 2009).

3.4.3.2 Shorelines and Wetlands

A sandy intertidal zone is present along the shoreline throughout much of the Site (Figure 2–1). Minimal habitat is present in the upland sand separation area, as demolition and closure of this area created a denuded upland with a covering of crushed cement and sand. The sandy shoreline of the sand separation area is littered with rip-rap, metal debris, and piles of cement fragments. An estimated 34 acres of estuarine and marine wetlands are found within USEPA's preliminary Site perimeter (Figure 3-3). Throughout the broader surrounding area, there are approximately 55 additional acres of freshwater, estuarine, and marine wetlands (Figure 3-1).

A wetland delineation for areas of the Site to the north of I-10 completed in 2010 prior to implementation of the TCRA (BESI 2010) identified a large portion of the area within the 1966 northern impoundment perimeter above high water as emergent intertidal wetlands. In addition, some patchy areas with wetland characteristics were identified around the margin of the northern impoundments, most of which were narrow in width and a few hundred feet in length, including fringing wetlands between the open water of the San Jacinto River and upland portions of the Site, and emergent wetlands associated with roadside ditches north of I-10 (Figure 3-3). Major vegetation found in association with fringing wetland areas included broadleaf cattail (*Typha latifolia*), saltmeadow cordgrass (*Spartina patens*), saltmarsh aster (*Symphyotrichum divaricatus*), marshelder (*Iva annua*), and saltgrass (*Distichlis spicata*). Other aquatic and wetland plants that could occur in the wetland habitats on the Site are listed in Table 3-6. The vegetation associated with the estuarine intertidal wetland documented on the north impoundment (Figure 3-3) is no longer present on the site as a result of the TCRA (Figure 2-3), discussed further below.

Wetland habitats to the south of I-10 along the eastern side of the channel include a narrow stretch of vegetation along the shoreline and the shoreline habitats of three small islands south of I-10. The vegetation on the islands mainly consists of shrubs and small trees. The shrubs and small trees which overhang the water line may provide some shelter and in-water habitat structure for juvenile and baitfish. This area also provides limited foraging habitat for mammals such as raccoons, opossums, skunks, and birds.

3.4.3.3 Aquatic Wildlife

Aquatic birds and semiaquatic mammals that are found in the vicinity of the Site include ducks, shorebirds, wading birds (herons and egrets), diving piscivores, and various others (Table 3-7). There are a number of migratory bird species known to winter in the vicinity of the Site. They include belted kingfisher (*Megaceryle alcyon*), red breasted merganser (*Mergus serrator)*, greater yellowlegs (*Tringa melanoleuca)*, western sandpiper (*Calidris mauri*), and dabbling ducks including gadwall (*Anas strepera*) and teal. Herons and closely related birds that use wetland and estuarine habitats and that may be present in the Site vicinity include the green (*Butorides virescens*), tri-colored (*Egretta tricolor*), and little blue (*E. cerulea*) herons, and also the black-crowned (*Nycticorax nycticorax*) and yellow-crowned

(*N. violacea*) night-herons. Raptors, rails, pelicans, gulls, ducks, and sandpipers are also among the aquatic-dependent and aquatic-associated bird species that use the aquatic habitat that is present in the vicinity of the Site. Sandpipers, egrets, and herons are wading birds that forage along shallow intertidal areas for benthic macroinvertebrates and small fish. Piscivorous bird species that may forage in the open waters of the river include cormorants, osprey, and pelicans. Omnivores including gulls and ducks may forage at the river's edge as well as in the water column. Mammals using both aquatic and wetland habitats that could occur in the vicinity of the Site include the marsh rice rat, muskrats, nutria, and raccoon (Table 3-8).

3.4.3.4 Effect of TCRA Construction

Prior to implementation of the TCRA, estuarine riparian vegetation lined the upland area that runs parallel to I-10 to the north. As a result of the TCRA, that area now includes a dirt road. The western cell of the waste impoundments north of I-10 was occupied by estuarine riparian vegetation until the recent implementation of the TCRA, when the vegetation was removed (Figure 2-3). The eastern cell, also completely covered as a result of the TCRA, lies within intertidal and subtidal habitats.

Under baseline conditions (prior to the implementation of the TCRA), the estuarine riparian vegetation present on the western cell was made up of a mixed shrub and tree canopy (Figure 2-3), could have provided habitat for foraging, nesting and shelter to a variety of bird and mammal species (Tables 3-7 and 3-8). Prior to TCRA implementation, clam shells were observed on the site, indicating a food source for animals such as raccoons, coyotes, wading birds, gulls, and corvids. As part of the TCRA construction, nearly all vegetation was removed from the entire western cell (Figure 2-3), leaving only small amounts of plant material on the western edge of the cell, and eliminating opportunities for upland foraging, nesting and refuge. The shoreline habitat in the TCRA footprint is now devoid of cover, and the exposed surface has limited habitat value for birds and small mammals, and in turn by their predators, such as coyote and raccoon. Some shoreline wading birds may still be expected to use the sandy, shallow intertidal zone for foraging in the post-TCRA conditions. However, over time the area affected by the TCRA cap would be expected to undergo some sedimentation, resulting in the development of plant habitat and plant community development, and subsequent use of the area by birds and mammals. Wildlife uses of this area in the future could be very similar to those of the baseline condition.

3.4.4 Endangered and Threatened Species at the Site

Wildlife that are state-listed as threatened and endangered and have the potential to be found in the general vicinity of the Site are:

- Timber rattlesnake
- Smooth green snake
- Alligator snapping turtle
- White-faced ibis
- Brown pelican
- Rafinesque's big-eared bat.

In addition to these listed species, the American bald eagle, protected under the federal Bald and Golden Eagle Protection Act and listed as threatened by the State of Texas, may be found in the vicinity of the Site.

The two snakes that are listed above are unlikely to occur on the Site. Available information on habitat for these snakes indicates that they prefer upland forested habitats, prairies, and fields or mesic habitats with good vegetative cover. They are not considered common occupants of estuarine or marine wetlands.

The alligator snapping turtle is found in a variety of aquatic habitats including lakes, oxbows, deep rivers, creeks, ponds and brackish estuaries (Appendix A). This species is an opportunistic carnivore, feeding primarily on fish but also on a range of other aquatic animals and occasionally aquatic plants. They spend most of their time in water, usually in the deepest part of their habitat. They are primarily a freshwater species, though they may occasionally use low salinity environments (Appendix A). It is therefore possible that alligator snapping turtles may use aquatic habitats in the Site vicinity, even though their use of the Site is expected to be low relative to their use of freshwater habitats.

The white faced ibis prefers freshwater wetlands, but can be found in estuarine habitats. It is intermediate to the surrogate receptors sandpiper and great blue heron in terms of both body size and diet. Its foraging strategy of visual hunting and tactile probing of sediments for invertebrates is similar to that of the sandpiper, making sandpiper an appropriate representative for this species (Ryder and Manry 1994). The ibis is omnivorous and opportunistic, consuming aquatic insects (in freshwater), fish, amphibians, and crustaceans. The extent to which this bird would use the Site is unknown, but it has been observed as on occasional visitor in summer and fall and rarely in winter and spring at the nearby Baytown nature center. There is limited information regarding the home range or foraging range of this species, though an estimate based on expert opinion and limited information about dispersal to foraging sites suggests that this species may require habitat patches greater than 12 km^2 (Appendix A). The type of habitat present at the Site is not like the foraging and nesting habitats preferred by the ibis, which primarily forages in shallow freshwater habitats with emergent vegetation like rushes and cattails (Ryder and Manry 1994). The white-faced ibis would be only an occasional visitor to the Site, and its exposure potential is considered low. Because sandpiper is assumed to use only the Site while the ibis is an occasional visitor, the sandpiper is a conservative representation of shorebirds such as the ibis.

The brown pelican is a marine piscivore that preys on small surface-schooling fishes. The brown pelican may range up to 20 km from nesting colonies during the breeding season and as far as 75 km from the nearest land during the non-breeding season (Shields 2012). Its diet is similar to that of the neotropic cormorant, making neotropic cormorant an appropriate representative for this species. Although there is little information regarding the foraging area of brown pelican, and information was insufficient to estimate a home range, given the wide-ranging, pelagic nature of the pelican, it is reasonable to assume that its foraging area is likely to be greater than the area of the Site used by the neotropic cormorant.

The American bald eagle may hunt for fish, or eat carrion found on terrestrial and shoreline areas. Foraging ranges for the bald eagle vary widely, from less than 10 km^2 to thousands of square kilometers depending on season and breeding status of the bird (Appendix A). The great blue heron is an appropriate representative for the bald eagle, as it is an omnivore feeding on a range of fish prey. In addition, the great blue heron's foraging strategy and diet make it likely that its association with sediments and rate of incidental sediment ingestion is likely to be higher than that of the bald eagle, making it a conservative choice for evaluating risks to bald eagle from the sediment exposure pathway.

The Rafinesque's big-eared bat may possibly use bridge structures or abandoned buildings in the vicinity of the Site for roosting (TPWD 2012c), but is not expected to forage in the habitats found in the vicinity of the Site because it feeds primarily on emergent aquatic insects, which are generally restricted to freshwater systems and are uncommon in brackish estuarine waters.

In light of this information the white-faced ibis, brown pelican, and American bald eagle are the protected species with a reasonable likelihood of occurring and possibly foraging on the Site.. Because the white-faced ibis, pelican, and eagle have foraging ranges significantly greater than the range of the selected surrogates, and greater than the area of the Site, the selected surrogates represent a much greater exposure potential than these three species. Therefore, risk to these species is considered negligible for a given $COPC_E$ when all of the avian receptor surrogates have negligible risks for that $COPC_E$. In cases where risk could be present for the surrogate species, differences between the home range of each of these protected species and the exposure unit used for modeling exposure to the surrogates provide the basis for evaluation of exposure and risk to the protected species. The method for the exposure evaluation is described in Section 4.3.1.6, and the approach to interpretation of results in presented in Section 6.1.

3.5 Ecological Receptors and Receptor Surrogates

Selection of receptors for this BERA was documented in the SLERA (Appendix B to the RI/FS Work Plan), and summarized in this section. Ecological receptors for the south impoundment are selected in Appendix E to this document.

Ecological receptor surrogates are selected to be representative of the trophic and ecological relationships known or expected at the Site. In selecting receptor surrogates for evaluation in the BERA for the Site, the following criteria were considered:

- The receptor is or could potentially be present at the Site.
- The receptor is representative of one or more feeding guilds.

- The receptor is known to be either sensitive or potentially highly exposed to COPCes at the Site.
- Life history information is available in the literature or is available for a similar species that can be used to inform life history parameters for the receptor.

Many species of aquatic-dependent wildlife may nest in, forage in, and/or migrate through the lower San Jacinto River system. Detailed listings of the species of plants, benthic invertebrates, reptiles, fish, birds, and mammals that could use the habitats on the Site or in the vicinity of the Site are provided in Tables 3-3 through 3-8.

Given that sediments, upland soils, and surface water are the primary environmental media determining the fate and transport of Site-related chemicals, the choice of receptors focused on aquatic-dependent species, or those species which use aquatic resources to a substantial extent. Fish and aquatic-dependent wildlife species for which there are potentially complete exposure pathways to Site-related chemicals include those with direct contact with contaminated soil, sediment, and water and those that prey on benthic macroinvertebrates or on fish that consume benthic macroinvertebrates. Few amphibians that are potentially present in the region are tolerant of brackish or saline waters, with the possible exception of the southern leopard frog. Amphibians are therefore not likely to be in contact with contaminants at the Site, are probably not an ecologically important component of the ecosystem expected at the Site, and are not considered relevant to the BERA.

Terrestrial species are also represented by avian and mammalian surrogate receptors that use upland habitats. The receptors selected for this BERA to address ecological risks for the north impoundment and surrounding aquatic environment are summarized in Table 3-9. More detailed discussion of their life histories in support of evaluating exposures is provided in Appendix A.

3.6 Assessment Endpoints and Risk Questions

An assessment endpoint is "an explicit expression of the environmental value to be protected, operationally defined as an ecological entity and its attributes" (USEPA 2003b). An assessment endpoint addresses a value of ecological significance, has an unambiguous operational definition, and is readily measured or predicted (Suter 1993). Ecological properties identified in assessment endpoints should be those which are susceptible to the chemical stressors and relevant to management goals. Clearly defined assessment endpoints help structure the assessment to address risk management and the primary concerns of stakeholders. Assessment endpoints discussed in this section were derived to conform to these guidelines. A summary of assessment endpoints and risk questions for each receptor group addressed by this SLERA is provided in Table 3-10.

The available literature, and the specific types of information it provides, determine to some extent how the assessment endpoints are defined. For example, although a wildlife *population* is often the level of ecological organization of significance to management, literature to provide measures of effects generally reports on effects on *individuals*. Metrics to assess attributes of individuals (i.e., individual survival, growth, and reproduction) are more generally available in the literature used to support ecological risk assessment than metric to address populations. As a result, some of the assessment endpoints presented in the SLERA have been slightly modified to more closely reflect the ecological attributes that are more commonly reported in the available toxicity literature for the COPC$_{ES}$, and to link the attributes addressed (i.e., individual-level) to the attributes relevant to risk management (i.e., population- or community-level). The fundamental ecological values expressed by the assessment endpoints have not been changed.

In the absence of site-specific population data, performing a series of actual population assessments for an ERA at a site is generally impractical. Because assessment of exposure and effects in this BERA relies on models, the assessment endpoints, for receptors other than the protected species, generally are population viability as indicated by survival, growth, and reproduction of individuals. Units for exposure estimates are selected to match the expression of exposure used in the available toxicity literature.

3.7 Ecological Conceptual Site Models

In the context provided by the CSMs, the receptors and exposure routes evaluated by this BERA include:

- The benthic macroinvertebrate community exposed through direct contact with the benthic environment (sediment, porewater, and surface water)
- Bivalve molluscs exposed through direct contact with the benthic environment (sediment, porewater, and surface water)
- Fish (in all feeding guilds) exposed through ingestion of sediment and food, and respiration of water
- Reptiles exposed through ingestion of sediment or soils, water, and food
- Birds (in all feeding guilds) exposed through ingestion of sediment or soils, water (for seabirds only), and food
- Mammals exposed through ingestion of sediment or soils and food.

Table 3-11 outlines each line of evidence used to address risk to these taxa and exposure groups.

3.8 Ecological Risk Analysis Plan

The problem formulation provides a complete description of the context for evaluation of ecological risks at the Site. In this context, this BERA uses standard methods provided for by USEPA guidance (USEPA 1997, 1998, 2008), including evaluation of uncertainty for results that reflect both reasonably conservative assumptions and realism when Site-specific information is available. This section provides a synopsis of the risk assessment approach and methods used to address the assessment endpoints and risk questions listed in Table 3-10. The purpose of this section is to summarize the overall approach to the risk assessment, to identify the measures of exposure and effects used, to outline the analytical steps for each selected analysis tool, to describe the approach to compilation of information on potential effects to ecological receptors, and to identify the means to characterize risk and evaluate uncertainty. Subsequent sections report the specific calculations, assumptions and related selection of data for the computation of exposure parameter estimates and supporting rationale, and report the outcome of each analysis step.

3.8.1 Ecological Risk Assessment Approach

According to USEPA guidance, a baseline risk assessment should be realistic, so that results accurately represent risks at the Site prior to remedial action (USEPA 1988). Unlike a screening level evaluation, the conclusions of a BERA should reflect realistic representations of exposure and toxicity, and the supporting analysis should not be overly conservative. This approach is appropriate because the baseline risk assessment results are used to inform selection of a risk management approach that is cost-effective.

This BERA uses a tiered approach to the risk analysis and characterization of risks under baseline conditions: an initial assessment of risk is performed using a deterministic model for each receptor and each $COPC_E$ to which that receptor may be significantly exposed via a major exposure pathway ("receptor–$COPC_E$ pair"). The initial assessment employs reasonably conservative but realistic assumptions for each receptor and exposure pathway. In some cases, screening level benchmarks are used for comparison with Site-specific data. This reasonable worst case analysis provides a gross evaluation of risk, resulting in a hazard quotient (HQ) for each receptor–$COPC_E$ pair. HQs are calculated for each receptor–$COPC_E$ pair using a no-observed-adverse-effects level (NOAEL) for the $COPC_E$ to derive the HQ_N, and a LOAEL for the HQ_L. HQs are reported to one significant figure. For each receptor–$COPC_E$ pair, the need for risk analyses subsequent to calculation of the HQ depends on the value of the HQ_L, with one of three possible outcomes, as shown in Figure 3-4. Interpretation of HQs is described in Section 3.8.4.1, below.

When the HQ_L is equal to or greater than 1, subsequent analyses include:

- A probabilistic exposure evaluation
- Evaluation of post-TCRA risk
- Consideration of background.

These three analyses are performed to support risk management decision-making. Use of probabilistic assessment tools results in a more complete and transparent characterization of risks and uncertainties than is possible using an HQ alone. The post-TCRA risk condition existing on the Site is evaluated for those receptors considered to have an unacceptable risk under baseline conditions. Although the TCRA is not considered part of the baseline condition, the purpose of this evaluation is to see what impact the TCRA has on ecological risk, using the general assumption that $COPC_E$ concentrations in sediments within the TCRA footprint are equal to the median concentration of the chemical in the upstream background sediment dataset. This information will inform consideration of the TCRA in the evaluation of remedial alternatives in the FS. Background ecological risks are characterized to describe the incremental risks due to the Site.

An overview of each step in the analytical approach is provided below, with additional details provided in subsequent sections.

3.8.2 Exposure Assessment

According to the CSM, aquatic receptors may be exposed to COPCES via transport of dissolved chemicals across the gills, ingestion, and direct contact. In many cases, the specific route of exposure cannot be discerned from the available literature, or it is not important to the interpretation of the potential for toxicity, because exposures in the literature are expressed simply as concentrations in water, sediment, or organism tissue (see Appendix B of the RI/FS Work Plan, Section 4). Exposures to birds, mammals, and reptiles occurring through respiration (inhalation) or dermal absorption are not evaluated in the BERA as these are generally considered to be minor pathways of exposure relative to the ingestion pathway (USEPA 2003a), although there is uncertainty about this assumption for reptiles (Weir et al. 2010).

3.8.2.1 Measures of Exposure

Measures of exposure selected to address benthic macroinvertebrate and fish receptors include concentrations of COPCs in the following general categories:

- Surface water (mg/L)
- Bulk sediment (mg/kg dry weight [dw])
- Tissue of whole fish, or benthic macroinvertebrates (mg/kg wet weight [ww]; mg/kg lipid weight).

Measures of exposure selected to address bird, reptile, and mammal receptors were the concentrations of COPCs in the following general categories:

- Surface water (mg/L)
- Sediment (mg/kg dw)
- Soils (mg/kg dw), for terrestrial receptors

- Tissue of whole fish (mg/kg dw)
- Tissue of benthic macroinvertebrates (mg/kg dw)
- Bird egg tissue (mg/kg ww; mg/kg lipid weight), estimated from concentrations in diet of birds.

To the maximum extent possible, empirical Site-specific data are used to compute the EPCs for each of these measures of exposure. In some cases, modeling is required to derive exposure concentrations. Models are used to estimate $COPC_E$ concentrations for the following:

- Surface water concentrations of $COPC_Es$ other than dioxins and furans (because empirical data are available for dioxins and furans)
- Terrestrial invertebrate prey and plant foods ingested by killdeer, marsh rice rat, alligator snapping turtle, and raccoon
- Concentrations of dioxin, furan, and dioxin-like PCB congeners in bird eggs.

The specific models and datasets used to derive estimates for these parameters are provided in Section 4.

3.8.2.2 Exposure Units and Calculation of Exposure Point Concentrations

As for the human health exposure assessment (Integral 2012), exposure units are identified for each ecological receptor prior to calculation of EPCs. An exposure unit reflects the area within which a receptor may contact an exposure medium. Spatially defined exposure areas are used to identify the specific set of samples needed to calculate the EPCs for each exposure unit.

For each $COPC_E$ in each exposure medium within each exposure unit, an expression of the central tendency (CT) of the dataset, and an expression of the reasonable maximum (RM) concentration are prepared. The means to estimate these two expressions of concentration depend on the distribution of the dataset, and may include the mean, median, or other expression for the CT; and the 95 percent UCL (95UCL), 95th percentile, or maximum for the RM. Using these two expressions of the EPC for any given $COPC_E$ enables presentation of the most likely (CT) exposure, along with the upper bound (RM) exposure condition, and reflects the middle and upper extent of the exposure profile for receptors. This profile is biased high, because the reasonable minimum (RMin) calculated as the 95 percent lower confidence limit on the mean (or a similar statistic) is just as likely to occur as the RM. An illustration of the importance of this bias in interpreting risks calculated with the RM is provided in Section 7.

For the probabilistic exposure assessment, the probability distribution of the chemical or TEQ concentration is derived from Site-specific data. For parameters estimated probabilistically but with no Site specific information, simple assumptions were made about the data distributions using information from the literature. Details are provided in Section 4.

3.8.2.3 Exposure Algorithms

Following derivation of EPCs for each exposure medium, one or more of the following are performed for the deterministic risk evaluation:

- Concentrations in water, sediment, or tissue are directly compared to benchmarks and/or TRVs expressed in the same exposure units.
- Concentrations in various exposure media are integrated for an individual receptor to compute a cumulative (for all exposure media) total daily ingestion rate.

For the latter, standard exposure algorithms commonly used in ecological risk assessments are used, and described in detail in Section 4. If a probabilistic evaluation of exposure is required, Site-specific data are used to define the probability distribution of several exposure parameters, including life history parameters and the concentrations of the $COPC_E$ in each exposure medium within each exposure unit.

3.8.2.4 Exposure Assumptions

Exposure algorithms for birds and mammals require a number of assumptions about the aspects of receptor biology that affect $COPC_E$ exposure, such as body weight, food or soil ingestion rate, and home range area. Estimates of relevant exposure parameters were taken from the primary literature or from USEPA's (1993) Exposure Factors Handbook. Information on the habits and life history of each receptor is provided in Appendix A, and a summary of related exposure parameters used in exposure algorithms is provided in Table 3-12.

3.8.3 Effects Assessment

This BERA relies on published literature to identify and describe toxicity of COPCES to ecological receptors. No Site-specific toxicity studies were conducted in support of this BERA. The effects assessment therefore consists of a review and compilation of TRVs and benchmarks. For the purposes of this document, a TRV is a species-specific value derived from a controlled experimental study at environmentally realistic exposure levels. The study underlying the TRV defines the specific type of toxic effect at a defined exposure level and exposure route or condition. A TRV can be either an NOAEL (or analogous, e.g., no-observed-effects concentration) at which no effect is expected, or an LOAEL (or analogous), which is the threshold level at or above which effects are expected. A benchmark is a derived value that reflects a broad array of information, potentially encompassing several species, and considered generally protective of a group of species or a community type. An example of a benchmark is USEPA's ambient water quality criteria for the protection of aquatic life (AWQC). Either a TRV or a benchmark can be considered a measure of effect.

The individual effects measures derived from the literature were those that could clearly be related to population- or community-level effects, consistent with the selected assessment endpoints (Table 3-10). Each selected measure of effects on ecological receptors addresses changes in survival, growth, or reproduction resulting from exposure to one or more COPCES. Survival, growth, and reproduction (including developmental inhibition leading to juvenile mortality) can clearly be related to population impacts. For invertebrates, the literature and some benchmarks address higher levels of organization such as populations and communities. Studies addressing endpoints below the organism level (e.g., cellular or biochemical alterations or gene expression), which are difficult or impossible to relate to population- or higher-level effects, were not used to establish TRVs for the BERA.

When using published toxicity literature to establish measures of effect, the specific effects measure depends on the experimental design that was used. For example, a toxicity study may provide a threshold dose above which a reduction in the hatchability of bird eggs occurs. In this case, the effect category is reproduction, and the measure is the LOAEL at or above which effects are observed. TRVs, which encompass both LOAEL and NOAEL values, can be expressed in several ways. The methods for selecting TRVs and benchmarks as well as the values used in this BERA are summarized in Section 5 and described in detail in Appendix B.

3.8.4 Risk Characterization and Uncertainty Analysis

One or more measures of exposure and one or more measures of effects are used to address each assessment endpoint and address the risk questions related to that endpoint. Measures of exposure and effect together define each line of evidence to address each assessment endpoint. Table 3-11 provides a summary of each receptor, assessment endpoint and the measures of exposure and effects that will be used for each line of evidence.

3.8.4.1 Calculation of Hazard Quotients

For each receptor–$COPC_E$ pair, risks are initially described using an HQ, calculated as the estimated exposure as an environmental concentration or daily dose based on the CT of the exposure data distribution, divided by the measure of effect. The ratio of the exposure estimate to the TRV or benchmark is calculated using the following equation:

$$HQ = E \div TRV \qquad \text{(Eq. 3-1)}$$

Where:

HQ	=	hazard quotient
E	=	estimated exposure
TRV	=	toxicity reference value or benchmark.

Units used for the exposure estimate and for the TRV may vary among lines of evidence, but must be the same for the numerator and denominator in the HQ equation. Individual HQs are calculated for each chemical, or TEQ.

To interpret results of HQ calculations, the following guidelines are used:

- Risk to individuals of any receptor from any $COPC_E$ to which the receptor is exposed at a level lower than the NOAEL (i.e., $HQ_N < 1$) is characterized as negligible.

- Risk to individuals of any receptor from any $COPC_E$ to which the receptor is exposed at a level between the NOAEL and LOAEL (i.e., $HQ_N > 1 > HQ_L$) is characterized as very low, and is discussed in the context of the toxicity data supporting the NOAEL and LOAEL values.
- Risk to individuals of any receptor from any $COPC_E$ to which the receptor is exposed at a level higher than the LOAEL (i.e., $HQ_L > 1$) is considered to be present. Risk to the assessment endpoint, which may be a population or community, is evaluated and discussed further in the context of the data supporting the TRV.

Measures of effect (TRVs) typically describe effects on individuals. To inform the risk assessment, HQs must be interpreted to describe risk to the assessment endpoint, which is generally a community or population of organisms (Table 3-10). Therefore, interpretation of $HQ_N > 1$ when $HQ_L < 1$ involves professional judgment and is informed by the basis for the TRV used in the HQ calculation. In these cases, conclusions about risk incorporate relevant context about assumptions and the toxicity information supporting the calculation. Supporting information is described in risk conclusions in Section 6.

Additivity of toxicity and risk for an individual receptor exposed to multiple chemicals (other than dioxin-like compounds) is not systematically considered or reported in this BERA. The absence of relevant information to address this issue is discussed in Section 7.

3.8.4.2 Probabilistic Risk Evaluation

A probabilistic evaluation of exposures is performed for ecological receptors that are potentially exposed at levels equal to or greater than the effects threshold according to one or more lines of evidence in the deterministic evaluation. The probabilistic assessment assigns probability distributions for exposure parameters to yield an output probability distribution for the exposure estimate. Risks to receptors with $HQ_L > 1$ are characterized as the probability that an individual (conforming to the exposure scenario represented by the exposure assumptions) is exposed at or above a level known to have a specified effect. This method allows risk to be expressed as the likelihood that exposures associated with adverse effects can occur under the exposure assumptions. An example of a risk statement of this type is: "there is a 3 percent probability that raccoons will be exposed to iron at a level that has been observed to result in the reduced growth of juvenile mammals."

3.8.4.3 Evaluation of Post-TCRA Risk

For ecological receptor–$COPC_E$ pairs for which $HQ_L > 1$, the post-TCRA risks are also considered. To evaluate the degree to which implementation of the TCRA reduced the baseline ecological risks, values for exposure parameters associated with sediment and soil that were capped by the TCRA are recalculated using a simple framework. For calculation of post-TCRA EPCs, sediment or soil samples collected from within the original 1966 perimeter of the impoundments north of I-10 are eliminated from the dataset used to estimate EPCs, and replaced with the median concentration of the chemical in the upstream background sediment dataset or from the background soil dataset, as appropriate. Samples collected outside the 1966 impoundment perimeter are considered to be the same as for the baseline condition. This approach assumes that birds and mammals that could be using the area affected by the TCRA cap under baseline condition will use the area the same way in the future, and that concentrations of COPCs in sediment in the future will be equivalent to the background condition established for the RI.

3.8.4.4 Risks to Populations of Ecological Receptors

Population-level and community-level assessment endpoints have been selected for the BERA, consistent with USEPA guidance (USEPA 2003b), but TRVs from the available literature providing measures of effects generally represent individual-level endpoints (i.e., those related to survival, growth and reproduction of individual organisms), particularly for birds and mammals. Population-level assessment endpoints are addressed qualitatively in the risk conclusions.

3.8.4.5 Comparison of Site Risks to Background

Background ecological risks are characterized using data from background areas to provide perspective on risks associated with the Site, and to gain an assessment of the incremental risks due to the Site. Only the incremental increase in risk relative to background can potentially be directly affected by controls at the Site.

Background risks are not calculated for all receptors and COPCEs, but are performed when it is concluded that there is an unacceptable baseline risk to an assessment endpoint from a COPCE. Evaluation of risks in background areas was conducted using the same general lines of evidence as for evaluation of Site specific risks. The Site-specific background dataset generated for this RI is used. A summary of that dataset is listed in Section 2.2.1. Details describing background sampling are provided in the Field Sampling Reports (FSRs) for this project, submitted to USEPA in July 2011 (Integral 2011e; Integral and Anchor QEA 2011a,b). Additional background sediments were collected in 2011 and are not described in the FSRs; a complete description of the RI data set will be provided in the RI Report.

4 EXPOSURE ASSESSMENT

Measures of exposure include concentrations of individual $COPC_{E}s$ in water, foods of fish and wildlife, sediment, soil and the eggs of birds, and daily ingestion rates of $COPC_{E}s$ for reptiles, birds and mammals. Because some of the fundamental concepts and resulting selection of methods differ substantially by receptor group, this section is organized by receptor (or receptor group). Each subsection presents methods including algorithms and supporting assumptions, provides a summary statement of the data used, and presents results of the exposure assessment to address each line of evidence for the receptor. Summary information to describe the results of the exposure assessment is presented for each receptor group at the end of each subsection, or in an appendix.

4.1 Exposure of Benthic Macroinvertebrates

Lines of evidence to evaluate risk to benthic macroinvertebrates include:

1. Comparison of bulk sediment concentrations of each $COPC_{E}$ to literature-based benchmarks or TRVs expressed as a concentration in sediment
2. Comparison of estimated concentrations of each $COPC_{E}$ in porewater to literature-based benchmarks or TRVs expressed as a concentration in water
3. For dioxins and furans only:
    a. Comparison of the concentration of 2,3,7,8-TCDD in tissue of whole clams to critical tissues residues (CTRs) expressed as a concentration in tissue of molluscs
    b. Comparison of the concentration of 2,3,7,8-TCDD in sediments with an NOAEL for sediment.

For all of the $COPC_{E}s$, the primary line of evidence is comparison of sediment concentrations at individual stations to benchmarks or TRVs expressed as a bulk sediment concentration. For any $COPC_{E}$ for which the first line of evidence could not be used because a benchmark or TRV is not available in that form, a TRV expressed as a concentration in water is used, and compared to estimated porewater concentrations for that chemical. Comparison of the concentration of 2,3,7,8-TCDD in tissue collected from the Site with the CTR for molluscs is only used to address risks of dioxin exposure to clams. For these lines of evidence, exposure to benthic macroinvertebrates can be characterized as a concentration in sediment, porewater, or tissue, and each approach has a different spatial context:

- Lines of evidence 1 and 3b are empirically based, and describe exposure to benthic macroinvertebrates at each sampling station where a surface sediment sample was collected.
- Line of evidence 2, which is only used for those chemicals lacking benchmarks or TRVs expressed as bulk sediment concentrations, requires an estimate of porewater concentration for each $COPC_E$. To evaluate risk to benthic invertebrates on the basis of exposures via sediment porewater, the water concentration is estimated using the concentration of each $COPC_E$ in sediment at each sampling location.[3]
- Line of evidence 3a addresses molluscs only, and relies on empirical data for 2,3,7,8-TCDD in clam tissue, representing small groups of organisms (those making up a composite). For clams, the composite of several individuals into a single tissue sample represents exposure across a small area.

None of these lines of evidence are reasonable for two metals, aluminum and vanadium, because their geochemistries in estuarine environments strictly limit their bioavailability and toxicity to benthic invertebrates (discussed further below). Concentrations of each of these two metals at each surface sediment sampling station were compared to their respective reference envelope values (REVs).

4.1.1 Estimated Porewater Concentrations

To evaluate exposure of benthic macroinvertebrates to $COPC_E$s via porewater, methods based on principles of equilibrium partitioning are used to extrapolate bulk sediment concentrations of each $COPC_E$ to estimate a water concentration. By using very conservative parameters, this method provides an upper limit estimate of porewater concentrations. It assumes that the sediment porewater is a limited volume of water in direct contact with sediment solids, and is in a two-phase equilibrium with the sediment solids.

[3] Empirical data are available to describe concentrations in water for dioxins and furans, but these data were not used because the first and third lines of evidence are preferred for assessment of risk to benthic macroinvertebrate communities due to TCDD.

The methods to extrapolate from sediment to porewater are as follows. For the metals that are $COPC_Es$ for benthic invertebrate communities but that lack TRVs expressed as bulk sediment concentrations, the concentration of each $COPC_E$ in porewater was estimated from the sediment concentration and its respective soil–water partitioning coefficient, or K_d (Table 4-1):

$$C_{PW} = C_s \div K_d \qquad \text{(Eq. 4-1)}$$

Where:

C_{PW} = concentration in porewater (mg/L)
C_S = concentration in sediment (mg/kg dry weight)
K_d = sediment–water partitioning coefficient (L/kg dry weight).

For organic $COPC_Es$, the role of organic carbon is considered. Because organic carbon binds to many organic chemicals, it can alter the rate of partitioning from the sediment matrix into water and limit the amount of the COPC that can be dissolved into water or porewater. For organic $COPC_Es$, the following algorithm was used to first convert the dry weight sediment concentration to an organic-carbon normalized concentration for each sample:

$$C_{s,oc} = C_s \div f_{oc} \qquad \text{(Eq. 4-2)}$$

Where:

$C_{s,oc}$ = organic carbon (OC)-normalized concentration in the sediment (mg/kg dw)
C_s = sediment concentration of the specific COPC (mg/kg dw)
f_{oc} = fraction of organic carbon in the sediment sample (unitless).

The organic carbon-normalized concentration for each sample is then used to estimate porewater concentrations at that sample location as follows:

$$C_{pw} = C_{s,oc} \div K_{oc} \qquad \text{(Eq. 4-3)}$$

Where:

C_{pw} = concentration in water (mg/L)
$C_{s,oc}$ = organic carbon-normalized concentration in sediment (mg/kg OC)
K_{oc} = organic carbon–water partitioning coefficient (L/kg OC).

Values for K_{OC} are provided in Table 4-1. To evaluate risk to macroinvertebrates that make up benthic infaunal communities, porewater concentrations are estimated for each sediment station from the individual $COPC_E$ concentration in sediment at that station. Porewater concentrations of organic $COPC_E$s for each station were estimated using the fraction of organic carbon in sediment at that station to express sediment concentrations on an organic carbon basis, per Equation 4-2.

Overall, the use of sediment concentrations and partitioning coefficients derived for soils (used for metals) to estimate porewater concentrations is highly conservative. For two of the metals, aluminum and vanadium, the use of partitioning coefficients derived for soils (RAIS 2011) to estimate estuarine porewater concentrations was considered inappropriate because of the geochemical conditions in the estuarine environment. While most COPCs are trace constituents, aluminum is a major rock forming mineral that is an important constituent of clay and feldspar minerals that constitute a large fraction of the inorganic constituents of the sediments. As a reference, aluminum is the third most abundant mineral in both the Earth's crust in general (Krauskopf 1979) and sediments in the nearby Mississippi Delta (Clark 1924) following only oxygen and silica. As a rock forming mineral, aluminum concentration is controlled by mineral solubility, which is affected by the sediment composition and pH. Generally, at neutral pH, aluminum solubility is very low (less than 1 μg/L). Vanadium pore water chemistry is controlled by the redox behavior of vanadium in sediments. In reduced sediments, V is generally found in the highly insoluble V^{4+} form, not the more soluble V^{6+} valence (Fox and Doner 2003). For example, the low solubility of V^{4+} in sediments resulted in water vanadium in lagoon sediments with higher levels of total vanadium being a maximum of 45 μg/L and no V^{6+} being detected in either sediment or pore water (Nicholson et al. 2011). For these reasons, the analysis of aluminum and vanadium consisted only of comparisons of concentrations at individual sediment sampling locations to the REV.

4.1.2 Datasets Used to Evaluate Risk to Benthic Macroinvertebrates

To address each line of evidence for benthic macroinvertebrates, the following datasets are used:

- Concentrations of $COPC_E$s in sediment (e.g., mg/kg) for each of the sediment samples collected from 0 to 15 cm (0 to 6 inches) for all aquatic portions of the Site. Where the elevation of the western cell is above the mean high tide, sediment samples are not included in the benthic invertebrate risk evaluation. This area is only occasionally inundated and does not provide appropriate habitat for a benthic macroinvertebrate community.

- Concentrations of TCDD (ng/kg ww) in individual clam samples collected from on the Site. Locations of transects at which clams were collected is shown in Figure 4-1.

4.1.3 Results of the Benthic Macroinvertebrate Exposure Evaluation

Summary statistics for estimated porewater concentrations for those chemicals evaluated using this line of evidence are presented in Table 4-2; their magnitudes relative to benchmarks are shown for each surface sediment sampling location on the Site in figures described in the risk characterization section. Estimated concentrations of phenol in porewater are not presented because phenol was not detected in 16 of 18 samples, and was J- or UJ-qualified in the other two.

4.2 Exposure of Fish

Lines of evidence to evaluate risk to fish include one of the following for each $COPC_E$:

1. Comparison of $COPC_E$ concentrations in the prey of fish to a TRV expressed as a concentration in food.
2. Comparison of estimated concentrations of $COPC_E$s in surface water to literature-based TRVs or benchmarks expressed as a concentration in water.
3. Comparison of the concentrations of total PCBs, $TEQ_{DF,F}$ and $TEQ_{DFP,F}$ in tissue of whole fish to CTRs expressed as a concentration in whole fish.

Data are not available to evaluate all lines of evidence for each receptor–$COPC_E$ pair. Information to address at least one line of evidence for each $COPC_E$ is presented.

For the second line of evidence, concentrations of $COPC_E$s in surface water were estimated from surface area-weighted average concentrations (SWAC) in sediment using the methods described in the previous section.

4.2.1 $COPC_E$ Concentrations in Fish Diets

For the first line of evidence, evaluation of exposure of fish through food and sediment ingestion, concentrations of $COPC_E$s in each ingested medium (food and sediment) is compared to the TRVs expressed as dietary concentrations (mg/kg diet). Where multiple prey types and sediment may be ingested by a fish (e.g., small fish and invertebrates), a concentration in the overall material ingested (food and sediment) was calculated using the following algorithm:

$$[COPC]_{diet} = \sum f1[COPC]_1 + f2[COPC]_2 \ldots + fn[COPC]_n \quad \text{(Eq. 4-4)}$$

Where:

$[COPC]_{diet}$	=	concentration of COPC in the overall diet (µg/kg dw)
$[COPC]_{1 \ldots n}$	=	concentration of COPC in ingested items 1 through n (µg/kg dw). Ingested items include both biological tissue and incidentally ingested sediment, if any
$f_{1 \ldots n}$	=	fraction of prey items 1 through n in the overall diet (unitless), based on mass, the sum of which does not exceed 1.

The result of this calculation is a concentration in material ingested by fish weighted according to the proportion of each material type in the fish diet (Table 4-3). The result is directly comparable to TRVs expressed as a dry weight concentration in food of fish.

To evaluate species-specific exposures for each of the fish receptor surrogates, information on the proportions of each prey type in their diets was compiled from the literature (Table 4-3). Where the literature reported a prey type for which no Site-specific tissue chemistry data are available, the fraction of the diet consisting of that prey type is added to the fraction of an ecologically similar aquatic prey type for which Site-specific chemistry data are available. In Table 4-3, there are two columns showing the proportion of each prey type used by the fish: the proportion of the diet for each prey type reported in the literature, and the modified proportion of different prey types that are used in the algorithm above. Those prey types not represented in the baseline dataset, such as terrestrial invertebrates, are thus reassigned to an aquatic animal category: molluscs, crustaceans, or fish. For fish that ingest aquatic plants, the plant portion of the diet reported in the literature is distributed evenly among the prey types for which data are available. In this way, the range of prey ingested by each fish receptor is evaluated on the basis of empirical information, and results are reasonably conservative. Together, the three fish receptor diets realistically reflect different feeding guilds: the killifish (omnivore), black drum (benthic invertivore), and southern flounder (benthic piscivore) (Table 3-9).

This method is used to characterize exposure of fish to metals, because reliable TRVs expressed as CTRs for metals are generally not appropriate (Meador et al. 2010). USEPA (2007c) cautions against the use of CTRs for assessment of risk to aquatic organisms from exposure to metals (with the exception of organometals such as tributyltin and methylmercury), unless a toxicologically valid residue-response relationship supports the use of the CTR threshold. Metals are sequestered by many aquatic animals, and metals CTRs for fish are generally not reliable (Meador et al. 2011).

4.2.2 Estimated Concentrations of Selected COPC$_E$s in Surface Water

A TRV expressed as concentrations of bis(2-ethylhexyl)phthalate (BEHP), and of nickel in foods of fish were not found, so surface water concentrations of these two COPC$_E$s were estimated, and compared to TRVs. The approach used to estimate BEHP and nickel in water is analogous to the method to estimate porewater concentrations of some metals for the benthic invertebrate risk assessment (Section 4.1.1). Representation of surface water concentrations using an equilibrium partitioning approach is highly conservative, because the surface water has less direct contact with sediment than porewater, and is more dynamic making it less likely to reach equilibrium with sediment. Surface water is also diluted with instream and tidal flows. Regardless, using equilibrium partitioning methods to estimate surface water chemistry (as well as porewater chemistry) is a simplification of the aquatic and sediment environments, and the result is a highly conservative representation of water chemistry.

For this evaluation, the Site-wide concentration of selected COPC$_E$s is required, and the sediment concentration used is the SWAC.[4] To calculate the SWAC values for each COPC$_E$ requiring a water estimate, a set of Thiessen polygons was created using data from 0 to

[4] In addition to surface water concentrations for nickel, surface water concentrations of several other COPC$_E$s were estimated for use in the wildlife exposure model.

12 inches (including smaller increments such as 0 to 6 inch grab samples) below surface from each sediment sample location within USEPA's preliminary Site perimeter. The polygons were fitted to the perimeter boundary, and the area of each polygon and the total area were calculated. The concentration of the $COPC_E$ in each sample (in mg/kg dw for metals and in mg/kg OC for organic compounds) was then multiplied by the sample's corresponding Thiessen polygon area divided by the total area of the Site, or the fraction of the Site area represented by that sample. The sum of these surface area-weighted concentrations is the total SWAC for the $COPC_E$. The SWAC was used to represent C_S in Equations 4-1 and 4-2 (Section 4.1.1), producing an estimate of the overall surface water concentration on the Site. This result was used to evaluate exposure of fish to BEHP and nickel in the water, and to evaluate $COPC_E$ exposures through ingestion of water to wildlife, as necessary (below). $COPC_E$ SWACs are shown in Table 4-4.

4.2.3 Concentrations of PCBs and Dioxin-Like Compounds in Whole Fish

The third line of evidence is comparison of concentrations of total PCBs, $TEQ_{DF,F}$ and $TEQ_{DFP,F}$ in whole bodies of fish to TRVs expressed in the same terms (µg/kg ww or ng/kg lipid weight [lw]). Site-specific data are used for evaluating exposure of fish to dioxins, furans, and PCBs using these metrics. Composite whole killifish samples were collected on the Site from a series of transects, and 10 whole hardhead catfish samples were collected across all three fish collection areas (FCAs) on the Site (Figure 4-1), with four composites collected in FCA 2, the location of the northern impoundments. Background samples of Gulf killifish were collected from upstream of the Site (Figure 4-2), and background samples of whole hardhead catfish were collected only from Cedar Bayou (Figure 4-3).

Total PCBs in whole killifish and whole catfish from the Site were calculated as the sum of all 209 congeners, with the nondetects substituted at one-half the detection limit. The total PCB concentration in each fish sample as µg/kg ww is shown in Table 4-5 for each individual sample on the Site and for background. The wet weight concentration is used because it is compatible with available toxicity information (Appendix B).

Exposure of fish to dioxin-like compounds is expressed in a manner compatible with relevant toxicity information presented by Steevens et al. (2005): as the probability distribution of lipid-normalized, whole-body TEQ concentrations in Gulf killifish and hardhead catfish. TEQ_F concentrations are expressed as the $TEQ_{DF,F}$, or with $TEQ_{P,F}$ added ($TEQ_{DFP,F}$) in ng/kg lw. Steevens et al.'s (2005) SSD characterizes the distribution of threshold effect levels from tests with fish early life stages using exposures characterized as concentrations in eggs, developing young and maternal whole body of multiple fish species exposed to TCDD (Steevens et al. 2005; Appendix B). Use of all three exposure metrics is appropriate and may be conservative. A study of maternal transfer of TCDD in brook trout established a ratio of 0.39 between whole body and egg concentrations (Tietge et al. 1998). Studies of maternal transfer of other non-polar organic compounds support an approximately 1:1 egg to adult fish ratio (Russell et al. 1999).

Concentrations of $TEQ_{DF,F}$ and $TEQ_{DFP,F}$ in whole Gulf killifish and in whole hardhead catfish from each FCA on the Site and from background areas are shown for each sample in Table 4-6. Probability density functions for $TEQ_{DF,F}$ and $TEQ_{DFP,F}$ in whole Gulf killifish are shown in Figures 4-4 and 4-5, respectively, and probability density functions for $TEQ_{DF,F}$ and $TEQ_{DFP,F}$ in whole catfish are shown on Figures 4-6 and 4-7, respectively.

4.2.4 Unit Conversions

The following computations were used to convert data reported by the laboratory as wet weight to either a dry weight concentration (for calculation of prey concentrations for fish) or to a lipid weight concentration:

- To convert between concentrations expressed as wet weight and dry weight for tissue: mg $COPC_E$/kg dw = mg $COPC_E$/kg ww ÷ (1 – fractional moisture content)
- To convert concentrations expressed as wet weight to lipid-normalized concentrations: mg $COPC_E$/kg lipid = mg $COPC_E$/kg ww ÷ fractional lipid content.

Before calculating EPCs for tissue on a dry weight basis, wet-weight concentrations in individual samples were first converted to dry-weight concentrations using the fractional solids data (i.e., 1 – fractional moisture content) for the same sample if available; if solids data were not available, the average fraction of solids data for the given species was applied for the conversion.

4.2.5 Datasets Used to Evaluate Exposure to Fish

Fish were sampled from the three FCAs on the Site (Figure 4-8) and from Cedar Bayou (Figure 4-9) to represent background. On the Site, composite whole catfish samples were generated in FCA 1 (3 samples) FCA 2 (4 samples) and FCA 3 (3 samples) for a total of 10 whole catfish samples. In Cedar Bayou, 8 composite whole catfish samples were collected. Crabs were also sampled from the three FCAs on Site (Figure 4-10), and from Cedar Bayou (Figure 4-11). On the Site, composite whole crab samples were generated in FCA 1 (3 samples) FCA 2 (3 samples) and FCA 3 (3 samples) for a total of 9 whole blue crab samples. In Cedar Bayou, 3 composite whole blue crab samples were collected. Except for the association with a specific FCA, crab and catfish composites were not spatially referenced; each composite was created from samples across the FCA. Killifish and clam tissues were collected along transects (Figure 4-1), with composites representing the entire length of the transect. Additional information on tissue sampling is provided in the Tissue FSR (Integral 2011e).

In addition to the use of whole fish TEQ_F concentrations used noted above, the following data were used to perform the exposure evaluation for fish:

- **$COPC_{ES}$ in sediment (e.g., mg/kg) for sediment samples collected from 0 to 15 cm (0 to 6 inches) for all aquatic portions of the Site.** This dataset was used for estimating exposure to black drum and southern flounder, which would be expected to move around the entire Site and therefore be exposed to sediments throughout the Site.
- **$COPC_{ES}$ in sediment (e.g., mg/kg) associated with transects used for collection of Gulf killifish.** Killifish primarily move and forage in shallow nearshore intertidal habitats or inundated marsh surface habitats (Lotrich 1975). Home ranges have been estimated from 36 m to 0.15 km^2 across a marsh surface at low tide (Lotrich 1975; Teo-Able 2003). Most movements within tidal creeks in a mid-Atlantic estuary were within 200 m, with the majority of recaptures in a release-recapture study occurring within 50 m (Teo-Able 2003). An intermediate distance of 75m was selected to create a buffer around fish collection transects for selecting sediments to include in exposure assessment of Gulf killifish on a transect-specific basis. Buffers for Transects 1 and 2 largely overlapped and so were combined for determining a sediment dataset.

For calculating the prey-weighted concentrations in the diets of fish, the following were used:

- **COPC$_{ES}$ in whole crabs from the Site.** COPC$_{ES}$ were evaluated in crab tissue from the Site as a whole for drum and flounder, and using the nearest FCA for the transect-specific evaluation of exposure to Gulf killifish (crabs from FCA 1 were used to evaluate exposure to killifish in Transects 1 and 2, crabs from FCA 2 were used to evaluate exposure to killifish in Transects 3, 4, and 5, and crabs from FCA 3 were used to evaluate exposure to killifish in Transect 6).
- **COPC$_{ES}$ in edible tissue of clams from the Site.** Analogous to the approach for sediment, COPC$_{ES}$ in clam tissue from the Site as a whole were used in exposure calculations for drum and flounder, and on a transect-specific basis for the evaluation of exposure to Gulf killifish.
- **COPC$_{ES}$ in whole killifish from the Site.** Analogous to the approach for sediment and clam tissue, COPC$_{ES}$ in killifish from the Site as a whole were used in exposure calculations for drum and flounder, and on a transect-specific basis for the evaluation of exposure to Gulf killifish.

Exposure of fish to PCBs, dioxins, and furans was evaluated using empirical data on concentrations in whole catfish and killifish. Concentrations in individual samples used from Site and background are shown in Tables 4-5 for total PCBs and in Table 4-6 for dioxins and furans as $TEQ_{DF,F}$ and for dioxins, furans and PCBs as $TEQ_{DFP,F}$.

4.2.6 Results of Fish Exposure Assessment

Results of the exposure assessment for fish include:

- Weighted concentrations of the metals in fish diets (Table 4-7)
- Total PCB concentrations in individual whole fish samples (Table 4-5).
- TEQ concentrations in fish whole bodies for each sample (Table 4-6)
- Probability density functions for $TEQ_{DF,F}$ and $TEQ_{DFP,F}$ in whole fish (Figures 4-4 through 4-7).

4.3 Exposure of Reptiles, Mammals, and Birds

Two lines of evidence are used to evaluate risks to birds:

1. Calculation of an individual's cumulative daily ingested dose of each $COPC_E$, expressed as mg $COPC_E$/kg body weight (bw) per day (mg/kg-day), and comparison of this estimate to a TRV expressed in the same terms.
2. Calculation of the $TEQ_{DF,B}$ and $TEQ_{P,B}$ concentration in bird eggs (ng/kg egg, ww), and comparison to TRVs expressed in the same terms.

The first line of evidence is also used to address all $COPC_E$s in reptiles and mammals. Ingested media may include plant and animal foods, water, and soil or sediment as appropriate for each receptor's diet. The cumulative daily dose of each $COPC_E$ to an individual through ingestion of all relevant media each day was calculated using a wildlife exposure model (Section 4.3.1).

Application of this model requires designation of exposure units for each of these receptor groups, and calculation of EPCs within each exposure unit for use in the ingestion model. In addition, estimated concentrations of $COPC_E$s in foods for which empirical data were not available was required for the marsh rice rat and killdeer (exposure via ingestion of terrestrial invertebrates) and for raccoon (exposure via ingestion of plants and terrestrial invertebrates) (Table 4-8).

The second line of evidence is used to evaluate risk to the blue heron, neotropic cormorant, and spotted sandpiper from exposures to dioxins, furans, and dioxin-like PCB congeners. Modeling was used to estimate $TEQ_{DF,B}$ and $TEQ_{P,B}$ concentrations in eggs, and their sum, from concentrations in media ingested by these birds (foods and sediment). Methods for this calculation are also presented below. This evaluation was not performed for killdeer because Site-specific empirical data were not available for the foods of killdeer.

The following sections describe the wildlife exposure model and algorithms necessary for making unit conversions, address the designation of exposure units, describe methods for estimating tissue concentrations as necessary and methods for calculation of EPCs to be used in the wildlife exposure model, and describes the models and assumptions used to estimate $TEQ_{DF,B}$ and $TEQ_{P,B}$ concentrations in bird eggs.

4.3.1 Wildlife Exposure Model

To estimate the cumulative daily dose for reptiles, mammals, and birds through ingestion of food and water, including incidental soil or sediment ingestion, the following general equation was used:

$$\text{Daily Dose} = \left((\text{FIR} \times C_{food} \times \text{RBA}_{food}) + (\text{WIR} \times C_{water}) + (\text{SIR} \times C_{sed} \times \text{RBA}_{sed})\right) \times \text{AUF} \quad \text{(Eq. 4-5)}$$

Where:

Daily Dose	=	COPCEs ingested per day via food, water, and sediment (mg/kg bw-day)
FIR	=	food ingestion rate (kg food dw/kg bw-day)
C_{food}	=	concentration in the overall diet (mg/kg food dw)
RBA_{food}	=	bioavailable fraction absorbed from ingested prey items (unitless)
WIR	=	water ingestion rate (L water/kg bw-day)
C_{water}	=	concentration in water (mg/L water)
SIR	=	Sediment or soil ingestion rate (kg sediment dw/kg bw-day)
C_{sed}	=	concentration in sediment or soil (mg/kg dw)
RBA_{sed}	=	bioavailable fraction absorbed from ingested sediment or soil (unitless)
AUF	=	area use factor (unitless); fraction of time that a receptor spends at the Site relative to the entire home range.

Given that surface waters of the Site are brackish, wildlife other than seabirds and aquatic reptiles are not expected to ingest surface water at the Site, and the WIR term is set to zero for these other receptors. For those estuarine and marine receptors that could ingest Site water (great blue heron, neotropic cormorant, spotted sandpiper, and alligator snapping turtle), a WIR is provided based on allometric equations in USEPA (1993) and included in the exposure algorithm (Table 3-12).

Estimated values for those exposure parameters pertaining to receptor life histories were identified for each species using data compiled in the USEPA's Wildlife Exposure Factors Handbook (USEPA 1993) and other ERAs conducted within USEPA Region 6. Food ingestion rates were estimated using allometric equations presented in Nagy (2001) and USEPA (1993). Receptor profiles detailing the life history information providing the basis for wildlife exposure assumptions are provided in Appendix A. A summary of the selected exposure assumptions for use in the wildlife exposure model is presented in Table 4-8.

4.3.1.1 Ingestion of Multiple Prey Types

For those receptors likely to eat more than one prey type, the portion of the dose derived from the diet incorporates the proportion of each prey type within a typical diet for that receptor. This was done by weighting the $COPC_E$ concentration in each component of the diet by the fraction of the total diet consisting of that prey type. For example, the concentration of a $COPC_E$ in the diet of a receptor which ingests fish, benthic invertebrates and plants is estimated as follows:

$$C_{food} = (C_f \times F_f) + (C_i \times F_i) + \left(C_p \times F_p\right) \qquad \text{(Eq. 4-6)}$$

Where:

C_{food} = concentration of the $COPC_E$ in the overall diet (mg $COPC_E$/kg food dw)

C_f = concentration in fish tissue (mg COPC/kg tissue dw), where

- C_{fs} = concentration in small fish tissue (e.g., killifish)
- C_{fl} = concentration in large fish tissue (e.g., catfish)

F_f = fraction of the diet consisting of fish (kg fish/kg food), where F_{fs} and F_{fl} are used to denote fractions of small and large fish in the diet, analogous to C_{fs} and C_{fl} above

C_i = concentration in invertebrate tissue (mg $COPC_E$/kg tissue dw), where

- C_{ic} = concentration in crustacean tissue (e.g., blue crabs)
- C_{im} = concentration in mollusc tissue (e.g., common rangia)
- C_{it} = concentration in terrestrial invertebrate tissue

F_i = fraction of the diet consisting of invertebrates (kg invertebrates/kg food), where F_{ic}, F_{im}, and F_{it} are used to denote the fractions of various types of invertebrate tissue in the diet, analogous to C_{ic}, C_{im}, and C_{it} above

C_p = concentration in plant tissue (mg $COPC_E$/kg tissue dw)

F_p = fraction of the diet consisting of plants (kg plants/kg food).

Receptor-specific assumptions about the fraction of each prey type in the diet and the information sources supporting each assumption are shown in Table 3-12. It is recognized that individuals of these receptor species may ingest prey types in proportions different from that shown here. The proportions of each type of food in the diet of any given receptor are intended to broadly represent the receptor and its feeding guild.

4.3.1.2 Relative Bioavailability Adjustment Factor

Except for the calculation of daily ingestion rates of dioxins and furans by birds, wildlife exposure calculations for all $COPC_{Es}$ conservatively assume that bioavailability in all media ingested in the field is the same as in the laboratory toxicity study that provides the basis for the TRV. This is a conservative assumption, because laboratory toxicity studies are often conducted using a highly soluble or dissolved form of the chemical in water or food, while the exposure in the field is to the chemical bound in a particular matrix (e.g., food or sediment), in multiple compartments within ingested prey species (such as muscle, bones, and blood), or is otherwise in a form that is not analogous to the laboratory exposure. However, given the variety of mechanisms used to administer test substances in laboratory studies, it is not appropriate to apply a single adjustment factor for all aspects of exposure and all chemicals. In the absence of compelling information for individual chemicals in specific ingested media, this exposure assessment does not apply relative bioavailability adjustment (RBA_{food} and RBA_s) factors in the wildlife exposure model.

One study was found to support application of an RBA in calculation of ingestion exposure by birds. Nosek et al. (1992a) tested the oral bioavailability of TCDD to adult pheasant hens. They mixed radiolabeled TCDD into a suspension of worms, a suspension of crickets, a suspension of paper mill sludge, and a suspension of soil, and administered a fixed amount of the chemical in each suspension into the crops of tested bird in a single dose. After dosing, the birds were allowed to eat normal feed *ad libitum*. After 24 hours, birds were sacrificed, the entire digestive tract removed, and the radioactivity remaining in the bird carcass was measured. Nosek et al. (1992a) report the following absorption rates from the different materials tested: earthworms, 30 percent; soil, 33 percent; paper mill solids 41 percent; and crickets, 58 percent. These were used to derive RBA factors for foods and soil or sediment.

A separate study by Nosek et al. (1992b) provides the basis for derivation of the TRV used in this BERA to interpret daily ingestion rates of dioxins and furans. In that study, adult pheasant hens were administered specific doses of TCDD via intraperitoneal injection. To derive a TRV, the weekly injections were converted to a daily rate and used as an approximation of the daily ingestion rate (Appendix B). While providing a very precise measure of dose, using intraperitoneal injection as the basis for an estimate of an ingestion rate is highly conservative, representing an assumption of 100 percent absorption of TCDD in ingested media through the gut.

In addition to the evidence provided by the robust and systematic analysis by Nosek et al. (1992a), there is a substantial body of evidence to support the assumption that oral bioavailability of TCDD is less than 100 percent in most (and possibly all) vertebrates. Limitations on uptake of all dioxin and furan congeners can be explained by both biological factors and physicochemistry of dioxins and furans in abiotic environmental exposure media. Opperhuizen and Sijm (1990) postulated that the relatively large size of dioxin molecules limits uptake across gill and gut membranes in fish, and that the limitation on uptake rates for dioxin and furan congeners increases with increasing chlorination, which corresponds to molecular size. This conceptual model was confirmed by evaluation of several independent lines of evidence in an analysis for this project (Integral 2010b). Moreover, USEPA (2010b) recently summarized several experimental studies with mammals demonstrating limited uptake of dioxins and furans from weathered soils (i.e., soils contaminated in an environmental context, not spiked for the test). They conclude that although there is variability among species, there is substantial evidence of limited oral bioavailability of dioxins and furans. Budinsky et al. (2008) provide an excellent review of the literature in their introduction, citing a range of experimental data on both limited absorption of dioxins and furans in mammalian gastrointestinal systems, but also limited desorption from ingested media within the gut, an additional factor controlling uptake independent of membrane pore size.

Although the experimental data for mammals and fish, as well as Nosek et al.'s (1992a) bioavailability data, conform to the conceptual framework advanced by Opperhuizen and Sijm (1990) and supported by Budinsky et al. (2008), experimental studies with birds are limited. In one study, Stephens et al. (1995) mixed clean soil, soil contaminated in the field with all 17 dioxin and furan congeners, and dioxin- and furan-spiked soil into the feed of chickens and periodically monitored concentrations of all congeners in various tissue types. Tissues were sampled during an exposure period of 178 days and during depuration of another 100 days. They used a mass balance approach to estimate overall uptake and retention during this experiment. They find that for all congeners, less than 100 percent of mass could be accounted for by their mass balance, and attribute this limited mass to constraints on bioavailability. Although likely correct in this interpretation, their results do not provide the basis for quantitative estimates of RBA because both their dose estimates and their mass balance calculations were imprecise. Nonetheless, they report that after 164 days, less than 65 percent of the ingested TCDD was present in tissues of chickens. This result was comparable to that for TCDF, and all other congeners were present at even lesser percentages of the total mass ingested by the chickens. In addition to this study, Nosek et al. (1992a) cite a study by Martin et al (1989), in which European starlings were dosed with radiolabeled TCDD in an experiment similar to that of Nosek et al. (1992a). Martin et al. (1989) tested oral bioavailability in starlings from suspensions of earthworms, paper mill sludge, soft-bodied invertebrates, and hard-bodied invertebrates, finding bioavailability of TCDD in these suspensions to be 14, 17, 37, and 44 percent, respectively. These results are generally consistent with the RBAs used in this risk assessment, but they indicate even more attenuated uptake in the guts of the starling than the pheasant. However, this paper was published in the grey literature and Nosek et al.'s (1992a) interpretation could not be independently confirmed.

Overall, the weight of evidence presented by Opperhuizen and Sijm (1990), USEPA (2010), Budinsky et al. (2008), Stephens et al. (1995) and Nosek et al. (1992a) clearly supports application of a bioavailability adjustment factor for TCDD. The values presented by Nosek et al. (1992a) and used in this risk assessment are technically robust and appropriately conservative.

To use the media-specific information on relative bioavailability provided by Nosek et al. (1992a), RBA factors for 2,3,7,8-TCDD in soil, sediment and invertebrate tissue were derived (Table 4-9). The RBA factor for invertebrates is the average of the two absorption rates reported for earthworms and crickets (30 percent and 58 percent, respectively), or 0.44. Nosek et al.'s (1992a) result for paper mill sludge is used as the RBA for sediment, and the result for soil is used for soils. For the wildlife exposure model, the 2,3,7,8-TCDD concentration was multiplied by the medium-specific RBA factor prior to calculation of the TEQ for this congener. For terrestrial invertebrate tissue concentrations estimated from soils, the RBA was multiplied by the concentration of 2,3,7,8-TCDD estimated for tissue from a regression relationship (Appendix D). The resulting adjusted TCDD concentration was then multiplied by the TEF to calculate the adjusted TEQ for TCDD in tissue, and added to TEQ concentrations for the other congeners within the sample to calculate a $TEQ_{DF,B}$ or $TEQ_{DFP,B}$. To evaluate the effect of this adjustment on the risk calculations, a sensitivity analysis was conducted using TEQs which were calculated without the application of the RBA for 2,3,7,8-TCDD. The results of this sensitivity analysis are discussed in the uncertainty evaluation (Section 7).

4.3.1.3 Unit Conversions

It is conventional for laboratories to report analytical results for tissue in wet weight concentrations. However, total food ingestion rates, which form the basis for the wildlife exposure model, are estimated on the basis of energy requirements, which are computed from the dry mass of different food types. To convert concentrations expressed as wet weight to dry weight concentrations for tissue, the following equation is used:

$$\text{mg } COPC_E/\text{kg dw} = \text{mg } COPC_E/\text{kg ww} \div (1 - \text{fractional moisture content}) \quad \text{(Eq. 4-7)}$$

Before calculating EPCs for tissue on a dry weight basis, wet weight concentrations in individual samples are first converted to dry weight concentrations using the fractional solids data for the same sample if available; if solids data is not available, the average fraction of solids data for the given species is used.

4.3.1.4 Estimation of Concentrations in Whole Fish

Ecological receptors that eat fish are assumed to ingest the entire fish. Whole hardhead catfish and whole crabs were not sampled for the RI, but remainder samples were collected with samples of edible tissue from both species. Tissue masses of both edible and remainder tissues for catfish and crab were measured in the laboratory. For each individual sample, a mass-weighted whole fish concentration was calculated as follows:

$$C_{wb} = \left[C_{fi} \times \frac{W_{fi}}{W_{wb}}\right] + \left[C_r \times \frac{W_r}{W_{wb}}\right] \quad \text{(Eq. 4-8)}$$

Where:

C_{wb} = chemical concentration in the whole body of the fish (mg/kg)
C_{fi} = chemical concentration in the fillet tissue (mg/kg)
C_r = chemical concentration in the remainder tissue (mg/kg)
W_{fi} = weight of the fillet (kg)
W_{wb} = weight of the whole body (kg)
W_r = weight of the remainder (kg).

Resulting "whole fish" concentrations were the only fish and crab tissue data used for the BERA; no calculations were performed with just edible crab or just catfish fillet.

Gulf killifish and clams were collected whole. Clams were briefly held in buckets after sampling, allowing them to excrete any sediment in their gut prior to chemical analysis. All soft tissues were extracted from the clam (everything inside the shell) for analysis, and it was these soft tissues that were used for the BERA.

4.3.1.5 Concentrations of $COPC_E$s in Foods of Alligator Snapping Turtle, Killdeer, Raccoon, and Marsh Rice Rat

Empirical data are available to describe concentrations of $COPC_E$s in soils, sediments, fish and aquatic invertebrate tissue from the Site and from background areas. Methods for calculation of EPCs for these media, for use in the wildlife exposure model, are presented in Section 4.3.1.7, below. There are no empirical data to describe $COPC_E$ concentrations in terrestrial invertebrates and plants from the Site. As a result, it was necessary to estimate $COPC_E$ concentrations in terrestrial invertebrates from soil concentrations, and to estimate

$COPC_E$ concentrations in aquatic plants from sediment concentrations and in terrestrial plants from soil concentrations. To do this, simple bioaccumulation factors or regression equations describing relationships between concentrations in sediment or soil and the concentrations in invertebrate prey or plant tissue were used. For the majority of $COPC_Es$, these equations take the following forms:

$$C_{prey} = C_s \times BAF_{prey} \qquad \text{(Eq. 4-9)}$$

$$C_{prey} = B1_{prey} \times C_s + B0_{prey} \qquad \text{(Eq. 4-10)}$$

Or

$$\ln(C_{prey}) = B1_{prey}(\ln[C_s]) + B0_{prey} \qquad \text{(Eq. 4-11)}$$

Where, for those chemicals lacking a statistically significant regression model:

BAF_{prey} = bioaccumulation factor (kg dw soil/kg dw prey)

And for those chemicals for which a significant regression model is available:

$B1_{prey}$ = slope of the regression of the concentration of the chemical in the prey against the concentration of the chemical in soil or sediment

$B0_{prey}$ = *y*-intercept term, describing the concentration in the prey when $C_s = 0$

The BAF is simply the ratio of the concentration of chemical in the prey (C_{prey}) to the concentration of the chemical in sediment or soil (Cs). As discussed by Integral (2011), regression models have several technical advantages over simple ratios, and are considered the most appropriate method for analysis and characterization of relationships between abiotic media and tissue. Generally, regression models were preferred for making the predictions of chemical concentrations in tissue necessary for wildlife exposure modeling, consistent with Integral (2011).

For chemicals other than dioxins and furans, the primary source of models for this analysis was USEPA's Guidance for Developing Ecological Soil Screening Levels (USEPA 2007c), which provides regression equations describing relationships between measured concentrations of chemicals in soils and plants, and between concentrations in soil and terrestrial invertebrates. When BAFs or regression equations were not available from USEPA

(2007c), other sources were reviewed, including Sample et al. (1998), USEPA (1999b), and RAIS (2011). Staples et al. (1997) describe the environmental chemistry of BEHP, and provided the relevant information for predicting BEHP in tissue. Burton et al. (2006) was used to establish BAFs for estimating tissue concentrations of mercury in terrestrial invertebrates from Site soils. For dioxins and furans, no model was selected for estimating concentrations in plants. A dataset for a Superfund site in Minnesota was analyzed to derive regression models for estimating concentrations of individual congeners in terrestrial invertebrates, as described below.

4.3.1.5.1 Estimating $COPC_E$ Concentrations in Plants

To estimate metals concentrations in plant tissue, BAFs or regression models from USEPA (2007c) were available for cadmium, chromium, cobalt, copper, lead, nickel, vanadium, and zinc (Table 4-10). Plant BAFs were not available from USEPA (2007c) for dioxins and furans, PCBs, BEHP, or mercury. For mercury, soil-to-plant BAFs were selected from USEPA (1999b). The BAF for mercury is the recommended BAF for mercuric chloride. Results of this evaluation are not presented in tabulated form.

For dioxins and furans, PCBs and BEHP, uptake from sediments into plants is considered negligible (Wild and Jones 1992; Bromilow and Chamberlain 1995; Staples et al. 1997). Plants are exposed to $COPC_{ES}$ in sediment primarily through porewater (Kabata-Pendias and Pendias 1992). Chemicals with low water solubility may adsorb to the roots of plants (e.g., lead), but uptake into the plant's vascular system and transport to leaves and fruit is limited for low-solubility chemicals. Lipophilic compounds are taken up by the roots or by foliage, and are transported in plant xylem. This transport is slowed by partitioning of lipophilic compounds to the lipid-like matter in plant tissue (Bromilow and Chamberlain 1995), and fruits are not affected. If taken up by plants, lipophilic chemicals tend to accumulate in the leaf margins and interveinal spaces. As a general indicator of the transport of lipophilic compounds within plants, Travis and Arms (1988) reviewed BAFs for plant foliage for 29 chemicals with log K_{ow} values ranging from 1 to 10, and found that the BAF was inversely proportional to the square root of the K_{ow}.

Although USEPA (1999b) provides a relationship for BEHP based on its K_{ow}, these types of simplified relationships based solely on chemical hydrophobicity are limited because they do not take into account processes such as metabolism which play an important role in modulating phthalate bioaccumulation (Staples et al. 1997). Staples indicates that most soil-to-plant BAFs for phthalates are <0.1 and typical soil-to-plant BAFs from their literature review are <0.01. Staples et al. (1997) also note that these BAF values are based on studies with radiolabeled carbon, and exposures to multiple phthalates. As such, resulting BAFs overestimate final tissue concentrations of the phthalate because metabolites in tissues are not distinguishable from the parent compound using this method. Other studies discussed by Staples et al. (1997) suggest there is no appreciable uptake of BEHP to plants from soils.

Much of the data concerning plant-uptake of organic chemicals from soil comes from studies investigating soil-to-plant transfers of chemicals derived from sludge-amended soils. Regarding these studies, Wild and Jones (1992) provide the following general comments including PCBs, polycyclic aromatic hydrocarbons (PAHs), and other organochlorine compounds typically studied, "these compounds are generally not taken up into the aboveground portion of crop plants" and, that "there is some evidence of slight enrichment of some compounds in some root crops, but the transfers are very inefficient, and consequently the BCFs [bioconcentration factors] are very low." They also note that enrichments are generally confined to the root skin or peels, and not the fruits, leaves, and stems that may be eaten by wildlife (Wild and Jones 1992).

For these reasons, plant tissue concentrations of dioxins, furans, PCBs, and BEHP are assumed to be zero in the wildlife exposure model.

4.3.1.5.2 Estimating $COPC_E$ Concentrations in Soil Invertebrates

BAFs to estimate terrestrial invertebrate tissue concentrations were available from USEPA (2007c) for cadmium, chromium, cobalt, copper, lead, vanadium, and zinc (Table 4-10). Soil-to-invertebrate BAFs were available for nickel in USEPA (1999b). For PCBs, a regression equation from Sample et al. (1998) was selected to estimate total PCB concentrations in soil invertebrates from total PCB concentrations in soil. Congener-specific models were not used because there are no PCB congener data for soils at the Site with the exception data for soils collected from the Texas Department of Transportation right-of-way.

Sample et al. (1998) and other compendia do not provide robust regression relationships for mercury, so Burton et al. (2006) was used to establish BAFs for estimating tissue concentrations from Site soils. Burton et al. (2006) conducted a 28-day study evaluating uptake by earthworms of total and organic mercury from soils across a range of mercury concentrations. Their study established that mercury concentrations in earthworm tissue were higher relative to concentrations in soils at lower soil concentrations, and lower relative to concentrations in soils at higher soil concentrations. Burton et al. (2006) report a BAF of 3.1 for low soil concentrations (0.156 mg/kg) and a BAF of 0.6 and 0.7 for intermediate (2.83 mg/kg) and high (11.54 mg/kg) soil concentrations. The method to estimate mercury concentrations in terrestrial invertebrate tissue for the wildlife exposure model recognizes this differential in mercury BAF with concentration in soil because mercury concentrations in soils are either less than 2 mg/kg or greater than 10 mg/kg (Figure 4-12).

Burton et al. (2006) did not establish a measure of variance around their surface soil concentrations, so the division between low and intermediate concentrations was established as the median value between 0.156 and 2.83 mg/kg, or 1.5 mg/kg. Therefore, to estimate concentrations of mercury in terrestrial invertebrates for the wildlife exposure model, the BAF of 3.1 was applied at stations with mercury concentrations below 1.5 mg/kg, and the BAF of 0.7 was applied to Site surface soils with concentrations greater than 1.5 mg/kg. Burton et al.'s (2006) BAFs for intermediate and high soil concentrations were not significantly different, so the choice of the higher BAF of 0.7 for estimating tissue concentrations at the stations with mercury concentrations in soil greater than 1.5 mg/kg was conservative (Figure 4-12).

At least one recent study both supports the use of this approach and illustrates that the use of a BAF simplifies a likely complex system. Fengxiang et al. (2012) reports on earthworm bioaccumulation studies using soils with and without cinnabar mercury, and with mature and immature worms. They report that cinnabar mercury, tightly bound to sulfur, does not correlate with mercury in worm tissue, while non-cinnabar mercury correlates well. This example illustrates the importance of local soil conditions. Fengxiang et al.'s (2012) soil-to-earthworm BAFs for mercury ranged from 0.32 to 1.75, indicating that the BAFs selected for this BERA are conservative (i.e., above the upper end of that range for uptake from soils with low concentrations of mercury).

Staples et al. (1997) presents information to indicate that BEHP does not bioaccumulate in soil invertebrate tissue at environmentally realistic concentrations in soil. Only one study (Hu et al. 2005) reported biological transfer of BEHP from soils to soil invertebrates, but exposure concentrations were much higher than the range of BEHP concentrations in Site soils. Therefore, the reported soil-to-invertebrate bioaccumulation relationship reported by Hu et al. (2005) is not appropriate for application to Site conditions. On the basis of the review provided by Staples et al. (1997), invertebrate tissue concentrations of BEHP are assumed to be zero in the wildlife exposure model.

None of the literature sources listed above provides sufficient information for use in estimating concentrations of dioxin and furan congeners in terrestrial invertebrates. Although Sample et al. (1998) present a model for TCDD, important details are missing from the analysis, and only one congener is addressed. To develop soil-to-invertebrate relationships for predicting dioxin and furan concentrations in tissue, data from a bioaccumulation study using earthworms (*Eisinia fetida*) at a Superfund site in Minnesota were used. In two locations from this study, naive earthworms were exposed to samples of field-collected soils contaminated with dioxins and furans for 28 days. At the end of the test, animals were purged and tissues analyzed for dioxins and furans. The study was conducted according to specifications of the American Society for Testing and Materials Method 1976–04. In an additional five locations, co-located soils and earthworm tissue were collected. One worm froze in transit and could not be depurated, and therefore was not used for developing soil-to-tissue relationships. All worms used from this dataset were depurated prior to analysis. All worm and co-located soils were analyzed for dioxins and furans using USEPA Method 8290. All chemistry data were validated according to CERCLA validation protocols. More information on this study, and the data used, are provided in Appendix D. Results include a series of regression and correlation relationships for dioxin and furan congeners, summarized in Table 4-11, that were used to estimate dioxin and furan concentrations in soil invertebrate tissue for use in the wildlife exposure model for killdeer and raccoon. Additional methodological details and results of statistical evaluations and resulting tissue concentration estimates are provided in Appendix D.

Soil-to-earthworm BAFs or regression relationships are summarized in Table 4-10.

4.3.1.6 Wildlife Exposure Units

An exposure unit is the area in which a receptor may be exposed to contaminants in environmental media. The exposure unit provides an organizing concept for selection of data to be used in estimating wildlife exposures. It defines the spatial area from which data were selected for calculation of EPCs for each medium, using samples collected according to the DQOs presented in relevant SAPs. An individual receptor is assumed to be equally likely to be exposed to media within all subareas of the exposure unit.

Wildlife exposure units for this BERA were defined to reflect the possible foraging areas and habitats for the surrogate receptor species at the Site (Appendix A). The exposure units for reptiles, birds, and mammals included the following areas of the Site:

- Upland habitat for evaluation of exposures to raccoon and killdeer within USEPA's preliminary Site perimeter, including soils and foods in:
  - All upland habitat north of I-10 (for killdeer, Figure 4-13)
  - All upland habitat on the peninsula within and adjacent to the impoundments, both north and south of I-10 (for raccoon, Figure 4-14)
- Shoreline habitat within USEPA's preliminary Site perimeter, including sediments, surface water (ingested by aquatic birds and reptiles) and prey within these exposure units (Figure 4-15)
- Aquatic habitat within USEPA's preliminary Site perimeter, including sediments, surface water (ingested by aquatic birds and reptiles) and prey within these exposure units (Figure 4-16 and 4-17).

When using the Site, a given receptor may be present in one or more of these exposure units, depending on its life history and foraging habits. Although receptors would use an area according to its habitat quality and resources provided by the habitat (forage, refugia), the approach used to establish EPCs conservatively assumes that receptors will be more likely to encounter contaminated areas than other areas on the Site, regardless of habitat quality. Concentrations in exposure media were not spatially weighted; instead, each sample was given equal weight, even though there is a higher spatial density of samples directly adjacent to the impoundments north of I-10 than elsewhere within USEPA's preliminary Site perimeter. Samples were not collected evenly across all habitats, such as the vegetated area to the west of the sand separation area, or the eastern shoreline. This spatial distribution of samples reflected DQOs described by approved SAPs.

Table 4-8 outlines the way in which exposure units and media are assigned to each receptor for the wildlife exposure assessment. Figures are presented to graphically illustrate the exposure units for each receptor surrogate, as follows:

- Figure 4-16, Alligator snapping turtle
- Figure 4-17, Neotropic cormorant
- Figure 4-15, Great blue heron, spotted sandpiper, and marsh rice rat
- Figure 4-14, Raccoon
- Figure 4-13, Killdeer.

Each of these figures shows the sediment and/or soil samples, and the transects for tissue collection where applicable, used for calculating EPCs for the estimate of exposure to each wildlife receptor. Because most samples were collected in locations near or adjacent to the impoundments north of I-10, regardless of habitat quality, and because all samples were given equal weight in exposure statistics, regardless of the spatial area represented, the selection and definition of exposure units was conservative. For the post-TRCA scenarios, all samples collected from within the original 1966 perimeter of the impoundments north of I-10 were removed from the data before performing calculations, and replaced with one value equal to the median concentration of the upstream background sediment or the background soil data, as appropriate, for the chemical of interest.

Data selected for calculating exposures in the aquatic environment were selected by clipping the hydrologic unit polygon for the San Jacinto River to the preliminary Site perimeter boundary. The hydrologic unit polygon was received from the Harris County Public Infrastructure Department Architecture and Engineering Division. This polygon was transformed into a line feature which was clipped appropriately and used to represent and calculate total length of the shoreline within the site boundary.

Data for calculating exposures in terrestrial areas were selected using digitized polygons based on 0.5-m 2008/2009 Digital Orthophoto Quarter Quads from the Texas Strategic Mapping Program (StratMap) that most closely represent the habitat of the organism of interest. Only soils collected from 0 to 6 inches depth were used. The habitat area calculations were used to estimate Site exposure unit sizes for each receptor (Table 4-12).

For protected species that could occur on the Site (white-faced ibis, bald eagle, and pelican), if the estimated exposure of their respective avian receptor surrogates (Section 3.3) to a $COPC_E$ exceeds the NOAEL or the LOAEL, then the exposure of each of the protected species to that chemical is calculated by adjusting the exposure area assumed for surrogate (as described above) by the relative size of the protected species' home range. Because the home range of each surrogate for the protected species that could occur on the Site is conservatively assumed to be equal to the exposure unit, this calculation consists of multiplying the dose by the ratio of the surrogate's exposure unit area to the protected species' home range area (Table 4-12). Results are addressed for relevant $COPC_{ES}$ in Section 6.

4.3.1.7 Calculation of Exposure Point Concentrations

Consistent with USEPA guidance (USEPA 1997) which directs ecological risk assessors to consider an exposure profile for each receptor, EPCs were generated for each exposure medium within each exposure unit for use in the wildlife exposure model described above. CT and RM exposure concentrations were generated for each $COPC_E$. Selection of the appropriate statistic to represent the CT and RM for each EPC was based on the statistical distribution of the data supporting that EPC for each $COPC_E$ within a given medium (sediment, soil, and tissue) and exposure unit. All analyses of data distributions and generation of distribution parameters were performed using the software R for Windows version 2.9.0 (R Development Core Team 2008).

Treatment of censored data in EPC calculations is discussed in Section 2.2.2. Decisions for generation of the statistical representations of the EPCs for a given data distributions were as follows (Appendix C):

- For normal data distributions, the arithmetic mean was chosen as the CT and the 95UCL based on a Gaussian data distribution was selected as the RM.

- For lognormal distributions, the geometric mean was chosen as the CT and the 95UCL based on a lognormal data distribution was selected as the RM.
- For unknown data distributions (i.e., those distributions that were not normal and could not be transformed to a log-normal distribution), the arithmetic mean was chosen as the CT and 95UCL was calculated using nonparametric statistics, consistent with ProUCL (USEPA 2007b).

In all cases, if the 95UCL was greater than the maximum value for the dataset, the maximum was selected as the RM. Results of all EPC calculations are presented in Appendix C.

For a few datasets (e.g., TEQ_P in soil and shoreline sediments), the sample size was so small ($N < 4$) that a distribution of the data could not be calculated and a UCL could not be generated with confidence; in these cases, the maximum value was used as an estimate of the RM. For a few other datasets (BEHP in clams and Gulf killifish), there were no detected values, so the CT and RM in these cases were set equal to one-half the detection limit. Concentrations of PCBs in water were not estimated, and the PCB doses via ingestion of water for seabirds were not calculated, because the dose via water ingestion is assumed to be minor relative to dose via ingestion of foods due to the low solubility and relatively high potential for bioaccumulation and biomagnification of PCBs.

To estimate concentrations of $COPC_{ES}$ other than dioxins and furans in terrestrial invertebrate and plant tissue, a soil or sediment EPC calculated using data from within the exposure unit of the subject receptor is multiplied by the BAFs or is used in the regression equations (Table 4-10) to generate CT and RM EPCs for input to the wildlife exposure model.

Where an analysis of the post-TCRA wildlife exposure is needed, all samples for stations within the original 1966 impoundment perimeter are removed and replaced with a single value representative of the possible post-TCRA condition. The value used in these substitutions is the median concentrations of the $COPC_E$ in the upstream sediment dataset or in the background soil dataset, depending on whether the exposure scenario involves exposures to sediments or soils. All of the analyses to describe the data distribution and to calculate CT and RM EPCs were repeated using this substituted dataset prior to their use in the wildlife exposure model. Results are presented in Appendix C. No substitutions were performed for tissue concentrations, so pre-TCRA tissue concentrations were used in post-TCRA analyses.

4.3.1.8 Data Used

The data used in the wildlife exposure model include:

- Sediment and soil samples collected from 0 to 6 inches shown in Figures 4-13 through 4-17
- Sediment from 0 to 6 inches from the upstream sediments background study
- Soil from 0 to 6 inches from the Site-specific background study
- All clam samples collected for the RI (Figure 4-1)
- All killifish samples collected for the RI (Figure 4-1)
- Whole hardhead catfish samples and whole blue crab samples from on the Site (Figure 4-1)
- Tissue samples collected from the upstream background (Figure 4-2) and Cedar Bayou (Figure 4-3) background tissue study
- Surface water samples collected by TCEQ for analysis of dioxins and furans (URS 2010).

Soil from the Site specific background study, sediment data from the upstream background area, and tissue data from background areas were used only when the $HQ_L \geq 1$ (Section 3.8).

4.3.1.9 Results

Summary presentations of results of the wildlife exposure model and supporting calculations are provided as follows:

- Results of calculations using BAFs and regression models for invertebrates and plants were not tabulated, but were incorporated directly into the wildlife exposure model
- The EPCs used in the ingestion model are presented in Appendix C, Table C1.
- Final estimates of the daily ingestion rate of each $COPC_E$ for each bird, mammal, and reptile receptor surrogate are shown in Table 4-13.

4.3.2 Estimated TEQ Concentrations in Bird Eggs

Concentrations of dioxin-like compounds in bird eggs were estimated as part of the exposure assessment because substantial toxicity information in the literature for birds is expressed as egg concentrations (RI/FS Work Plan, Appendix B, Attachment B2), and because comparison of TEQ concentrations in eggs to TRVs expressed as egg concentrations is the risk assessment method recommended by USEPA (2003b; 2008). Site-specific data to described TEQ concentrations in bird eggs were not developed for the RI, so modeling was performed to derive estimates of egg concentrations. Methods for modeling were different for dioxins and furans than for dioxin-like PCBs, due to differences in the information available in the literature. Each method is described below.

4.3.2.1 Estimating Dioxins and Furans in Bird Eggs

The uptake of dioxins and furans from dietary sources into bird tissue and subsequent transference into eggs is both species- and congener-specific. This process can be considered as occurring in two steps: 1) the uptake and retention of dioxins and furans by the egg laying female, and 2) the maternal transfer of dioxins and furans into the egg. Although uptake and retention of dioxin and furans in vertebrates is species- and congener-specific, general trends can be found in the literature (Integral 2010b). In contrast, maternal transfer of dioxins and furans from egg laying female birds to their eggs has been less well studied, and sufficient information for mechanistically modeling egg concentrations stepwise through these two process steps was not found. As a result, the simple bioaccumulation from foods ingested by the parent bird into eggs provides the conceptual basis for estimating egg concentrations for this evaluation.

Simple estimation methods such as biomagnification factors (BMFs) calculated as the ratio of a food concentration to an egg concentration, can lead to significant error in predicted egg tissue chemistry. This potential for error is due to congener- and species-specific differences in retention and distribution of dioxins and furans (Integral 2010b). If appropriate data are available, use of statistical regression models overcomes several weaknesses in the ratio method.

4.3.2.1.1 Identification of a Prey-to-Egg Regression Model

A literature search was conducted to identify studies describing statistical models of prey-to-egg relationships, and only one paper was found to support this analysis (Elliott et al. 2001). The study presented by Elliott et al. (2001) provides a set of regression models for estimating dioxin and furan concentrations in bird eggs from concentrations in foods of the birds. Elliott et al. (2001) focus on the great blue heron, a piscivore and one of the avian receptors at the Site. Congener-specific and homologue-based regression models reported by Elliott et al. (2001) for log-transformed dioxin and furan data in prey tissue were used to estimate egg tissue TEQ concentrations from dioxins and furans in ingested media from the Site and background areas, and to estimate post-TCRA exposures.

Elliott et al. (2001) monitored dioxins and furans in eggs from 21 great blue heron rookeries and in prey fish from 1983 to 1998. They developed linear regression models showing strong positive relationships between congener families and TCDD and TCDF in prey fish species and in heron egg tissue (Table 4-14). A review of the literature and subsequent reanalysis of published data showed linear relationships between diet and egg dioxin and furan concentrations from two other studies:

- **Tree swallows in Woonasquatucket River, Rhode Island (Custer et al. 2005):** Evaluation of data for this site by Integral indicated a linear relationship between dioxin and furan concentrations in pooled diet samples and egg tissue (when nondetects in diet samples are excluded).
- **Herring gulls in Lake Ontario (Braune and Nordstrom 1989):** Evaluation of data from this study indicated a moderate linear relationship between alewife prey and egg tissue concentrations of dioxins and furans.

Although both of these studies support the selected approach for modeling egg tissue concentrations, the data were insufficient for use in developing a model for egg tissue estimates. As a result, only the regression models reported by Elliott et al. (2001) were used. Results of the studies with herring gulls and swallows improve confidence in the conceptual basis for the selected approach.

4.3.2.1.2 Implementation of the Prey-to-Egg Model

The linear regression models for each congener or homologue group from Elliott et al. (2001) (Table 4-14) were used to estimate egg concentrations for three bird receptor surrogate species: blue heron, cormorant, and sandpiper. The independent variable used in each model, which was fish tissue concentration in Elliott et al. (2001), was estimated to reflect the aggregate of ingested media by the receptor surrogates for the Site. Ingestion of contaminated prey with and without ingestion of contaminated sediment was evaluated. Ingested media in this model for each bird receptor were as follows:

- **Blue heron:** Whole catfish, whole blue crab, Gulf killifish, and shoreline sediment (Figure 4-15)
- **Cormorant:** Gulf killifish and bottom sediment (Figure 4-17)
- **Sandpiper:** Clams, whole blue crabs, and shoreline sediment (Figure 4-15).

The regression models required individual congener or homologue data for each ingested medium. Using the CT and RM for each individual congener was considered overly conservative because to do so would result in a combination of dioxin and furan congeners for each ingested medium that would not be representative of the congener composition and TEQ concentrations in the natural environment and in actual tissue samples. Moreover, this approach would be inconsistent with exposure profiles represented by the CT and RM of TEQ elsewhere in this risk assessment. Instead, an individual sample of each medium was selected to represent the CT and RM exposures. To do this, the CT and RM $TEQ_{DF,B}$ concentrations of each medium within each exposure unit were calculated. Because the result is a statistic, and not a specific sample, the actual sample with the $TEQ_{DF,B}$ concentration closest to the CT and the sample with the $TEQ_{DF,B}$ concentration closest to the RM were identified. For each medium, the sample number and congener concentrations under each scenario are shown in Table 4-15. The physical locations of the samples in Table 4-15 are referenced in Figures 4-15 and 4-17. The specific congener concentrations within these samples were used in subsequent calculations. Examples of several specific calculations for the cormorant, heron, and sandpiper are presented in Exhibits 2A and 2B.

To estimate the congener or homologue concentrations in eggs that accounted for all ingested media, a mass-weighted concentration in the total mass of ingested media was calculated for each congener or homologue group for each receptor. The method for calculation is analogous to the approach used for calculation of exposure via food ingestion for fish (Section 4.3.1.1), shown in Equation 4-4. The congener or homologue concentration in each ingested medium was weighted according the fractional contribution of the media type to the total mass of ingested media (Table 3-12). Resulting congener concentrations or homologue concentrations were then used as input into the regression models (Table 4-14). Egg concentrations for each congener or homologue group were then estimated using the regression equations published by Elliott et al. (2001) (Table 4-14). Because calculations were conducted using laboratory-reported homologue concentrations, and not sums of individual congeners, resulting egg $TEQ_{DF,B}$ concentrations are expected to have an upward bias. The degree of bias is unknown due to variability in the results for individual samples, but is higher when all congeners in a homologue group were detected, and indeterminate when congeners and homologues were reported as nondetects. Uncertainties associated with use of the Elliott et al. (2001) regression models are discussed further in Section 7.

Concentrations of congeners in homologue groups for which Elliott et al. (2001) did not publish regression equations were estimated by using regression parameters for the most closely associated homologue group (e.g., HpCDF was modeled using the equation for HxCDF). This substitution allowed prediction of congeners or homologue concentrations in eggs for all congeners except the octachlorinated congeners. Octachlorinated congeners have rarely been reported in bird tissue (see Table 15 of Integral 2010b), and have very low TEFs (Table 3-2). Moreover, they are the largest among the dioxin and furan congeners and therefore the least bioaccumulative (Integral 2010b). For these reasons, the lack of predicted egg concentrations of octachlorinated congeners is expected to have a negligible effect on the final egg $TEQ_{DF,B}$ concentration estimates. Further, because regression parameters in Elliott et al.'s (2001) models for PeCDD and HxCDD are very similar (Table 4-14), the model substitutions that were made were considered appropriate. However, extending model substitutions for the octachlorinated congeners using one of Elliott et al.'s (2001) models was considered too uncertain because of the known differences from other congeners in the bioaccumulation patterns of the octachlorinated congeners.

Finally, estimated CT and RM concentrations of each congener or homologue in egg tissue were multiplied by the appropriate TEF to compute the final $TEQ_{DF,B}$ in eggs. Because two homologue groups include congeners with different TEFs (Table 3-2), a conservative estimate of egg $TEQ_{DF,B}$ was calculated assuming the maximum TEF for all congeners in the group (Table 4-16). To estimate a lower bound on the estimated egg $TEQ_{DF,B}$ concentrations, a second calculation was performed using the lowest TEF for the homologue group, which resulted in a change to TEF for ∑HxCDD and ∑PeCDF (Table 4-16).

$TEQ_{DF,B}$ concentrations in eggs were calculated for prey consumption only, as well as with the inclusion of incidental sediment ingestion. The role of incidental sediment ingestion was evaluated under both baseline and post-TCRA conditions. In the post-TCRA exposure evaluation, concentration of dioxins and furans in the foods of birds were not changed from the pre-TCRA (baseline) scenario. This model was used only to estimate egg concentrations in the cormorant, heron, and sandpiper. Concentrations of dioxins and furans in the foods of killdeer were estimated using soil concentrations (Section 4.3.1.5.2 and Appendix D), and were regarded as an insufficiently robust foundation for further modeling to estimate egg concentrations. No estimates of killdeer eggs were prepared.

In response to USEPA comments on the draft of this report, example calculations showing each step and each parameter used in each example were prepared, and are presented as Exhibit 2A and 2B. Also in response to comments (Appendix F), estimates for egg concentrations were added for background conditions for sandpipers and herons consuming prey and shoreline sediments. Results of the TEQ calculations using the regression models from Elliott et al. (2001) to estimate concentrations in eggs of the neotropic cormorant, the great blue heron, and the spotted sandpiper are provided in Table 4-17.

The original models developed by Elliott et al. (2001) were based on concentrations in prey of piscivorous birds. Application of models to predict egg tissue concentrations from a mixture of different media including both prey and sediment is associated with some uncertainty. This uncertainty is discussed in Section 7.

4.3.2.2 Estimating PCB Concentrations in Bird Eggs

Although there is a wealth of literature on the biomagnification of PCBs from dietary sources into bird tissues, there have been few studies documenting specific biomagnification relationships for dioxin-like PCBs from foods of birds to their eggs. No studies were found that provide regression models of egg tissue on fish tissue or other food concentrations for dioxin-like PCB congeners. As a result, prey-to-egg BMFs are used to estimate dioxin-like PCB congeners in bird eggs. Moreover, no one study provides BMF for all dioxin-like PCB congeners, and no set of studies provides data for the full suite of dioxin-like PCB congeners for any one bird species. Given the uncertainties that already result from the use of BMFs, combining BMFs for different species across different studies to generate a suite of BMFs for all dioxin-like PCB congeners was considered prohibitively uncertain. Instead, estimates of concentrations of only a subset of dioxin-like PCB congeners in bird eggs are developed and presented in this BERA. The result underestimates the role of PCBs in risks to birds, but a means to comprehensively address dioxin-like PCBs in bird eggs was not available. The degree of the underestimate is likely small, because the selected congeners are those with the highest TEFs.

4.3.2.2.1 Overview of Literature Found

On the basis of the available literature, prey and egg data or congener-specific BMFs were extracted from the literature to estimate concentrations of selected PCB congeners in bird eggs. Although several papers address the PCB congeners with the highest dioxin-like potency (PCB77, PCB81, and PCB126), two of these congeners were detected rarely in sediments collected from the Site (Table 4-18), and in some cases, were not detected at all outside of the original 1966 perimeter of the impoundments north of I-10. Six of the 12 dioxin-like PCB congeners were ultimately selected for modeling egg concentrations: the three with the highest TEFs regardless of detection frequency: PCB77, PCB81, and PCB126; and three with relatively high detection frequencies in Site sediments and relatively high TEFs: PCB105, PCB114, and PCB118. Of the six selected congeners, concentrations of four of them correlate with concentrations of TCDD and TCDF in Site sediments (Integral 2011c), so measures to address risks from dioxins will address these congeners. Those that do not correlate with TCDD and TCDF were rarely detected in sediments.

Three sets of BMFs were used in this evaluation, to reflect the three different bird receptor surrogates. BMFs for herring gull were taken from Braune and Norstrom (1989). Braune and Norstrom (1989) data include only a limited set of PCB congeners, and only two of the dioxin-like PCBs, PCB105 and PCB118. Results were considered analogous to eggs of the omnivorous cormorant (Table 4-18). Braune and Norstrom (1989) did not provide BMFs for all relevant congeners, but they provide data for three tetrachlorinated PCB congeners and five pentachlorinated PCB congeners. To estimate BMFs for the congeners selected, an average of BMFs within these homologue groups was calculated, and applied to the PCB congener within the same homologue group. This was necessary to estimate BMFs for PCB77 and PCB81 (tetrachlorinated PCBs) and for PCB114 (a pentachlorinated PCB) (Table 4-18).

Congener-specific BMFs for the gray heron (*Ardea cinerea*), a bird nearly as large as the great blue heron, and for the kingfisher (*Alcedo atthis*), a smaller piscivore, were compiled by Naito and Murata (2007) from Murata (2003) and Murata et al. (2003), and used to represent the great blue heron and sandpiper, respectively. Although important differences from the receptor surrogates are recognized, the results are considered to reflect general estimates of $TEQ_{P,B}$ for the various bird eggs, and to generally represent the variability in this parameter.

Selection of data for input to the BMF models was conducted in the same manner as selection of data for input into the regression models for dioxins and furans: An individual sample of each medium was selected to represent the CT and RM exposures. To do this, the CT and RM $TEQ_{P,B}$ concentrations of each medium within each exposure unit were calculated. The actual sample with the $TEQ_{P,B}$ concentration closest to the CT and the sample with the $TEQ_{P,B}$ concentration closest to the RM were identified. The specific congener concentrations within these samples were used in subsequent calculations.

Similarly, the prey-weighted average concentration of each PCB congener for the total mass ingested by each bird receptor was calculated, using the same approach used to compute the final input for the dioxin and furan egg model. Once a total ingested concentration of the PCB congener was calculated, it was multiplied by its respective BMF (Table 4-18) and the resulting TEQs were summed for a total $TEQ_{P,B}$ concentration. All TEF values are presented on a ng/kg ww basis, in Table 4-19. At the request of USEPA in its comments on the draft BERA, a series of examples of these calculations for each requested combination have been prepared and are presented in Exhibit 2A and 2B.

Given the manner in which the BMFs are derived, the range of studies used to provide parameter estimates, the variety of analytical methods and the general uncertainties associated with the use of BMFs, results of these calculations should be regarded as general estimates, useful only to provide perspective on the relative importance of PCBs in risks to birds across a range of bird species and feeding guilds. Results underestimate the $TEQ_{P,B}$ concentration in bird eggs because not all congeners could be modeled. Results provide a perspective on the relative importance of PCBs in TEQ risk to birds under the baseline condition, with and without the influence of sediment ingestion for cormorants, herons, and sandpipers as well as under post-TCRA and background conditions (cormorant only).

4.3.2.3 Egg Exposure Scenarios

A total of five different scenarios were modeled to determine baseline risks to birds from dioxins, furans, and PCBs. The details of each scenario are:

- **Prey:** Ingestion of prey is the only source of dioxins and furans or PCBs and follows ingestion parameters detailed in Section 4.3.1. Evaluation of exposure to dioxin-like chemicals via prey ingestion is useful only for determining the relative importance of sediment exposures.
- **Prey and sediment:** Results of this analysis represent the baseline exposure assessment for this line of evidence. The sediment ingestion rate for sandpipers is appreciable, with lesser sediment ingestion rates for great blue heron and cormorants (Table 3-12). In all birds, sediment ingestion will contribute to the overall intake of dioxins, furans, and PCBs. This scenario takes sediment ingestion into account and uses the same exposure assumptions and exposure units as for the wildlife exposure model (Section 4.3.1). Cormorants were assumed to ingest sediment from 0 to 6 inches from the aquatic and shoreline areas of the Site, excluding sediments from the western cell of the impoundments. All shoreline sediment (0 to 6 inches) samples for the site were included for the great blue heron and sandpiper.
- **Prey and sediment (post-TCRA):** For analysis of post-TCRA exposure, all samples for stations within the original 1966 impoundment perimeter are removed and replaced with a single CT or RM value of TEQ_{DF} or TEQ_{P} to represent the possible post-TCRA condition (Appendix C). Regression models were not available to estimate post-TCRA whole crab and whole catfish concentrations, so the baseline dataset for tissue

was used in models of the post-TCRA egg concentrations, resulting in a conservative assessment of post-TCRA exposure. Because there are no data for PCB congeners in shoreline sediments from upstream, post-TCRA estimates for $TEQ_{P,B}$ on the Site, which represents the post-TCRA sediment condition using median background concentrations, could not be made for the herons and sandpipers. Upstream benthic sediments were included in background calculations for cormorant.

- **Background (prey only):** For comparison with the Site, background exposures were modeled. Background analysis used all tissue data collected from the background areas. The CT and RM of TEQ_{DF} and TEQ_{P} from the background dataset were used for data selection and were determined independently from upstream data (Appendix C).
- **Background (prey and sediment):** Background egg concentrations were also estimated including ingestion of both prey and sediment. Tissue and sediment (0 to 6 inches) data from upstream background study were used. The reference TEQ_{DF} or TEQ_{P} used for data selection was calculated independently for background conditions (Appendix C). Because there are no data for PCB congeners in shoreline sediments from upstream, background estimates with exposure to PCBs from both sediment and prey could not be made for the herons and sandpipers. Upstream benthic sediments were included in background PCB calculations only for the cormorant.

The regression models used for the dioxin and furan estimates and the BMFs used for the PCB calculations were based on concentrations in prey of piscivorous birds. Modeling using regression equations or BMFs based on exposure via fish ingestion to predict egg tissue concentrations from exposure via a mixture of different media including both prey and sediment is associated with some uncertainty. This uncertainty is discussed in Section 7.

4.3.2.3.1 Data Used to Estimate Bird Egg Concentrations

Data used was identical to that used in the wildlife exposure model detailed in Section 4.3.1 with the exception that no soil data were included, all tissue data were used on a wet weight basis (sediment values were dry weight) and all scenarios were tested regardless of resultant HQ values.

4.3.2.3.2 Results of Bird Egg Models

Estimated $TEQ_{DF,B}$ concentrations in the eggs of the neotropic cormorant, great blue heron, and spotted sandpiper are shown in Table 4-17. Estimates of $TEQ_{P,B}$ concentrations in the eggs of the neotropic cormorant, great blue heron, and spotted sandpiper are shown in Table 4-19. The $TEQ_{DFP,B}$ for eggs of each receptor, showing the relative importance of PCBs and dioxins and furans for each scenario, are provided in Table 4-20.

4.4 Probabilistic Exposure Assessment

Probabilistic exposure assessment was performed for receptors whose estimated exposure for one or more $COPC_Es$ equaled or exceeded the associated LOAEL in the deterministic risk assessment (i.e., spotted sandpiper, killdeer, and marsh rice rat). The probabilistic exposure assessment involved assigning probability distributions to certain exposure parameters to yield a probability distribution for $COPC_E$ exposure. This exposure distribution was then compared to the TRV, and the likelihood that exposure exceeded the TRV (under the assumptions used) was determined. Exposure distributions were developed for the site as a whole for each relevant receptor–$COPC_E$ pair.

Probabilistic analyses of exposure and risk were developed using Oracle® Crystal Ball software (Gentry et al. 2005). Crystal Ball employs Monte Carlo analysis, a commonly used probabilistic numerical technique where the uncertainty and variability in exposure (and HQ) estimates are characterized by estimating the exposure (and HQ) distributions. To develop each exposure distribution, the exposure estimate for a receptor–$COPC_E$ pair is repeatedly calculated by Crystal Ball, with each iteration of the exposure model using different sets of parameter values determined by random sampling of the probability distributions for those input parameters treated probabilistically (USEPA 2001). Those parameters modeled probabilistically and the means used to estimate the exposure probability distributions for applicable receptor–$COPC_E$ pairs are discussed further below.

4.4.1 Parameters to be Estimated in a Probabilistic Analysis

Certain receptor-specific exposure parameters identified on the basis of life histories, and Site-specific EPCs used in the wildlife exposure model were treated probabilistically to increase understanding of ecological risk. Parameters treated with probability distributions included EPCs, body weight, feeding rate, prey fraction for each prey type, water ingestion rate (when relevant), and rate of incidental ingestion of soil or sediment. Additional life history information from the literature was required to perform the probabilistic analyses, because parameter estimates for the probabilistic analyses require measures of variance and range. Results of the information search to obtain the required data are presented in Table 4-21.

Because EPCs associated with terrestrial prey for upland receptors were estimated using BAFs from Site soils, the contribution of prey in these cases is also dependent on the underlying Site soil data used to derive the prey component of the diet. Therefore, $COPC_{ES}$ in terrestrial invertebrate tissue were varied probabilistically with soils.

4.4.2 Derivation of Parameter Distributions

To derive parameters for distributions of Site-specific EPCs, relevant $COPC_E$ soil and sediment concentration data were compiled and imported into Crystal Ball for distribution goodness-of-fit testing. Goodness-of-fit testing employed Anderson-Darling, Chi square, and Kolmogorov-Smirnov analyses for ranking the fit of each $COPC_E$ dataset against 14 available distribution types. Distributions selected by Crystal Ball for each dataset were compared to distributions selected by other means (e.g., R [R Development Core Team 2008]), if available. If no other distribution information was available for a given dataset, only Crystal Ball was used to evaluate its distribution. If the distribution recommended by Crystal Ball for a particular dataset differed from the recommendations of other software programs, professional judgment was used to select the best fitting dataset distribution. Following selection of an appropriate distribution for each soil or sediment concentration dataset, distribution parameters were estimated by Crystal Ball and incorporated into the probabilistic model.

To derive parameter distributions for life history parameters, the CT and range of each parameter were determined from the literature. Where assignment of a normal distribution was appropriate (e.g., for body weight of a receptor), the mean and standard deviation were derived from the literature. For the prey fraction for each prey type and the rate of incidental ingestion of soil or sediment, a triangular or uniform distribution was assigned using the estimates for the CT and the range (only the range was needed in the case of the uniform distribution). For these exposure model terms, professional judgment was used to derive the range when data were not readily available in the literature. A normal distribution is defined by a mean and a variance (or standard deviation). A triangular distribution is defined by a mode, a minimum, and a maximum. A uniform distribution is defined by a minimum and a maximum. The triangular and uniform distributions are used when information is limited and the form of the distribution is unknown. For feeding rate and water ingestion rate, allometric equations were applied to determine the appropriate value corresponding to the body weight value randomly selected during a given iteration of the Monte Carlo.

Distribution characteristics used in probabilistic risk analysis are summarized in Appendix C for EPCs and in Table 4-21 for other exposure parameters.

5 EFFECTS CHARACTERIZATION

Lines of evidence in this BERA employ both TRVs, which are intended to denote no-effects/effects thresholds for survival, growth, and reproduction of individuals; and benchmarks such as the AWQC, considered protective of a broader group of taxa (e.g., aquatic macroinvertebrates or aquatic communities). Detailed information on the methods used and data considered in selection or derivation of NOAEL and LOAEL TRVs and benchmarks used in this BERA is provided in Appendix B. This section provides an overview of the types of TRVs and benchmarks used in calculating HQs, methods to aggregate toxicity data, approaches to selection of each TRV needed, the general meaning of different types of TRVs or benchmarks, and types of uncertainties common to these approaches.

5.1 Types of Toxicity Information Used

Selection of TRVs and benchmarks for use in ecological risk assessment involves consideration of several factors: types of receptors under evaluation and assessment endpoints for each, whether the analysis calls for a screening or a more realistic risk description, the data and methods available for estimating exposure to receptors, and the availability of toxicity information that meets basic data quality standards. To address all of the lines of evidence for each receptor to be used in this BERA, effects measures consisting of TRVs or benchmarks expressed in the following terms are needed:

- Bulk sediment concentrations (mg/kg) that are protective of the benthic macroinvertebrate community
- Concentrations in water (mg/L) that are protective of benthic macroinvertebrate communities and fish
- Concentrations of metals in media ingested by fish (mg/kg)
- CTR values for TCDD (or other organic compounds) as concentrations in whole fish (mg/kg ww)
- CTR values for TCDD (or other organics) expressed as concentration in whole clams (mg/kg ww)
- CTR values for dioxins, furans and PCBs expressed as a TCDD or TEQ concentration in eggs of birds (ng/kg ww)
- Daily ingested dose NOAELs and LOAELs (mg/kg bw-day) for reptiles, birds, and mammals for all COPCs.

When using published toxicity literature to establish measures of effect, the specific meaning of the effects measure depends on the experimental design used and the test endpoints. For example, a toxicity study may provide a threshold dose above which a reduction in the hatchability of bird eggs occurs, or a reduction in the growth of juveniles. Exceedance of TRVs from such studies would have different meanings to the risk assessment. In cases where the estimated exposure to an ecological receptor is greater than an LOAEL and risk to the receptor cannot be considered negligible, the specific endpoint represented by the TRV or benchmark is considered in the description of risk. A risk estimate based on a TRV denoting an LOAEL for effects on survival is interpreted to have a potentially more severe effect on the receptor than exceedance of a TRV denoting an LOAEL for individual growth rate or reproduction.

In some cases, the application of an uncertainty factor to conservatively estimate the benchmark or TRV was required (e.g., Table 5-1). In a review of the types and uses of uncertainty factors, Chapman et al. (1999) conclude that an uncertainty factor should account for the uncertainty in the extrapolation, but should not be so large that it renders the resultant value meaningless for assessing risk. Although uncertainty associated with estimating an NOAEC from an LC_{50} [median lethal concentration], which was required for this risk assessment in some cases, may be substantial, Chapman et al. (1999) do not support the use of uncertainty factors greater than 10. They also clearly avoid specific recommendations for uses of uncertainty factors, focusing instead on general technical considerations in their use, and point out that their use does not specifically resolve uncertainty, it can only compensate for a lack of empirical information. Chapman et al.'s (1999) discussion is summarized in Appendix B, and related uncertainties are addressed in Section 7.

5.2 Methods Used for Aggregation of Toxicity Data

As described in Appendix B, many TRVs used in this risk assessment were those presented in compendia of values prepared by federal agencies (e.g., Sample et al. 1996; T&N Associates 2002; USEPA 2005a) or from USEPA-approved, final risk assessments conducted for other CERCLA sites. In most cases, the final selected TRV (NOAEL or LOAEL) was either the geometric mean of data from studies of acceptable quality (e.g., the TRVs developed by USEPA and others for the ecological soil screening levels [EcoSSLs]), or in cases where insufficient information was available to calculate a geometric mean, the TRV was the lowest LOAEL and the highest NOAEL from among studies of acceptable quality. If the highest NOAEL was greater than the lowest LOAEL, then the highest NOAEL that did not exceed the lowest LOAEL was selected.

This approach results in fairly conservative estimates of toxicity and a fairly protective risk assessment overall. For dioxins and furans, and for PCBs because the toxicity of some congeners is considered to be additive with that of TCDD, the relatively extensive literature available was reviewed in greater detail. In both cases, more than one TRV of acceptable quality are often available for certain species. For example, there are several studies of PCB toxicity in mink, and there are several studies of TCDD toxicity to birds following injection into eggs. In cases such as these, if fewer than 10 values with a common endpoint and route of administration were found, the following steps were taken to derive a TRV, for example, a LOAEL:

1. Within-species LOAELs are grouped
2. The geometric means of the within-species LOAELs are calculated
3. Resulting geometric mean LOAEL values are pooled. No individual species is represented by more than one value, although some values are the results of only one study.
4. The geometric mean of the pool of data for multiple species is calculated, and that value becomes the LOAEL for the $COPC_E$ and receptor.

NOAELs were treated in the same way in cases of more than one acceptable study. This approach is consistent with calculation of TRVs for use in development of EcoSSLs (USEPA 2005a), and generally consistent with derivation of the benchmarks used. It results in values that are both representative of multiple taxa within broad categories of receptors, and reasonably conservative without being overly so.

The RI/FS Work Plan indicates that cumulative distribution functions derived from multiple effects-level metrics within a species, or SSDs, would be developed using multiple literature values for several species. This is a tool that can be used to clearly define the risk and the uncertainty associated with a risk calculation. However, sufficient data for a set of related taxa that have similar exposure and effects metrics were not found, except for the SSD for early life stage fish developed by Steevens et al. (2005). This SSD was the only one used in this BERA.

5.3 Benthic Macroinvertebrate Communities

For most $COPC_{ES}$, risks to benthic macroinvertebrate communities were estimated using benchmarks, either sediment quality guidelines (SQGs) or AWQC. SQGs expressed as bulk sediment concentrations for marine and estuarine environments were derived by Long et al. (1995) using data for a large number of contaminated sediment sites.

Although other sources of marine SQGs are available (MacDonald et al. 1996) and may be more robust on the basis of the methods used for their derivation, Long et al. (1995) is the same source of information used by TCEQ in establishing sediment screening benchmarks for benthos. TCEQ interprets sediment chemistry in terms of risk to benthic invertebrate communities relative to Long et al.'s (1995) sediment benchmarks as follows:

- The effects range-low (ER-L) values are concentrations below which adverse effects on benthic communities rarely occur
- The effects range-median (ER-M) values are concentrations above which adverse effects on benthic macroinvertebrate communities are "probable"
- At concentrations between the ER-L and ER-M, adverse effects on benthic invertebrates are considered possible.

Although Long et al.'s (1995) ER-L and ER-M values have technical flaws (e.g., Sampson et al. 1996a, 1996b; Becker and Ginn 2008), they are regarded by TCEQ as protective of benthic communities. Therefore, in this risk assessment and consistent with the role of SQGs as screening benchmarks, ER-Ls were used to identify $COPC_{ES}$ and stations posing negligible risk to benthic macroinvertebrate communities. When concentrations of a $COPC_E$ in sediment exceeds its respective ER-M value, the number of exceedances and area involved are considered to determine whether additional toxicity information is warranted to better describe risk.

AWQC are derived using a minimum dataset for at least 8 major aquatic taxa including invertebrates, fish, algae and vascular plants (USEPA 1985). AWQC are expressed as the criterion maximum concentration (CMC) for evaluation of short term (1-hour) spikes in chemical concentrations, and the criterion continuous concentration (CCC), a concentration that can be present for long periods with no adverse effects. These criteria are considered to be protective of 95 percent of aquatic species when concentrations of a chemical are not present in surface waters above the CMC and CCC within their specific time limits. The CCC was used for comparison to estimated concentrations in porewater for those chemicals lacking ER-L and ER-M values.

Neither type of benchmark is available for carbazole, phenol, BEHP, barium, cobalt, or manganese. For these chemicals, information searches were conducted as described by Appendix B, primarily using USEPA's ECOTOX database. The ECOTOX database includes results of studies that could be used to derive water quality criteria if all of the required taxa were represented. For the most part, available data included LC_{50} concentrations, and uncertainty factors were used to derive concentrations below which no effect was expected. Details are provided in Appendix B.

Attachment B2 to the RI/FS Work Plan provides a detailed summary of studies testing the toxicity of dioxins and furans to benthic macroinvertebrates, and that information is presented again in Appendix B to this document (summarized in Table B-4 of Appendix B). From this information, a no-observed-adverse-effects concentration (NOAEC) for TCDD in sediments was derived as the geometric mean of all NOAECs found in the literature for a wide range of invertebrate taxa.

In addition, a CTR for interpretation of reproductive risk to molluscs was found, and is included as the TRV for comparison to concentrations in clams. Wintermeyer and Cooper (2007) report that at 2 ng/kg in female eastern oysters (*Crassostrea virginica*), initial development of follicular structures and oocytes were notably different than in controls. Males had normal gametogenesis at this concentration. At 10 ng/kg ww, marked effects on gonad development and gamete maturation relative to controls were observed in both male and female oysters, as well as morphological lesions in females leading to resorption of oocytes. Effects were more evident at lower doses in females and were more pronounced in both males and females at 10 ng/kg than at 2 ng/kg. Cooper and Wintermyer (2009) report that in clams, the majority of TCDD is found in tissue of gonads 28 days after exposure, supporting the observation that TCDD affects these tissues. Other publications by these authors provide added detail. Wintermyer and Cooper (2003) present both a field study and a laboratory experiment with the eastern oyster. Although they report that 2 ng/kg is also associated with reduced veliger larval survival, this may overstate the effect of TCDD in the field study, because Wintermyer and Cooper (2003) used test subjects that were field-collected and field-exposed. Exposures related to their tests occurred in an urban estuary in New Jersey, but Wintermyer and Cooper (2003) did not document exposures to other environmental pollutants there, which could include PAH, estrogenic compounds, and physical stressors such as siltation. Therefore, the effects levels they report from their field study could overestimate the role of TCDD. However, Wintermyer and Cooper (2003) also exposed oysters to TCDD (without other chemicals) in a controlled experiment, and found reduced egg fertilization success, and reduced larval survival at the lower concentration in the adult tissue (2 ng/kg ww). Therefore, although the field study cannot account for the effects of the chemical mixtures, the laboratory study reported in this paper demonstrates that 2 ng/kg ww in whole eastern oyster tissue causes reduced fertilization of eggs and reduced larval survival in eastern oysters.

For this BERA, 2 ng/kg is considered the LOAEL for effects on reproduction in individual molluscs, as required by USEPA in comments (Appendix F). Because this tissue concentration is associated with a small but measureable histological effect that occurred only in females, reduced egg fertilization and reduced larval survival, this is a conservative TRV. A corresponding NOAEC was not available, and was not estimated. CTRs or other types of TRVs for other dioxin and furan congeners were not found.

More detailed information on the results of literature searches, derivation of TRVs, and all benchmarks is provided in Appendix B, and a summary of selected values is in Table 5-1.

5.4 Fish

The effects characterization for fish involved use of TRVs expressed as concentrations in foods of fish and in water For most metals, TRVs for interpreting concentrations in foods of fish were selected from literature reviews generated for ecological risk assessments approved for other CERCLA sites. Recent risk assessments for Portland Harbor in Portland, Oregon, and the Lower Duwamish Waterway in Seattle, Washington, provide extensive literature reviews. TRVs for BEHP and nickel expressed as concentrations in food were not available. Results of an acute toxicity test with sheepshead minnow were multiplied by an uncertainty factor to derive a no-effects concentration of BEHP for fish expressed as a concentration in water. For nickel, the results of tests with marine fish were combined to determine a chronic TRV for nickel expressed as a concentration in water. Details are provided in Appendix B. A summary of results of reviews to identify TRVs and benchmarks for fish is provided in Table 5-2.

To address the potential effects of dioxins and furans on fish, Integral used results of a study by Steevens et al. (2005). Steevens et al. (2005) developed an SSD to describe the toxicity of TCDD to several fish species, compiling multiple studies of TCDD and dioxin-like compounds with salmonids and other teleost fishes expressed as concentrations in fish eggs or embryos, a life stage that is sensitive to the effects of TCDD. Concentrations selected were those associated with no-observable effects, or the lowest concentration producing observable effects on egg survival. Steevens et al. (2005) selected the lowest paired effect levels available for a given species, calculated geometric means of the no-effect and lowest observable effect residue concentrations, and used the resulting 10 data points to derive the SSD. This risk assessment uses TEQ_F concentrations in whole body samples of fish for comparison to the CTRs of Steevens et al. (2005). This approach is conservative. Tietge et al. (1998) found that TCDD concentrations in eggs of brook trout (*Salvelinus fontinalis*) were 39 percent of the concentrations in the whole fish. Heiden et al. (2005) reported an even lower level of egg accumulation of TCDD relative to female whole bodies in zebrafish, with egg concentrations of just 5 percent of whole adults. This risk assessment is conservative because it assumes a 1 to 1 ratio of whole adult fish to egg concentrations. Additional details are provided in Appendix B, including data used by Steevens et al. (2005).

5.5 Reptiles

Integral conducted a literature review to identify toxicity information useful for evaluation of risk to reptiles; details of the search methods and resources used are provided in

Appendix B. The majority of available studies report chemical concentrations in field-collected specimens, and provide no means of interpreting exposure in terms of the potential for harmful effects. There are studies describing the concentrations of PCBs, dioxins, and furans in tissue of turtles in which the authors evaluate correlations of chemical concentrations with embryo deformities. However, the presence of other chemicals in the animals studied, including organochlorine pesticides, confounds interpretation, and TRVs for reptiles could not be derived.

Some risk assessments have addressed this data gap by assuming that birds are an appropriate model for reptiles, and that TRVs derived for birds can be used to interpret exposure to reptiles. A recent publication by Weir et al. (2010) examines this assumption by comparing results of controlled laboratory tests on birds and reptiles for chemicals for which representatives of both groups have been tested, which are mostly pesticides and ordnance compounds (explosives). Weir et al. (2010) find that reptiles were more sensitive than birds in 5 of 15 cases and less sensitive in 3 of 15. The rest of the comparisons (7 of 15) were inconclusive, or birds and reptiles were approximately equivalent.

For these reasons, the absence of reptile-specific toxicity studies for the $COPC_{ES}$ at this Site and the uncertainties about their sensitivities relative to other receptors, this document does not specify TRVs or benchmarks for interpreting estimated reptile exposures. Risks to reptiles are addressed qualitatively, by considering their estimated exposures relative to exposures by other receptors, and by considering the overall patterns in risk estimates observed for the other receptors.

5.6 Birds and Mammals

Lines of evidence used to evaluate risk to birds and mammals include comparison of estimated daily ingestion rates for individual $COPC_{ES}$ at the Site to TRVs expressed in the same terms. Comparison of estimated concentrations of $TEQ_{DF,B}$ and $TEQ_{P,B}$ in bird eggs is also used to evaluate risks to birds. The methods to identify measures of effect for both of these lines of evidence are detailed in Appendix B and summarized below. Results of the process to identify and select TRVs for birds are summarized in Table 5-3; TRVs for mammals are summarized in Table 5-4.

5.6.1 Daily Ingestion Rate

The primary literature available to interpret estimated daily ingestion rates of COPC$_{E}$s at the Site is highly variable in terms of age, quality, numbers and types of species studied, depth and completeness. Moreover, many ecological risk assessments have previously been conducted at CERCLA sites and these tend to draw from the same sets of studies, although there are some differences in the data quality considered acceptable among sites. Finally, USEPA and related federal agencies have compiled toxicity data for use in risk assessment (e.g., Sample et al. 1996) and for development of EcoSSLs (USEPA 2005a). For all of the COPC$_{E}$s except dioxins, furans, and PCBs, this BERA initially draws from compendia of literature prepared by USEPA or affiliates, including Sample et al. (1996) and USEPA (2005; 2012), and two recent BERAs accepted by USEPA for CERCLA sites in Portland, Oregon, and Seattle, Washington. These were considered a reasonable starting place for identification of wildlife TRVs. The general literature accessible through standard search tools like PubMed, Biosis, Google Scholar and others was also consulted when established TRVs were lacking. A detailed description of how these resources were used is provided in Appendix B.

Although the assessment endpoints for this BERA are expressed in terms of populations, the vast majority of literature including studies employed by prior risk assessments address endpoints on the level of the individual organism. The types of individual effects measures derived from the literature for this BERA were limited to those clearly relating to population-level effects, generally the survival, growth, and reproduction of tested individuals. Effects on reproduction are interpreted to include developmental effects, when it is clearly related to the reduced survival of young. Studies addressing unrelated endpoints (e.g., cellular or biochemical alterations or modified gene expression) were not used to establish TRVs for the BERA, because these effects cannot be related to population-level assessment endpoints.

5.6.2 Egg Concentrations

Use of egg-exposure based TRVs is the recommended approach to risk assessment for birds by both TN & Associates (2002) and USEPA (2003b). USEPA (2003b) provides a compilation of results of toxicity tests in which exposures as concentrations in eggs were documented, building on the detailed literature review conducted by TN & Associates (2002) for USEPA's Office of Research and Development. Both laboratory and field studies were compiled by USEPA (2003b). A paper was only selected for use in USEPA's (2003b) analysis if it included all of the following:

- Evaluation of more than one quantitative dose or exposure level. Studies evaluating only one dose or exposure level were considered to have too much uncertainty.
- One or more quantifiable toxicological endpoint.
- Appropriate statistical tests showing significant changes in response with changes in dose or exposure levels.
- Evaluation of the potential for co-contaminants to affect results (for field studies).

USEPA's (2003b) compilation of TRVs expressed as TCDD (or TEQ) concentrations in eggs includes NOAELs for developmental impairment from laboratory studies ranging from 66 ng TEQ/kg egg for the chicken to 50,000 ng TEQ/kg egg for several other bird species, including two gull species, the Graylag goose, and the goldeneye (a duck). Corresponding LOAELs range from 150 to 4,400 ng TEQ/kg egg. Integral did not use all of these studies for developing egg tissue TRVs, as discussed in Appendix B.

Because of the selection criteria used by USEPA (2003b), Integral used studies compiled by USEPA (2003b) as a starting point. Toxicity data selected for interpretation of estimated bird egg concentrations were taken only from controlled laboratory studies in which TCDD was injected into yolks during the earliest stages of embryo development. Because there is known to be substantial inter- and intraspecies variability in response to TCDD and other dioxin-like compounds, and because there is evidence that the existing TEFs for birds may not fully describe the relative toxicity of various dioxin-like compounds (e.g., Cohen-Barnhouse et al. 2011), egg toxicity studies with other dioxin-like compounds were not used. To do so would have introduced variability in the estimate of toxicity to bird eggs with unknown effects and uncertainties. Finally, for development of the final TRV, only studies performed using yolk injection were used, because TCDD transferred from hens to eggs occurs only in the yolks (Nosek et al. 1992a). Selected TRVs from egg yolk injection studies are summarized in Table 5-5. Data from studies in which TCDD is injected into the albumin or the air cell were compiled and are discussed, but were not incorporated into the final TRV for eggs. Details on those studies, and relevant field studies are discussed and presented in Appendix B.

6 RISK CHARACTERIZATION

Risk characterization combines the information developed in the exposure and effects characterizations to provide quantitative and qualitative descriptions of the likelihood that hazardous materials at the Site are causing adverse ecological effects under the baseline condition for the Site. According to USEPA (1997) guidance, risk characterization should also present information important to interpreting risks (USEPA 1997).

Each risk question represents an independent line of evidence that was applied to address risks to each receptor. All lines of evidence involve the evaluation of whether estimated exposures on the Site exceed an exposure level or concentration associated with effects (Table 3-11). Factors contributing to interpretation of the exceedance include the adverse effect(s) represented by the benchmark or TRV exceeded, and the type of threshold exceeded (i.e., LOAEL, NOAEL, EC_{10}), and the quality of the toxicity data used. This section presents the results of these basic comparisons, and at the request of USEPA in comments on the draft BERA (Appendix F), includes a section devoted to evaluation of risks to threatened and endangered species and a discussion of bioaccumulation. These discussions are followed by a section providing analysis of uncertainty. The final section of this document provides a summary statement of risk that incorporates all lines of evidence for a given receptor to address risk questions, and addresses qualitative and/or quantitative analysis of uncertainty for each receptor.

6.1 Overview of Risk Characterization

As described in Section 3.8.1, this BERA uses a tiered approach to the analysis and characterization of risks: an initial assessment of risk is performed deterministically for each receptor–$COPC_E$ pair. The initial assessment is a reasonable worst case evaluation, resulting in an HQ for each receptor–$COPC_E$ pair. For each receptor–$COPC_E$ pair, subsequent analyses depend on the value of the HQ_L, with one of the following possible outcomes (Figure 3-4):

- Risk to individuals of any receptor from any $COPC_E$ to which the receptor is exposed at a level lower than the NOAEL (i.e., $HQ_N < 1$) is characterized as negligible.
- Risk to individuals of any receptor from any $COPC_E$ to which the receptor is exposed at a level between the NOAEL and LOAEL (i.e., $HQ_N > 1 > HQ_L$) is characterized as very low, depending on the toxicity data supporting the NOAEL and LOAEL values

- Risk to individuals of any receptor from any $COPC_E$ to which the receptor is exposed at a level higher than the LOAEL (i.e., $HQ_L > 1$) is considered to be present. Risk to the assessment endpoint, which may be a population or community, is evaluated and discussed further in the context of the data supporting the TRV.

An HQ_L equal to or greater than 1 is interpreted to indicate a need for further evaluation of risk to the receptor using refined methods (e.g., more realistic exposure assumptions or probabilistic analysis) and/or additional data, and is considered in context of the specific toxicity information used to derive the TRV. In this case, subsequent analyses include:

- A probabilistic exposure evaluation
- Evaluation of post-TCRA risk
- Consideration of background.

For avian receptor–COPC pairs that are surrogates for protected species, potential exposures evaluated according to the method described in Section 4.3.1.6 are discussed in Section 6.7.

Deterministic COPC and receptor-specific HQs were calculated for the initial evaluation of risk, as described in Section 3.8. Methods to perform the probabilistic exposure analysis are presented in Section 4.4, and results are provided below for those receptor–$COPC_E$ pairs for which the deterministic HQ analysis suggests a potentially unacceptable risk. Evaluation of population-level risks is addressed qualitatively, and incremental risk relative to background is evaluated when the HQ_L is equal to or greater than 1.

6.2 Risks to Benthic Macroinvertebrate Communities

A summary of results for each line of evidence to assess risk to benthic macroinvertebrate communities is provided in this section.

6.2.1 $COPC_E$s in Sediment Relative to Benchmarks and the TCDD NOAEC

$COPC_E$s that were evaluated by comparing their concentrations in individual sediment samples with SQGs include copper, lead, mercury, and zinc. Concentrations of TCDD were compared to the NOAEC for sediments. Results are summarized in Table 6-1, and below:

- Copper does not exceed the ER-M at any sampling station. Copper exceeded the ER-L at one station within the 1966 perimeter of the northern impoundments, and at two stations adjacent to the Southwest Shipyard property, south of I-10 and to the east of the southern peninsula (Figure 6-1).
- Lead does not exceed the ER-M at any sampling station. Lead exceeded the ER-L at two stations south of I-10 and to the east of the southern peninsula (Figure 6-2). One station exceeding the ER-L is adjacent to the Southwest Shipyard property; the other in a shoreline sample across the channel, on the east bank of the San Jacinto River.
- Mercury does not exceed the ER-L or ER-M in any location outside of the 1966 impoundment perimeter. Mercury concentrations exceed the ER-L in four locations, and exceed the ER-M in two locations within the impoundment perimeter (Figure 6-3).
- Zinc does not exceed the ER-M at any sampling station. Zinc exceeds the ER-L at one station within the 1966 perimeter of the northern impoundments, and at two stations adjacent to the Southwest Shipyard property, south of I-10 and to the east of the southern peninsula (Figure 6-4). Zinc exceedances of the ER-L occurred in the same samples as copper (Figure 6-1).
- The concentration of TCDD in two 0- to 6-inch sediment samples exceeded the no-effects level for sediments, at Stations SJB1 and SJC1, both within the footprint of the TCRA. The NOAEC was calculated as the geometric mean of no-effects concentrations from spiked-sediment bioassays with a range of invertebrate taxa including polychaetes, bivalves, insects, and molluscs with growth and mortality endpoints. Exceedances of NOAECs are not interpreted to indicate risk; Table B-4 in Appendix B shows no-effects concentrations ranging up to 25,000 ng/kg.

Results for this line of evidence indicate that risks to benthic macroinvertebrates from copper, lead, zinc, and TCDD are negligible because concentrations exceed no-effects levels in very few locations, and do not approach the effects threshold (ER-M).

Exceedance of the ER-M for mercury in two 0- to 6-inch sediment samples within the impoundment perimeter does not indicate a widespread risk to benthos from mercury. Exceedance of the ER-M is not predictive of effects, but is interpreted by TCEQ to suggest that adverse effects are probable. Given the very limited area within which mercury concentrations exceed the ER-M, risks to benthic macroinvertebrates due to mercury are considered very low to negligible.

6.2.2 Estimated Concentrations in Sediment Porewater Relative to TRVs

$COPC_{ES}$ that were evaluated by estimating porewater concentrations at individual sampling locations and comparing these to AWQC or available TRVs include BEHP, phenol, cobalt, manganese, and thallium. Results are summarized in Table 6-1, and below:

- Estimated sediment porewater concentrations of BEHP do not exceed the NOAEC for BEHP at any location (Figure 6-5).
- Estimated sediment porewater concentrations of phenol exceed the NOAEC for phenol at five locations (Figure 6-6). However, phenol was not detected in the sixteen of the eighteen 0- to 6-inch sediment samples shown in Figure 6-6; in the remaining two sediment samples, phenol concentrations are estimated (J- or UJ-qualified). Because porewater concentrations for organic compounds were estimated on the basis of OC-normalized concentrations in sediment, exceedances of the phenol NOAEC in porewater at all five locations is an artifact of the low OC content in these samples. Phenol was not detected in any of these five locations.
- Estimated sediment porewater concentrations of cobalt do not exceed the NOAEC for cobalt at any location (Figure 6-7).
- Estimated sediment porewater concentrations of manganese exceed the estimated no-effects concentration at 12 locations distributed around the Site (Figure 6-8). Three of those locations correspond to exceedances of ER-Ls for copper and zinc, including 1 within the 1966 impoundment perimeter, and two adjacent to and to the east of the Southwest Shipyards, and an additional location adjacent to the Shipyards. Other locations where this occurs are distributed randomly around the site.
- Estimated sediment porewater concentrations of thallium do not exceed the NOAEC for thallium at any location (Figure 6-9).

Results for this line of evidence indicate that risks to benthic macroinvertebrates from BEHP, cobalt, and thallium are negligible because concentrations do not exceed no-effects levels in any locations. Risks due to phenol are also negligible because phenol was generally not detected or could only be estimated in sediment. Whether manganese presents a risk to benthic invertebrates is uncertain, because the only TRV was a no-effects level; there is no corresponding effect level to enable interpretation of potential effects. Spatial correspondence of the relatively elevated manganese with concentrations of copper and zinc above ER-L values suggest that sediments in small areas of the impoundments north of I-10, and sediments adjacent to the Shipyards contain metals at concentrations elevated relative to very conservative screening levels, but below concentrations that indicate risk. Results for copper, magnesium, and zinc do not indicate unacceptable risk.

6.2.3 TCDD in Clam Tissue Relative to the Critical Tissue Residue for Molluscs

Composite clam samples were collected at five transects on the Site (Figure 4-1) and two transects upstream (Figure 4-2). Concentrations of TCDD in clams on the Site, where detected, ranged from 0.647 ng/kg ww (J-qualified) in Transect 6 to 17.6 ng/kg in a sample from Transect 3 (Table 6-2). The five clam samples from Transect 3, which is directly adjacent to the impoundments north of I-10, have the five highest concentrations of TCDD among all clam samples, ranging from 5.79 to 17.6 ng/kg. The next highest concentrations are at Transect 5, collected directly adjacent to the upland sand separation area to the west of the northern impoundments, where the maximum TCDD concentration in clam tissue was 2.43 ng/kg. TCDD concentrations in two of five clam samples collected from Transect 5 were greater than 2 ng/kg ww, the lower threshold of effects on reproduction in molluscs (Appendix B). Concentrations of TCDD in clam tissue are highest where sediment concentrations under the baseline condition are highest, consistent with the finding reported in the PSCR (Table 6-61; Integral and Anchor QEA 2012) that, for the tetrachlorinated congeners, concentrations in sediment correlate significantly and relatively strongly with those in clam tissue (i.e., tau-b values of 0.67 and 0.71 for TCDD and TCDF, respectively at $p < 0.05$).

The TRVs available to interpret tissue concentrations in molluscs are based on a series of studies in which oysters were injected with TCDD at various doses, and reproductive tissues were analyzed to determine if adverse effects on gametogenesis would result from TCDD exposure. In separate studies, clams were collected from the field and tissues observed, and were injected with radiolabeled TCDD to evaluate toxicity and bioaccumulation (Wintermeyer and Cooper 2003; Cooper and Wintermyer 2009). Wintermeyer and Cooper

(2007) report that at 2 ng/kg in female oysters, initial development of follicular structures and oocytes were notably different than controls. Males had normal gametogenesis at this concentration. At 10 ng/kg ww, effects were more pronounced, and were evident in both males and females. Female reproductive tissues were more sensitive to TCDD exposures; effects were more pronounced in both males and females at 10 ng/kg than at 2 ng/kg. Cooper and Wintermyer (2009) found that in clams, the majority of TCDD is found in tissue of gonads 28 days after exposure, supporting the observation that TCDD affects these tissues.

Cooper and Wintermyer (2009) also summarize other studies on this subject, including Wintermeyer and Cooper (2003), which involved field and laboratory components. In the field study, the authors transplanted adult eastern oysters to Newark Bay, the Arthur Kill area of Raritan Bay, and Sandy Hook, New Jersey. Results suggest that oysters with TCDD (ng/kg)/TCDF (ng/kg)/total PCB (μg/kg) concentrations of 3.2/2.1/68 and of 1.3/1.7/65 had reduced survival of veliger larvae. Conditions of this study are not analogous to conditions at the SJRWP because of the relatively high levels of PCBs in the oyster tissue, which could have been the cause of reductions in larval survival. Also, the field study reported by Wintermeyer and Cooper (2003) exposed test organisms in complex urban estuaries, where sediment and water quality are influenced by oil refineries, urban runoff, combined sewer overflows, sewage treatment plants, and other sources of anthropogenic pollutants. The effects of estrogenic compounds and other chemicals in addition to TCDD, TCDF, and PCBs were not considered or discussed by Wintermyer and Cooper (2003), and exposures of test organisms to other chemicals were not evaluated.

However, in the laboratory, Wintermyer and Cooper (2003) injected eastern oysters with TCDD, with resulting nominal tissue concentrations reported at 2 and 20 ng/kg ww. Oysters exhibited a dose-dependent reduction in egg fertilization success and in larval survival. Therefore, this paper demonstrates that 2 ng/kg ww in whole bivalves causes reproductive effects in addition to the histopathological effects observed in female oysters at this exposure level (Wintermyer and Cooper 2007).

All five clam samples collected adjacent to the northern impoundments at Transect 3 had tissue concentrations higher than 2 ng/kg, and four out of five at this location had tissue concentrations higher than 10 ng/kg. Two out of five (40 percent) clam samples next to the upland sand separation area were just above the 2 ng/kg, the LOAEL for histological effects in individual females and reduced egg fertilization and larval survival (Table 6-2). Although it is not possible to specify the effect on mollusc populations, individual clams from the area represented by Transect 3, assuming they are as sensitive as the oysters of Wintermyer and Cooper (2007), are at risk of reproductive impairment. Because of uncertainty associated with the use of literature-based TRVs, the field logbooks for collection of the clams were consulted. Field notes for the clam sampling for this project indicate no difficulty in capturing clams at Transect 3; clam collection required about 30 minutes at each of the transects (Integral 2011e), regardless of where they were collected. Although this is not the result of systematic study, any long term population level effects due to reproductive impairment in clams would suggest that capture of clams at Transect 3 should be more difficult, which was not the case. In light of this anecdotal information, although reproductive risk to individual clams collected from Transect 3 is present, risk to mollusc populations is considered low.

TCDD concentrations in three of five samples of clams collected adjacent to the upland sand separation area (Transect 5) exceed the reproductive LOAEL for oysters, but TCDD concentrations in 60 percent of clam samples from Transect 5 were below concentrations associated with effects on reproduction in individuals. The concentrations in the remaining 40 percent were just above the lower threshold of effects, indicating a substantially lower risk than at Transect 3. Therefore, risks to individual molluscs collected from Transect 5 appear to be low, and risks to populations are negligible. Risk to molluscs collected at Transects 2, 4, and 6 are negligible, because TCDD concentrations in clam tissues of these transects are below the LOAEL for the histological endpoint identified by Wintermeyer and Cooper (2007) and reproductive endpoints reported by Wintermyer and Cooper (2003).

It is not possible to evaluate post-TCRA risk to clams in the vicinity of Transect 3, but there is a statistically significant correlation between sediment TCDD and clam tissue TCDD for the site (Integral and Anchor QEA 2012). Because the concentrations of TCDD decline rapidly with distance from the impoundment, it is likely that the baseline risk of reproductive effects in individual molluscs is highly localized adjacent to the impoundments, and possibly only within the original 1966 impoundment perimeter. Because Transect 3 was within the TCRA footprint (Figure 4-1), it is also likely that risk to molluscs in the vicinity of

Transect 3 is greatly reduced as a result of the TCRA, which contained the area with the most contaminated sediments.

6.2.4 Miscellaneous $COPC_E$s

$COPC_E$s lacking a TRV or benchmark expressed as a concentration in porewater, tissue or sediment include carbazole and barium. $COPC_E$s for which reliable estimates in porewater could not be made are aluminum and vanadium. Results are summarized in Table 6-1, and below:

- Carbazole (Figure 6-10) in surface sediment samples from the site does not exceed concentrations upstream, although the maximum concentration upstream is estimated from the detection limit. Carbazole also was not detected in most locations on the Site and those concentrations that were detected are J-qualified (estimated).
- Aluminum (Figure 6-11) in surface sediment samples on the Site does not exceed the REV for aluminum except at one station, just beneath the I-10 bridge.
- Barium (Figure 6-12) in surface sediment samples on the Site exceeds the REV in 31 locations on the Site. The spatial pattern in sediments is random, with the highest concentrations from stations outside of the original 1966 impoundment perimeter. Barium in sediments does not appear to be associated with the impoundments north of I-10.
- Vanadium (Figure 6-13) in sediment samples on the Site exceeds the REV in 32 locations on the Site. The spatial pattern in sediments is random, and like barium, the highest concentrations are not within the impoundments north of I-10. Vanadium does not appear to be associated with the waste in the impoundments north of I-10.

Although specific toxicity information for carbazole is not available, the relatively small number of detects and small area with barely detectable concentrations suggests that carbazole does not present a risk to benthic invertebrates. Risks to benthic invertebrates on the Site from aluminum are not elevated over background. Although barium and vanadium are present in multiple locations on the Site at concentrations above the REV, the spatial distribution of samples with concentrations above the REV is random, and does not show an association with the impoundments. The highest concentrations of both barium and vanadium are outside of the impoundments, suggesting that the wastes in the impoundments are not the source of these metals. Any risks to benthic macroinvertebrates resulting from barium and vanadium are not associated with wastes.

6.2.5 Summary: Lines of Evidence for Benthic Macroinvertebrate Communities

Results of these analyses address the related risk question identified in Table 3-10: whether the concentrations of COPCES in whole sediment from benthic habitats of the Site are greater than threshold concentrations relating to the survival, growth or reproduction of benthic invertebrates, or the productivity or viability of invertebrate populations or communities. Analysis results for benthic macroinvertebrates indicate that generally, they do not, although in localized areas adjacent to the former waste impoundments, tissue concentrations of TCDD in clams may affect reproduction of individuals. The area of impact, however, is small relative to the Site, so overall there is low risk to populations of molluscs, and only in a limited area, directly adjacent to the impoundments.

Risks to benthic macroinvertebrate communities from BEHP, phenol, copper, cobalt, lead, thallium and zinc are negligible. Risks due to carbazole and aluminum are no greater than in upstream areas. Risks to benthic invertebrate communities from barium, manganese, and vanadium, if any, have random spatial patterns not associated with the impoundments, and are therefore not a result of the presence of the impoundments.

Exceedance of the ER-M for mercury in two isolated surface sediment samples within the original impoundment perimeter does not indicate risks to the assessment endpoint for the overall benthic invertebrate community. Samples adjacent to affected samples are either below the ER-M or below the lower SQG, the ER-L. The isolation of these two samples, and the relatively small area affected, indicate negligible risk to benthic macroinvertebrate communities from mercury. In the post-TCRA environment, there are no risks to benthic invertebrates from mercury.

Risk to benthic macroinvertebrate communities from TCDD in sediments is negligible, according to the comparison of TCDD concentrations in surface sediments to the geometric mean of the NOAEC values (Appendix B). Since NOAECs available in the literature are a random assortment of values that are an artifact of the study designs of the publications from which they are drawn, an exceedance of the NOAEL is not considered to predict a potential effect. None of the studies of TCDD toxicity to invertebrates identified an effects concentration in sediment, even when 25,000 ng/kg was tested, so concentrations of TCDD in surface sediments from the Site cannot be compared to an effects level. Risks to the benthic community from TCDD overall is therefore considered negligible. Other dioxin and furan congeners cannot be evaluated because of a lack of toxicity data.

The analyses presented also address the following risk question (Table 3-10): whether concentrations of organic primary $COPC_{ES}$ (dioxins and furans) in tissue of field collected clams equal to or greater than concentrations considered threshold levels of reproductive effects in molluscs. Individual molluscs directly adjacent to the impoundment north of I-10 are at risk of reproductive effects from exposure to TCDD, and risks to populations at Transect 3 are considered low. Because tissue concentrations in all clams from this area (Transect 3) exceed the concentrations associated with effects in both male and female oysters, some effect on the reproductive productivity of clams or other molluscs in the area of very high concentrations of TCDD in sediment is possible. Although a precise estimate of the effect on the populations of molluscs on the Site is not possible, risks to molluscs from exposure to TCDD appear to be localized, and do not extend to other areas sampled elsewhere on the Site. Risks to a fraction of the individual molluscs near the upland sand separation area are very low, and risk to populations there are negligible. Risks to molluscs elsewhere on the Site are negligible. There are no toxicity data available to interpret tissue concentrations of the other dioxin and furan congeners.

Wintermyer and Cooper (2007) discuss possible mechanisms of the toxicity of TCDD to reproductive tissues of the oysters in their study, and acknowledge that the mechanism is AhR-independent. They are silent on the question of whether other congeners might have similar effects, but the absence of an AhR that binds dioxin in invertebrates indicates that the toxicity observed in oysters is not scalable to other congeners, as it is in birds, fish, and mammals. The potential effects of the other congeners are uncertain.

6.3 Risks to Fish

A summary of results for each line of evidence to address risk to fish is provided in this section.

6.3.1 Estimated Concentrations of Metals in Fish Diets Relative to TRVs

COPCEs evaluated by estimating the prey-weighted concentration in foods of fish (sediment ingestion was included) are those for which corresponding TRVs are available, and include cadmium, copper, mercury, and zinc. The analysis was conducted for the black drum and the southern flounder, expected to move around throughout the Site, and for Gulf killifish, on a smaller scale because these and related species are expected to have more localized foraging ranges. HQs for fish exposed to cadmium, copper, mercury, and zinc in foods and incidentally ingested sediment are summarized in Table 6-3. In no cases do the concentrations in ingested media exceed NOAELs or LOAELs for fish for these metals. Therefore, risks to all three fish receptors from cadmium, copper, mercury and zinc are negligible on the basis of this line of evidence.

6.3.2 Estimated Concentrations in Surface Water Relative to TRVs

COPCEs evaluated by estimating concentrations in surface water and comparing to TRVs for water are BEHP and nickel. Results are summarized in Table 6-4. An estimate of the Site-wide concentration of BEHP in water from a SWAC of surface sediments does not exceed the TRV for BEHP in water, which is a NOAEC. The estimated Site-wide concentration of nickel in water does not exceed the TRV for nickel, which was derived from several studies of marine fish (Appendix B). Therefore, risks to fish from BEHP and nickel are negligible on the basis of this line of evidence.

6.3.3 Total PCB Concentrations in Whole Fish Relative to the TRV for Fish

None of the whole hardhead catfish samples or Gulf killifish samples had total PCB concentrations above the NOAEC of 5.0 mg/kg ww or LOAEC of 16 mg/kg ww for total PCBs in fish (Table 4-5). Even the highest total PCB concentration, in a whole catfish from FCA 2, was more than a factor of 5 below the NOAEC. Risks to fish from total PCBs on the Site are negligible on the basis of this line of evidence.

6.3.4 TEQ Concentrations in Whole Fish Relative to the TEQ SSD for Fish

The analysis of toxicity data for fish eggs prepared by Steevens et al. (2005) and resulting SSD was used as the basis for comparison to Site-specific concentrations of $TEQ_{DF,F}$ and $TEQ_{DFP,F}$ (ng/kg lipid weight) in whole Gulf killifish and hardhead catfish. Results are considered representative of the fish receptors, the Gulf killifish, the black drum, and the southern flounder.

Representativeness of the hardhead catfish of the receptor fish species was evaluated by considering the available data for $TEQ_{DFP,F}$ in all fish tissues from the Site. Among all samples of fish from the Site (only the RI dataset includes whole fish, all other data are for fillet samples), which includes samples of the southern flounder, black drum, and several other fishes, hardhead catfish, gafftopsail catfish, and spotted sea trout dominate the upper end of the range of $TEQ_{DFP,F}$ concentrations (ng/kg ww) in fillet tissue. The relatively elevated concentrations in these species edible tissue samples (for which lipid data are not available for lipid normalization) could be caused by higher lipid content in edible tissue and not by greater exposures, but hardhead catfish are among the species with the highest $TEQ_{DFP,F}$ concentrations in fillet, suggesting that the hardhead catfish is a reasonably conservative representation of the southern flounder and black drum.

6.3.4.1 Killifish

There is no overlap in the distribution of concentrations of $TEQ_{DF,F}$ in whole killifish (Figure 6-14) with concentrations represented by Steevens et al.'s (2005) SSD. Therefore, there is no risk to Gulf killifish from dioxins and furans. When dioxin-like PCBs are included in the TEQ calculation, risks to Gulf killifish appear to be slightly increased (Figure 6-15; Table 4-6). One sample of whole killifish from Transect 4 has a concentration of $TEQ_{DFP,F}$ of 503 ng/kg lw, but this concentration is an artifact of high detection limits, and the true concentration is unknown. No dioxins and furans were detected in this sample, and PCB81, PCB123, and PCB169 were all not detected. If the estimated concentrations of these PCB congeners are removed, the $TEQ_{P,F}$ is only slightly reduced, to 193 from 196 ng/kg lw. The $TEQ_{DFP,F}$ concentration is below the concentration considered by Steevens et al. (2005) to be protective of 90 percent of fish species, but this comparison overstates risk, because none of the dioxins and furans in this sample were detected.

Therefore, there are negligible risks to Gulf killifish and those species that it represents resulting from dioxin and furan exposures alone, and there are generally negligible risks to these fish from the combination of all dioxin-like compounds. There is a small chance that the presence of dioxin-like compounds in one of two samples at Transect 4 could result in early life stage effects in killifish, but the available result for this sample is confounded by high detection limits, and a suggestion of risk is most likely an analytical artifact, because the other fish from this transect has a $TEQ_{DFP,F}$ concentration of 9.07 ng/kg lw, well below any risk threshold (Table 4-6). However, the $TEQ_{P,F}$ concentration of approximately 196 ng/kg lw is relatively high for this parameter. Transect 4 is near an outfall, which may affect the exposure of fish to PCBs in that area. Risk to Gulf killifish collected near the impoundments (Transect 3) is negligible.

6.3.4.2 Hardhead Catfish

There is no overlap in the distribution of concentrations of $TEQ_{DF,F}$ (ng/kg lw) on the Site with concentrations represented by Steevens et al.'s (2005) SSD for fish (Figure 6-16). Two samples of whole catfish from FCA 1 and one from FCA 3 have $TEQ_{DF,F}$ concentrations that slightly exceed Steevens et al.'s (2005) best estimate of the concentration at which 95 percent of fish species are protected (Table 4-6), and all samples are within the range of error of that calculation, suggesting a low to negligible risks to large fish represented by hardhead catfish from dioxins and furans. The result does not change appreciably when dioxin-like PCBs are added to the exposure estimate, except that $TEQ_{DFP,F}$ in two samples from FCA 2 also are equal to or slightly exceed the concentration protective of 95 percent of fish species (Table 4-6; Figure 6-17).

Given the conservatism of the Steevens et al. (2005) SSD for TCDD (because it is largely based on salmonids, which are known to be relatively sensitive to this and other toxicants), the conservatism of the approach, which assumes a 1 to 1 ratio of dioxin, furan, and PCB concentrations in whole fish to those of egg tissue (Tietge et al. 1995; Heiden et al. 2008), and that $TEQ_{DFP,F}$ in all samples is within the range of error of Steevens et al.'s estimate of the level protective of 95 percent of all fish species, risks to fish from exposure to dioxin-like compounds is very low to negligible.

6.3.5 Summary: Lines of Evidence for Fish

Risk questions for fish (Table 3-10) address whether the concentrations of $COPC_E$s in waters of the Site, concentrations of inorganic $COPC_E$s in the diet of fish, or concentrations of organic $COPC_E$s in fish tissue from the Site are greater than the concentrations of $COPC_E$s associated with the survival, growth or reproduction of fish. Analyses presented in this section indicate that they are not. Risks to all of the fish receptors from exposures to cadmium, copper, mercury, and zinc in the diet, including incidentally ingested sediment, are negligible. Risks to fish following exposure through water to BEHP and nickel are negligible. Risks to fish as indicated by total PCB concentrations in whole body samples are negligible.

Concentrations of $TEQ_{DF,F}$ (ng/kg lw) and $TEQ_{DFP,F}$ (ng/kg lw) in both whole Gulf killifish and whole catfish are generally below concentrations associated with adverse effects on fish early life stages. One Gulf killifish sample seems to exceed risk thresholds, but this is an artifact of elevated detection limits for dioxin and furan congeners. For five whole catfish, the $TEQ_{DFP,F}$ is slightly above the concentration protective of 95 percent of fish species, but within the margin of error, and below the concentration protective of 90 percent of species. Because the SSD derived by Steevens et al. (2005) is largely biased towards salmonids which are known to be among the most sensitive fish taxa for many toxicants, this evaluation is considered conservative. Overall, risks to fish on the Site are negligible.

6.4 Risks to Birds

Risks to birds were evaluated by comparing estimated daily ingestion rates of each $COPC_E$ to their respective TRVs expressed in the same terms. Risks to birds from exposures to dioxin-like compounds were also evaluated by comparing estimated egg concentrations to TRVs expressed as concentrations in eggs, providing a second and independent line of evidence to evaluate risks to birds from exposure to dioxins, furans and dioxin-like PCBs.

6.4.1 Estimated Daily Ingestion Rates Relative to TRVs

Results of the comparison of estimated daily ingestion rates of each $COPC_E$ by each avian receptor to its respective TRV are summarized in Table 6-5. For great blue heron and neotropic cormorant, daily ingestion rates of all $COPC_Es$ do not exceed NOAELs nor do they exceed LOAELs. This line of evidence indicates that risks to great blue heron, neotropic cormorant, and the species they represent, from ingestion exposure to cadmium, copper, mercury, nickel, zinc, BEHP, total PCBs, $TEQ_{DF,B}$, and $TEQ_{DFP,B}$ are negligible.

Estimated daily ingestion rates of cadmium, mercury, nickel, zinc, BEHP, and total PCBs by the spotted sandpiper also indicate that risks to sandpipers and the species they represent from these $COPC_Es$ are negligible. Estimated daily ingestion rates of copper by the spotted sandpiper could exceed the NOAEL, but neither the CT nor the RM exposures to copper for this receptor exceed the LOAEL. The avian TRV for copper was taken from the literature compilation in USEPA's EcoSSL for copper (USEPA 2007d), which identified over 3,000 papers and generated 393 copper TRVs for birds for a range of endpoints. The selected NOAEL of 4.05 mg/kg-day was the highest bounded NOAEL that was also lower than the lowest bounded LOAEL. The associated LOAEL from the study reporting the NOAEL of 4.05 mg/kg-day was 12.1 mg/kg-day for reproduction in chickens. Among the dataset compiled by USEPA (2007d), this NOAEL is among the lowest overall, and dozens of survival, growth, and reproduction NOAELs that are both higher than this and bounded by LOAELs are reported for sensitive endpoints in chickens as well as other species. The selected NOAEL for this risk assessment is from a study in which chickens were administered copper in food for 84 days and those exposed at the LOAEL exhibited a reduction in fecundity. Therefore, the selected TRV was a highly conservative representation of copper toxicity in individual birds, and exceedance of the NOAEL by a factor of 2 does not indicate a risk to sandpiper populations. Risks to this and other avian receptor populations from ingestion of copper are negligible.

The CT and RM of estimated daily ingestion rates of $TEQ_{DF,B}$ and $TEQ_{DFP,B}$ by the sandpiper exceed both the NOAEL and the LOAEL (Table 6-5). The HQ_L of 1 for CT exposure and the HQ_L of 3 for RM exposure indicates that there is a possibility that exposure of a shorebird foraging on the Site to dioxin-like compounds will be at levels that exceed effects levels for these chemicals. The very low HQs for $TEQ_{P,B}$ indicate that the risk to sandpipers is driven primarily by dioxins and furans, and not PCBs. Risks to sandpiper due to ingestion of dioxin-like compounds are evaluated further below.

Estimated daily ingestion rates of cadmium, copper, nickel, BEHP, and total PCBs by the killdeer also indicate that risks to killdeer and the species they represent from these $COPC_{ES}$ are negligible. Estimated daily ingestion rates of mercury exceed the NOAEL, but not the LOAEL. The study supporting the NOAEL (Heinz 1979) found no reproductive effects in the first generation of mallard ducks administered methylmercury dicyandiamide in the diet. Reproductive endpoints evaluated included fecundity and duckling survival. The study supporting the LOAEL used Japanese quail and reported reproductive effects at 0.9 mg/kg-day. Heinz (1979) administered methylmercury, which is highly bioavailable and is the toxic form of mercury, in the diet. In the killdeer exposure model, more than half of the daily mercury dose is derived from soil ingestion. However, methylmercury, the more toxic form of mercury, is generally not a large proportion of total mercury in soils, and thus, the Heinz study is not a realistic model of environmental conditions. Therefore, exceedance of the NOAEL by a factor of 2 (Table 6-5) does not indicate reproductive risk to individual killdeer. Risk to killdeer populations from mercury is negligible.

The RM ingestion rates of $TEQ_{DF,B}$ and $TEQ_{DFP,B}$ by killdeer are about equal to the LOAEL, indicating that risk to individual killdeer reproduction from dioxin-like compounds is present. The RM of the daily ingestion rate of zinc is about equal to the LOAEL for zinc in birds. The HQ_L of 1 for killdeer exposed to zinc indicates that there is a low probability that exposure of an individual terrestrial invertivorous bird foraging on the Site (prior to implementation of the TCRA) could occur at the effects level for zinc. Additional evaluation to describe risks to killdeer from zinc and dioxin-like compounds, including an evaluation of the probability that zinc and dioxin-like compounds exposures will exceed the LOAEL, is provided below.

6.4.2 Estimated TEQ Concentrations in Bird Eggs Relative to TRVs

Results of the evaluation of TEQ concentrations in the eggs of neotropic cormorant, great blue heron and spotted sandpiper relative to TRVs for egg mortality are summarized as HQs in Table 6-6, for all of the exposure scenarios modeled (Section 4.3.2). Concentrations of

TEQ in eggs of killdeer were not estimated because empirical data on the concentrations of PCBs, dioxins, and furans in their foods are not available. Results of risk calculations using this line of evidence are largely consistent with the results of risk calculations using estimated ingestion rates. Estimated concentrations of $TEQ_{DF,B}$ and $TEQ_{DFP,B}$ in the eggs of neotropic cormorant and great blue heron do not exceed the LOAEL concentration for egg mortality. Estimated concentrations in the eggs of cormorant do not exceed the field- or laboratory-based NOAELs for cormorants, except for the RM exposure that includes pre-TCRA sediment ingestion (Table 6-6, Tables 5-1 and 5-2). Similarly, estimated concentrations of $TEQ_{DFP,B}$ in eggs of great blue heron only exceed the NOAEL when ingestion of sediment is considered (Table 4-20). HQ_N values for great blue heron and cormorant ingesting prey and sediment are 2 to 3, but the egg-based HQ_L values for these scenarios are below 1 (Table 6-6). Results of several technically sound studies were used in deriving the egg TRV (Tables B-6 and B-8 of Appendix B). All of them report egg mortality as the endpoint. The final NOAEL for bird eggs was less than half of the lower of the two NOAELs available for cormorants, and the lowest effects level for cormorants was almost 10 times higher than the NOAEL (Table B-6). Therefore, an HQ_N of 2 for cormorants does not indicate risk of egg mortality in individual cormorants, and risk to cormorants is negligible.

There were no species-specific LOAELs for great blue heron, but a NOAEL of 207 ng/kg ww in eggs was reported for this species (Appendix B, Table B-9). The robust studies evaluating TCDD or TEQ in bird egg yolks report concentrations associated with actual effects that are from 2.2 to 12 times greater than NOAEL of 450 ng/kg. Also, there is substantial interspecies variability in the sensitivity to dioxin toxicity, and the relative sensitivity of herons is unknown. As a result, the HQ_N of 2 (or 3 at the RM exposure) is not a definitive indicator of risk or lack of risk to the mortality of eggs laid by individual birds. However, given the very conservative assumption that herons forage exclusively within its exposure unit on the Site, the inherent spatial bias of the associated sediment data set, and the conservatism of the egg model (Section 4.3.2.1.2), the egg exposure estimate is probably higher than the actual egg exposure. This is a key consideration given the uncertainty in the actual effects threshold for herons and that the exposure estimate is between the NOAEL and LOAEL. In light of the conservative representation of exposure, the egg-based HQ_N values for great blue heron are not interpreted to specifically indicate risk of egg mortality to individual herons.

The estimated post-TCRA egg concentrations for these two receptors indicate that implementation of the TCRA has a substantial effect on the potential exposures of these types of birds, reducing estimated egg concentrations. Baseline risks to neotropic cormorant and great blue heron from exposure to dioxin-like compounds is negligible.

The HQ_L values calculated using the baseline (prey plus sediment) CT and RM egg exposures for spotted sandpiper are consistent with results of those based on ingestion exposure: the CT and RM HQ_L values for this receptor are 1 and 2, respectively. This result indicates that the average egg exposure to shorebirds whose foraging habits result in extensive contact with sediments could equal the concentrations resulting in egg mortality, and the upper bound on the average egg concentration could be two times the LOAEL for egg mortality. The TRV used in the HQ_L calculation is the geometric mean of two other geometric mean LOAEL egg concentrations indicating egg mortality, one for ring-necked pheasants (1,215 ng/kg ww) and the other for double-crested cormorants (4,648 ng/kg ww) (Table 5-1). Egg mortality in these four studies ranged from 10 percent to 50 percent above that of controls. The results of a field study with spotted sandpipers indicated a NOAEL for egg mortality of 732 ng/kg ww (Appendix B), which is higher than the NOAEL used as a TRV, and higher than the NOAEL for pheasants, suggesting that the spotted sandpiper is not among the bird species considered highly sensitive to dioxin-like egg toxicity.

Results of both lines of evidence (estimated ingestion rate and estimated egg concentrations) are consistent in indicating some risk of egg mortality to the spotted sandpiper and the birds it represents from exposures to dioxin-like PCBs, dioxins, and furans. Risks to spotted sandpiper are considered in greater detail below.

6.4.3 Probability that Exposure Exceeds Effects Thresholds

A probabilistic analysis of exposure was conducted for those receptor–$COPC_E$ pairs for which the HQ_L is greater than or equal to 1. Probabilistic exposure analyses were conducted using only the wildlife exposure model, and not the egg exposure model. The exposure scenarios modeled probabilistically include zinc for killdeer, and $TEQ_{DFP,B}$ for spotted sandpiper.

6.4.3.1 Killdeer

A probabilistic exposure model for killdeer was performed for zinc and $TEQ_{DF,B}$ using the methods described in Section 4.4. Each of the resulting exposure levels generated by the Monte Carlo analysis was divided by the LOAEL for zinc. Results are presented as the cumulative probability distribution of the HQ_L for killdeer.

The result of the probabilistic exposure analysis indicates that there is an 8.3 percent probability that baseline exposures of killdeer to zinc will exceed the LOAEL (Figure 6-18).

The result of the probabilistic exposure analysis indicates that there is a 4.7 percent probability that baseline exposure of individual killdeer and the birds it represents to $TEQ_{DF,B}$ will exceed the LOAEL (Figure 6-19).

6.4.3.2 Spotted Sandpiper

Probabilistic exposure models for the spotted sandpiper were performed for ingestion of $TEQ_{DFP,B}$. The result of this analysis indicates that there is a 13.7 percent probability that baseline exposure of spotted sandpiper and the birds it represents to $TEQ_{DF,B}$ will exceed the LOAEL for wasting syndrome in adults and mortality of their eggs (Figure 6-20).

6.4.4 Post-TCRA Risks to Killdeer and Spotted Sandpiper

Under baseline conditions, zinc HQ_L values for killdeer equal 1, and TEQ HQ_L values for killdeer and spotted sandpiper exceed 1. These HQ_Ls were also calculated under post-TCRA conditions to determine whether implementation of the TCRA affects risk, and if the post-TCRA environment no longer presents risks to these receptors.

Table 6-7 provides a summary of pre-TCRA and post-TCRA HQ_Ls for these receptor–$COPC_E$ pairs. Risks to spotted sandpiper from exposures to $TEQ_{DF,B}$ using the line of evidence based on ingested dose are negligible in the post-TCRA scenario. The line of evidence based on estimated TEQ concentrations in eggs is consistent with the HQ_L results (Table 6-6). Therefore, implementation of the TCRA has eliminated risks to spotted sandpipers from exposure to $TEQ_{DF,B}$.

As for great blue heron baseline risks, the HQ_N for killdeer in the post-TCRA exceeds 1, but the HQ_L does not (Table 6-7). Similarly, although the absence of a species-specific threshold of effects for egg mortality results in some uncertainty, the several layers of conservatism in the exposure model for killdeer suggest that risks to individual killdeer from exposures to $TEQ_{DF,B}$ using the line of evidence based on ingested dose are very low in the post-TCRA scenario. Therefore, risk to the assessment endpoint, bird populations, is negligible. Risks to killdeer from exposure to zinc are not affected by implementation of the TCRA. This suggests that sources other than the waste impoundments are the primary source of this metal resulting in exposure to killdeer. Spatial patterns in surface soil concentrations of zinc within the exposure unit for killdeer support this conclusion: the samples with highest concentrations occur outside of the northern impoundments (Figure 6-21).

6.4.5 Risks to Killdeer and Sandpiper in Background Areas

The zinc and $TEQ_{DF,B}$ HQ_L values for killdeer equal 1, and $TEQ_{DF,B}$ HQ_L values for spotted sandpiper exceed 1 under baseline conditions. Risks in background areas are presented to provide perspective on the incremental risk to these receptors due to the Site. For the killdeer, the zinc HQ_L at the CT and RM background exposures are 87 and 71 percent, respectively, of the corresponding HQ_L values for the Site. This indicates that the incremental increase in exposure of killdeer to zinc at the Site is small, ranging from only about 13 to 29 percent, and suggests a substantial role of background conditions in the exposures of killdeer to zinc. The $TEQ_{DF,B}$ HQ_L at the CT and RM background exposures are 23 and 22 percent, respectively, of the corresponding HQ_L values for the Site, indicating that the incremental exposure in background areas is nearly a quarter of the exposure of killdeer to dioxin-like compounds on the Site.

For the spotted sandpiper, the $TEQ_{DF,B}$, $TEQ_{P,B}$ and $TEQ_{DFP,B}$ HQ_L values for background are low, regardless of whether the ingestion rate or the egg concentrations are considered (Tables 6-8 and 6-6). For both $TEQ_{DF,B}$ (and $TEQ_{DFP,B}$) baseline HQ_L values for background are about 1 percent of those on the Site, indicating that baseline (pre-TCRA) exposures of spotted sandpiper to dioxins and furans on the Site are substantially elevated over background. If background PCBs are considered on their own (as $TEQ_{P,B}$), the background $TEQ_{P,B}$ HQ_L values for CT and RM exposures are 26 and 21 percent, respectively, of those on the Site. This suggests that, although sandpipers (and related birds) are exposed to dioxin-like PCBs on the Site at levels higher than background, PCB exposures are a more important contributor to overall exposures of sandpipers to all of the dioxin-like compounds in background areas than they are on the Site.

6.4.6 Summary: Lines of Evidence for Birds

The analysis presented in this section addresses two risk questions (Table 3-10): 1) whether the total daily ingested dose (mg/kg-day) of $COPC_{ES}$ is greater than doses known to cause effects on the survival, growth, or reproduction in birds; and 2) whether the estimated concentration of dioxins and furans, expressed as TEQ_{B}, in bird eggs is greater than threshold concentrations for reproductive effects in birds. Results presented in this section indicate that there is a low probability that ingestion rates of zinc by killdeer, and ingestion rates of $TEQ_{DF,B}$ by the spotted sandpiper will exceed ingestion rates associated with adverse effects on bird reproduction. Results also indicate that $TEQ_{DFP,B}$ concentrations in eggs of sandpiper could also exceed those resulting in egg mortality. Ingestion rates of these and other chemicals by other bird receptors and estimated egg concentrations in the great blue heron and neotropic cormorant do not exceed effects thresholds.

Overall, baseline risks to individual birds on the Site are very low to negligible for most chemicals, and are low for dioxins and dioxin-like compounds. Baseline risks to cormorant and great blue heron are negligible for all of the $COPC_{ES}$, including dioxins, furans and dioxin-like PCBs, although there is some uncertainty about risks to heron due to a lack of species-specific effects thresholds for TEQ in eggs. Baseline risks to killdeer are negligible for all chemicals except zinc and dioxins and furans for which they are very low, and not much greater than background for zinc. Baseline risks to spotted sandpiper are negligible for all metals, BEHP, and total PCB as well as $TEQ_{P,B}$.

The probability that exposures of killdeer to zinc will exceed the effects level is low (8.3 percent). Background exposures to zinc are a substantial fraction of the overall exposure of killdeer to zinc. The probability that exposures of killdeer to dioxins and furans will exceed the ingestion-based LOAEL is low (4.7 percent). Background exposures to dioxins and furans represent about a quarter of the overall exposure of killdeer to dioxins and furans.

There is a moderate risk to spotted sandpiper from dioxins and furans under the baseline condition, as indicated by two independent lines of evidence, a wildlife exposure model of ingestion rate, and a model of egg concentrations. On the basis of a probabilistic evaluation of ingestion exposure, the probability that spotted sandpiper will be exposed to $TEQ_{DF,B}$ at levels exceeding TRVs is a moderate 13.7 percent. Although dioxin-like PCBs are additive with the $TEQ_{DF,B}$, the contribution of $TEQ_{P,B}$ to the exposure of spotted sandpiper is small, as indicated by both the estimated ingestion rate and the estimated egg concentration. Risks to the spotted sandpiper were reduced to negligible as a result of implementation of the TCRA.

6.5 Risks to Mammals

Risks to mammals were evaluated by comparing estimated daily ingestion rates of each COPC to their respective TRVs expressed in the same terms. Results are discussed below.

6.5.1 Estimated Daily Ingestion Rates Relative to TRVs

Results of comparisons of estimated daily ingestion rates of the $COPC_{ES}$ to their respective TRVs for the raccoon and the marsh rice rat are summarized in Table 6-9. Estimated daily ingestion rates of all $COPC_{ES}$ by raccoon are below LOAELs, regardless of whether the CT or RM is considered. Therefore, risks to raccoon, and the terrestrial mammals that it represents, are negligible.

Estimated daily ingestion rates of cadmium, copper, nickel, zinc, BEHP, and $TEQ_{P,M}$ and total PCBs by the marsh rice rat are all below their respective NOAELs and LOAELs, indicating negligible risk to the marsh rice rat for these $COPC_{ES}$. Estimated daily ingestion rates of mercury exceed the NOAEL but not the LOAEL (Table 6-9). The $TEQ_{DF,M}$ and $TEQ_{DFP,M}$ HQ_L values are both 2, indicating that marsh rice rats could be exposed to dioxins and furans at levels exceeding those resulting in reduced pup survival and effects on other reproductive endpoints in laboratory rats.

6.5.2 Probability that Exposure Exceeds Effects Thresholds

A probabilistic analysis of exposure of marsh rice rat to $TEQ_{DF,M}$ was conducted using the methods described in Section 4.4, and results are illustrated in Figure 6-22. There is a

14.3 percent probability that exposure of marsh rice rat to $TEQ_{DF,M}$ will exceed the level associated with effects on reproduction in mammals.

6.5.3 Post-TCRA Risks to Marsh Rice Rat

Exposures to the marsh rice rat following implementation of the TCRA is reduced to levels below those associated with effects, and the resulting $TEQ_{DF,M}$ and $TEQ_{DFP,M}$ HQ_L values are below 1 (Table 6-7), although NOAELs are still exceeded. The post-TCRA analysis conservatively assumes that concentrations in foods of rice rats do not change as a result of the TCRA (because post-TCRA food concentrations were not available). The reduction of the HQ_L to a value below 1 indicates that the majority of the rice rat exposures to dioxins and furans were associated with exposure to sediments within the impoundments.

6.5.4 Risks to Marsh Rice Rat in Background Areas

Exposure of marsh rice rat in background areas to $TEQ_{DF,M}$ as indicated by the HQ_L for background is very low (Table 6-8). The CT and RM exposures of marsh rice rat to $TEQ_{DF,M}$ in background areas are about 3 percent and 1 percent, respectively, of the CT and RM exposures on the Site, indicating that the incremental exposure of marsh rice rat to dioxins and furans at the Site is about the same as it is for the spotted sandpiper. Also like the sandpiper, the CT and RM exposures to $TEQ_{P,M}$ by the rice rat in background areas are about 37 and 29 percent of those on the Site, indicating that exposure of marsh rice rat to dioxin-like PCBs plays a larger role to the entire $TEQ_{DFP,M}$ exposure in background areas than it does on the Site.

6.5.5 Summary: Lines of Evidence for Mammals

Analyses presented in this section address the following risk question whether the total daily ingested doses (mg/kg-day) of $COPC_{ES}$ are greater than doses known to cause effects on the survival, growth, and reproduction of mammals. Results of the exposure and risk analyses indicate that, for all $COPC_{ES}$ except $TEQ_{DF,M}$, they are not, and that rates of ingestion of $TEQ_{DF,M}$ by raccoon do not exceed effects thresholds. Risks to raccoon are negligible for all $COPC_{ES}$. Risks to marsh rice rat are negligible for all $COPC_{ES}$ except TEQ. Risks due to $TEQ_{P,M}$ only are negligible, and dioxin-like PCBs do not contribute substantially to the $TEQ_{DFP,M}$ exposures. Marsh rice rats on the Site are at risk of reproductive effects and reduced survival of pups as a result of exposure to $TEQ_{DF,M}$. The probability that the exposure of marsh rice rat to $TEQ_{DF,M}$ exceeds the LOAEL for these effects is 14.3 percent. However, the risks to this receptor are reduced to negligible as a result of implementation of the TCRA.

6.6 Risks to Reptiles

Appendix B describes the literature search for information to support TRVs for reptiles. No information was found to interpret reptile exposures. Extensive literature searches by other authors corroborate this result. Because TRVs needed to interpret exposure estimates for reptiles could not be developed, HQs cannot be calculated, and risks to reptiles cannot be addressed using the same approaches used for other receptors. The risk question presented in Table 3-10, "whether the total daily ingested doses (mg/kg-day) of $COPC_{ES}$ greater than doses known to cause effects on the survival, growth and reproduction of reptiles," cannot be addressed with the available information.

However, exposure estimates for reptiles can be compared to those for other receptors. Table 4-13 shows the CT and RM exposure in mg/kg-day of all wildlife receptors to each $COPC_E$. The estimated daily ingested dose of the alligator snapping turtle is included in this summary. Generally speaking, the estimated exposures to alligator snapping turtle for all of the $COPC_{ES}$ are consistently and substantially lower than for other receptors. This is a reflection of the ingestion rate assumption for the alligator snapping turtle, which is based on the field metabolic rate provided by Nagy et al. (1999). Because reptile metabolic demands are lower than those of birds and mammals, use of an allometric model to estimate ingestion rates, and application of those ingestion rates as the basis for exposure estimates for reptiles, will generally result in lower estimates of ingested doses, assuming reptiles are eating the same types of foods on the Site as birds and mammals.

Because the HQs are generally very low for the other receptors at the Site, this general difference in the level of exposure of reptiles would suggest that risk to reptiles are also negligible for metals, BEHP and PCBs. However, it is not possible to conclude with confidence that risks to alligator snapping turtle and other reptiles from exposure to dioxins and furans are also negligible because risks to molluscs, birds, and mammals from dioxins and furans are present in localized areas adjacent to the northern impoundment, and the relative sensitivity of reptiles to dioxins and furans is unknown. Risk to killdeer from zinc is not an indicator of risk to the alligator snapping turtle because risk to killdeer is a result of exposures originating in soils, and the turtle will be exposed mainly in the aquatic environment.

Uncertainties about risk to reptiles from dioxins and furans also arise from likely differences in the relative importance of the dermal exposure route. Because reptiles lack fur and feathers, and because the skin of some reptiles can have a relatively large lipid content, Weir et al. (2010) have suggested that dermal exposure may be the most important exposure route in reptiles, contributing significantly more of the daily dose of lipid soluble compounds than other exposure routes. There are no means to evaluate this aspect of reptile exposure for the Site, and there are no toxicity data to interpret resulting exposure estimates.

In conclusion, risks to reptiles from metals, BEHP, and PCBs are considered negligible, because risks due to these $COPC_{ES}$ were generally negligible for all other receptors. Even PCBs, which are lipid soluble, are present only at low levels on the Site and are not likely to contribute significantly to reptile risk. Risks to reptiles due to dioxins and furans are unknown, because there are no means to estimate reptile exposures, and no toxicity information to interpret exposures. Because other receptors are exposed to dioxins and furans at levels above those associated with effects in laboratory animals, it may also be true that reptiles using the site at the same frequencies and in the same manner as these other wildlife would have comparable risks.

6.7 Threatened and Endangered Species

Because the evaluation of risk to sandpiper, cormorant, and heron resulted in HQ_N values greater than 1, risks to the white-faced ibis, brown pelican, and bald eagle were evaluated as described in Section 4.3.1.6.

These comparisons were conducted for the white-faced ibis for copper, $TEQ_{DF,B}$, and $TEQ_{DFP,B}$; for brown pelican for $TEQ_{DF,B}$ and $TEQ_{DFP,B}$; and for the bald eagle for $TEQ_{DF,B}$ and

$TEQ_{DFP,B}$. Results are summarized in Table 6-10. No other $COPC_{ES}$ exceeded HQ_N for the surrogate receptors representative of protected species.

Estimated exposures of individuals among protected species that could occur on the Site to $COPC_{ES}$ (i.e., those for which $HQ_N \geq 1$ for the surrogate receptors) do not exceed NOAELs. Therefore, risks to protected species that could occur on the Site are negligible.

6.8 Bioaccumulation and Biomagnification of $COPC_E$s

In its comments on the draft BERA (Appendix F), USEPA requires that "the report shall provide/expand its description and evaluation of food chain implications…." Evaluation of bioaccumulation and biomagnification was not included among the DQOs in any of the SAPs for the RI, and data were not collected specifically for that purpose. To effectively evaluate patterns in bioaccumulation or biomagnification using field studies, certain parameters are necessary, such as stable isotopes of nitrogen in various tissue types, including in tissue of primary producers in the study area. Alternatively, controlled experiments can be conducted to evaluate bioaccumulation. None of these types of information are necessary for an RI, and so were not developed for this Site or for this report. To address the USEPA comment for dioxins and furans, a synthesis of information presented in the Technical Memorandum on Bioaccumulation Modeling (Integral 2010b) is presented below. The reader is referred to that report for a detailed discussion.

From the data analyses and the literature review presented in the Technical Memorandum on Bioaccumulation Modeling, including evaluation of region-specific multivariate statistical correlations, it was concluded that the majority of dioxin and furan congeners do not consistently bioaccumulate in fish and invertebrate tissue. Moreover, systematic predictions of bioaccumulation from concentrations of dioxins and furans in abiotic media are difficult and uncertain for some congeners and impossible for others. Uptake efficiencies vary by congener, exposure medium, exposure route, and species. The ability of organisms to transform and eliminate the different dioxin and furan congeners, and the differences in transformation and elimination rates for different congeners adds complexity to patterns of dioxin and furan bioaccumulation across the range of taxa evaluated for this Site. The literature on these subjects is extensive and largely observational. A common conclusion in the literature is that bioaccumulation is controlled more by physiological mechanisms such as the limitations on rates of uptake across gill and gut membranes imparted by the size of dioxin and furan molecules (Opperhuizen and Sijm 1990), and the metabolism and excretion of dioxins and furans, than by chemical properties such as log K_{ow}.

Because rates of uptake and excretion of dioxins and furans are dynamic, species- or taxa-specific, and not described for several congeners, broad generalities are not available to interpret tissue concentrations of dioxins and furans in site-specific samples in terms of the position of each sampled species in the food web. USEPA's (2009) *National Study of Chemical Residues in Lake Fish Tissue* found that benthic fish species overall had higher concentrations of dioxins and furans than predatory fish species, supporting a conclusion that concentrations of dioxins and furans are not predicted by position in the food chain, but are accumulated more as a function of proximity to contaminated sediments. On the Site, whole hardhead catfish have the highest $TEQ_{DF,M}$ concentrations among all tissue types collected for the RI, and hardhead catfish fillet tend to have higher TEQ concentrations than other fish caught on the Site (Exposure Assessment Memo, Appendix B [Integral 2012]), including spotted seatrout and southern flounder, which both eat fish and invertebrates. However, the mean and 95UCL concentrations of $TEQ_{DF,M}$ in whole catfish from FCA2, in which the northern impoundments is located, are the lowest among the three FCAs on the Site. Therefore, results for hardhead catfish suggest that their tissue concentrations of dioxins and furans are higher than for other fish species caught on Site, and more than other species sampled for the RI, but that there is not enough information about the mobility and spatial use patterns, degree of contact with sediment, ages of fish, and other factors to explain the differences. It is notable that clams have the second-highest $TEQ_{DF,M}$ concentrations among tissue collected for the RI on the Site, and that concentrations of individual congeners in clam tissue correlate reasonably well with concentrations in sediments adjacent to where they were collected (PSCR Section 6.2.2.3). Whether clams and catfish occupy similar trophic positions is unknown, but both are more closely associated with the benthic environment than other species for which data are available.

In the absence of specific data to the define trophic structure of the food web on the Site (such as stable nitrogen isotopes or stomach content analysis), no specific conclusions can be drawn about the reasons for higher concentrations of $TEQ_{DF,M}$ in catfish than in other species. Therefore, there are no known "food chain implications" of dioxins and furans in the tissue of species collected for the RI.

7 UNCERTAINTY ANALYSIS

Ecological risk assessments are inherently imprecise and uncertain, and any ecological risk analysis provides only a simplified model of a natural environment that is complex and dynamic. Risk assessors can compensate for uncertainties by using conservative assumptions, but an overly conservative analysis does not effectively inform risk management decisions, and baseline risk assessments should incorporate realism wherever possible. In this section, the following broad categories of uncertainty are described, specific examples from this risk assessment are addressed in detail, and the effects of such uncertainties on the risk evaluation are discussed:

- Data gaps and limitations
- Model uncertainty
- Toxicity information.

Not all of these uncertainties can be addressed by conservatism, so the discussion of each includes a clear statement of whether the resulting bias is conservative, not conservative, or unknown. Finally, several underlying methods and assumptions provide an overall conservatism to the analysis, and these are outlined and described within the categories listed above.

7.1 Data Gaps and Data Limitations

Although a significant number of analytical samples have been collected for the remedial investigation and risk assessments, there are some data gaps that affect the degree of certainty associated with risk estimates: the absence of data for surface water or porewater chemistry; the absence of data for some tissue types that are potentially ingested by receptors; the actual chemical concentrations of sediment on the TCRA cap, now and in the future; a limit to the number of samples that can be collected for the RI; an absence of detailed information about use of the Site by certain protected species; and the lack of information on the toxicity of $COPC_{ES}$ to reptiles. Each of these is discussed below.

7.1.1 Surface Water Chemistry

There are no empirical data in the baseline dataset to describe concentrations of most $COPC_{ES}$ in water, and there are limited water data only for dioxins and furans and PCBs. As a result, a simple model was used to generate conservative estimates of water concentrations for use in the wildlife exposure model, and to estimate porewater chemistry for those chemicals lacking SQGs expressed as bulk sediment concentrations. The results of these simple partitioning models are conservative estimates of $COPC_E$ concentrations in water for several reasons. First, the partitioning models represented by the K_d and K_{oc} values used (Table 4-1) assume a two-phase system in equilibrium, and the resulting prediction is for a dissolved concentration. In reality, estuarine surface waters are complex multi-phase systems including several constituents such as dissolved organic carbon and other materials that bind chemicals, preventing them from entering solution. Moreover, it is also unlikely that water and sediment are at equilibrium, given the tidal dynamics in an estuarine environment. Also, because the sediment is in direct contact with only a limited volume of water, most of the surface water would mix with and dilute any metals or other COPCs partitioning from sediment. Such dilution was not accounted for by the simple models used.

The simple models were used because of the absence of empirical data. Because they generate a very conservative representation of water chemistry, they are useful for screening. That is, when these conservative estimates are below levels of concern, the exposure pathway or receptor–COPC pair can be eliminated from further consideration with a high degree of confidence. If an actual estimate of metals or other COPC in surface water were needed, a much larger set of information would be brought to bear, bringing greater realism to the estimate for this particular environment.

7.1.2 Tissue Chemistry for Plants and Terrestrial Invertebrates

Similarly, there are no data to describe concentrations of $COPC_{ES}$ in tissue of terrestrial invertebrates and in plants. For all estimates except for dioxins and furans in terrestrial invertebrate tissue, simple models derived for other sites and published in the literature, or by USEPA or the Oak Ridge National Laboratory for use in risks assessments were applied. These “off-the-shelf” models for estimating plant and invertebrate tissue concentrations provide a reasonable estimate of tissue concentrations, but they cannot account for the Site conditions affecting bioavailability, the particular species studied relative to those on the Site, seasonality of the data providing the basis for the model, local geochemistry and other factors that could affect uptake rates in plants and invertebrates. For the most part, the direction of bias created by using these models is unknown.

7.1.3 Post-TCRA Conditions

To analyze post-TCRA risk, it was necessary to make an assumption about the concentrations and mixture of dioxins and furans in sediments within the area inside the original 1966 impoundment perimeter. To perform the post-TCRA risk analysis, this risk assessment assumes that the area provides the same habitat function that it had up until the TCRA cap was constructed, and that dioxins and furans in sediments within the 1966 perimeter are at the median concentration in the background sediment dataset. This approach assumes that animals will continue to use the area as they did prior to implementation of the TCRA, and that the sediment coming onto the Site, and which becomes deposited on the TCRA cap, is from a broad area similar or equivalent to the background area. The approach also assumes that the conditions remain static, and does not attempt to evaluate the dynamics of sediment deposition and erosion on the TCRA cap in future years. Whether local conditions within USEPA's preliminary Site perimeter would have a greater or lesser effect than the upstream background conditions is unknown. It is also unknown whether the conditions will be static or dynamic, and whether state regulatory programs aimed at controlling releases of dioxins and furans in the region will result in a general lowering of dioxin and furan concentrations in background sediment, which could lower post-TCRA concentrations at the location of the cap.

The selected approach is appropriate because it is not speculative about these details of future conditions. However, if sediment conditions in the area directly adjacent to the 1966 impoundment perimeter do have a disproportionate impact on post-TCRA sediment conditions because of their proximity to the TCRA cap, then the post-TCRA evaluation could slightly underestimate risk. However, the overall conclusion that the TCRA has resulted in significant risk reduction would not change.

7.1.4 Sample Numbers

Designs of all of the studies supporting the RI were developed in collaboration with USEPA, and data collection was performed with USEPA approval of each SAP, with specific DQOs articulated according to the four study elements used to structure the investigation. In each study, the sampling design was directed towards characterizing conditions adjacent to and within the impoundments north of I-10, which is appropriate because the wastes in those impoundments are most likely the primary source of COPCs in the environments addressed by this BERA (a BERA for the southern impoundments is to be presented with the RI). As a result, there is a spatial bias that emphasizes conditions near and in the northern impoundments in the dataset, resulting in a relative over-representation of chemical conditions there in the sediment, soil, and tissue data. This is the most important limitation to the existing data that introduces a conservative bias. Combined with the definition of exposure units that encompass and emphasize the area of the former waste impoundments, the spatially biased sampling designs result in a representation of risk that may overstate exposure and risk to all receptors.

The shoreline exposure unit for great blue heron, spotted sandpiper, and marsh rice rat (Figure 4-14) provides an example of how the spatial bias in the sampling design results in a conservative bias in the exposure analysis. For these receptors, a large proportion of the sediment samples used to calculate EPCs were collected from within the impoundments (more than 50 percent), including several from directly within the wastes, even though the fraction of shoreline on the Site represented by the impoundments is less than 5 percent. Because there was no spatial weighting to normalize the area represented by each sample, the spatial bias in sampling resulting from a focus on the waste impoundments skews the CT and RM exposure statistics upward, directly affecting exposure and risk estimates. Results indicating that implementation of the TCRA resolves risks due to dioxins and furans illustrate the importance of the spatial bias in driving risk estimates.

7.1.5 Threatened and Endangered Species

There are no systematic observations of threatened or endangered species occurring on or near the Site, so use of the Site by those species was inferred on the basis of habitat availability on and near the Site (Section 3.4). As described in Section 3.4.4, among the six threatened or endangered species that could occur in the vicinity of the Site, only the brown pelican would be likely to use the Site for resting and foraging. The bald eagle and white-faced ibis could also visit occasionally. All of these birds have foraging ranges much larger than the Site area, and would therefore be expected to be exposed to a lesser degree than any of the modeled receptors. Evaluation of white-faced ibis, brown pelican, and bald eagle using surrogate receptors and adjustments to exposure areas showed an absence of risk to all three protected bird species.

7.2 Model Uncertainty

Several model forms are used to support the risk assessment, including ratios such as BAFs and BMFs, regression models to predict dioxin and furan concentrations in worm tissue or in bird eggs, and ingested-dose models to estimate exposure of individuals. Each type of model can introduce both bias and inaccuracies.

7.2.1 Prediction Using Ratios

Ratios are the simplest representation of the relationship between chemical concentrations in a source medium, such as sediment or prey, and in tissue. Ratios are calculated as the concentration in the tissue of interest, divided by the concentration in a single exposure medium, which may be sediment, water, or food. Underlying the use of ratios is the assumption of a strictly proportional relationship between concentrations in the two media. Although ratios are widely used, the assumption of proportionality is rarely demonstrated to be justified. For this reason, ratios are the most likely to introduce inaccuracies. Because truly linear relationships between abiotic media or prey and biological tissue are rare, use of ratios introduces a conservative bias, which is made worse at higher concentrations in the abiotic medium. The range and variability of BMFs found linking PCB congeners in foods of fish to those in bird eggs (Table 4-18) illustrates the random variability in predictions that occurs when ratios are used.

7.2.2 Prediction Using Regression Models

Regression analysis of concentrations in tissue and soil or sediment, or between different tissue types (e.g., a consumer and its prey), is a straightforward method using well-established statistical procedures. Regression analysis has several advantages over ratios, specifically the ability to incorporate non-zero intercepts and to produce a statistically sound measure of uncertainty. Because it is a strictly empirical method, regression analysis does not require any information on the mechanisms of exposure and uptake, and thus can be applied to the sort of site characterization data typically collected in an RI/FS. Several guidance documents supporting the use of regression modeling in the process of developing risk assessments and remedial goals have been published (Exponent 1998; Corl 2001; University of Florida 2005). Application of regression models in this risk assessment avoids the sort of overly conservative bias introduced by ratios, because regression models produce predictions within the known variance of the dataset used to build the model. Although Site-specific data are preferred for making predictions for the Site, regression models which limit prediction error on the basis of an empirical dataset are preferred over the use of ratios because they better control the uncertainty in predicted concentrations, at both the lower and upper ends of the predicted range.

7.2.2.1 Fish-to-Bird Egg Models for Dioxins and Furans

Bioaccumulation of dioxins and furans in birds is poorly described in the literature with paucity in available data to guide model development for the prediction of accumulation and transfer from exposure media to tissues of birds. As a result, any estimates of bioaccumulation will result in uncertainties. Use of regression models provides the most straightforward means to limit uncertainty, and requires only the use of data that are available for the Site, instead of multiple variables that would have to be derived from the literature to implement other types of models, such as mechanistic models.

As detailed in the Technical Memorandum on Bioaccumulation Modeling (Integral 2010b), the relationships between exposure concentrations and tissue concentrations in birds is a complex process balancing absorption rates, metabolism, excretion, and maternal transfer, all of which operate on a congener- and species-specific basis. The literature review performed for this risk assessment produced only one study providing a regression based model for estimating concentrations of dioxins and furans in bird eggs (Elliott et al. 2001). Uncertainties associated with this model arise from the derivation of the regression relationship for the individual congener groups. We were unable to directly test whether our input data follows a distribution similar to that of the fish data used to generate the regression. However, visual inspection of a subset of graphically presented data showed a similar distribution of concentrations in prey for PeCDD and HxCDD. Further uncertainty exists for TCDF, which did not result in a strong linear association between prey fish and measured egg concentrations although a positive and potentially nonlinear relationship was evident. Hence, modeled values for this congener are subject to the uncertainty of the regression analysis, which shows that the relationship applies 93 percent of the time (p=0.07) and therefore provides a reasonable level of confidence in the estimated bird egg concentrations. Further, we are unable to assess the uncertainty of congeners for which the regression modeled was derived from other homologous groups (e.g., HpCDD and HpCDF).

Uncertainties also exist in species specific differences in metabolism and excretion of dioxins and furans. The regression model of Elliott et al. (2001) was developed for the great blue heron. This risk assessment applies the models to predict egg concentrations for herons as well as cormorant and sandpipers for which differences in dioxin and furan metabolism may exist. This same difference in metabolism extends to the absorption of compounds from the different tissues. Also, the model was derived using fish as the ingested exposure medium; however, this risk assessment has extended the models to estimate the transfer of dioxins and furans from crab, clam, and sediments. Nosek et al. (1992a) demonstrated differences in the oral bioavailability of TCDD from different matrices in pheasant hens. Similar oral bioavailability was determined for earthworms (30 percent) and soils (33 percent) while higher availability was found for paper mill sludge (41 percent) and a suspension of crickets (58 percent). Application of Elliott et al.'s (2001) regression models to a mixed media diet assumes that the uptake of dioxins and furans from sediments, crab, and clams occurs at the same rate as for those in fish. The importance of this assumption is unknown, but likely somewhat conservative in light of the 30 to 33 percent bioavailability from soils and higher bioavailability from foods demonstrated by Nosek et al. (1992a). However, the assumption was necessary because sediment and shoreline sediment concentrations account for the highest dose of dioxins and furans to birds (Appendix C).

7.2.2.2 Soil-to-Invertebrate Tissue Models for Dioxins and Furans

In modeling dioxin and furan congener uptake into invertebrate tissue, it cannot be assumed that all congeners behave similarly, and a congener-specific approach is needed (Integral

2010a; Appendix D). The regression approach selected to estimate dioxin and furan concentrations in earthworm tissue from concentrations in colocated soils relies on a relatively small sample size (N=6) of colocated soil and earthworm tissue, and there is uncertainty in developing regressions from this small sample size. 2,3,7,8-TCDD was not detected in several of the soil–earthworm pairs in the available dataset, which also introduces uncertainty in the use of these data. However, uncertainties were reduced by limiting selections to statistically significant relationships on a congener-specific basis, and selecting congener correlates to minimize underprediction of tissue concentrations in a soil environment characterized by significant spatial variability in concentrations of some dioxin and furan concentrations. This approach affords substantive advantages in terms of providing an empirically based estimation of invertebrate uptake of dioxins and furans over simplified approaches using biota–sediment accumulation factors or extrapolating widely across aggregate variables such as TEQ.

7.2.3 RM Exposure and the Risk Profile

A CT and RM EPC were generated for each exposure medium within each exposure unit for use in the fish and wildlife exposure models. Using these two expressions of the EPC for any given $COPC_E$ enables presentation of the most likely (CT) exposure, along with the upper bound (RM) exposure condition, and is intended to reflect the "exposure profile" for receptors recommended by USEPA guidance (USEPA 1997, 1998). However, this profile is biased high, because the RMin exposure, calculated as the 95 percent lower confidence limit on the mean (or a similar statistic) is as likely to occur as the RM. An illustration of the importance of this bias in interpreting risks is provided in Table 7-1. This table shows the CT, RM, and RMin $TEQ_{DF,B}$ EPCs for soils used to estimate exposure to killdeer. The RMin is a more than a factor of 7 below the CT. A proportionate decrease in the final HQ_N would lead to a conclusion of negligible risk to killdeer. This example illustrates how the use of the RMin exposure estimate, which is equally as likely as the RM exposure, leads to no finding of risk.

7.2.4 Wildlife Exposure Model

The wildlife exposure model uses fixed parameter values, set at realistic or conservative levels, to make predictions about contaminant intake. Use of conservative assumptions, such as the redistribution of plant matter (likely less contaminated) in the diet of omnivorous benthic fish into a compartment representing animal matter (likely more contaminated) (Table 4-3) is an example of this type of conservatism in the wildlife exposure model. The bird egg exposure models for dioxins and furans also make several conservative assumptions, including the use of total homologue concentrations (not the sum of 2,3,7,8-substituted congeners within a homologue) as the basis for exposure model for several congeners. This approach allows a conclusion of negligible risk to be made with confidence. In some cases, information from the literature that informs specific details of the exposure analysis are employed. While incorporation of this information is considered carefully on the basis of the technical merits of the study or studies reporting the data, it may or may not reflect actual conditions in the field. Examples of this is the use of the RBAs described in Section 4.3.1.2 in the exposure assessment for birds, the choices about proportions of each prey type in the diets of fish and wildlife receptors, and the timing and duration of exposures of each receptor. Generally, these choices are demonstrably conservative; for example, the BERA assumes that blue heron consume a significant amount of catfish, the species with the highest concentrations of dioxins and furans in fillet among those captured on the Site (see Appendix B of the Exposure Assessment Memorandum; Integral 2012), when they likely eat a range of fish sizes and species. The BERA also assumes continuous exposures exclusively in the study area. But not all assumptions are clearly conservative, and are instead included to impart realism to the extent possible. Use of RBAs in the exposure assessment for birds is an example of this.

At USEPA's request in comments on the draft BERA (Appendix F), a sensitivity analysis was conducted to evaluate the importance of the RBAs in risk conclusions for birds. To do this, exposures of birds to dioxins and furans (as $TEQ_{DF,B}$) was recalculated without using RBAs, (i.e., assuming that ingested TCDD was 100 percent bioavailable to the birds). Details of this sensitivity analysis are discussed below.

In addition, a deterministic risk calculation can oversimplify the risk conclusions, by suggesting a black and white risk/no risk conclusion. To improve the depth of the evaluation, risk was evaluated probabilistically when the deterministic models suggested that exposures could exceed the LOAEL. Use of the probabilistic food web exposure model is also discussed below.

7.2.4.1 Alternative Assessment of Exposure of Birds to for Dioxins and Furans

A sensitivity analysis was conducted to evaluate the importance of the RBAs in risk conclusions for birds. To do this, exposures of birds to dioxins and furans (as $TEQ_{DF,B}$) were recalculated without using RBAs (i.e., assuming that ingested TCDD was 100 percent bioavailable to the birds). The results of these analyses are presented in Table 7-2. Tables 7-3 and 7-4 also provide a comparison of the effect of the RBA on pre-and post-TCRA analyses and site and background analyses, respectively. Probabilistic risk analyses of sandpiper and killdeer exposure to $TEQ_{DF,B}$ without the RBA are provided in Figures 7-1 and 7-2.

The results of these analyses are similar to results with the RBA; the probability that exposures of killdeer to dioxins and furans will exceed the effects level is still low (5.6 percent without the use of the RBA [Figure 7-1] compared to 4.7 percent with the RBA [Figure 6-19]), and the probability that the exposures of spotted sandpiper to dioxins and furans will exceed the effects level is still low (14.7 percent without the use of the RBA [Figure 7-2] compared to 13.7 percent with the RBA [Figure 6-20]). There are no changes to the risk conclusions relative to the outcome of the risk analysis using the RBA. Thus, while the use of the RBA is considered an appropriate adjustment to TCDD bioavailability, its use does not have a substantive effect on the outcome of the risk evaluation for avian receptors.

7.3 Use of Probabilistic Exposure Models

In this risk assessment, when predicted ingestion exposures exceeded LOAELs, the probabilistic exposure assessment incorporates the variability of the exposure parameters in the deterministic model, providing a more precise statement of probability of adverse outcomes (e.g., an 8.3 percent chance that exposure of killdeer to zinc will exceed the LOAEL). This statement uses empirical information about the Site to more accurately reflect likelihood of an effect on an individual.

7.4 Toxicity Information

Ecological risk assessments rely on a very limited set of toxicity information, usually developed with very few species derived from domestic stocks. Often, these domestic species are less fit than wild species, which benefit from greater genetic diversity in each generation.

The advantage of using controlled laboratory studies is certainty in the dose-response relationship that is derived from highly controlled exposures such as injection or oral administration by gavage. However, loss of realism is significant to risk assessment, because toxicity studies cannot represent variability in individual fitness, variable resistance or sensitivity among species, physical controls on bioavailability that exist in the field, and a host of other factors affecting potential toxicity in the environment. Generally, the bias resulting from the use of laboratory-based toxicity studies is considered conservative.

Toxicity information for PCBs in fish, birds, and mammals relied largely on toxicity studies in which the test subjects were administered Aroclor 1254. To achieve a risk assessment protective of the ecological receptors addressed by this BERA, concentrations of total PCBs were used in exposure estimates. For sediments, PCB congener data are insufficient for calculating total PCBs as the sum of congeners. The concentrations of total PCBs in sediment were therefore estimated by summing the concentrations of Aroclors with nondetects set to one-half the detection limit. Because data for PCB congeners are available for tissue samples (and Aroclor data are not), total PCBs in prey was estimated by summing the concentrations of 43 congeners analyzed in prey tissue.

Finally, very conservative assumptions were applied in calculating total PCBs in sediment as the sum of Aroclors with nondetects set to one-half the detection limit. For the sediment study, PCBs were analyzed and reported as Aroclors, consistent with the Sediment SAP.[5] However, in several samples of material from within the 1966 impoundment perimeter, matrix interferences resulted in elevated detection limits for Aroclors. The use of these elevated detection limits for the sum of Aroclors likely results in a substantial overestimate of the sediment EPC for total PCBs. This is a conservative assumption because no Aroclors were detected in surface sediment within the 1966 impoundment perimeter during the sediment study for this RI, and only a single detected concentration of Aroclor 1254 was measured at depth (2 to 4 feet) within this area (i.e., Station SJGB014, 1,400 µg/kg [qualifier – J]). This estimated concentration is lower than the elevated detection limit for this Aroclor in two of the stations where detection limits were elevated. Moreover, in the Screening Site

[5] The USEPA comment requiring evaluation of exposures to total PCBs as the sum of 43 specific congeners was first articulated in the comments on the Tissue SAP, which was produced after the Sediment SAP was final and implemented. See Appendix C of the Tissue SAP.

Assessment Report (TCEQ and USEPA 2006), which reports Aroclor results for several samples from within the wastes in the western cell of the northern impoundments, Aroclors were never detected, and detection limits were much lower in that study (<90 µg/kg). In summary, there is uncertainty about the actual Aroclor concentrations in the materials collected from within the 1966 impoundment, but the estimated concentration of Aroclor 1254 at Station SJGB014, and results of TCEQ and USEPA (2006) confirm that the approach taken to estimating total PCBs in sediment is conservative.

In addition to uncertainty in calculation of sediment EPCs for total PCBs, there is uncertainty associated with the use of Aroclor 1254 toxicity information in combination with total PCBs as the exposure metric. The mixture of PCB congeners in sediments and tissue at the Site may not reflect the same congener composition as Aroclor 1254. Nevertheless, the assessment approach should be protective because Aroclor 1254 is expected to be among the Aroclors most toxic to aquatic organisms (Nebeker and Puglisi 1974; Mayer et al. 1977; Johnson and Finley 1980), and dechlorination of PCBs by natural processes at the Site would likely lead to mixtures with toxicity less than or equal to Aroclor 1254.

7.4.1 Use of Uncertainty Factors for Deriving TRVs

The preferred approach for selecting TRVs is to find values that meet acceptability criteria (Section 1.4 of Appendix B) and are taxonomically relevant and appropriate to the receptors of concern, but in some cases, data may not be available for the receptor and COPC of interest. In these cases, the application of an uncertainty factor to conservatively estimate the benchmark or TRV may be considered. In a review of the types and uses of uncertainty factors, Chapman et al. (1999) conclude that an uncertainty factor should account for the uncertainty in the extrapolation, but should not be so large that it renders the resultant value meaningless for assessing risk.

Chapman et al.'s (1999) review emphasizes the importance of evaluating the substance and context of the uncertainty. They caution against the extrapolation of LOAELs to NOAELs because there can be substantial uncertainty in moving from effects to no-effects concentrations. They provide several examples that support the use of uncertainty factors of 10 or less for individual extrapolations, including extrapolation of acute lethality toxicity tests to thresholds for sublethal effects in aquatic systems, and lowest-observed-effect concentration to no-observed-effect concentration ratios for wildlife criteria (Chapman et al. 1999). This review points out that uncertainty factors are essentially screening tools for which the imprecision cannot be quantified, and should not be regarded as mathematical absolutes. These recommendations were used as a basis for the application of uncertainty factors in deriving TRVs where relevant effects level values were missing but related values were available.

7.4.2 Toxicity of Mixtures

Organisms inhabiting or using the Site are exposed to more than one chemical. They may be exposed to COPCs and other chemicals in locations other than the Site. Exposures of organisms to chemicals other than COPCs off of the Site cannot be estimated. Each contaminated area and each individual receptor results in a unique exposure profile, characterized by both the specific chemical mixtures and the magnitude of each chemical. For most chemicals, there are simply no published studies to evaluate these unique mixtures.

Some chemicals have known additivity, such as dioxins and furans, and mixtures are evaluated on the basis of the best available science. For other COPCs, whether effects associated with one COPC are additive with another cannot be addressed without significant effort, and may not be resolved in any case. In this evaluation, because very few COPCs are associated with unacceptable risk, evaluation of mixtures other than dioxin, furan, and dioxin-like PCBs was not conducted. The related uncertainty is considered minor for this Site.

7.4.3 Bivalve Toxicity for Dioxins and Furans Other than TCDD

Appendix B provides an overview of the technical literature available for the evaluation of dioxin and furan toxicity to invertebrates. Several studies have found no adverse effects in freshwater or marine invertebrates following exposure to TCDD; studies to provide systematic toxicity data for the other dioxin and furan congeners are rare. The literature and related analyses find that invertebrates are relatively insensitive to TCDD toxicity. Although AhR homologues have been identified in various invertebrate species, invertebrate AhR homologues lack the ability to bind dioxins (Hahn et al. 1992; Butler et al. 2001), which may explain the relatively low sensitivity of invertebrates to TCDD toxicity.

The histological and developmental toxicity of TCDD in eastern oysters documented by Wintermyer and Cooper (2007) illustrates that TCDD toxicity to bivalves is likely through non-AhR mediated pathways. Therefore, this study and related literature do not inform the question of whether other dioxin and furan congeners may also have similar effects in oysters, other additional effects, or no effects at all. Because the mechanism of toxicity to oyster reproductive tissues is not described, it cannot be concluded that other dioxin and furan congeners may have the same effects. Because these other congeners have not been tested, it cannot be concluded that they do not have any effect. Therefore, the absence of information on the toxicity of dioxin and furan congeners other than TCDD results in uncertainty about risks to bivalves on the Site from these chemicals.

7.4.4 Toxicity to Reptiles

Finally, the absence of information on toxicity of $COPC_E$s to reptiles and the inability to estimate actual exposures to reptiles, which may include dermal uptake, create important uncertainties. Even if reptile tissue samples from the Site had been collected, interpretation of these in terms of risks would not be possible. Calculation of exposure in terms of daily ingested dose for each $COPC_E$, and comparison with birds and mammals, which generally showed little to no risks for the majority of $COPC_E$s, suggest that there are no risks to reptiles for most $COPC_E$s. However, a remaining uncertainty that cannot be resolved is whether baseline exposures of reptiles to dioxins and furans were at levels that could result in unacceptable risks.

7.5 Summary of Uncertainties

Uncertainty in an ecological risk assessment is unavoidable, because the risk assessment attempts to model the natural environment, which is highly variable and complex. Although a baseline ecological risk assessment should incorporate realism to the maximum extent possible, conservative choices are made throughout the process, making the risk assessment generally conservative. Even so, data gaps such as the lack of toxicity information for reptiles, and randomness, such as that introduced by use of ratios to make predictions, cannot be resolved, and their bias cannot be said to be conservative or otherwise. The analyses presented here result in a high level of confidence when there is a conclusion of no risk.

8 SUMMARY OF ECOLOGICAL RISKS AND RISK CONCLUSIONS

This section synthesizes the results of the risk characterization and uncertainty analysis to provide an overall conclusion about the ecological risks at the Site.

8.1 Characterization of Risks to Benthic Invertebrates

A conservative assessment of risks to benthic invertebrates indicates no risks to the assessment endpoint of the abundance and diversity of benthic macroinvertebrate communities from exposure to BEHP, phenol, cobalt, copper, lead, thallium, and zinc. Carbazole and aluminum concentrations in surface sediments of the Site are not greater than in background areas, and risks associated with these metals are therefore not greater than background risks. Barium and vanadium, for which information on toxicity to benthic macroinvertebrates is lacking, and manganese are randomly distributed in sediments, and therefore appear not to be associated with the source material in the impoundments. Concentrations of mercury exceed a conservative SQG in two locations, but these exceedances do not equate to a prediction of effects. If effects exist at these two locations, the affected areas are isolated and small, and do not adversely affect the assessment endpoint, abundance and diversity of the overall benthic macroinvertebrate community.

Concentrations of TCDD in sediment exceed the NOAEL in only two locations, within the original impoundment perimeter, but there were no studies identifying benthic invertebrate LOAELs for dioxins and furans in sediment. NOAEL values as high as 25,000 ng/kg have been reported (Appendix Table B-4), suggesting that concentrations of TCDD in sediments are not sufficiently high to negatively impact the benthic macroinvertebrate community.

Clam tissue concentrations of TCDD are sufficiently elevated in samples collected directly adjacent to the impoundments to indicate reproductive risks to individual molluscs in that area. Concentrations of TCDD in clam tissue from two of five samples at Transect 5, directly adjacent to the upland sand separation area, exceed a threshold of reproductive effects in individual oysters. These localized effects do not adversely affect the assessment endpoint, stable or increasing populations of bivalves within the Site, because the affected area is limited to the immediate vicinity of the impoundments north of I-10.

8.2 Characterization of Risks to Fish

Assessment of baseline risks to fish considered the concentrations of cadmium, copper, mercury, and zinc in the diets of fish, the concentrations of BEHP and nickel in water, and the concentrations of total PCBs, $TEQ_{DF,F}$, $TEQ_{P,F}$, and $TEQ_{DFP,F}$ in whole fish. Results indicate that baseline risks to the assessment endpoint, stable or increasing populations of benthic omnivorous fish, benthic invertivorous fish, and benthic piscivorous fish on the Site are negligible.

8.3 Characterization of Risks to Birds

Baseline risks to the assessment endpoint of stable or increasing populations of great blue heron and neotropic cormorant, and the birds in their feeding guilds that are represented by these receptor surrogates and that could use the Site are negligible. Exceedance of the egg tissue based NOAEC for great blue heron and cormorant ingesting prey and sediment at the Site are noted, but do not indicate risk to the assessment endpoints for piscivorous birds. Baseline risks to terrestrial invertivorous birds such as the killdeer are also negligible for all $COPC_{ES}$ except zinc and dioxins and furans. Baseline risks to spotted sandpiper and similar shorebirds, which ingest substantial amounts of sediment as a result of their foraging habit, are negligible for all $COPC_{ES}$ except for dioxins and furans.

There is a low probability (8.3 percent) that exposures of individual killdeer to zinc could exceed levels affecting reproduction, indicating negligible risk to the assessment endpoint of stable or increasing populations of terrestrial invertivorous birds. Uncertainties about the bioavailability of zinc from site soils, and of the form of this metal in foods and soils on the Site relative to the form used in toxicity tests result in a conservative bias in the risk assessment for zinc in killdeer. Exposures of killdeer to zinc on the Site are only slightly greater than exposures in background areas. There is also a low probability (4.7 percent) that exposures of individual killdeer to $TEQ_{DF.B}$ could exceed the LOAEL at the Site. Overall, risks to terrestrial invertivorous bird populations on the Site from zinc are very low to negligible.

There is a probability of 13.7 percent that exposure of individual spotted sandpipers and the species it represents to dioxins and furans exceeds exposures associated with reproductive effects in individual birds under baseline (pre-TCRA) conditions. Although probability of this exposure level was only calculated using the ingestion rate of birds, results of the modeling to estimate egg concentrations also indicate some baseline risk of reproductive effects from dioxins and furans in the spotted sandpiper. Among all vertebrate ecological receptors for this risk assessment, the sandpiper ingests the largest amount of sediment (per unit body weight), which is the most important source of their exposure. Implementation of the TCRA reduced risk to spotted sandpiper to negligible.

8.4 Characterization of Risks to Mammals

Baseline risks to raccoon and mammals in the same feeding guild as the raccoon that could use the Site are negligible. There is negligible risk to the assessment endpoint of stable or increasing populations of omnivorous mammals from any $COPC_E$. Baseline risks to the marsh rice rat, representative of aquatic mammals, are also negligible for all $COPC_{ES}$ except dioxins and furans. There is a 14.3 percent probability that an individual marsh rice rat using the Site under baseline conditions could be exposed to $TEQ_{DFP,M}$ at levels exceeding those associated with reproductive effects on mammals. Given the spatial bias in the dataset towards areas containing the most contaminated sediment on the Site, and given that these rodents can rear more than one litter each year (Appendix A), and that the probability of exposure at the effects level is low, baseline risks to the assessment endpoint of stable or increasing populations of omnivorous mammals on the Site as a whole are negligible. Implementation of the TCRA eliminated risks to the marsh rice rat and the mammals it represents.

8.5 Characterization of Risks to Reptiles

There is insufficient information on the toxicity of $COPC_{ES}$ to specifically address risks to the assessment endpoint of stable or increasing populations of reptiles using the Site. Although there are substantial uncertainties about dermal absorption of $COPC_{ES}$, in addition to uncertainties about toxicity, comparison of the alligator snapping turtle's ingested doses with those of bird and mammal receptors indicates that exposure potential of reptiles via ingestion is very low. For this reason, and because risks to $COPC_{ES}$ other than dioxins and furans are low for some but more often negligible for these other receptors, risks to reptiles to $COPC_{ES}$ other than dioxins and furans are also considered to be low. However, risks to reptiles living in close association with the former waste impoundments from exposure to dioxins and furans could exist under baseline conditions, because risks to spotted sandpiper and marsh rice rat are present, and because reptiles may be more susceptible to dermal uptake of dioxins and furans, increasing their exposure over estimates presented herein. Similarly, because implementation of the TCRA resolves risks to sandpiper and marsh rice rat, any risk to reptiles, if present, would be similarly reduced. Risks to reptiles from exposure to dioxins and furans are unknown.

8.6 Ecological Risk Assessment Conclusions

Baseline risks to benthic macroinvertebrate communities and populations of fish, birds, mammals, and reptiles resulting from the presence of metals, BEHP, PCBs, carbazole, and phenol on the Site are negligible. Risks to fish populations from all COPC$_E$s are negligible. There are negligible risks to populations of wading birds represented by the great blue heron, and to populations of diving birds like the neotropic cormorant. There are negligible risks to populations of terrestrial mammals such as the raccoon.

There are low to negligible risks to individual terrestrial invertivorous birds like the killdeer from exposure to zinc, and negligible risks to populations of such birds. Although the upper bound of estimated daily intakes of zinc by individual killdeer is about equal to conservative effects thresholds, the exposure estimate is influenced by the use of generic models to estimate zinc concentrations in the foods of the killdeer, and this model likely overestimates ingested tissue concentrations, resulting in overestimates of exposure and risk. The highest exposures of killdeer to zinc occur outside of the northern impoundment perimeter, and background exposures less than 30 percent lower than on the Site. In addition, the low probability of individual exposures exceeding effects levels indicates low risk to populations. There are also low to negligible risks to individual terrestrial invertivorous birds from exposure to dioxins and furans.

Baseline risks to ecological receptors associated with the wastes in the impoundments north of I-10 are the result of exposures to dioxins and furans localized to the immediate vicinity of the impoundments. Baseline ecological risks include reproductive risks to molluscs from exposure to TCDD, but primarily in the area of Transect 3, which surrounds the former waste impoundments, and low risks of reproductive effects in individual molluscs in sediments adjacent to the upland sand separation area, but not to populations of molluscs. Baseline risks include moderate risks to individual birds like the killdeer or spotted sandpiper whose foraging area could regularly include the shoreline adjacent to the impoundments north of I-10, but low risk to populations because of the low to moderate probability that individual exposures reach effects levels. Baseline risks include risks to individual small mammals with home ranges that include areas adjacent to the impoundments such as the marsh rice rat, but low to negligible risks to small mammal populations because of the moderate probability that exposures will reach levels associated with reproductive effects in individuals, and because small mammals reproduce rapidly.

To the extent that risks from chemicals other than dioxins and furans occur on the Site, they are not associated solely with hazardous substances that may have been released from the wastes in the former impoundments. Substantial exposure of killdeer to zinc, and a variable fraction of the exposures of several receptors to PCBs, occur in background areas.

Implementation of the TCRA has reduced individual and population-level risks associated with dioxins and furans to negligible, but does not affect risks to killdeer from zinc, suggesting that the wastes in the northern impoundments are not the primary source of exposures of killdeer to zinc. Results of the evaluation of post-TCRA ecological risks support the conclusion that localized exposures of ecological receptors to the wastes in the northern impoundments is the primary driver of baseline ecological risk at the Site, and that therefore risks are localized, resulting from direct contact with the wastes in the northern impoundments.

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TABLES

Table 3-1
Chemicals of Potential Ecological Concern

Chemical	Receptors North of I-10 and Aquatic Environment: Benthic Invertebrates	Receptors North of I-10 and Aquatic Environment: Fish and Wildlife
Dioxins/Furans		
Dioxins and Furans	X	X
Polychlorinated Biphenyls		
Polychlorinated Biphenyls		X
Semivolatile Organic Compounds		
Bis(2-ethylhexyl)phthalate	X	X
Carbazole	X	
Phenol	X	
Metals		
Aluminum	X	
Barium	X	
Cadmium		X
Cobalt	X	
Copper	X	X
Lead	X	
Manganese	X	
Mercury	X	X
Nickel		X
Thallium	X	
Vanadium	X	
Zinc	X	X

Notes
$COPC_E$ = chemical of potential ecological concern

Table 3-2
Toxicity Equivalency Factors for Dioxins and Furans and Dioxin-Like PCBs

Compound	TEF-M (WHO 2005)[a]	TEF-Fish (WHO 1998)	TEF-Bird (WHO 1998)
Chlorinated Dibenzo-*p* -dioxins			
2,3,7,8-TCDD	1	1	1
1,2,3,7,8-PeCDD	1	1	1
1,2,3,4,7,8-HxCDD	0.1	0.5	0.05
1,2,3,6,7,8-HxCDD	0.1	0.01	0.01
1,2,3,7,8,9-HxCDD	0.1	0.01	0.1
1,2,3,4,6,7,8-HpCDD	0.01	0.001	0.001
OCDD	0.0003	0.0001	0.0001
Chlorinated Dibenzofurans			
2,3,7,8-TCDF	0.1	0.05	1
1,2,3,7,8-PeCDF	0.03	0.05	0.1
2,3,4,7,8-PeCDF	0.3	0.5	1
1,2,3,4,7,8-HxCDF	0.1	0.1	0.1
1,2,3,6,7,8-HxCDF	0.1	0.1	0.1
1,2,3,7,8,9-HxCDF	0.1	0.1	0.1
2,3,4,6,7,8-HxCDF	0.1	0.1	0.1
1,2,3,4,6,7,8-HpCDF	0.01	0.01	0.01
1,2,3,4,7,8,9-HpCDF	0.01	0.01	0.01
OCDF	0.0003	0.0001	0.0001
Non-ortho Substituted PCBs			
3,3',4,4'-Tetrachlorobiphenyl (PCB 77)	0.0001	0.0001	0.05
3,4,4',5-Tetrachlorobiphenyl (PCB 81)	0.0003	0.0005	0.1
3,3',4,4',5-Pentachlorobiphenyl (PCB 126)	0.1	0.005	0.1
3,3',4,4',5,5'-Hexachlorobiphenyl (PCB 169)	0.03	0.00005	0.001
Mono-ortho Substituted PCBs			
2,3,3',4,4'-Pentachlorobiphenyl (PCB 105)	0.00003	0.000005	0.0001
2,3,4,4',5-Pentachlorobiphenyl (PCB 114)	0.00003	0.000005	0.0001
2,3',4,4',5-Pentachlorobiphenyl (PCB 118)	0.00003	0.000005	0.00001
2',3,4,4',5-Pentachlorobiphenyl (PCB 123)	0.00003	0.000005	0.00001
2,3,3',4,4',5-Hexachlorobiphenyl (PCB 156)	0.00003	0.000005	0.0001
2,3,3',4,4',5'-Hexachlorobiphenyl (PCB 157)	0.00003	0.000005	0.0001
2,3',4,4',5,5'-Hexachlorobiphenyl (PCB 167)	0.00003	0.000005	0.00001
2,3,3',4,4',5,5'-Heptachlorobiphenyl (PCB 189)	0.00003	0.000005	0.00001

Sources

WHO (1998) corresponds to Van den Berg et al. (1997)

WHO (2005) corresponds to Van den Berg et al. (2006)

Notes

PCB = polychlorinated biphenyl
TEF-M = mammalian toxicity equivalency factor
a - Endorsed by USEPA (2010a)

Table 3-3
Reptiles and Amphibians That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations[a]	Federal and/or State Listing	Source of Information
Amphibians				
Gulf Coast toad	*Bufo valliceps valliceps*	From coastal prairies and barrier beaches along the Gulf of Mexico to roadside and irrigation ditches to urban/suburban sewers and backyard gardens		University of Texas (1999)
Southern leopard frog	*Rana sphenocephala utricularia*	All types of shallow freshwater habitats, including temporary pools, cypress ponds, ponds, lakes, ditches, irrigation canals, and stream and river edges; will inhabit slightly brackish coastal wetland		USFWS (2009c); TPWD (2009b)
Reptiles				
American alligator	*Alligator mississippiensis*	Alligators are found in or near water. They are common in swamps, rivers, bayous, and marshes. While typically found in fresh-water, they can tolerate brackish water as well.		USFWS (2009c)
Western cottonmouth	*Agkistrodon piscivorus leucostor*	Western cottonmouths prefer lowland swamps, lakes, rivers, sloughs, irrigation ditches, rice fields and salt marshes, but are not confined to living in moist habitats		USFWS (2009c)
Gulf Salt Marsh snake	*Nerodia clarkii*	Just as the name indicates, gulf salt marsh snakes prefer brackish and saltwater estuaries, salt marshes and tidal mud flats.	R	USFWS (2009c)

Table 3-3
Reptiles and Amphibians That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations[a]	Federal and/or State Listing	Source of Information
Texas garter snake	*Thamnophis sirtalis annectens*	Wet or moist microhabitats are conducive to the species occurrence, but is not necessarily restricted to them; hibernates underground or in or under surface cover; breeds March-August	R	TPWD (2010)
Timber rattlesnake	*Crotalus horridus*	Swamps, floodplains, upland pine and deciduous woodlands, riparian zones, abandoned farmland; limestone bluffs, sandy soil or black clay; prefers dense ground cover, i.e., grapevines or palmetto	T	TPWD (2010)
Smooth green snake	*Liochlorophis vernalis*	Gulf Coastal Plain; mesic coastal shortgrass prairie vegetation; prefers dense vegetation	T	TPWD (2010)
Common snapping turtle	*Chelydra serpentina*	The snapping turtle can be found in waters ranging from slow moving rivers to stagnant ponds.		USFWS (2009c)
Alligator snapping turtle	*Macrochelys temminickii*	Alligator snapping turtles generally live in the deepest water within their habitat: large rivers, canals, lakes, swamps, and rivers.	T	USFWS (2009c)
Western chicken turtle	*Deirochelys reticularia maria*	Chicken turtles are semi-aquatic turtles, found both in water and on land. They prefer water with dense vegetation and soft substrate.		USFWS (2009c)

Table 3-3
Reptiles and Amphibians That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations[a]	Federal and/or State Listing	Source of Information
Eastern river cooter	*Psuedemysconcinna metteri*	The river cooter is primarily a river turtle, but can be found in ditches and saltwater areas near river mouths. Rivers with slow to moderate currents, abundant aquatic vegetation, and rocky bottoms are preferred. Other less frequently used habitats include lakes, ponds, deep springs, floodplain river pools, and swamps.		USFWS (2009c)
Common musk turtle	*Sternotherus odoratus*	The habitat of the common musk turtle includes any kind of permanent body of water, like shallow streams, ponds, rivers, or clear water lakes, and it is rare to find the turtle elsewhere.		USFWS (2009c)
Red-eared sldier	*Trachemys scripta elegans*	The red-eared slider enjoy large areas where they are free to swim. These turtles also require a basking area, where they can completely leave the water and enjoy the light provided for them.		USFWS (2009c)
Texas spiny softshell turtles	*Trionyx spiniferus emoryi*	Soft-shelled turtles are almost entirely aquatic powerful swimmers, fond of basking and rarely venture far from aquatic margins.		USFWS (2009c)

Table 3-3
Reptiles and Amphibians That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations[a]	Federal and/or State Listing	Source of Information
Diamondback terrapin	*Malaclemys terrapin littoralis*	Diamondback terrapins prefer brackish or salt water. They are the only turtle found in estuaries, tidal creeks, and saltwater marshes where the salinity comes close to that of the ocean.	R	TPWD (2009b)

Notes
a - Additional habitat Information accessed at www.amphibiaweb.org and http://animaldiversity.ummz.umich.edu/site/index.html

Federal or State Listing
LE/LT = Federally Listed Endangered/Threatened
E/T = State Endangered/Threatened
R = Rare

Table 3-4
Fish That May Be Found in the Vicinity of the San Jacinto River Waste Pit Site

Common Name	Scientific Name	Habitat Associations and Diet[a]	Source of Information
Benthic			
Omnivores			
Pinfish	*Lagodon rhomboides*	Commonly found on vegetated bottoms, occasionally over rocky bottoms and in mangrove areas. Enters brackish water and even freshwaters. Feeds mainly on small animals, especially crustaceans, but also takes mollusks, worms and occasionally small fishes that are associated with the grassy habitat.	Osborn et al. (1992)
Atlantic croaker	*Micropogonias undulatus*	Occurs usually over mud and sandy mud bottoms in coastal waters and in estuaries where the nursery and feeding grounds are located. Feeds mainly on worms, crustaceans and fishes.	Osborn et al. (1992)
Hardhead catfish	*Ariopsis felis*	Inhabits continental waters and enters estuaries. Found in turbid waters over muddy bottoms. Commonly captured from catwalks, bridges and piers, particularly in passes and inland waterways. It has a varied diet including detritus, invertebrates, and fish.	Crocker and Young (1990)
Carnivores			
Blue catfish	*Ictalurus furcatus*	Diet is variable, tends to eat fish earlier in life. Also uses invertebrates Inhabits deep water of impoundments, main channels, and backwaters of medium to large rivers, over mud, sand and gravel.	TPWD (2009a)
Channel catfish	*Ictalurus punctatus*	Estuaries, lagoons, brackish seas, rivers, streams, lakes and ponds. Feed primarily on small fish, crustaceans (e.g., crayfish), clams and snails; also feed on aquatic insects and small mammals.	TPWD (2009a)

Table 3-4
Fish That May Be Found in the Vicinity of the San Jacinto River Waste Pit Site

Common Name	Scientific Name	Habitat Associations and Diet[a]	Source of Information
Southern flounder	*Paralichthys lethostigma*	Found mostly over mud bottoms in estuaries and coastal waters to about 40 m depth. A cryptic species; tolerates low salinities; occurs frequently in brackish bays and estuaries, even on occasion in fresh water. This species moves to deeper water in winter, but is still easily accessible. Feeds chiefly on fishes, also on crabs and shrimps. Juveniles take mainly small bottom-living invertebrates	Osborn et al. (1992)
Bowfin [b]	*Amia calva*	Found in swampy, vegetated lakes and rivers. An air-breather that can withstand high temperatures, which enables it to survive in stagnant areas and is even known to aestivate; lethal temperature is 35.2°C. A voracious and opportunistic feeder, it uses scent as much as site and subsists on fish, frogs, crayfish, insects, and shrimps.	TPWD (2009b)
Pelagic			
Omnivore			
Grass carp	*Ctenopharyngodon idella*	Occurs in lakes, ponds, pools and backwaters of large rivers, preferring large, slow-flowing or standing water bodies with vegetation. Tolerant of a wide range of temperatures from 0° to 38°C, and salinities to as much as 10 ppt and oxygen levels down to 0.5 ppm. Feeds on higher aquatic plants and submerged grasses; takes also detritus, insects and other invertebrates.	USFWS (2009a)

Table 3-4
Fish That May Be Found in the Vicinity of the San Jacinto River Waste Pit Site

Common Name	Scientific Name	Habitat Associations and Diet[a]	Source of Information
Invertivore			
Gulf killifish	*Fundulus grandis*	Small fish species common in estuaries and rivers of the Gulf Coast. They do not migrate, remaining in the same location for their entire life. They eat various small invertebrates. Tolerates a wide range of salinities, from freshwater to estuarine.	Hassan-Williams et al. (2007)
Carnivore			
Dollar sunfish [b]	*Lepomis marginatus*	Inhabits sand-bottomed and mud-bottomed, usually brushy, pools of creeks and small to medium rivers; and also swamps. Feeds on midge larvae and microcrustacean.	TPWD (2009a)
Red drum	*Sciaenops ocellatus*	Occurs usually over sand and sandy mud bottoms in coastal waters and estuaries. Abundant in surf zone. Feeds mainly on crustaceans, mollusks and fishes.	TPWD (2009a)
Black drum	*Pogonias cromis*	Usually found over sand and sandy mud bottoms in coastal waters, especially in areas with large river runoffs. Juveniles often enter estuaries. Primarily a benthic feeder, mainly on crustaceans, molluscs and fishes.	Osborn et al. (1992)
Spotted seatrout	*Cynoscion nebulosus*	Inhabits river estuaries and shallow coastal marine waters over sand bottoms, often associated with seagrass beds. Also occurs in salt marshes and tidal pools of high salinity. Feeds mainly on crustaceans and fishes.	Osborn et al. (1992)

Table 3-4
Fish That May Be Found in the Vicinity of the San Jacinto River Waste Pit Site

Common Name	Scientific Name	Habitat Associations and Diet[a]	Source of Information
Bay anchovy	*Anchoa mitchilli*	More commonly found in shallow tidal areas with muddy bottoms and brackish waters, tolerates a wide range of salinities (virtually fresh to fully saline or hypersaline). Feeds mostly on Mysis and copepods, also small fishes, gastropods, and isopods.	Osborn et al. (1992)

Notes

a - Additional habitat association information from www.fishbase.org

b - Found rarely in estuaries

Table 3-5
Invertebrates That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations	Source of Information
Blue crab	*Callinectus sapidus*	Blue crabs are benthic in every type of habitat from the saltiest water of the gulf to the almost fresh water of the back bays and estuaries, from the low tide line to waters 120 ft (36 m) deep. It is considered a scavenger, eating dead or dying organisms, but will also take live prey.	Crocker and Young (1990)
Oyster	*Crassostrea virginica*	Eastern oysters are abundant in shallow saltwater bays, lagoons and estuaries, in water 8 to 25 feet (2.5 to 7.5 m) deep and between 28 and 90 degrees F. Have been collected in the vicinity of the Site.	Crocker and Young (1990), Broach (2010)
Stone crab	*Menippe mercenaria*	Stone crabs prefer bottoms of bays, oyster reefs and rock jetties where they can burrow or find refuge from predators. Juveniles do not usually dig burrows, but instead hide among rocks or in seagrass beds.	TPWD (2009b)
Hermit crab	*Clibanarius vittatus*	Benthic scavengers found in the intertidal.	GBIC (2009)
Fiddler crab	*Uca longisignalis*	Fiddler crabs are most often found in soft sand or mud near or around the edges of shallow salt marshes.	TPWD (2009b)
Asian clam	*Corbicula fluminea*	Sand and clay, salinities up to 13 ppt.	USGS (2009)
Common rangia	*Rangia cuneata*	Low salinity estuaries, <19 ppt, most found in 5 - 15 ppt. Found in sandy, muddy, and vegetated areas. Species has been collected from the vicinity of the Site.	USFWS (1983), Broach (2010)
Brown rangia	*Rangia flexuosa*	Typically found in the intertidal zone at the water's edge. Species has been collected from the vicinity of the Site.	Broach (2010)
Dark false mussel	*Mytilopsis leucophaeata*	Typically found in brackish waters.	Broach (2010)
Dwarf surf clam	*Mulinia lateralis*	The dwarf surf clam is normally found in the soft strata in benthic communities.	Broach (2010)
Surf clam	*Macoma mitchelli*		Young (2010)

Table 3-5
Invertebrates That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations	Source of Information
Hooked mussel	*Ischadium recurvum*	Typically found in the intertidal zone at the water's edge. Species has been collected from the vicinity of the Site.	Culbertson (2010)
Southern quahog	*Mercenaria texana*		Culbertson (2010)
Grass shrimp	*Palaemonetes pugio*	A small shrimp species common to the estuaries of the Gulf Coast. Short life span (6-12 months). Limited commercial, recreational, or consumptive value for humans, but is a food source for many other species. Inhabits low salinity areas with grassy shorelines.	GBIC (2009)

Table 3-6
Aquatic and Wetland Plants That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Federal and/or State Listing	Source of Information
Water-milfoil	*Myriophyllum pinnatum*		USFWS (2008)
Threeflower broomweed	*Thurovia triflora*	R	TPWD (2010)
Eastern woodland sedge	*Carex blanda Dewey*		USFWS (2008)
Thinfruit sedge	*Carex flaccosperma Dewey*		USFWS (2008)
Frank's sedge	*Carex frankii Kunth*		USFWS (2008)
Shoreline sedge	*Carex hyalinolepis Steud.*		USFWS (2008)
Greater bladder sedge	*Carex intumescens Rudge*		USFWS (2008)
Cypress swamp sedge	*Carex joorii L.H. Bailey*		USFWS (2008)
Blunt broom sedge	*Carex tribuloides Wahlenb.*		USFWS (2008)
Fox sedge	*Carex vulpinoidea Michx.*		USFWS (2008)
Common spikerush	*Eleocharis palustris*		USFWS (2008)
Blunt spikerush	*Eleocharis obtusa*		USFWS (2008)
Shortbristle horned beaksedge	*Rhynchospora corniculata*		USFWS (2008)
Scouring-rush	*Equisetum hyemale*		USFWS (2008)
Carolina foxtail	*Alopecurus carolinianus*		USFWS (2008)
Giant cutgrass	*Zizaniopsis miliacea*		USFWS (2008)
Jungle rice	*Echinochloa colona*		USFWS (2008)
Field paspalum	*Paspalum laeve Michx.*		USFWS (2008)
Southern canary grass	*Phalaris caroliniana*		USFWS (2008)
Cattail	*Typha latifola*		USFWS (2008)
Tapertip rush	*Juncus acuminatus*		USFWS (2008)
Forked rush	*Juncus dichotomus*		USFWS (2008)
Common rush	*Juncus effusus*		USFWS (2008)
Inland rush	*Juncus interior*		USFWS (2008)
Grassleaf rush	*Juncus marginatus*		USFWS (2008)
Path rush	*Juncus tenuis*		USFWS (2008)
Flat rush	*Juncus validus*		USFWS (2008)
Common duckmeat	*Spirodela polyrrhiza*		USFWS (2008)
Duckweed	*Lemna aequinoctialis*		USFWS (2008)
Water-meal	*Wolffia brasiliensis*		USFWS (2008)
Water-meal	*Wolffia columbiana*		USFWS (2008)
Marsh purslane	*Ludwigia palustris*		USFWS (2008)
Hairy water primrose	*Ludwigia grandiflora*		USFWS (2008)
Texas prairie dawn	*Hymenoxys texana*	LE, E	TPWD (2010)
Water lettuce	*Pistia stratiotes*		Gonzalez et al. (2006)
Common water hyacinth	*Eichornnia crassipes*		Gonzalez et al. (2006)

Federal or State Listing

LE/LT = Federally Listed Endangered/Threatened
E/T = State Endangered/Threatened
R = Rare (State; this does not indicate a regulatory listing status)

Table 3-7
Aquatic-Dependent Birds That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations and Diet	Federal and/or State Listing
Omnivore			
Gadwall	*Anas strepera*	Dabbling duck, primarily herbivorous but feeds on invertebrates during breeding season. Wetlands, ponds, marshes and lakes with heavily vegetated margins.	
Green winged teal	*Anas crecca*	Opportunistic feeder; seeds of aquatic vegetation, also invertebrates. Found in shallow ponds and marshes with abundant vegetation, tidal creeks and mudflats.	
Northern pintail	*Anas acuta*	Nests in open country with shallow, seasonal wetlands and low vegetation. Winters in wide variety of shallow inland freshwater and intertidal habitats.	
Blue-winged teal	*Anas discors*	Variable diet, including aquatic invertebrates, seeds and algae. Shallow ponds and wetlands.	
Mallard	*Anas platyrhynchos*	From large marshes to small river bends and bays; found in a wide variety of habitats. Variety of vegetation, increased feeding on invertebrates during breeding season.	
Black-bellied whistling duck	*Dendrocygna autumnalis*	Primarily feeds on plant material, but also consumes insects and molluscs. Breeds in coastal Texas. Primarily breeds in shallow freshwater ponds and lakes.	
Northern shoveler	*Anas clypeata*	Freshwater marshes, tidal bays in winter	
Lesser scaup	*Aythya, affinis*	Salt marshes, estuaries and lakes	
Ring-billed gull	*Larus delawarensis*	Breeds on islands in inland lakes, in winter along seacoasts	
Laughing gull	*Larus atricilla*	Nests in marshes, on beaches, and on islands along coast Found along coasts, in estuaries, bays, and inland lakes. Feeds along the ocean, on rivers, at landfills, and in urban parks.	

Table 3-7
Aquatic-Dependent Birds That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations and Diet	Federal and/or State Listing
Gull-billed tern	*Sterna nilotica*	Breeds on gravelly or sandy beaches. Winters in salt marshes, estuaries, lagoons and plowed fields, less frequently along rivers, around lakes and in fresh-water marshes.	
Roseate spoonbill	*Platalea ajaja*	Marsh habitat. Omnivore with a wide diet inluding plants, invertebrates and fish.	
Killdeer	*Charadrius viciferous*	Fields, coastal fields, beaches, lawns. Insects make up the majority of the killdeer's diet, but they will also eat berries and crustaceans.	
Invertivore			
Pied-billed grebe	*Podilymbus podiceps*	Breeds on seasonal or permanent ponds with dense stands of emergent vegetation, bays and sloughs. Uses most types of wetlands in winter.	
Least sandpiper	*Calidris minutilla*	Breeds in mossy or wet grassy tundra, occasionally in drier areas with scattered scrubby bushes. Migrates and winters in wet meadows, mudflats, flooded fields, shores of pools and lakes, and, less frequently, sandy beaches.	
Mottled duck	*Anas fulvigula*	Freshwater wetlands, ditches, wet prairies, and seasonally flooded marshes.	
Black-necked Stilt	*Himantopus mexicanus*	Shallow fresh and saltwater wetlands, including salt ponds, rice fields, shallow lagoons, and mangrove swamps	
Greater yellowlegs	*Tringa melanoleuca*	Breeds in muskeg, wet bogs with small wooded islands, and forests (usually coniferous) with abundant clearings. Winters in wide variety of shallow fresh and saltwater habitats.	

Table 3-7
Aquatic-Dependent Birds That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations and Diet	Federal and/or State Listing
Lesser yellowlegs	*Tringa flavipes*	Breeds in open boreal forest with scattered shallow wetlands. Winters in wide variety of shallow fresh and saltwater habitats.	
Spotted sandpiper	*Actitis macularia*	Breeds in a variety of habitats, such as shoreline, sagebrush, grassland, forest, lawn, or park. Winters wherever water is present. The spotted sandpiper is a shorebird that obtains much of its diet by probing or "mining" soft sediments along shorelines. Spotted sandpipers feed on a wide variety of benthic invertebrates and appear to be relatively common winter residents in coastal Texas.	
Western sandpiper	*Calidris mauri*	Breeds in coastal sedge-dwarf tundra. Migrates and winters along mudflats, beaches, shores or lakes and ponds, and flooded fields.	
White-faced ibis	*Plegadus chihi*	Primarily freshwater wetlands, but can also be found in estuarine habitats. Feeds on crustaceans, earthworms and insects	T
Carnivore			
Brown pelican	*Pelacanus occidentalis*	Oceans, inshore waters; stands on pilings or rocks	E
Double crested cormorant	*Phalacrocorax auritus*	Found in diverse aquatic habitats, such as ponds, lakes, rivers, lagoons, estuaries, and open coastline; more widespread in winter	
Neotropic cormorant	*Phalacrocorax brasilianus*	Various wetlands, including fresh, brackish, and saltwater habitats. Nests and roosts mostly in trees, but also on cliffs and human-made structures. Feeds primarily on fish <8cm in length.	

Table 3-7
Aquatic-Dependent Birds That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations and Diet	Federal and/or State Listing
Great blue heron	*Ardea herodias*	Wetlands where tall trees, rock ledges or extensive reeds provide a safe site for the heronry. Feeds on fish but also crustaceans, amphibians.	
Great egret	*Casmerodius albus*	Marshes where deeper water is edged with low , vegeatated banks. Nesting colonies may be in reeds or cattails, but more commonly in trees.	
Tricolored heron	*Egretta tricolor*	Breeds primarily in coastal habitats; feeds mainly on small fishes.	
Little blue heron	*Egretta caerulea*	Swamps, estuaries, rivers, ponds, and lakes	
Snowy egret	*Egretta thula*	Near freshwater lakes or estuaries	
Cattle egret	*Bubucus ibis*	Extensive marshes, wooded marshes	
Green heron	*Butorides virescens*	Breeds in swampy thickets. Forages in swamps, along creeks and streams, in marshes, ponds, lake edges, and pastures. Winters mostly in coastal areas, especially mangrove swamps.	
Black-crowned night-heron	*Nycticorax nycticorax*	Various wetland habitats, including salt, brackish, and freshwater marshes, swamps, streams, lakes, and agricultural fields.	
Yellow-crowned night-heron	*Nyctanassa violacea*	Marsh	
White ibis	*Eudocimus albus*	Large marshes	
Red-breasted merganser	*Mergus serrator*	Lakes rivers, winters on saly water	
Osprey	*Pandion haliaetus*	Coasts and inland lakes and rivers	
Forster's tern	*Sterna forsteri*	Breeds in marshes, generally with lots of open water and large stands of island-like vegetation.Winters in marshes, coastal beaches, lakes, and rivers.	
Least tern	*Sterna antillarum*	Beaches, bordering, shallow water along rivers, lakes, or coasts	

Table 3-7
Aquatic-Dependent Birds That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations and Diet	Federal and/or State Listing
Belted kingfisher	*Megaceryle alcyon*	Breeds along streams, rivers, lakes, and estuaries with banks for nest holes. Winters along coast, streams, and lakes.	
Bald eagle	*Haliaeetus leucocephalus*	Coasts and inland lakes and rivers	BGEPA
Reddish egret	*Egretta rufescens*	Marsh habitat	

Notes

Birds are all listed on the bird checklist of the Baytown Nature Center (2006).
Additional habitat information from Cornell Lab of Ornithology's 2009 Bird Search. Accessed at
http://www.allaboutbirds.org/NetCommunity/Page.aspx?pid=1189 Accessed on December 30 2009, and from Birds of North America Online,

Federal or State Listing

BGEPA = Bald and Golden Eagle Protection Act
LE/LT = Federally Listed Endangered/Threatened
E/T = State Endangered/Threatened
R = Rare

Table 3-8
Aquatic-Dependent Mammals That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations	Federal and/or State Listing	Source of Information
Herbivore				
Marsh rice rat	*Oryzomys palustris*	Marsh rice rats are semi-aquatic rodents that eats aquatic plants, and some invertebrates such as crabs and snails. This animal nests in cattatils and bulrushes, and is prey to hawks and owls.		eNature.com
Nutria	*Myocastor coypus*	Nutria are an invasive species that spend most of their time in or near the water. Favored foods for nutria include rushes, reeds, cattails, arrowhead, square-stem spike rush and sawgrass.		USFWS (2009)
American beaver	*Castor canadensis*	Herbivore found in ponds, lakes, or large streams.		USFWS (2009)
Omnivore				USFWS (2009)
Virginia opossum	*Didelphis virginiana*	Opossums are omnivorous, primarily woodland creatures, but are also frequently found in prairies, marshes, and farmlands. Although they prefer to live in hollow trees and logs, opossums will also shelter in woodpiles, rock piles, crevices in cliffs, under buildings, in attics, and in abandoned underground burrows dug by other animals.		USFWS (2009)
Northern raccoon	*Procyon lotor*	Raccoons prefer brushy or wooded areas near streams, lakes or swamps, although they can live close to developed areas if sufficient food, water and cover are provided. Though they prefer woodlands, raccoons can live practically anywhere and have adapted well to human habitats.		USFWS (2009)

Table 3-8
Aquatic-Dependent Mammals That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations	Federal and/or State Listing	Source of Information
Muskrat	*Ondatra zibethicus*	Muskrats primarily inhabit wetlands, areas in or near salt and fresh-water marshlands, rivers, lakes, or ponds.		USFWS (2009)
Carnivore				USFWS (2009)
River otter	*Lutra canadensis*	River otters prefer to live near bodies of water such as lakes, large rivers, and streams. Along the Texas Gulf Coast region, otters also live in marshes, bayous, and brackish inlets.		USFWS (2009)
Insectivore				
Rafinesque's big-eared bat	*Corynorhinus rafinesquii*	Roosts in cavity trees of bottomland hardwoods, concrete culverts, and abandoned man-made structures	T	TPWD (2010)
Southeastern myotis	*Myotis austroriparius*	Roosts in cavity trees of bottomland hardwoods, concrete culverts, and abandoned man-made structures	R	TPWD (2010)

Federal or State Listing

LE/LT = Federally Listed Endangered/Threatened
E/T = State Endangered/Threatened
R = Rare

Table 3-9
Summary of Ecological Receptor Surrogates

Receptor Group	Receptor Surrogate	Feeding Guild	Potentially Present	Representative of One or More Feeding Guilds	High Site Fidelity/Residential	Sensitive or Potentially Highly Exposed	Life History Information Is Readily Available	Additional Considerations
Benthic macroinvertebrates								
	Benthic macroinvertebrate community	All	X	X	X	X	X	Close association with sediment; much of the toxicological literature addresses community level endpoints.
	Molluscs	Filter feeders	X	X	X	X[a]	X	Close association with sediment
Fish								
	Gulf killifish	Omnivore	X	X	X		X	Common prey for other fish and bird species
	Black drum	Benthic invertivore	X	X	X		X	Popular sport fish; limited range, limited interbay movement
	Southern flounder	Benthic piscivore	X	X	X[b]	X	X	Supports commercial and recreational fisheries
Reptiles								
	Alligator snapping turtle	Omnivore	X	X	X	X	X	Sensitive species (rare in estuaries)
Birds								
	Neotropic cormorant	Piscivore (diving)	X	X			X	
	Great blue heron	Piscivore (wading)	X	X			X	
	Spotted sandpiper	Invertivore (probing)	X	X		X	X	As a sediment-probing invertivore, expected to be closely associated with sediment exposure pathway
	Killdeer	Invertivore (terrestrial)	X	X	X		X	Feeds on invertebrate fauna closely associated with soils
Mammals								
	Marsh Rice Rat	Omnivore	X	X	X		X	Semi-aquatic, diet consists of aquatic and emergent plants, and invertebrates
	Raccoon	Omnivore	X	X			X	Representative of both aquatic and terrestrial omnivorous feeding guilds

Notes

a - Sensitive reproductive endpoint

b - Site fidelity is probably high except in winter, when this species moves into more saline waters to spawn.

Table 3-10
Summary of Assessment Endpoints and Risk Questions for the BERA

Receptor Class	Assessment Endpoint	Risk Questions
Benthic macroinvertebrates	Abundance and diversity of benthic macroinvertebrate communities	Are the concentrations of chemicals of potential concern ($COPC_E$s) in whole sediment from benthic habitats of the Site greater than threshold concentrations relating to the survival, growth, or reproduction of benthic invertebrates, or the productivity or viability of invertebrate populations or communities?
Bivalve molluscs	Stable or increasing populations of bivalves within the Site	Are concentrations of organic primary $COPC_E$s in tissue of field collected clams equal to or greater than concentrations considered threshold levels of reproductive effects in molluscs?
Fish	Stable or increasing populations of fish in the following feeding guilds:	Are the concentrations of $COPC_E$s in waters of the Site greater than threshold concentrations relating to the survival, growth, or reproduction of fish?
	- Benthic omnivore - Benthic invertivore - Benthic piscivore	Are the concentrations of inorganic $COPC_E$s (metals) in the diet of fish greater than threshold effect levels for survival, growth, or reproduction of fish?
		Are concentrations of organic $COPC_E$s in fish tissue from the Site greater than the concentrations of $COPC_E$s associated with effects on the survival, growth or reproduction of fish?
Reptiles	Stable or increasing populations of omnivorous reptiles	Is the total daily ingested dose (mg/kg bw-day) of $COPC_E$s greater than doses known to cause effects on the survival, growth and reproduction of reptiles?
Birds	Stable or increasing populations of birds (that may be exposed to $COPC_E$s from the Site) in the following feeding guilds:	Is the total daily ingested dose (mg/kg bw-day) of $COPC_E$s greater than doses known to cause effects on the survival, growth, and reproduction of birds? Is the estimated concentration of dioxins and furans, expressed as TEQs, in bird eggs greater than threshold concentrations for reproductive effects in birds?
	- Invertivore (aquatic and terrestrial)	
	- Omnivorous wading bird	
	- Piscivorous diving bird	
Mammals	Stable or increasing populations of omnivorous mammals	Is the total daily ingested dose (mg/kg bw-day) of $COPC_E$s greater than doses known to cause effects on the survival, growth and reproduction of mammals?

Table 3-11
Summary of Lines of Evidence for Each Receptor and Assessment Endpoint

Receptor	Assessment Endpoint	Lines of Evidence	Measure of Exposure	Measure of Effect	Comments/Rationale
Benthic Macroinvertebrates	Abundance and diversity of benthic macroinvertebrate communities	Comparison of $COPC_E$ concentrations in sediment to literature-based effects levels	$COPC_E$ Concentrations in sediment (mg/kg dw)	Toxicity reference values for sediment (mg/kg dw)	
		Comparisons of $COPC_E$ concentrations in sediment porewater to literature-based effects levels	$COPC_E$ concentrations in porewater (µg/L)	Toxicity reference values for estuarine and marine waters (µg/L)	Porewater concentrations are modeled using sediment concentrations and Kd or Koc values from the literature (Table 4-5)
Bivalve Molluscs	Stable or increasing populations of bivalves within the site	Comparisons of $COPC_E$ concentrations in clam tissue to literature-based reproductive effect values for molluscs	$COPC_E$ concentrations in clam tissue	Toxicity reference values for invertebrate tissue (ng/kg ww)	
Fish	Stable or increasing populations of fish in the following guilds: benthic omnivore, benthic invertivore, benthic piscivore	Comparison of $COPC_E$ concentrations in surface water to literature-based effects levels	$COPC_E$ concentrations in water (µg/L)	Toxicity reference values for estuarine and marine surface waters ((µg/L)	Surface water concentrations of nickel and BEHP are modeled using sediment concentrations and Kd or Koc values from the literature (Table 4-5)
		Comparison of $COPC_E$ concentrations (metals) in the diet of fish to literature-based effects levels associated with concentrations in the diet of fish	$COPC_E$ concentrations (metals) in food items of fish (mg/kg dw)	Toxicity reference values for concentrations of $COPC_E$s (metals) in food items of fish (mg/kg dw)	
		Comparisons of $COPC_E$ concentrations (PCBs, dioxins, and furans) in fish tissue to literature-based effects levels	$COPC_E$ concentrations (PCBs, dioxins, and furans) in fish tissue (µg/kg lw or ww)	Toxicity reference values for concentrations of $COPC_E$s (PCBs, dioxins, and furans) in fish tissue (ug/kg lw or ww)	
Reptiles	Stable or increasing populations of omnivorous reptiles	Comparison of estimated ingested $COPC_E$ dose to literature-based effects levels expressed on a dose basis	$COPC_E$ doses that account for all ingested media (mg/kg bw-day)	Toxicity reference values for concentrations of COPCEs as ingested doses (mg/kg bw-day)	
Birds	Stable or increasing populations of birds that may be exposed to $COPC_E$s from the site in the following feeding guilds: invertivore (aquatic and terrestrial), omnivorous wading bird, piscivorous diving bird	Comparison of estimated ingested $COPC_E$ dose to literature-based effects levels expressed on a dose basis	$COPC_E$ doses that account for all ingested media (mg/kg bw-day)	Toxicity reference values for concentrations of COPCEs as ingested doses (mg/kg bw-day)	
		Comparison of estimated concentrations of $COPC_E$s (dioxins and furans) in bird eggs to literature-based effects levels for associated with reproductive effects in birds	$COPC_E$ (dioxins and furans) concentration in bird eggs (ng/g ww)	Toxicity reference values for $COPC_E$s (dioxins and furans) in bird eggs (ng/g ww)	Exposure concentrations are estimated using data for concentrations of $COPC_E$s in ingested media (prey and sediment)
Mammals	Stable or increasing populations of omnivorous mammals	Comparison of estimated ingested $COPC_E$ dose to literature-based effects levels expressed on a dose basis	$COPC_E$ doses that account for all ingested media (mg/kg bw-day)	Toxicity reference values for concentrations of COPCEs as ingested doses (mg/kg bw-day)	

Notes
bw = body weight
$COPC_E$ = chemical of potential ecological concern
dw = dry weight

Table 3-12
Receptor-Specific Life History Parameters for the WildlifeExposure Model

Species	Parameter	Units	Value	Source	Notes
Birds					
Great Blue Heron	bw	kg	2.2	USEPA (1993)	Average of adult males and females, eastern US, Quinney (1982) in USEPA (2003)
	FIR	kg diet/kg bw-day	0.044	USEPA (1993)	Based on Kushlan (1978); converted from wet weight to dry weight using dietary composition provided by Alexander (1977) and average moisture contents of major prey types provided in USEPA (1993) (FIR dw = FIR ww*(1-% moisture))
	WIR[a]	L/kg bw-day	0.045	Calder and Braun (1983) in USEPA (1993)	Water ingestion rate
	F_s	kg sediment/kg diet	0.033	Beyer et al. (1994)	Mallard sediment fraction in diet used as surrogate
	HR	km	2.7	Custer and Galli (2002)	Median flight distance for a Minnesota population
	F_{it}	kg food/kg diet	0		
	F_{ic}	kg food/kg diet	0.01	Alexander (1977)	Fish in diet split equally between large and small fish. Percent vertebrate prey items in diet (5%) reassigned to fish category)
	F_{im}	kg food/kg diet	0	Alexander (1977)	
	F_{fs}	kg food/kg diet	0.495	Alexander (1977)	
	F_{fl}	kg food/kg diet	0.495	Alexander (1977)	
	F_p	kg food/kg diet	0	Alexander (1977)	
Killdeer	bw	kg	0.088	UMMZ (2011)	
	FIR	kg diet/kg bw-day	0.19	Nagy (2001)	Allometric equation for Charadriiformes, dry-weight basis: DMI g/day = 0.522* (g bw) $^{0.769}$, divided by kg bw and converted to kg diet basis by * kg/1,000 g
	F_s	kg sediment/kg diet	0.10	Beyer et al. (1994)	Value for American woodcock used, as most ecologically similar species available (terrestrial invertivore) in Beyer et al. (1994)
	HR	km^2	0.06	Jackson and Jackson (2000)	Average of home ranges for N=10 in ne CA population; defended breeding territories are considerably smaller and feeding may also take place at much greater distances.
	F_{it}	kg food/kg diet	0.98	Jackson and Jackson (2000)	MO population 98% insects, predominantly terrestrial; Puerto Rico N=20 stomachs 98% animal material, primarily terrestrial invertebrates
	F_{ic}	kg food/kg diet	0	Jackson and Jackson (2000)	
	F_{im}	kg food/kg diet	0	Jackson and Jackson (2000)	
	F_{fs}	kg food/kg diet	0	Jackson and Jackson (2000)	
	F_{fl}	kg food/kg diet	0	Jackson and Jackson (2000)	
	F_p	kg food/kg diet	0.02	Jackson and Jackson (2000)	2% plant material in gut contents of Puerto Rico study; 1.3% in Missouri study
Neotropic Cormorant	bw	kg	1.3	Telfair and Morrison (2005)	Average of adult males and females
	FIR	kg diet/kg bw-day	0.067	Nagy (2001)	Allometric equation for food intake rates for all birds: DMI g/day = 0.638*g bw $^{0.685}$, divided by bw (kg), multiplied by kg/1000 g to convert to kg diet basis
	WIR[a]	L/kg bw-day	0.054	Calder and Braun (1983) in USEPA (1993)	Water ingestion rate
	F_s	kg sediment/kg diet	0.02	Beyer et al. (1994)	Value of <0.02 given for ring-necked duck, as a diving duck is the ecologically most similar species available in Beyer et al. 1994
	HR	N/A	ND	Telfair and Morrison (2005)	No home range information available. Dispersal of juveniles from natal area may be relatively limited, or up to hundreds of kilometers.
	F_{it}	kg food/kg diet	0		Diet almost entirely comprised of fish in local study. Primarily fish <8 mm taken (see exp areas worksheet). Small proportion of shrimp in diet added into fish category, as this is primarily a pelagic invertebrate pathway that would not be well-represented by benthic invertebrate tissue data
	F_{ic}	kg food/kg diet	0	King (1989)	
	F_{im}	kg food/kg diet	0	King (1989)	
	F_{fs}	kg food/kg diet	1	King (1989)	
	F_{fl}	kg food/kg diet	0	King (1989)	
	F_p	kg food/kg diet	0	King (1989)	

Table 3-12
Receptor-Specific Life History Parameters for the WildlifeExposure Model

Species	Parameter	Units	Value	Source	Notes
Spotted Sandpiper	bw	kg	0.043	USEPA (1993)	Average of males and females, Maxson and Oring (1980)
	FIR	kg diet/kg bw-day	0.22	Nagy (2001)	Allometric equation for Charadriiformes, dry-weight basis: DMI g/day = 0.522* (g bw) $^{0.705}$, divided by kg bw and converted to kg diet basis by * kg/1,000 g
	WIR[a]	L/kg bw-day	0.17	Calder and Braun (1983) in USEPA (1993)	Water ingestion rate
	F_s	kg sediment/kg diet	0.18	Beyer et al. (1994)	Average of data for four sandpiper species (range 7 to 30%)
	HR	km	1.5	Macwhirter et al. (2002)	No HR information for spotted sandpiper; this value is a home range for sanderling, a similarly sized invertivorous shorebird that winters in coastal Texas.
	F_{it}	kg food/kg diet	0	USEPA (1993)	
	F_{ic}	kg food/kg diet	0.5		
	F_{im}	kg food/kg diet	0.5		
	F_{fs}	kg food/kg diet	0		
	F_{fl}	kg food/kg diet	0		
	F_p	kg food/kg diet	0		
White-Faced Ibis[b]	HR	km^2	12	GBBO (2012)	Minimum recommended habitat patch size based on expert opinion and limited home range information
Bald Eagle[b]	HR	km^2	14.5; 145	Buehler (2012)	Average home ranges for breeding and non-breeding (wintering) populations, respectively, of bald eagles
Brown Pelican[b]	HR	N/A	see notes		No home range information available. Foraging radius from nesting sites described as within 20 km radius of nesting colony during breeding season, up to 75 km from nearest land during non-breeding season
Mammals					
Marsh Rice Rat	bw	kg	0.051	Davis and Schmidly (1994)	Average of range of adult weights
	FIR	kg diet/kg bw-day	0.19	Nagy (2001)	Allometric equation for mesic rodents, dry-weight basis: DMI g/day = 0.614* (g bw) $^{0.705}$, divided by kg bw and converted to kg diet basis by * kg/1,000 g
	F_s	kg sediment/kg diet	0.02	Beyer et al. (1994)	Value of < 0.02 given for white-footed mouse, most ecologically similar mammal available in Beyer et al. 1994
	HR	km	0.075	Wolfe (1982)	Average of Maryland (75 m) and Florida (68 and 82m) range lengths
	F_{it}	kg food/kg diet	0		Estimated assignments based on Wolfe's summary of multiple studies, which indicates multiple food sources, with roughly equal amounts of plant and animal materials. Small fish, clams, crabs, snails, bird eggs among common gut contents.
	F_{ic}	kg food/kg diet	0.2	Wolfe (1982)	
	F_{im}	kg food/kg diet	0.2	Wolfe (1982)	
	F_{fs}	kg food/kg diet	0.2	Wolfe (1982)	
	F_{fl}	kg food/kg diet	0		
	F_p	kg food/kg diet	0.4	Wolfe (1982)	
Raccoon	bw	kg	5.1	USEPA (1993)	Average of adult males and females from an Alabama population and a Missouri population
	FIR	kg diet/kg bw-day	0.041	Nagy (2001)	Allometric equation for placental mammals,: DMI g dw/day= 0.299 (g bw) 0.767, divided by bw in kg and kg/1,000 g to convert to kg diet basis
	F_s	kg sediment/kg diet	0.094	Beyer et al. (1994)	
	HR	km^2	0.52	USEPA (1993)	Average of male and female year-round ranges on a Georgia coastal island (Lotze 1979)
	F_{it}	kg food/kg diet	0.05	Alexander (1977)	Dietary composition by % of wet weight for 29 raccoons: % vertebrates in diet were reassigned to invertebrates and fish categories; percent unidentified material reassigned to plant category
	F_{ic}	kg food/kg diet	0.24	Alexander (1977)	
	F_{im}	kg food/kg diet	0.05	Alexander (1977)	
	F_{fs}	kg food/kg diet	0.20	Alexander (1977)	
	F_{fl}	kg food/kg diet	0.20	Alexander (1977)	
	F_p	kg food/kg diet	0.26	Alexander (1977)	

Table 3-12
Receptor-Specific Life History Parameters for the WildlifeExposure Model

Species	Parameter	Units	Value	Source	Notes
Reptiles					
Alligator Snapping Turtle	bw	kg	51.5	National Geographic (2011)	Average of bw for male and female alligator snapping turtles of 80 and 23 kg, respectively.
	FIR	kg diet/kg bw-day	0.01	Nagy (2001)	Allometric equation for carnivorous reptiles, dry-weight basis: DMI g/day = 0.00865* (g bw) $^{0.963}$, divided by kg bw and converted to kg diet basis by * kg/1,000 g
	WIR[a]	L/kg bw-day	0.02	USEPA (1993)	Water ingestion rate
	F_s	kg sediment/kg diet	0.05	Beyer et al. (1994)	Value for sediment in the diet of box turtle, as snapping turtle data not available.
	HR	km	0.778	Riedle (2008)	
	F_{it}	kg food/kg diet	0		Assignments to prey categories based on indices of relative importance for invertebrates calculated from Elsey. Vertebrates in diet were reassigned to fish (fish as proportion of diet split equally between large and small); unidentified material was reassigned to plant category.
	F_{ic}	kg food/kg diet	0.03	Elsey (2006)	
	F_{im}	kg food/kg diet	0.01	Elsey (2006)	
	F_{fs}	kg food/kg diet	0.35	Elsey (2006)	
	F_{fl}	kg food/kg diet	0.35	Elsey (2006)	
	F_p	kg food/kg diet	0.26	Elsey (2006)	

Notes

AUF= area use factor
bw = body weight (kg)
Fs = fraction of the diet that is sediment
F_{fs} = fraction of the diet consisting of small fish (kg fish/kg food)
F_{fl} = fraction of the diet consisting of large fish (kg fish/kg food)
F_{it} = fraction of the diet consisting of terrestrial invertebrates (kg invertebrates/kg food)
F_{ic}= fraction of diet consisting of crustacea (kg invertebrates/kg food)
F_{im} = fraction of the diet consisting of molluscs (kg invertebrates/kg food)
F_p = fraction of the diet consisting of plants (kg plants/kg food)
FIR = food ingestion rate (kg food dw/day)
HR = home range
WIR = water ingestion rate

a - allometric equation for birds, WIR (L/day) = (0.059*BW(kg)0.67)/kg bw
b - state or federally listed species, evaluated in cases where risk to surrogate receptors $HQ_N \geq 1$

Table 4-1

Partition Coefficients for Chemicals of Potential Ecological Concern

Chemical	K_d [a] (L/kg)	K_{OC} [b] (L/kg)
Metals		
Barium	41	NA
Cadmium	75	NA
Cobalt	45	NA
Copper	35	NA
Lead	900	NA
Magnesium	5	NA
Manganese	65	NA
Mercury	52	NA
Nickel	65	NA
Thallium	71	NA
Zinc	62	NA
Semivolatile Organic Compounds		
Bis(2-ethylhexyl)phthalate	NA	120,000
Carbazole	NA	9,160
Phenol	NA	187

Notes

NA = not applicable

a - Soil-water partition coefficient from the Risk Assessment Information System (USDOE 2012)

b - Organic carbon partition coefficient from the Risk Assessment Information System (USDOE 2012)

Table 4-2

Summary Statistics for Estimated Porewater Concentrations of $COPC_E$s

$COPC_E$	Units	N	Min	Mean	Max
Metals					
Cobalt	mg/L	97	0.0044	0.092	0.30
Manganese	mg/L	97	0.025	3.7	23
Thallium	mg/L	97	0.0028	0.019	0.049
Semivolatile Organic Compounds					
Bis(2-ethylhexyl)phthalate	µg/L	97	0.00804	0.0809	1.85

Notes

$COPC_E$ = chemical of potential ecological concern

Table 4-3
Receptor-Specific Dietary Assumptions for Fish

Species	Parameter	Literature-Based Value	Modeled Value[a]	Sources for Dietary Estimates	Notes
Gulf killifish	Fs	0.01	0.01	Windward (2007)	Sediment ingestion portion of diet for Pacific staghorn sculpin adopted for Lower Duwamish Waterway Group BERA (Windward 2007)
	F_{it}	0.19		Hassan-Williams and Bonner (2012); USGS (2009)	Omnivorous: feeds throughout the water column, consuming benthic algae, vascular plants, grass shrimp, microcrustaceans, terrestrial insects that fall onto the water surface, mosquito larvae and pupae, bivalve molluscs, and small fishes (e.g., killifishes and anchovies). For modeling exposure, terrestrial invertebrate portion of diet reassigned to aquatic invertebrates, and plants component reassigned to invertebrate and fish components of diet.
	F_{ap}	0.2			
	F_{ic}	0.2	0.36		
	F_{im}	0.2	0.36		
	F_{fs}	0.2	0.27		
Black drum	Fs	0.01	0.01	Windward (2007)	Sediment ingestion portion of diet for English sole and Pacific staghorn sculpin adopted for Lower Duwamish Waterway Group BERA (Windward 2007).
	F_{ap}	0.04		TPWD (2012a); Sutter et al. (1982); LSU (2012); Smithsonian (2012)	Benthic invertivore: young black drum (< 20 cm) feed on marine worms and small fish, shrimp and crab and larger young (8 to 20 cm) eat small fish (36 percent) and polychaetes (32 percent). Larger drum (> 20 cm) consume molluscs, small crabs, worms and algae. In Texas estuaries, the dominant food of black drum (21 to 50 cm) is the mollusc *Mulinia* sp. (33 percent). The largest drum ate mostly molluscs (74 percent) and crabs (16 percent). For modeling exposure, polychaetes and plants portion of diet reassigned to aquatic invertebrates.
	F_{ip}	0.05			
	F_{ic}	0.16	0.176		
	F_{im}	0.7	0.814		
	F_{fs}	0			
Southern flounder	Fs	0.01	0.01	Windward (2007)	Sediment ingestion portion of diet for English sole adopted for Lower Duwamish Waterway Group BERA (Windward 2007).
	F_{it}	0		Hassan-Williams and Bonner (2012), TPWD (2012b)	Benthic piscivore: small fishes (e.g., anchovies, juvenile striped mullet, menhadens, Atlantic croaker, spot, pinfish, and fat sleeper) and to a lesser extent, crustaceans (e.g., mysids, isopods, amphipods, penaeid shrimp, and portunid crabs) constitute most of the southern flounder diet. Ninety-five percent of the food items of juvenile southern flounder (10 to 150 mm) from Texas consisted of invertebrates. Juvenile southern flounder (>80 mm) consume progressively larger food items as they grow. Fish make up 70 percent of the adult (>150 mm) diet.
	F_{ic}	0.29	0.29		
	F_{im}	0			
	F_{fs}	0.7	0.7		

Notes

Fs = fraction (unitless) of the diet that is sediment
F_{it} = fraction (unitless) of the diet consisting of terrestrial invertebrates
F_{ap} - fraction (unitless) of diet consisting of aquatic plants
F_{ip} = fraction (unitless) of diet consisting of polychaetes
F_{ic}= fraction (unitless) of diet consisting of crustacea
F_{im} = fraction (unitless) of the diet consisting of molluscs
F_{fs} = fraction (unitless) of the diet consisting of small fish

a - The modeled value reassigns the literature-based proportion of prey that is in a category for which empirical tissue data are not available, to a category of prey which is ecologically similar and for which for which empirical data are available. A category for which empirical tissue data are not available is reassigned to categories for which data are available in the modeled diet, weighted by their relative abundance (e.g., polychaetes in the black drum diet are reassigned to crustacea as % polychaetes * (% crustacea in black drum diet/(%crustacea +%molluscs)); and to molluscs as %polychaetes*(%molluscs in black drum diet/(%molluscs+%crustacea))

Table 4-4
Surface-Area Weighted Average Concentrations of $COPC_E$s in Sediments of the Site and Estimated Concentrations in Surface Water

Analyte	Sediment SWAC	K_d [a] (L/kg)	K_{OC} [a] (L/kg)	Estimated Concentration in Surface Water [b]
SVOCs	**µg/kg OC**			**µg/L**
Bis(2-ethylhexyl)phthalate	16,400	N/A	120,000	0.14
Metals	**mg/kg dw**			**mg/L**
Cadmium	0.440	75	N/A	0.00586
Copper	11.4	35	N/A	0.326
Lead	11.8	900	N/A	0.0131
Mercury	0.0495	52	N/A	0.0010
Nickel	6.26	65	N/A	0.096
Zinc	55.0	62	N/A	0.887

Notes

$COPC_E$ = chemical of potential ecological concern
SVOC = semivolatile organic compound
SWAC = surface area-weighted average concentration

a - See Table 4-1 for source of Kd and Koc values.
b - These gross and highly conservative estimates of surface water chemical concentrations are calculated as sediment SWAC ÷ K_d for inorganics and SWAC ÷ K_{oc} for organics, per Eqn. 4-2.

Table 4-5
Total PCBs Concentrations in Whole Fish from Site and Background

Hardhead Catfish		*Gulf Killifish*	
	Total PCBs		**Total PCBs**
Sample ID	**(µg/kg ww)**	**Sample ID**	**(µg/kg ww)**
FCA1			
SJFCA1-LF1	588	GK-TTR2-1	33.5
SJFCA1-LF6	664	GK-TTR2-2	40.1
SJFCA1-LF10	759		
FCA2			
SJFCA2-LF1	793	GK-TTR3-1	187
SJFCA2-LF4	647	GK-TTR3-2	191
SJFCA2-LF8	563	GK-TTR4-1	24.2
SJFCA2-LF10	286	GK-TTR4-2	19.4
		GK-TTR5-1	44
		GK-TTR5-2	32.7
FCA3			
SJFCA3-LF1	469	GK-TTR6-1	51.9
SJFCA3-LF6	750	GK-TTR6-2	28.9
SJFCA3-LF10	942		
Background			
SJFCACB-LF1	137	GK-TTR7-1	13.1
SJFCACB-LF2	347	GK-TTR7-2	12.3
SJFCACB-LF4	163	GK-TTR7-3	13.9
SJFCACB-LF5	206	GK-TTR7-4	15.5
SJFCACB-LF6	251	GK-TTR8-1	12.2
SJFCACB-LF8	192	GK-TTR8-2	13
SJFCACB-LF9	460	GK-TTR8-3	11.9
SJFCACB-LF10	412	GK-TTR8-4	14.1

Table 4-6

Dioxins, Furans, and PCBs in Whole Fish Expressed as $TEQ_{DFP,F}$

	$TEQ_{DF,F}$ [a] ng/kg lw		$TEQ_{DFP,F}$ [b] ng/kg lw				$TEQ_{DF,F}$ [a] ng/kg lw		$TEQ_{DFP,F}$ [b] ng/kg lw	
Hardhead Catfish						***Gulf Killifish***				
Sample ID						**Sample ID**				
FCA1										
SJFCA1-LF1	**323**	J	**330**	J		GK-TTR2-1	147	U	197	J
SJFCA1-LF6	264	J	271	J		GK-TTR2-2	8.72	U	12	J
SJFCA1-LF10	**367**	J	**381**	J						
FCA2										
SJFCA2-LF1	315	J	**321**	J		GK-TTR3-1	137	J	142	J
SJFCA2-LF4	314	J	**326**	J		GK-TTR3-2	265	J	270	J
SJFCA2-LF8	205	J	212	J		GK-TTR4-1	5.98	J		J
SJFCA2-LF10	183	J	188	J		GK-TTR4-2	307	U	**503**	J
						GK-TTR5-1	160	J	169	J
						GK-TTR5-2	11	J	12.9	J
FCA3										
SJFCA3-LF1	221	J	229	J		GK-TTR6-1	3.38	J	3.88	J
SJFCA3-LF6	286	J	291	J		GK-TTR6-2	4.87	J	5.28	J
SJFCA3-LF10	**373**	J	**381**	J						
Background										
SJFCACB-LF1	25.7	J	26.6	J		GK-TTR7-1	4.29	J	4.75	J
SJFCACB-LF2	36.6	J	39.4	J		GK-TTR7-2	3.71	J	4.46	J
SJFCACB-LF4	23.2	J	26.6	J		GK-TTR7-3	3.52	J	3.74	J
SJFCACB-LF5	44.3	J	48.1	J		GK-TTR7-4	15.7	J	17.5	J
SJFCACB-LF6	41.4	J	45.5	J		GK-TTR8-1	2.11	U	2.95	J
SJFCACB-LF8	36.9	J	40.3	J		GK-TTR8-2	0.857	J	1.12	J
SJFCACB-LF9	68.0	J	76.8	J		GK-TTR8-3	3.67	J	4.11	J
SJFCACB-LF10	47.5	J	54.1	J		GK-TTR8-4	3.04	J	3.84	J

Notes

Bold indicates that the concentration is greater than that considered protective of 95 percent of fish species

lw = lipid weight

J = One or more congener used in calculation of TEQ was not detected

FCA = fish collection area

a - Toxicity equivalent for dioxins and furans calculated using fish toxicity equivalency factors with nondetects set at one-half the detection limit.

b - Toxicity equivalent for dioxins, furans, and polychlorinated biphenyls calculated using fish toxicity equivalency factorswith nondetects set at one-half the detection limit.

Table 4-7

Weighted Concentrations[a] of $COPC_E$s (mg/kg dw) in the Diets of Fish

$COPC_E$	Gulf Killifish - Transect 1 and 2 (FCA 1)									
	Sediment		Prey - Crustacea		Prey - Molluscs		Prey - Small fish		Total Diet[b]	
	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	0.004	0.005	0.099	0.108	0.093	0.103	0.002	0.003	0.199	0.219
Copper	0.1	0.1	16.3	20.1	6.3	6.7	1.4	1.5	24.2	28.4
Mercury	0.0003	0.0004	0.03	0.04	0.03	0.04	0.03	0.04	0.10	0.11
Zinc	0.4	0.5	42.3	44.1	38.9	42.0	44.8	46.9	126	134

$COPC_E$	Gulf Killifish - Transect 3 (FCA 2)									
	Sediment		Prey - Crustacea		Prey - Molluscs		Prey - Small fish		Total Diet[b]	
	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	0.003	0.010	0.115	0.133	0.099	0.110	0.002	0.003	0.220	0.256
Copper	0.1	0.3	17.4	21.2	13.8	15.0	1.5	1.6	32.8	38.1
Mercury	0.003	0.01	0.02	0.03	0.05	0.05	0.07	0.09	0.14	0.17
Zinc	0.4	1.1	40.7	44.4	35.9	37.6	45.3	46.6	122	130

$COPC_E$	Gulf Killifish - Transect 4 (FCA 2)									
	Sediment		Prey - Crustacea		Prey - Molluscs		Prey - Small fish		Total Diet[b]	
	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	0.002	0.006	0.115	0.133	0.082	0.088	0.002	0.002	0.201	0.229
Copper	0.0173	0.089	17.4		6.4	7.2	1.4	1.7	25.2	30.2
Mercury	0.0002	0.0005	0.02	0.03	0.03	0.04	0.06	0.10	0.11	0.16
Zinc	0.1	0.5	40.7	44.4	30.7	33.6	45.2	45.6	117	124

Table 4-7

Weighted Concentrations[a] of $COPC_E$s (mg/kg dw) in the Diets of Fish

$COPC_E$	Gulf Killifish - Transect 5 (FCA 2)									
	Sediment		Prey - Crustacea		Prey - Molluscs		Prey - Small fish		Total Diet[b]	
	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	0.001	0.004	0.115	0.133	0.076	0.082	0.002	0.003	0.194	0.222
Copper	0.1	0.1	17.4	21.2	6.4	7.7	1.6	1.9	25.5	30.8
Mercury	0.0001	0.0003	0.02	0.03	0.02	0.02	0.04	0.04	0.08	0.09
Zinc	0.2	0.5	40.7	44.4	34.8	37.0	45.6	46.7	121	129

$COPC_E$	Gulf Killifish - Transect 6 (FCA 3)									
	Sediment		Prey - Crustacea		Prey - Molluscs		Prey - Small fish		Total Diet[b]	
	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	0.001	0.001	0.079	0.099	0.095	0.104	0.002	0.002	0.177	0.206
Copper	0.01	0.04	16.4	17.7	11.6	11.8	1.6	1.7	29.6	31.2
Mercury	0.0001	0.0001	0.03	0.04	0.05	0.06	0.07	0.08	0.15	0.18
Zinc	0.03	0.1	38.2	40.5	33.7	35.8	50.8	52.0	123	128

$COPC_E$	Black Drum (Site-wide)							
	Sediment		Prey - Crustacea		Prey - Molluscs		Total Diet[b]	
	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	0.004	0.006	0.048	0.054	0.200	0.209	0.252	0.268
Copper	0.1	0.2	8.1	9.0	20.0	26.5	28.2	35.8
Mercury	0.001	0.003	0.013	0.015	0.078	0.088	0.092	0.106
Zinc	0.3	1.0	19.7	20.6	78.0	80.8	98.1	102

Table 4-7

Weighted Concentrations[a] of $COPC_E$s (mg/kg dw) in the Diets of Fish

$COPC_E$	Southern Flounder (Site-wide)							
	Sediment		Prey - Crustacea		Prey - Small fish		Total Diet[b]	
	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	0.004	0.006	0.078	0.088	0.006	0.006	0.088	0.101
Copper	0.1	0.2	13.4	14.9	4.0	4.4	17.5	19.5
Mercury	0.001	0.003	0.022	0.025	0.142	0.184	0.164	0.211
Zinc	0.3	1.0	32.4	33.9	122	126	155	161

Notes

CT = central tendency

RM = reasonable maximum

a - Weighted concentrations are the product of the transect-specific EPC for the prey item (Table C-2) and the prey item's estimated fraction of the total diet as described in Table 4-3. For example, the CT of the weighted concentration of cadmium in blue crab in the diet of gulf killifish in FCA1 is 0.273 mg/kg (CT EPC of cadmium in crab) x 0.36 (the fraction of crustacea in the diet of killifish) = 0.099 mg/kg dw.

b - Total diet is the sum of the weighted concentrations of prey and sediment.

Table 4-8
Assumptions for Parameterizing the Wildlife Exposure Model

Taxon	Feeding Guild	Receptor	Area of the Site/Data Source Used for Model Parameterization: Aquatic/ Terrestrial	Exposure Unit	Surface Water	Sediments (0- to 6-inch depth)	Soils (0- to 6-inch depth)	Fish	Terrestrial Invertebrates	Benthic Invertebrates	Plants	Notes
Birds	Piscivore (wading)	Great blue heron[a]	Aquatic	All accessible shorelines of the site	Estimated from sediment SWACs [b]	All site shoreline sediments	N/A	Small and large fish	N/A	Shoreline invertebrate tissue samples	N/A	Great blue heron is limited to fish < 25 cm, but will use large fish (30 to 45 cm) to estimate exposure to other receptors that may ingest this size range.
	Invertivore (terrestrial)	Killdeer	Terrestrial	All upland areas of the site north of I-10	N/A	N/A	Soil data for site north of I-10	N/A	BAFs from upland soils N of I-10 for non-dioxin COPCs; regression approach for dioxins and furans (Appendix D)	N/A	N/A	
	Piscivore (diving)	Neotropic cormorant	Aquatic	All aquatic areas of the site	Estimated from sediment SWACs [b]	Site-wide sediments	N/A	Small fish: <8 cm TL	N/A	N/A	N/A	Fold pelagic invertebrates (2% of diet) into fish so 100% fish modeled in diet
	Invertivore (probing)	Spotted sandpiper	Aquatic	All accessible shorelines of the site	Estimated from sediment SWACs [b]	All site shoreline sediments	N/A	N/A	N/A	Shoreline invertebrate tissue samples	N/A	
Mammals	Omnivore	Marsh rice rat	Aquatic	All accessible shorelines of the site	N/A	All site shoreline sediments	N/A	Small fish: <8 cm TL	N/A	Shoreline invertebrate tissue samples	BAFs from shoreline sediments	
	Omnivore	Raccoon	Aquatic and terrestrial	Non-island uplands and shorelines of accessible areas of the peninsula	N/A	Shoreline sediments of the peninsula	Soils of the peninsula	Small fish from the peninsula shoreline	BAFs from upland peninsula soils for non-dioxin COPCs; regression approach for dioxins (Appendix D)	Peninsula shoreline invertebrate tissue samples	BAFs from peninsula soils	Assumes receptor uses both upland and shorelines for foraging: soil and sediment ingestion each receive one-half of incidental ingestion rate
Reptiles	Omnivore	Alligator snapping turtle	Aquatic	All aquatic areas of the site	Estimated from sediment SWACs [b]	Site-wide sediments	N/A	Site-wide: all fish	N/A	Site-wide: all aquatic invertebrates	BAFs from shoreline sediments	

Notes
BAF = bioaccumulation factor
COPC = chemical of potential concern
N/A = not applicable (no exposure to this medium is expected)
SWAC = surface area-weighted average concentration
a - Surrogate receptor for bald eagle, for which assumptions are identical to this receptor except home range and area use factor (see Table 4-12)
b - Except for dioxins and furans, for which empirical data are used
c - Surrogate receptor for white-faced ibis, for which assumptions are identical to this receptor except home range and area use factor (see Table 4-12)

Table 4-9
Relative Bioavailability Adjustment Factors for TCDD in Soil, Sediment, and Food Ingested by Birds

Medium	Relative Bioavailability Adjustment Factor
Invertebrates[a]	0.44
Soils	0.33
Sediments[b]	0.41

Source

Nosek et al. (1992a)

Notes

a - Average of percent absorption from homogenate of earthworms and homogenate of crickets

b - Percent absorption from paper mill sludge solids

Table 4-10
Bioaccumulation Relationships for Soil-to-Invertebrates and Soil-to-Plant Tissue

Chemical	Concentration in Invertebrate Tissue [a,b] (mg/kg dw)	Concentration in Plant Tissue [a,b] (mg/kg dw)
Dioxins and Furans		
Dioxins and Furans	See Appendix D	0 [c]
Polychlorinated Biphenyls		
Polychlorinated Biphenyls	$Cs^{1.361} * e^{1.41}$ [d]	0 [c]
Semivolatile Organic Compounds		
Bis(2-ethylhexyl)phthalate	0 [e]	0 [c]
Metals		
Cadmium	$Cs^{0.795} * e^{2.114}$	$Cs^{0.546}/e^{0.475}$
Chromium	Cs*0.306	Cs*0.041
Cobalt	Cs * 0.122	Cs * 0.0075
Copper	Cs * 0.515	$Cs^{0.394}*e^{0.668}$
Lead	$Cs^{0.807}/e^{0.218}$	$Cs^{0.561}/e^{1.328}$
Mercury	Cs*3.1 for Cs ≤1.5 mg/kg; [f] Cs*0.7 for Cs > 1.5 mg/kg	Cs*0.0375 [g]
Nickel	(Cs*0.02)/0.16 [h]	$Cs^{0.748}/e^{2.223}$
Vanadium	Cs * 0.042	Cs * 0.00485
Zinc	$Cs^{0.328}*e^{4.449}$	$Cs^{0.554}*e^{1.575}$

Notes

BAF = bioaccumulation factor

Cs = concentration in soil (mg/kg)

dw = dry weight

a - Unless otherwise indicated, the source for values in this column is USEPA (2007c) Attachment 4-1: Exposure Factors and Bioaccumulation Models for Derivation of Wildlife EcoSSLs (Table 4a).

b - Natural log equations were transformed as follows:

$\ln(y) = a*\ln(x)-b$, transformed to $y = x^a/e^b$; or

$\ln(y) = a*\ln(x)+b$, transformed to $y = x^a*e^b$

c - Dioxins, PCBs, and BEHP are low-solubility, high molecular weight compounds which have a negligible potential for uptake into plant tissues (Staples et al. 1997; Bacci et al. 1992; McCrady et al. 1990, 1993); therefore, a BAF of zero is used for these $COPC_E$s.

d - Sample et al. (1998). Regression equation from Table 12 for total PCBs.

e - BEHP does not bioaccumulate in invertebrate tissue at environmentally realistic concentrations in soil (Staples et al. 1997).

f - Based on differential uptake by earthworms depending on soil concentrations: a higher BAF for soils with lower mercury concentrations, and a lower BAF for soils with higher mercury concentrations (Burton et al. 2006)

g - Recommended soil to plant bioconcentration factor from Table C-2 for mercuric chloride in USEPA (1999b).

h - Recommended soil-to-invertebrate bioconcentration factor from Table C-1 in USEPA (1999b). Because the BCF provided by USEPA (1999b) is on basis of kg dw soil/kg ww tissue, the resulting value is converted to dw tissue basis by dividing by (1-moisture content), where moisture content = 0.86 (USEPA 1993).

Table 4-11

Regression Equations Used to Estimate Dioxin and Furan Congener Concentrations in Terrestrial Invertebrate Tissue

Congener	Equation
2,3,7,8-TCDD	$\exp(-2.49 + 0.819*(\ln(C_{s2,3,7,8\text{-}TCDD}))$
1,2,3,7,8-PCDD	$\exp(-5.92+0.516*(\ln(C_{s1,2,3,7,8\text{-}PCDD}))$
1,2,3,4,7,8-HxCDD	$0.430*C_{e1,2,3,7,8,9\text{-}HxCDD}$
1,2,3,6,7,8-HxCDD	$\exp(-3.42+0.664*(\ln(C_{s1,2,3,6,7,8\text{-}HxCDD}))$
1,2,3,7,8,9-HxCDD	$\exp(-5.04+0.55*(\ln(C_{s1,2,3,7,8,9\text{-}HxCDD}))$
1,2,3,4,6,7,8-HpCDD	$\exp(-3.91+0.479*(\ln(C_{s1,2,3,4,6,7,8\text{-}HpCDD}))$
OCDD	$8.02*C_{e1,2,3,4,6,7,8\text{-}HpCDD}$
2,3,7,8-TCDF	$0.120*C_{e1,2,3,6,7,8\text{-}HxCDD}$ [a]; $0.250*C_{e1,2,3,4,7,8\text{-}HxCDF}$ [b]
1,2,3,7,8-PCDF	$\exp(-4.86+0.593*(\ln(C_{s1,2,3,7,8\text{-}PCDF}))$
2,3,4,7,8-PCDF	$0.108*C_{e1,2,3,6,7,8\text{-}HxCDD}$
1,2,3,4,7,8-HxCDF	$\exp(-4.29+0.616*(\ln(C_{s1,2,3,4,7,8\text{-}HxCDF}))$
1,2,3,6,7,8-HxCDF	$\exp(-4.50+0.609*(\ln(C_{s1,2,3,6,7,8\text{-}HxCDF}))$
1,2,3,7,8,9-HxCDF	$\exp(-5.74+0.671*(\ln(C_{s1,2,3,7,8,9\text{-}HxCDF}))$
2,3,4,6,7,8-HxCDF	$\exp(-5.22+0.576*(\ln(C_{s2,3,4,6,7,8\text{-}HxCDF}))$
1,2,3,4,6,7,8-HpCDF	$\exp(-3.69+0.593*(\ln(C_{s1,2,3,4,6,7,8\text{-}HpCDF}))$
1,2,3,4,7,8,9-HpCDF	$0.723*C_{e1,2,3,4,7,8\text{-}HxCDF}$
OCDF	$0.603*C_{e1,2,3,4,6,7,8\text{-}HpCDD}$

Notes

$C_{scongener}$= concentration of the given congener in soil

$C_{econgener}$ = concentration of the given congener in earthworms

a - Selected congener for estimating 2,3,7,8-TCDF tissue concentrations from soil samples outside of the impoundments.

b - Selected congener for estimating 2,3,7,8-TCDF tissue concentrations from soil samples inside of the impoundments.

Table 4-12
Area Use Factors Used to Evaluate Exposure of Wildlife Receptors

	Alligator Snapping Turtle	Neotropic Cormorant	Great Blue Heron	Spotted Sandpiper	Marsh Rice Rat	Raccoon	Killdeer	White-Faced Ibis [a]	Brown Pelican[a]	Bald Eagle [a]	
Exposure Unit [b]	All aquatic shorelines of the site	All aquatic areas of the site	All aquatic shorelines of the site	All aquatic shorelines of the site	All aquatic shorelines of the site	Terrestrial area of the peninsula	Terrestrial area north of I-10	All aquatic shorelines of the site [c]	All aquatic areas of the site	All aquatic shorelines of the site	
										breeding [d]	wintering [d]
Estimated Size of Exposure Unit	37.61 km	2.52 km^2	37.6 km	37.6 km	37.6 km	0.36 km^2	0.13 km^2	0.38 km^2	2.52 km^2	2.52 km^2	2.52 km^2
Home Range [e]	0.778 km	ND	2.7 km	1.5 km	0.075 km	0.52 km^2	0.06 km^2	12 km^2	1,257 km^2	14.5 km^2	125 km^2
AUF [f]	1	1	1	1	1	0.68	1	0.03	0.002	0.17	0.02

Notes

AUF = area use factor

a - Listed species; all other life history parameters are based on surrogate receptors. which are spotted sandpiper for ibis and great blue heron for bald eagle.

b - The exposure unit is calculated in units that match the units of the home range so that an AUF may be calculated. See Figures 4-13 through 4-17 for illustrations of these exposure units.

c - Home range for white-faced ibis is given on a km^2 basis, which was converted to relevant habitat area at the site by multiplying total shoreline length by a width of 10 m around the shoreline based or shallow water foraging strategy of this species (Safran et al. 2000).

d - Bald eagles have primarily been noted as wintering in site vicinity, but their breeding distribution may include the site vicinity, so AUFs are calculated for both breeding and non-breeding eagles.

e - Receptor home ranges are further described in Table 3-12.

f - Receptors whose home range is less than the exposure unit are assigned an AUF of 1; for receptors lacking home range data, an AUF of 1 is assumed.

Table 4-13
Daily Ingestion Rates of $COPC_{E}s$ for Aquatic and Upland (North of I-10) Wildlife Receptors

	Ingestion Rate (mg/kg bw-day)																					
	Great Blue Heron		**Spotted Sandpiper**		**Neotropic Cormorant**		**Killdeer**		**Marsh Rice Rat**		**Raccoon**		**Alligator Snapping Turtle**		**White-Faced Ibis**		**Bald Eagle: Breeding**		**Bald Eagle: Wintering**		**Brown Pelican**	
Analyte	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**
Cadmium	0.0015	0.0021	0.071	0.087	0.0013	0.0016	0.68	1.2	0.048	0.061	0.0059	0.0082	8.3×10^{-4}	0.0011	0.0022	0.0027	2.7×10^{-4}	3.6×10^{-4}	2.7×10^{-5}	3.6×10^{-5}	2.7×10^{-6}	3.2×10^{-6}
Copper	0.21	0.27	8.1	10	0.41	0.46	1.0	4.9	3.1	3.7	0.43	0.60	0.033	0.042	0.25	0.32	0.036	0.047	0.0036	0.0047	0.0008	0.0009
Mercury	0.010	0.013	0.027	0.042	0.014	0.018	0.19	0.54	0.016	0.021	0.0048	0.0089	9.7×10^{-4}	0.0012	0.0008	0.0013	0.0018	0.0023	0.00018	0.00023	3.0×10^{-5}	4.0×10^{-5}
Nickel	0.074	0.13	1.7	2.1	0.14	0.19	0.26	0.92	0.63	0.78	0.061	0.12	0.010	0.016	0.053	0.065	0.013	0.022	0.0013	0.0022	0.0003	0.0004
Zinc	18	20	24	28	12	12	56	100	17	21	6.0	8.1	1.8	2.0	0.75	0.89	3.1	3.5	0.31	0.35	0.02	0.02
Bis(2-ethylhexyl)phthalate	0.026	0.033	0.21	0.26	0.029	0.029	8.2×10^{-4}	0.0096	0.089	0.11	0.014	0.019	0.0026	0.0033	0.0066	0.0082	0.0046	0.0058	0.00046	0.00057	6.0×10^{-5}	6.0×10^{-5}
$TEQ_{DF, B}$ [a]	6.8×10^{-6}	1.6×10^{-5}	1.7×10^{-4}	3.8×10^{-4}	1.5×10^{-6}	7.8×10^{-6}	4.3×10^{-5}	1.3×10^{-4}	N/A	N/A	N/A	N/A	4.4×10^{-7}	1.2×10^{-6}	4.6×10^{-6}	1.2×10^{-5}	1.2×10^{-6}	2.7×10^{-6}	1.2×10^{-7}	2.7×10^{-7}	3.0×10^{-9}	1.6×10^{-8}
$TEQ_{P, B}$ [b]	1.0×10^{-6}	1.3×10^{-6}	5.7×10^{-6}	8.4×10^{-6}	6.3×10^{-7}	1.2×10^{-6}	4.2×10^{-8}	5.6×10^{-8}	N/A	N/A	N/A	N/A	9.7×10^{-8}	1.3×10^{-7}	1.8×10^{-7}	2.6×10^{-7}	1.8×10^{-7}	2.3×10^{-7}	1.8×10^{-8}	2.8×10^{-8}	1.3×10^{-9}	2.3×10^{-9}
$TEQ_{DF, M}$ [c]	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	6.3×10^{-6}	1.7×10^{-5}	3.5×10^{-6}	8.9×10^{-6}	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
$TEQ_{P, M}$ [d]	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	3.4×10^{-7}	6.8×10^{-7}	1.8×10^{-7}	2.5×10^{-7}	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Total PCBs	0.057	0.098	0.59	1.4	0.015	0.038	0.030	0.038	0.073	0.17	0.029	0.060	0.0037	0.0055	0.018	0.045	0.010	0.017	0.0010	0.0017	3.0×10^{-5}	0.0001

Notes

CT = central tendency

RM = reasonable maximum

a - Toxicity equivalent for dioxins and furans calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit.

b - Toxicity equivalent for dioxin-like PCBs calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit.

c - Toxicity equivalent for dioxins and furans calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection

d - Toxicity equivalent for dioxin-like PCBs calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection limit.

Table 4-14
Regression Models Developed by Elliott et al. (2001) for Predicting Dioxin and Furan Concentrations in Bird Eggs from Prey Fish of Birds

Congener	r^2	Slope	Intercept	p-value
2,3,7,8-TCDD	0.848	0.869	1.484	0.004
∑ PeCDD	0.904	0.647	1.832	0.002
∑HxCDD	0.917	0.662	1.757	<0.001
2,3,7,8-TCDF	0.628	0.407	0.333	0.07
∑PeCDF	0.847	0.741	1.4	0.008

Source

Elliott et al. (2001); Equation: log (egg [ ]) = slope x log (prey [ ]) + Intercept

Table 4-15
Sample Location Identification and Associated Dioxin and Furan Concentrations for Prey and Sediment Media at the Site and for Post-TCRA and Background Scenarios

	Ingestion Fraction for Each Receptor						Congener (ng/kg ww)							
Dietary Source	**Cormorant**	**Heron**	**Sandpiper**	**Scenario**	**Mode**	**Sample ID**	TCDD	ΣPeCDD	ΣHxCDD	ΣHpCDD	TCDF	ΣPeCDF	ΣHxCDF	ΣHpCFD
Blue crab	NA	0.01	0.5	Site	CT	SJFCA2-CR6	1.60	0.237	0.718	0.612	5.82	0.714	0.0872	0.0200
					RM	SJFCA1-CR6	3.34	1.16	1.67	0.864	10.2	2.61	0.259	0.0216
				Background	CT	SJFCACB-CR6	0.0668	0.0374	0.384	0.635	0.281	0.117	0.0186	0.0216
					RM	SJFCACB-CR1	0.124	0.0322	0.461	0.424	0.251	0.0840	0.0361	0.0153
Common rangia	NA	NA	0.5	Site	CT	CL-TTR5-001	1.58	0.0256	0.0379	0.403	7.73	0.0312	0.0120	0.0276
					RM	CL-TTR3-005	5.79	0.0261	0.0377	0.318	34.8	0.185	0.0249	0.0321
				Background	CT	CL-TTR8-002	0.0540	0.0660	0.0484	1.28	1.20	0.0580	0.0336	0.0414
					RM	CL-TTR7-001	0.132	0.0403	0.148	1.38	1.69	0.0312	0.0290	0.0328
Gulf killifish	1	0.495	NA	Site	CT	GK-TTR5-2	0.201	0.0123	0.0119	0.447	0.618	0.00880	0.0112	0.0110
					RM	GK-TTR3-2	9.53	0.00995	0.00950	0.348	4.46	0.0125	0.335	0.0165
				Background	CT	GK-TTR7-2	0.120	0.0232	0.0189	0.814	0.0895	0.0218	0.0195	0.0204
					RM	GK-TTR7-1	0.169	0.0795	0.0459	0.381	0.0850	0.0405	0.0331	0.0505
Hardhead catfish	NA	0.495	NA	Site	CT	SJFCA1-LF6	23.7	0.0235	4.51	4.47	3.78	2.22	0.0198	0.0213
					RM	SJFCA1-LF1	28.1	0.0236	3.26	3.84	2.83	1.62	0.0163	0.0184
				Background	CT	SJFCACB-LF6	1.62	0.544	1.35	2.14	0.227	0.251	0.495	0.0231
					RM	SJFCACB-LF5	1.67	0.492	1.23	2.32	0.517	0.201	0.0234	0.0190
Sediment	0.02	NA	NA	Site	CT	SJB2	269	3.99	33.5	235	898	127	118	45.8
					RM	SJE1	1020	10.2	14.1	73.2	3,590	225	142	43.6
				Post-TCRA	CT	SJNE052	24.4	2.95	0.0483	0.692	0.316	9.38	0.338	0.726
					RM	SJNE052	24.4	2.95	0.0483	0.692	0.316	9.38	0.338	0.726
				Background	CT	SJUP006	0.307	0.270	8.97	64.2	1.17	0.306	0.175	3.87
					RM	SJUP015	0.117	0.106	7.90	91.7	3.40	0.0920	0.726	3.41
Shoreline sediment	NA	0.033	0.18	Site	CT	TCEQ2009_03	680	130	95.0	220	2700	145	170	75.0
					RM	SJNE022-2	1600	13.4	12.8	80.6	4930	466	371	107
				Post-TCRA	CT	SJSH002	7.65	0.788	0.0160	0.235	0.163	2.75	0.118	0.337
					RM	SJSH021	7.69	6.41	0.0385	0.273	0.0351	25.2	0.703	0.130
				Background	CT	SJSH055	0.0342	0.0268	2.86	20.2	0.826	0.183	0.0178	0.790
					RM	SJSH049	0.0182	0.237	1.25	13.3	4.38	0.702	0.443	0.650

Table 4-16
Regression Models and TEF Substitutions Used To Estimate $TEQ_{DF,B}$ Concentrations in Bird Eggs

Exposure	Regression Model Used	Min. TEF Used	Max. TEF Used
TCDD	TCDD	1	1
PeCDD	PeCDD	1	1
∑HxCDDs	HxCDD	0.01	0.1
∑HpCDD [a]	HxCDD	0.001	0.001
TCDF	TCDF	1	1
∑PeCDF	PeCDF	0.1	1
∑HxCDF [a]	PeCDF	0.1	0.1
∑HpCDF [a]	PeCDF	0.01	0.01

Sources

Regression model: Elliott et al. (2001)

TEF: Van den Berg et al. (1998)

Notes

TEF = toxicity equivalence factor

a - Regression parameters not available; parameters used were for the most closely associated homologue group.

Table 4-17
Predicted TEQ Concentrations for Each Dioxin and Furan Congener and $TEQ_{DF,B}$ in Bird Eggs for the Site

Receptor	Congener	Scenarios																			
		Prey Only				Prey + Sediment				Prey + Post-TCRA Sediment				Background: Prey Only				Background: Prey + Sediment			
		CT		RM		CT		RM		CT		RM		CT		RM		CT		RM	
		TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax
Cormorant	TCDD	7.54	7.54	216	216	136	136	584	584	9.44	9.44	230	230	4.83	4.83	6.49	6.49	5.04	5.04	6.56	6.56
	PeCDD	3.95	3.95	3.44	3.44	14.5	14.5	25.0	25.0	6.80	6.80	8.15	8.15	5.94	5.94	13.2	13.2	6.80	6.80	13.4	13.4
	∑HxCDDs	0.0304	0.304	0.0262	0.262	0.444	4.44	0.253	2.53	0.234	2.34	0.213	2.13	0.0413	0.413	0.0743	0.743	0.196	1.96	0.199	1.99
	∑HpCDD	0.0368	0.0368	0.0312	0.0312	0.182	0.182	0.0847	0.0847	0.0997	0.0997	0.0778	0.0778	0.0545	0.0545	0.0332	0.0332	0.101	0.101	0.0967	0.0967
	TCDF	1.77	1.77	3.96	3.96	7.07	7.07	12.6	12.6	1.97	1.97	4.70	4.70	0.806	0.806	0.789	0.789	0.886	0.886	1.00	1.00
	∑PeCDF	0.0753	0.753	0.0977	0.977	5.02	50.2	7.67	76.72	0.143	1.43	0.545	5.45	0.147	1.47	0.233	2.33	0.177	1.77	0.241	2.41
	∑HxCDF	0.0897	0.0897	1.12	1.12	4.76	4.76	5.91	5.91	0.279	0.279	1.42	1.42	0.136	0.136	0.201	0.201	0.153	0.153	0.263	0.263
	∑HpCDF	0.00886	0.00886	0.0120	0.0120	0.237	0.237	0.230	0.230	0.0297	0.0297	0.0548	0.0548	0.0140	0.0140	0.0275	0.0275	0.0449	0.0449	0.0518	0.0518
	TEQ	**13.5**	**14.5**	**225**	**226**	**168**	**217**	**636**	**708**	**19.0**	**22.4**	**245**	**252**	**12.0**	**13.7**	**21.0**	**23.8**	**13.4**	**16.8**	**21.8**	**25.8**
Heron	TCDD	261	261	387	387	657	657	1240	1240	261	261	391	391	26.7	26.7	28.1	28.1	26.8	26.8	28.1	28.1
	PeCDD	5.42	5.42	6.75	6.75	175	175	41.7	41.7	6.69	6.69	8.72	8.72	29.9	29.9	30.0	30.0	29.9	29.9	30.6	30.6
	∑HxCDDs	0.977	9.77	0.791	7.91	1.74	17.4	0.921	9.21	1.03	10.3	0.851	8.51	0.444	4.44	0.423	4.23	0.484	4.84	0.441	4.41
	∑HpCDD	0.112	0.112	0.101	0.101	0.276	0.276	0.173	0.173	0.137	0.137	0.130	0.130	0.0802	0.0802	0.0756	0.0756	0.0944	0.0944	0.0837	0.0837
	TCDF	2.99	2.99	3.67	3.67	13.5	13.5	17.3	17.3	3.03	3.03	3.99	3.99	1.02	1.02	1.32	1.32	1.09	1.09	1.55	1.55
	∑PeCDF	2.72	27.2	2.20	22.0	9.35	93.5	19.8	198	2.76	27.6	2.30	23.0	0.573	5.73	0.524	5.24	0.592	5.92	0.596	5.96
	∑HxCDF	0.118	0.118	0.695	0.695	9.03	9.03	16.2	16.2	0.239	0.239	0.824	0.824	0.912	0.912	0.179	0.179	0.913	0.913	0.244	0.244
	∑HpCDF	0.0118	0.0118	0.0125	0.0125	0.494	0.494	0.642	0.642	0.0360	0.0360	0.0212	0.0212	0.0147	0.0147	0.0207	0.0207	0.0264	0.0264	0.0297	0.0297
	TEQ	**273**	**306**	**402**	**429**	**867**	**966**	**1340**	**1530**	**275**	**309**	**408**	**437**	**59.6**	**68.8**	**60.6**	**69.2**	**59.9**	**69.6**	**61.6**	**70.9**
Sandpiper	TCDD	45.6	45.6	114	114	2010	2010	4240	4240	49.1	49.1	139	139	2.66	2.66	5.10	5.10	2.89	2.89	5.21	5.21
	PeCDD	18.3	18.3	48.4	48.4	524	524	138	138	21.9	21.9	52.3	52.3	9.99	9.99	7.94	7.94	10.6	10.6	13.1	13.1
	∑HxCDDs	0.300	3.00	0.515	5.15	3.80	38.0	1.22	12.2	0.719	7.19	0.873	8.73	0.207	2.07	0.260	2.60	0.464	4.64	0.375	3.75
	∑HpCDD	0.0400	0.0400	0.0442	0.0442	0.700	0.700	0.369	0.369	0.185	0.185	0.200	0.200	0.0606	0.0606	0.0583	0.0583	0.157	0.157	0.126	0.126
	TCDF	4.69	4.69	7.65	7.65	26.8	26.8	34.5	34.5	4.83	4.83	8.24	8.24	1.90	1.90	2.13	2.13	2.05	2.05	2.71	2.71
	∑PeCDF	1.21	12.1	3.22	32.2	28.5	285	67.7	677	1.49	14.9	3.71	37.1	0.413	4.13	0.303	3.03	0.523	5.23	0.716	7.16
	∑HxCDF	0.271	0.271	0.591	0.591	31.7	31.7	56.6	56.6	0.732	0.732	1.25	1.25	0.168	0.168	0.198	0.198	0.184	0.184	0.497	0.497
	∑HpCDF	0.0158	0.0158	0.0172	0.0172	1.73	1.73	2.25	2.25	0.111	0.111	0.0541	0.0541	0.0194	0.0194	0.0159	0.0159	0.0687	0.0687	0.0588	0.0588
	TEQ	**70.4**	**84.0**	**175**	**208**	**2630**	**2920**	**4540**	**5160**	**79.0**	**98.9**	**205**	**247**	**15.4**	**21.0**	**16.0**	**21.1**	**16.9**	**25.8**	**22.8**	**32.7**

Notes

CT = central tendency
RM = reasonable maximum
TEF = toxicity equivalence factor
TEQ = toxicity equivalent (ng/kg)

Table 4-18
Fish-to-Egg Biomagnification Factors for Selected PCB Congeners

PCB Congener	TEF-Bird (WHO 1998)	Detection Frequency in Onsite Sediments	Correlates with TCDD and TCDF in Onsite Sediments?	Herring Gull BMF[a]	Gray Heron BMF[b]	Kingfisher BMF[b]
Assessment Species - Background				**Cormorant**	**Blue Heron**	**Sandpiper**
Non-ortho Substituted PCBs						
3,3',4,4'-Tetrachlorobiphenyl (PCB 77)	0.05	15/27 (56%)	Y	18.1	0.7	0.16
3,4,4',5-Tetrachlorobiphenyl (PCB 81)	0.1	5/27 (19%)	N	18.1	14.8	3.45
3,3',4,4',5-Pentachlorobiphenyl (PCB 126)	0.1	3/27 (11%)	N	18.7	20.4	4.74
Mono-ortho Substituted PCBs						
2,3,3',4,4'-Pentachlorobiphenyl (PCB 105)	0.0001	23/27 (85%)	Y	20	17.4	4.06
2,3,4,4',5-Pentachlorobiphenyl (PCB 114)	0.0001	15/27 (56%)	Y	18.7	14.4	3.36
2,3',4,4',5-Pentachlorobiphenyl (PCB 118)	0.00001	22/27 (81%)	Y	31	19.8	4.61

Notes

BMF = biomagnification factor

PCB = polyclorinated biphenyl

TEF = toxicity equivalence factor

a - Braune and Norstrom (1989). These authors present fish tissue (alewife) and egg data (herring gulls) for several congeners, but among dioxin-like PCB congeners, only two are represented: PCB 105 and PCB 118. BMFs shown for those not represented are an average for the relevant homologue group.

b - Naito and Murata (2007)

Table 4-19

Estimated $TEQ_{P,B}$ (ng/kg wet weight) for Selected PCB Congeners in Bird Eggs for Each Exposure Scenario [a]

		$TEQ_{P,B}$ (ng/kg wet weight)									
		Prey Only		Prey + Sediment		Prey + Post-TCRA Sediment		Background: Prey		Background: Prey + Sediment	
Receptor	**Congener**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**
Cormorant	PCB077	29.6	20.8	30.4	25.1	29.7	21.1	5.11	4.00	5.18	4.29
	PCB081	1.14	2.21	1.21	2.31	1.26	2.24	1.76	1.96	1.88	2.03
	PCB105	1.43	5.70	1.64	6.01	1.43	5.72	0.440	0.436	0.441	0.439
	PCB114	0.0819	0.380	0.0915	0.395	0.0820	0.381	0.0290	0.0316	0.0291	0.0317
	PCB118	0.654	2.27	0.733	2.39	0.655	2.28	0.216	0.213	0.216	0.214
	PCB126	12.8	36.5	13.3	36.8	13.0	36.6	2.50	10.2	2.64	10.2
	TEQ	**45.7**	**67.8**	**47.4**	**73.0**	**46.1**	**68.3**	**10.1**	**16.8**	**10.4**	**17.2**
Heron	PCB077	2.00	1.54	2.28	2.03	NA	NA	1.01	0.655	NA	NA
	PCB081	6.66	6.02	6.79	7.02	NA	NA	2.89	3.03	NA	NA
	PCB105	16.2	15.2	16.6	15.8	NA	NA	4.97	3.40	NA	NA
	PCB114	0.802	0.767	0.821	0.794	NA	NA	0.262	0.181	NA	NA
	PCB118	3.45	5.13	3.58	5.31	NA	NA	2.23	1.54	NA	NA
	PCB126	46.9	93.3	47.4	95.3	NA	NA	11.0	30.4	NA	NA
	TEQ	**75.9**	**122**	**77.4**	**126**	NA	NA	**22.4**	**39.2**	NA	NA
Sandpiper	PCB077	0.498	0.797	0.841	1.41	NA	NA	0.102	0.128	NA	NA
	PCB081	0.879	0.708	1.05	1.98	NA	NA	0.281	0.295	NA	NA
	PCB105	0.365	0.272	0.930	1.01	NA	NA	0.0329	0.0609	NA	NA
	PCB114	0.0199	0.0117	0.0443	0.0472	NA	NA	0.00195	0.00346	NA	NA
	PCB118	0.124	0.0902	0.282	0.312	NA	NA	0.0206	0.0224	NA	NA
	PCB126	2.37	1.94	3.04	4.40	NA	NA	0.429	0.508	NA	NA
	TEQ	**4.26**	**3.82**	**6.19**	**9.15**	NA	NA	**0.867**	**1.02**	NA	NA

Notes

CT = central tendency

RM = reasonable maximum

TEF = toxicity equivalence factor

a - Not all PCB congeners are represented because biomagnification factors for a full suite of dioxin-like PCB congeners are not presented by any one study, nor for any one species. Selected congeners are those with relatively high TEFs, or which were commonly detected in Site sediments (Table 4-18).

Table 4-20

Estimated Concentrations of $TEQ_{DF.B}$, $TEQ_{P.B}$, and $TEQ_{DFB.P}$ in Bird Eggs under Each Exposure Scenario [a]

Receptor	Scenario	Max $TEQ_{DF,B}$ (ng/kg wet wt)		$TEQ_{P,B}$ (ng/kg wet wt)		$TEQ_{DFP,B}$ (ng/kg wet wt)	
		CT	RM	CT	RM	CT	RM
Cormorant	prey (Gulf killifish)	14.5 (24.0%)	226 (76.9%)	45.7 (76.0%)	67.8 (23.1%)	60.2	294
	prey + sediment	217 (82.1%)	708 (90.7%)	47.4 (17.9%)	73.0 (9.74%)	265	781
	prey + post-TCRA sediment	22.4 (32.7%)	252 (78.7%)	46.1 (67.3%)	68.3 (21.3%)	68.5	320
	prey - background	13.7 (57.6%)	23.8 (58.6%)	10.1 (42.4%)	16.8 (41.4%)	23.7	40.6
	background prey + sediment	16.8 (61.8%)	25.8 (60%)	10.4 (38.2%)	17.2 (40.3%)	27.1	43.0
Great blue heron	prey (Gulf killifish, blue crab, hardhead catfish)	306 (80.1%)	429 (77.8%)	75.9 (19.9%)	122 (22.2%)	382	551
	prey + sediment	966 (92.6%)	1,530 (92.4%)	77.4 (7.42%)	126 (7.63%)	1,040	1,650
	prey + post-TCRA sediment	309 (100%)	437 (100%)	- [b]	- [b]	309	437
	prey - background	68.8 (75.5%)	69.2 (63.8%)	22.4 (24.5%)	39.2 (36.2%)	91.2	108
	background prey + sediment	69.6 (100%)	70.9 (100%)	- [b]	- [b]	69.6	70.9
Sandpiper	prey (common rangia, blue crab)	84.0 (95.2%)	208 (98.2%)	4.26 (4.83%)	3.82 (1.80%)	88.2	212
	prey + sediment	2,920 (99.8%)	5,160 (99.8%)	6.19 (0.211%)	9.15 (0.177%)	2,920	5,170
	prey + post-TCRA sediment	98.9 (100%)	247 (100%)	-	-	98.9	247
	prey - background	21.0 (96.0%)	21.1 (95.4%)	0.870 (3.98%)	1.02 (4.62%)	21.9	22.1
	prey - background	25.8 (100%)	32.7 (100%)	- [b]	- [b]	25.8	32.7

Notes

CT = central tendency

RM = reasonable maximum

TEQ = toxicity equivalent (ng/kg)

a - Percent contribution to $TEQ_{DFP,B}$ is shown.

b - There are no PCB congener data in upstream shoreline sediments

Table 4-21

Parameter Distributions Used for Probabilistic Exposure and Risk Assessment for Wildlife Receptors[a]

Receptor	Distribution Type	Central Tendency	SD	Range	Reference
Sandpiper					
Body Weight (kg)	Normal	0.0471	0.0018	0.043-0.050	DREBWQAT (1999), Maxson and Oring (1980)
Sediment Ingestion Rate (Fraction of Diet)	Triangular	0.18	NA	0.073-0.30	Beyer et al. (1994); mean and range for four sandpiper species
Diet – Crabs (Fraction of Diet)	Uniform	NA	NA	0-1	BPJ
Diet – Clams (Fraction of Diet)	Uniform	NA	NA	0-1	BPJ; fraction in diet for clams calculated in each iteration after random selection of fraction in diet for crabs
Killdeer					
Body Weight (kg)	Normal	0.101	0.0037	0.0922-0.107	Jackson and Jackson (2000) for CT of adult female; range and SD based on scaling sandpiper data to killdeer CT
Sediment Ingestion Rate (Fraction of Diet)	Triangular	0.10	NA	0.02-0.2	Beyer et al. (1994) for CT; BPJ for range
Diet – Terrestrial Invertebrates (Fraction of Diet)	Triangular	0.98	NA	0.5-0.99	Jackson and Jackson (2000) for CT; BPJ for range
Diet – Plants (Fraction of Diet)	Triangular	0.02	NA	0.01-0.5	Jackson and Jackson (2000) for CT; BPJ for range; fraction in diet for plants calculated in each iteration after random selection of fraction in diet for terrestrial invertebrates
Marsh Rice Rat					
Body Weight (kg)	Normal	0.0677	0.0134[b]	N/A	Fernandes (2011)
Sediment Ingestion Rate (Fraction of Diet)	Triangular	0.02	NA	0.01-0.1	Beyer et al. (1994) for CT based on <0.02 for white-footed mouse, and for range based on BPJ and values for black-tailed prairie dog, opossum, and raccoon

Table 4-21

Parameter Distributions Used for Probabilistic Exposure and Risk Assessment for Wildlife Receptors[a]

Receptor	Distribution Type	Central Tendency	SD	Range	Reference
Diet – Crabs (Fraction of Diet)	Triangular	0.2	NA	0.066-0.6	Wolfe (1982) for CT; BPJ for range[c]
Diet – Clams (Fraction of Diet)	Triangular	0.2	NA	0.066-0.6	Wolfe (1982) for CT; BPJ for range[c]
Diet – Plants (Fraction of Diet)	Triangular	0.4	NA	0.132-1	Wolfe (1982) for CT; BPJ for range[c]
Diet - Small Fish (Fraction of Diet)	Triangular	0.2	NA	0.066-0.6	Wolfe (1982) for CT; BPJ for range[c]

Notes

NA = not applicable

BPJ = best professional judgment

SD = standard deviation

a - Feeding rate and water ingestion rate were calculated from body weight value using allometric equations in each iteration of the Monte Carlo analysis. Home range was not used in the exposure model for these receptors because Area Use Factor was assumed equal to 1.0.

b - Standard deviation was calculated from the supplied standard error and population sample size provided in Fernandes (2011).

c - For each iteration of the Monte Carlo analysis, values for the dietary components were randomly selected from the specified distributions and then normalized so that all components summed to 1.0. The normalization process included dividing each dietary component value by the sum total of the dietary component values.

Table 5-1
Toxicity Reference Values and Benchmarks for Benthic Macroinvertebrates

Chemical	Sediment Concentration (ng/kg dw for organics; mg/kg dw for metals)		Ref	Water Concentration[a] (µg/L)		Ref	Endpoint/Comments
	TRV Type	Value		TRV Type	Value		
Organic Compounds							
2,3,7,8-TCDD	NOAEC	2,343		NA	NA		Geometric mean of NOAECs for a range of invertebrate taxa from Table B-4
Bis(2-ethylhexyl)phthalate	--	ND		NOAEC[b]	100	c	Opossum shrimp and amphipod mortality in 4 day lab test. NOAEC is LC_{50} ÷10.
Carbazole	--	ND					No marine invertebrate data were available in ECOTOX. No sediment or water TRVs were found in the literature.
Phenol	--	ND		NOAEC[b]	26	d	Mysid shrimp mortality in 4 day lab test. NOAEC is LC_{50} ÷ 10.
Metals							
Aluminum	--	ND		NOAEC[b]	1,000	e	Derived from 96-hour LC_{50} with Harpacticoid copepod. NOAEC is LC_{50} ÷ 10.
Barium	--	ND		--	ND		No marine invertebrate data were available in ECOTOX. No sediment or water TRVs were found in the literature.
Cobalt	--	ND		NOAEC[b]	450	e	Derived from 96-hour LC_{50} with Harpacticoid copepod. NOAEC is LC_{50} ÷ 10.
Copper	ER-L	34	f	--			
	ER-M	270	f	AWQC (CCC)	3.1	g	AWQC (CCC) values are concentrations at or below which unacceptable effects are not expected. [g]
Lead	ER-L	46.7	f	--			
	ER-M	218	f				
Manganese	--	ND		NOAEC[b]	7,000	e	Derived from 96-hour LC_{50} with Harpacticoid copepod. NOAEC is LC_{50} ÷ 10.
Mercury	ER-L	0.15	f	--			
	ER-M	0.71	f	AWQC (CCC)	0.94	g	AWQC (CCC) values are concentrations at or below which unacceptable effects are not expected. [g]
Thallium	--	ND		NOAEC[b]	213	h	Derived from acute toxicity to marine life . NOAEC is EC ÷ 10. Details unavailable.
Vanadium	--	ND		NOAEC	5	i	NOAEC is EC_{50} ÷10 in most sensitive species. Effect is development.
				LOAEC	10	i	LOAEC is EC_{50} ÷ 10 in most sensitive species. Effect is development.
Zinc	ER-L	150	f	--			
	ER-M	410	f	AWQC (CCC)	81	g	AWQC (CCC) values are concentrations at or below which unacceptable effects are not expected. [g]

Notes

-- = Risks were not evaluated using lines of evidence requiring this information.
AWQC = Ambient Water Quality Criteria. Criterion Continuous Concentrations shown
CCC = Criterion Continuous Concentration
CMC = Criterion Maximum Concentration
EC = effects concentration
ER-L = effect range-low: concentration below which effects are rarely observed or predicted among sensitive life stages and (or) species of biota
ER-M = effect range-median: concentration above which effects are frequently or always observed among most species of biota
USEPA = U.S. Environmental Protection Agency
WHO = World Health Organization

a - TRVs as concentrations in water for those chemicals with no AWQC (see Table B-3)
b - TRV is an LC_{50} divided by an uncertainty factor of 10.
c - Ho et al. (1997)
d - Kim and Chin (1995)
e - Bengtsson (1978)
f - Long et al. (1995)
g - Ambient Water Quality Criteria Website (http://water.epa.gov/scitech/swguidance/standards/current/index.cfm#altable)
h - USEPA (1986)
i - WHO (2001)

Table 5-2
Toxicity Reference Values and Benchmarks for Fish

Chemical	Water Concentration[a] (µg/L)		Ref	Fish Food[b] (mg/kg dw)		Ref	Fish Whole Body		Units	Ref	Comments
Organic Compounds											
TCDD (mg/kg lipid)	--	--		--	--		NOAEL	0.321	µg/kg lipid	c	From a species sensitivity distribution; protects 95 percent of fish species. Endpoint is egg survival.
PCBs	--	--		--	--		NOAEL	5.0	mg/kg ww	d	Geometric mean of NOAELs from 3 fish species.
	--	--		--	--		LOAEL	16	mg/kg ww	d	Geometric mean of LOAELs across 3 fish species.
Bis(2-ethylhexyl)phthalate	NOAEL	55,000	e	--	--		--	--	--		Derived from 4-day acute test with sheepshead minnow. NOAEL is LC_{50} ÷ 10. Endpoint is survival.
Metals											
Cadmium				LOAEL	14.1	f	--	--	--		
Copper				NOAEL	50	g	--	--	--		
				LOAEL	100	h	--	--	--		
Mercury				NOAEL	0.5	i	--	--	--		Endpoint is F_0 male survival in mummichog resulting from increased aggression due to neurotoxic effects. aquarium confinement, or both.
				LOAEL	1.9	i	--	--	--		
Nickel	NOAEL	3,600	j, k	ND			--	--	--		Geometric mean of NOECs for several marine fish. See Table B-17 and Appendix B text.
Zinc				NOAEL	1,900	l	--	--	--		Fish exposed to multiple metals in water as well as food. Fish fed live Artemia exposed to zinc chloride in water. Endpoints are growth and survival.
				LOAEL	2,000		--	--	--		Fish fed at same dose of zinc with 0.5% calcium experienced no adverse effects. Endpoint is growth.

Notes

AWQC = ambient water quality criteria
CCC = Criterion Continuous Concentration
CMC = Criterion Maximum Concentration
LOAEL = lowest observed adverse effect level
NOAEL = no observed adverse effect level
TRV = toxicity reference value
-- = Risks were not evaluated using lines of evidence requiring this information.

a - Includes AWQC and TRVs as concentrations in water for those chemicals with no AWQC (see Table B-3)
b - Windward (2011). Values presented are lowest NOAEC with a bounded LOAEC.
c - Steevens et al. (2005)
d - See Table B-11
e - TRV is an LC_{50} divided by an uncertainty factor of 10
f - Hatakayama and Yasuo (1987), as cited in Windward (2011b)
g - Windward (2011b)
h - Windward (2011b)
i - Matta et al. (2001)
j - Hunt et al. (2002)
k - USEPA (1988) Ambient Water Quality Criteria Document for Nickel
l - Windward (2007)

Table 5-3
Toxicity Reference Values for Birds

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Organic Compounds					
PCBs	NOAEL	2	a	Reproduction	Geometric mean of NOAELs for 5 bird species (Table B-11). See Appendix B.
	LOAEL	3			Geometric mean of LOAELs for 4 bird species (Table B-11). See Appendix B.
TCDD (ingested dose)	NOAEL ng/kg-d	14	b	Hen mortality and egg mortality	Ingested dose was estimated from weekly injected dose.
	LOAEL ng/kg-d	140			
TCDD (egg concentration ng/kg ww)	NOAEL	450	c	Egg mortality	Derived from multiple studies. See Appendix B
	LOAEL	2,400			
Bis(2-ethylhexyl)phthalate	NOAEL	74.9	d	Growth	Unbounded NOAEL for body weight
	LOAEL	--			
Metals					
Cadmium	NOAEL	1.47	f	Reproduction, growth	Geometric mean of NOAELs for reproduction and growth
	LOAEL	2.37		Reproduction	Minimum bounded LOAEL for a mortality/growth/repro endpoint
Chromium	NOAEL	2.66	g	Reproduction, growth	Geomean of NOAELs for reproduction and growth
	LOAEL	2.78			Minimum bounded LOAEL for a mortality/growth/repro endpoint
Copper	NOAEL	4.05	h	Reproduction, growth	Highest bounded NOAEL below the lowest bounded LOAEL for survival, growth, or reproduction
	LOAEL	12.1			
Lead	NOAEL	1.63	i	Reproduction	Highest bounded NOAEL below lowest bounded LOAEL
	LOAEL	1.94			Lowest bounded LOAEL
Mercury	NOAEL	0.078	j	Reproduction	One dose only tested. Unbounded NOAEL for first generation.
	LOAEL	0.9	k	Reproduction	Administered as methylmercury.

Table 5-3
Toxicity Reference Values for Birds

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Nickel	NOAEL	6.71	l	Reproduction, growth	Geomean of NOAELs for reproduction and growth
	LOAEL	11.5		Growth	Minimum bounded LOAEL for a mortality/growth/repro endpoint
Thallium	NOAEL	0.35	m	Survival	This is an LC50 multiplied by an uncertainty factor of 0.01. No LOAEC was available
Vanadium	NOAEL	0.344	n	Growth	Highest bounded NOAEL below the lowest bounded LOAEL for survival, growth, or reproduction
	LOAEL	0.413		Reproduction	Lowest bounded LOAEL for survival, growth, or reproduction
Zinc	NOAEL	66.1	o	Reproduction	Geomean of NOAELs for reproduction and growth
	LOAEL	86.6			Lowest bounded LOAEL for survival, growth, or reproduction

Notes

EcoSSL = Interim EcoSSL Documents by chemical. Available at: http://www.epa.gov/ecotox/ecossl/

LOAEL = lowest observed adverse effect level
NA = not available
NOAEL = no observed adverse effect level
PCB = polychlorinated biphenyl
TCDD = 2,3,7,8-tetrachlorodibenzo-*p* -dioxin
TRV = toxicity reference value
USEPA = U.S. Environmental Protection Agency

a - Risebrough and Anderson (1975)
b - Nosek et al. (1992a)
c - Appendix B
d - O'Shea and Stafford (1980)
e - Johnson et al. (1960)
f - EcoSSL (USEPA 2005b)
g - EcoSSL for Cr(III) (USEPA 2008)
h - EcoSSL (USEPA 2007d)
i - EcoSSL (USEPA 2005c)
j - Heinz (1979)
k -Hill and Schaffner (1976)
l - EcoSSL (USEPA 2007e)
m - USEPA (1999)
n - EcoSSL (USEPA 2005d)
o - USEPA (2007f)

Table 5-4
Toxicity Reference Values for Mammals

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Organic Compounds					
PCBs	NOAEL	0.98	a	Reproduction	Geometric means of NOAELs and LOAELs from toxicity studies with mice. See Appendix B.
	LOAEL	2			
TCDD	NOAEL	0.000001	b	Reproduction	Converted from dietary concentration to dose using assumed body weight and consumption rate.
	LOAEL	0.00001			
Bis(2-ethylhexyl)phthalate	NOAEL	5.8	c	Reproduction	Effects seen at 29 and 147 mg/kg/day doses might be age-related, in which case NOAEL and LOAEL would be under-estimated
	LOAEL	29			
Metals					
Cadmium	NOAEL	2	d	Geometric mean of bounded NOAELs for growth, mortality, repro	38 bounded NOAELs/LOAELs included in calculation
	LOAEL	10		Geometric mean of associated LOAELs	
Chromium	NOAEL	2.40	e	Reproduction, growth	Geomean of NOAELs for reproduction and growth
	LOAEL	2.82		Mortality	No unbounded LOAELs. This is the minimum unbounded LOAEL for a mortality/growth/repro endpoint.
Copper	NOAEL	5.6	f	Reproduction, growth, survival	Highest bounded NOAEL beneath the lowest bounded LOAEL
	LOAEL	9.34			
Lead	NOAEL	4.7	g	Survival	Highest bounded NOAEL below lowest bounded LOAEL
	LOAEL	5.0		Growth	Lowest bounded LOAEL
Mercury	NOAEL	0.015	h	Survival and growth	Converted from dietary concentration to dose using assumed body weight and consumption rate. Converted to chronic from subchronic exposure period. Administered as methylmercury chloride.
	LOAEL	0.025			
Nickel	NOAEL	1.7	i	Reproduction	Highest bounded NOAEL below the lowest bounded LOAEL for a mortality/growth/repro endpoint
	LOAEL	2.71			Minimum bounded LOAEL for a mortality/growth/repro endpoint

Table 5-4
Toxicity Reference Values for Mammals

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Thallium	NOAEL	0.071	j	Reproduction	No NOAEL was provided. This NOAEL is the LOAEL multiplied by 0.1. Rats were exposed in drinking water. TRV may overstate bioavailability.
	LOAEL	0.71			
Zinc	NOAEL	75.4	k	Reproduction	Geomean of NOAELs for reproduction and growth; lowest bounded LOAEL for survival, reproduction and growth
	LOAEL	75.9			

Notes

Eco-SSL = Interim Eco-SSL Documents by chemical. Available at: http://www.epa.gov/ecotox/ecossl/

LOAEL = lowest observed adverse effect level
NOAEL = no observed adverse effect level
PCB = polychlorinated biphenyl
TCDD = 2,3,7,8-tetrachlorodibenzo-*p*-dioxin
TRV = toxicity reference value
USEPA = U.S. Environmental Protection Agency

a - Aulerich and Ringer (1977)
b - Murray et al. (1979)
c - David et al. (2000)
d - EcoSSL (USEPA 2005b)
e - EcoSSL (USEPA 2008)
f - EcoSSL (USEPA 2007d)
g - EcoSSL (USEPA 2005c)
h - Sample et al. (1996)
i - EcoSSL (USEPA 2007e)
j - Formigli et al. (1986)
k - USEPA (2007f)

Table 5-5
Summary of Egg Mortality TRVs; Maternal Transfer and Yolk Injection Studies

Exposure Parameter	NOAEC ng/kg ww	LOAEC ng/kg ww	Egg Exposure	Ref	Comments
Ring-necked (or common) pheasant					
$[Egg]_{TCDD}$	328	1,477	MT	a	Egg concentrations estimated on the basis of maternal dose of 1µg/kg for no effects and an estimated 50 percent egg mortality at 4.5 µg/kg bw, assuming a 1 percent maternal transfer into eggs (mean egg wt = 30.5 g) Nosek et al. (1992a; 1993).
$[Egg]_{TCDD}$	100	1,000	YI	b	Egg concentration associated with 10 percent egg mortality
GeoMean for Pheasants	181	1,215			
Double crested cormorant					
$[Egg]_{TCDD}$	1,000	4,000	YI	c	LOAEL is associated with 23.3 percent increase in egg mortality over egg mortality in vehicle controls
$[Egg]_{TCDD}$	1,300	5,400	YI	d	LOAEL is associated with 25.5 percent increase in egg mortality over egg mortality in vehicle controls
GeoMean for Cormorants	1,140	4,648			
FinalGeoMean	450	2,400			Geometric means rounded to two significant figures for use as TRVs
Domestic Chicken					
$[Egg]_{TCDD}$	100	300	YI	e	LOAEL is associated with 100 percent egg mortality over control egg mortality
$[Egg]_{TCDD}$	80	160	YI	f	LOAEL is associated with 63.8 percent increase in egg mortality over egg mortality in vehicle controls
GeoMean for Chickens	89	220			
GeoMeanAll	260	1,100			

Notes

LOAEC = lowest-observed-adverse-effects concentration
LOAEL = lowest observed adverse effect level
MT = maternal transfer
NOAEC = no-observed-adverse effects concentration
TRV = toxicity reference value
YI = yolk injection

a - Nosek et al. (1992b)
b - Nosek et al. (1993)
c - Powell et al (1997a)
d - Powell et al. (1998)
e - Henschel et al. (1997a)
f - Powell et al. (1996)

Table 6-1
Summary of Results for Benthic Macroinvertebrates

Chemical	HQ > 1[a] at one or more sediment sample locations (Figures 6-1 to 6-13)
Semivolatile Organic Compounds	
Bis(2-ethylhexyl)phthalate	N[d]
Carbazole	N[b]
Phenol	Y[d]
Metals	
Aluminum	Y[c]
Barium	Y[c]
Cobalt	N[d]
Copper	Y
Lead	Y
Manganese	Y[d]
Mercury	Y
Thallium	N[d]
Vanadium	Y[c]
Zinc	Y

Notes

HQ = hazard quotient

N/A = not available (no TRV for this $COPC_E$ or addressed via sediment comparison)

Bold values are HQs ≥1

a - Individual sediment samples compared to a sediment TRV, unless otherwise noted

b - Compared to upstream maximum detection limit

c - Compared to upstream REV

d - Surface water TRV compared to estimated porewater at individual sample locations

Table 6-2
Concentrations of 2,3,7,8-TCDD in Clam Tissue (common rangia) from the Site and Background

Sample ID	2,3,7,8-TCDD (ng/kg ww)	
Site		
Transect 2 (FCA1)		
CL-TTR1-001	1.37	J
CL-TTR1-002	1.31	J
CL-TTR1-003	0.348	U
CL-TTR1-004	1.5	
CL-TTR1-005	1.42	J
Transect 3 (FCA2)		
CL-TTR3-001	***10.7***	
CL-TTR3-002	***17.6***	
CL-TTR3-003	***12.6***	
CL-TTR3-004	***13.3***	
CL-TTR3-005	**5.79**	
Transect 4 (FCA2)		
CL-TTR4-001	0.93	U
CL-TTR4-002	1.98	
CL-TTR4-003	1.64	
CL-TTR4-004	0.476	U
CL-TTR4-005	0.519	J
Transect 5 (FCA2)		
CL-TTR5-001	1.58	
CL-TTR5-002	1.18	J
CL-TTR5-003	***2.45***	
CL-TTR5-004	***2.33***	
CL-TTR5-005	1.89	
Transect 6 (FCA3)		
CL-TTR6-001	0.143	U
CL-TTR6-002	0.123	U
CL-TTR6-003	0.784	J
CL-TTR6-004	0.647	J
CL-TTR6-005	0.696	J

Table 6-2
Concentrations of 2,3,7,8-TCDD in Clam Tissue (common rangia) from the Site and Background

Sample ID	2,3,7,8-TCDD (ng/kg ww)	
Upstream Background		
CL-TTR7-001	0.132	U
CL-TTR7-002	0.244	U
CL-TTR7-003	0.454	J
CL-TTR7-004	0.261	U
CL-TTR7-005	0.175	U
CL-TTR8-001	0.0375	U
CL-TTR8-002	0.054	U
CL-TTR8-003	0.0481	U
CL-TTR8-004	0.0505	U
CL-TTR8-005	0.0625	U

Notes
Bold and *italicized* values are higher than the 2 ng/kg ww threshold in tissue associated with histology of reproductive tissues in individual female oysters.

J = Estimated value

U = Compound analyzed, but not detected above detection limit

Table 6-3
Hazard Quotients for Fish Exposed to $COPC_E$s in Food and Incidentally Ingested in Sediment

	Gulf Killifish - TTR1/TTR2				Gulf Killifish - TTR3				Gulf Killifish - TTR4				Gulf Killifish - TTR5				Gulf Killifish - TTR6				Gulf Killifish - Area-Wide			
	NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based	
$COPC_E$	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	NA	NA	<0.1	<0.1	NA	NA	<0.1	<0.1	NA	NA	<0.1	<0.1	NA	NA	<0.1	<0.1	NA	NA	<0.1	<0.1	NA	NA	<0.1	<0.1
Copper	0.5	0.6	0.2	0.3	0.7	0.8	0.3	0.4	0.5	0.6	0.3	0.3	0.5	0.6	0.3	0.3	0.6	0.6	0.3	0.3	0.5	0.6	0.3	0.3
Mercury	0.2	0.2	<0.1	<0.1	0.3	0.3	<0.1	<0.1	0.2	0.3	<0.1	<0.1	0.2	0.2	<0.1	<0.1	0.3	0.4	<0.1	<0.1	0.2	0.3	<0.1	<0.1
Zinc	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1

	Black Drum				Southern Flounder			
	NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based	
$COPC_E$	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	NA	NA	<0.1	<0.1	NA	NA	<0.1	<0.1
Copper	0.6	0.7	0.3	0.4	0.4	0.4	0.2	0.2
Mercury	0.2	0.2	<0.1	<0.1	0.3	0.4	<0.1	0.1
Zinc	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1

Notes

$COPC_E$ = chemical of potential ecological concern
CT = central tendency
HQ = hazard quotient
NA = not available; TRV not available
LOAEL = lowest observed adverse effect level
NOAEL = no observed adverse effect level
RM = reasonable maximum
Bold values are HQs >1

Table 6-4
Hazard Quotients for Fish Exposed to $COPC_E$s in Surface Water under Pre- and Post-TCRA Conditions

$COPC_E$	**Hazard Quotient**
Bis(2-ethylhexyl)phthalate	<0.1
Nickel	<0.1

Notes

$COPC_E$ = chemical of potential ecological concern

Table 6-5
Hazard Quotients for Avian Receptors North of I-10 and Aquatic Areas

	Great Blue Heron				Neotropic Cormorant				Spotted Sandpiper				Killdeer			
	NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based	
$COPC_E$	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**
Cadmium	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.5	0.8	0.3	0.5
Copper	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.1	< 0.1	< 0.1	**2**	**3**	0.7	0.8	0.3	**1**	< 0.1	0.4
Mercury	0.1	0.2	< 0.1	< 0.1	0.2	0.2	< 0.1	< 0.1	0.3	0.5	< 0.1	< 0.1	**2**	**7**	0.2	0.6
Nickel	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.2	0.3	0.1	0.2	< 0.1	0.1	< 0.1	< 0.1
Zinc	0.3	0.3	0.2	0.2	0.2	0.2	0.1	0.1	0.4	0.4	0.3	0.3	0.8	**2**	0.6	**1**
Bis(2-ethylhexyl)phthalate	< 0.1	< 0.1	NA	NA	< 0.1	< 0.1	NA	NA	< 0.1	< 0.1	NA	NA	< 0.1	< 0.1	NA	NA
$TEQ_{DF,\ B}$ [a]	0.5	**1**	0.05	0.1	0.1	0.6	0.01	0.06	**10**	**30**	**1**	**3**	**3**	**9**	0.3	**1**
$TEQ_{P,\ B}$ [b]	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.4	0.6	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
$TEQ_{DFP,\ B}$ [c]	0.6	**1**	< 0.1	0.1	0.2	0.6	< 0.1	< 0.1	**10**	**30**	**1**	**3**	**3**	**9**	0.3	**1**
Total PCBs	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.3	0.7	0.2	0.5	< 0.1	< 0.1	< 0.1	< 0.1

Notes

$COPC_E$ = chemical of potential ecological concern
CT = central tendency
NA = not available; LOAEL-based TRV not available
LOAEL = lowest observed adverse effect level
NOAEL = no observed adverse effect level
RM = reasonable maximum
Bold values are HQs≥1

Table 6-6
Hazard Quotients Based on Estimated Egg Concentrations for Birds Exposed to $TEQ_{DFP,B}$

		Max $TEQ_{DF,B}$ (ng/kg wet wt)				$TEQ_{P,B}$ (ng/kg wet wt)				$TEQ_{DFP,B}$ (ng/kg wet wt)			
Receptor	**Scenario**	**CT**		**RM**		**CT**		**RM**		**CT**		**RM**	
		NOAEL-based HQ	**LOAEL-based HQ**	**NOAEL-based HQ**	**LOAEL-based HQ**	**NOAEL-based HQ**	**LOAEL-based HQ**	**NOAEL-based HQ**	**LOAEL-based HQ**	**NOAEL-based HQ**	**LOAEL-based HQ**	**NOAEL-based HQ**	**LOAEL-based HQ**
Cormorant	prey (Gulf killifish)	< 0.1	< 0.1	0.5	< 0.1	0.1	< 0.1	0.2	< 0.1	0.1	< 0.1	0.7	0.1
	prey + sediment	0.5	< 0.1	**2**	0.3	0.1	< 0.1	0.2	< 0.1	0.6	0.1	**2**	0.3
	prey + post-TCRA sediment	< 0.1	< 0.1	0.6	0.1	0.1	< 0.1	0.2	< 0.1	0.2	< 0.1	0.7	0.1
	prey - Background	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
	Background prey + sediment	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Great blue heron	prey (Gulf killifish, blue crab, hardhead catfish)	0.7	0.1	**1**	0.2	0.2	< 0.1	0.3	< 0.1	0.8	0.2	**1**	0.2
	prey + sediment	**2**	0.4	**3**	0.6	0.2	< 0.1	0.3	< 0.1	**2**	0.4	**4**	0.7
	prey + post-TCRA sediment	0.7	0.1	**1**	0.2	NA	NA	NA	NA	0.7	0.1	**1**	0.2
	prey - background	0.2	< 0.1	0.2	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.2	< 0.1	0.2	< 0.1
	Background prey + sediment	0.2	< 0.1	0.2	< 0.1	NA	NA	NA	NA	0.2	< .01	0.2	< .1
Sandpiper	prey (common rangia, blue crab)	0.2	< 0.1	0.5	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.2	< 0.1	0.5	< 0.1
	prey + sediment	**6**	**1**	**10**	**2**	< 0.1	< 0.1	< 0.1	< 0.1	**6**	**1**	**10**	**2**
	prey + post-TCRA sediment	0.2	< 0.1	0.5	0.1	NA	NA	NA	NA	0.2	< 0.1	0.5	0.1
	prey - background	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
	Background prey + sediment	< 0.1	< 0.1	< 0.1	< 0.1	NA	NA	NA	NA	< 0.1	< 0.1	< 0.1	< 0.1

Notes

TEQ = toxicity equivalent

Values in bold are ≥ 1

Table 6-7

Hazard Quotients for Pre- and Post-TCRA Exposures-for Marsh Rice Rat, Spotted Sandpiper, and Killdeer when Pre-TCRA $HQ_L \geq 1$

		Pre-TCRA Exposures[a]				Post-TCRA Exposures, Median-Based[b]			
		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based	
Receptor	**$COPC_E$**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**
Marsh rice rat	$TEQ_{DF, M}$	**6**	**20**	0.6	**2**	**1**	**5**	0.1	0.5
Spotted sandpiper	$TEQ_{DF, B}$	**10**	**30**	**1**	**3**	0.8	**3**	< 0.1	0.3
Killdeer	$TEQ_{DF, B}$	**3**	**9**	0.3	**1**	0.8	**2**	< 0.1	0.2
	Zinc	0.8	**2**	0.6	**1**	0.8	**2**	0.6	**1**

Notes

$COPC_E$ = chemical of potential ecological concern

CT = central tendency

LOAEL = lowest-observed-adverse-effect level

NOAEL = no-observed-adverse-effect level

RM = reasonable maximum

TCRA = time-critical removal action

Bold values are HQs ≥ 1

a - Exposures based on concentrations in sediments prior to the TCRA.

b - Exposures based on estimated post-TCRA sediment or soil concentrations: median value from upstream background sediments used to replace sediment or soil samples within TCRA footprint, as appropriate.

Table 6-8
Hazard Quotients for Site and Background Exposures for Marsh Rice Rat, Spotted Sandpiper, and Killdeer when Site HQ_L ≥ 1

		Site Exposures				Background Exposures			
		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based	
Receptor	**$COPC_E$**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**
Marsh Rice Rat	$TEQ_{DF, M}$	**6**	**20**	0.6	**2**	0.2	0.2	< 0.1	< 0.1
	$TEQ_{P, M}$	0.3	0.7	< 0.1	< 0.1	0.1	0.2	< 0.1	< 0.1
	$TEQ_{DFP, M}$	**7**	**20**	0.7	**2**	0.3	0.4	< 0.1	< 0.1
Spotted Sandpiper	$TEQ_{DF, B}$	**10**	**30**	**1**	**3**	0.1	0.2	< 0.1	< 0.1
	$TEQ_{P,B}$	0.4	0.6	< 0.1	< 0.1	0.1	0.1	< 0.1	< 0.1
	$TEQ_{DFP, B}$	**10**	**30**	**1**	**3**	0.2	0.3	< 0.1	< 0.1
Killdeer	$TEQ_{DF, B}$	**3**	**9**	0.3	**1**	0.7	**2**	< 0.1	0.2
	$TEQ_{P,B}$	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
	$TEQ_{DFP, B}$	**3**	**9**	0.3	**1**	0.7	**2**	< 0.1	0.2
	Zinc	0.8	**2**	0.6	**1**	0.7	**1**	0.6	0.8

Notes

$COPC_E$ = chemical of potential ecological concern
CT = central tendency
LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
RM = reasonable maximum
TCRA = time-critical removal action
Bold values are HQs ≥ 1

Table 6-9
Hazard Quotients for Mammalian Receptors North of I-10 and Aquatic Areas

	Marsh Rice Rat				Raccoon			
	NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based	
$COPC_E$	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Copper	0.5	0.7	0.3	0.4	< 0.1	0.1	< 0.1	< 0.1
Mercury	**1**	**1**	0.6	0.8	0.3	0.6	0.2	0.4
Nickel	0.4	0.5	0.2	0.3	< 0.1	< 0.1	< 0.1	< 0.1
Zinc	0.2	0.3	0.2	0.3	< 0.1	0.1	< 0.1	0.1
Bis(2-ethylhexyl)phthalate	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
$TEQ_{DF, M}$ [a]	**6**	**20**	0.6	**2**	**4**	**9**	0.4	0.9
$TEQ_{P, M}$ [b]	0.3	0.7	< 0.1	< 0.1	0.2	0.2	< 0.1	< 0.1
$TEQ_{DFP, M}$ [c]	**7**	**20**	0.7	**2**	**4**	**9**	0.4	0.9
Total PCBs	< 0.1	0.2	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1

Notes

$COPC_E$ = chemical of potential ecological concern
CT = central tendency
LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
RM = reasonable maximum
bold values are HQs≥1
a - Toxicity equivalent for dioxins and furans calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection limit.
b - Toxicity equivalent for dioxin-like PCBs calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection limit.
c - Toxicity equivalent for dioxins, furans and dioxin-like PCBs calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection limit.

Table 6-10
Hazard Quotients for Endangered and Threatened Species when $HQ_N \geq 1$ for Surrogate Species

	Bald Eagle, breeding		**Bald Eagle, nonbreeding**		**Brown Pelican**		**White-Faced Ibis**	
	NOAEL-based		**NOAEL-based**		**NOAEL-based**		**NOAEL-based**	
$COPC_E$	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**
Copper	NA	NA	NA	NA	NA	NA	<0.1	<0.1
$TEQ_{DF, B}$ [a]	<0.1	0.20	<0.1	<0.1	<0.1	<0.1	0.3	0.8
$TEQ_{DFP, B}$ [b]	0.1	0.2	<0.1	<0.1	<0.1	<0.1	0.3	0.9

Notes

$COPC_E$ = chemical of potential ecological concern
CT = central tendency
NA = not available; LOAEL-based TRV not available
LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
RM = reasonable maximum
Bold values are HQs≥1

Table 7-1
Exposure Point Concentrations for TEQ in Soils Based on the Central Tendency, Reasonable Minimum, and Reasonable Maximum Exposures of Killdeer

	Concentration in soil, ng/kg dw		
	CT	**Rmin**	**RM**
$TEQ_{DF,B}$	1,650	230	5,190

Notes

CT = central tendency

RM = reasonable maximum

RMin = reasonable minimum

Table 7-2
Hazard Quotients for Avian Receptors North of I-10 and Aquatic Areas for $TEQ_{DF,B}$ With and Without Bioavailability Adjustment for 2,3,7,8-TCDD

	Great Blue Heron				Neotropic Cormorant				Spotted Sandpiper				Killdeer				White-faced Ibis		Bald Eagle, Breeding		Bald Eagle, Wintering	
	NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		NOAEL-based		NOAEL-based	
$COPC_E$	**CT**	**RM**	**CT**	**RME**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**
$TEQ_{DF,B}$ [a]	0.5	**1**	0.05	0.1	0.1	0.6	< 0.1	< 0.1	**10**	**30**	**1**	**3**	**3**	**9**	0.3	**1**	0.3	0.8	< 0.1	0.2	< 0.1	< 0.1
$TEQ_{DF,B}$ without RBA [b]	0.5	**1**	0.05	0.1	0.1	0.6	< 0.1	< 0.1	**10**	**30**	**1**	**3**	**4**	**10**	0.4	**1**	0.4	0.9	< 0.1	0.2	< 0.1	< 0.1

Notes

$COPC_E$ = chemical of potential ecological concern
CT = central tendency
NA = not available; LOAEL-based TRV not available
LOAEL = lowest observed adverse effect level
NOAEL = no observed adverse effect level
RBA = relative bioavailability adjustment factor
RM = reasonable maximum
Bold values are HQs≥1
a - Toxicity equivalent for dioxins and furans calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit, including the relative bioavailability factor adjustment for 2,3,7,8-TCDD (results as presented in Table 6-5).
b - Toxicity equivalent for dioxins and furans calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit, without the relative bioavailability factor adjustment for 2,3,7,8-TCDD.

Table 7-3
Hazard Quotients for Pre- and Post-TCRA Exposures of Spotted Sandpiper and Killdeer to TEQ $_{DF,B}$ with and without Bioavailability Adjustment for 2,3,7,8-TCDD

Receptor	$COPC_E$	Pre-TCRA Exposures[a]				Post-TCRA Exposures, Median-Based[b]			
		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based	
		CT	RM	CT	RM	CT	RM	CT	RM
Spotted Sandpiper	$TEQ_{DF, B}$[c]	**10**	**30**	**1**	**3**	0.8	**3**	< 0.1	0.3
Spotted Sandpiper	$TEQ_{DF, B}$ without RBA[d]	**10**	**30**	**1**	**3**	0.8	**3**	< 0.1	0.3
Killdeer	$TEQ_{DF, B}$[c]	**3**	**9**	0.3	**1**	0.8	**2**	< 0.1	0.2
Killdeer	$TEQ_{DF, B}$ without RBA[d]	**4**	**10**	0.4	**1**	**2**	**5**	0.2	0.5

Notes

$COPC_E$ = chemical of potential ecological concern
CT = central tendency
LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
RM = reasonable maximum
TCRA = time-critical removal action
Bold values are HQs ≥ 1
a - Exposures based on concentrations in sediments prior to the TCRA.
b - Exposures based on estimated post-TCRA sediment or soil concentrations: median value from upstream background sediments used to replace sediment or soil samples within TCRA footprint, as appropriate.
c - Toxicity equivalent for dioxins and furans calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit, including the relative bioavailability factor adjustment for 2,3,7,8-TCDD (results as presented in Table 6-7).
d - Toxicity equivalent for dioxins and furans calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit, without the relative bioavailability factor adjustment for 2,3,7,8-TCDD.

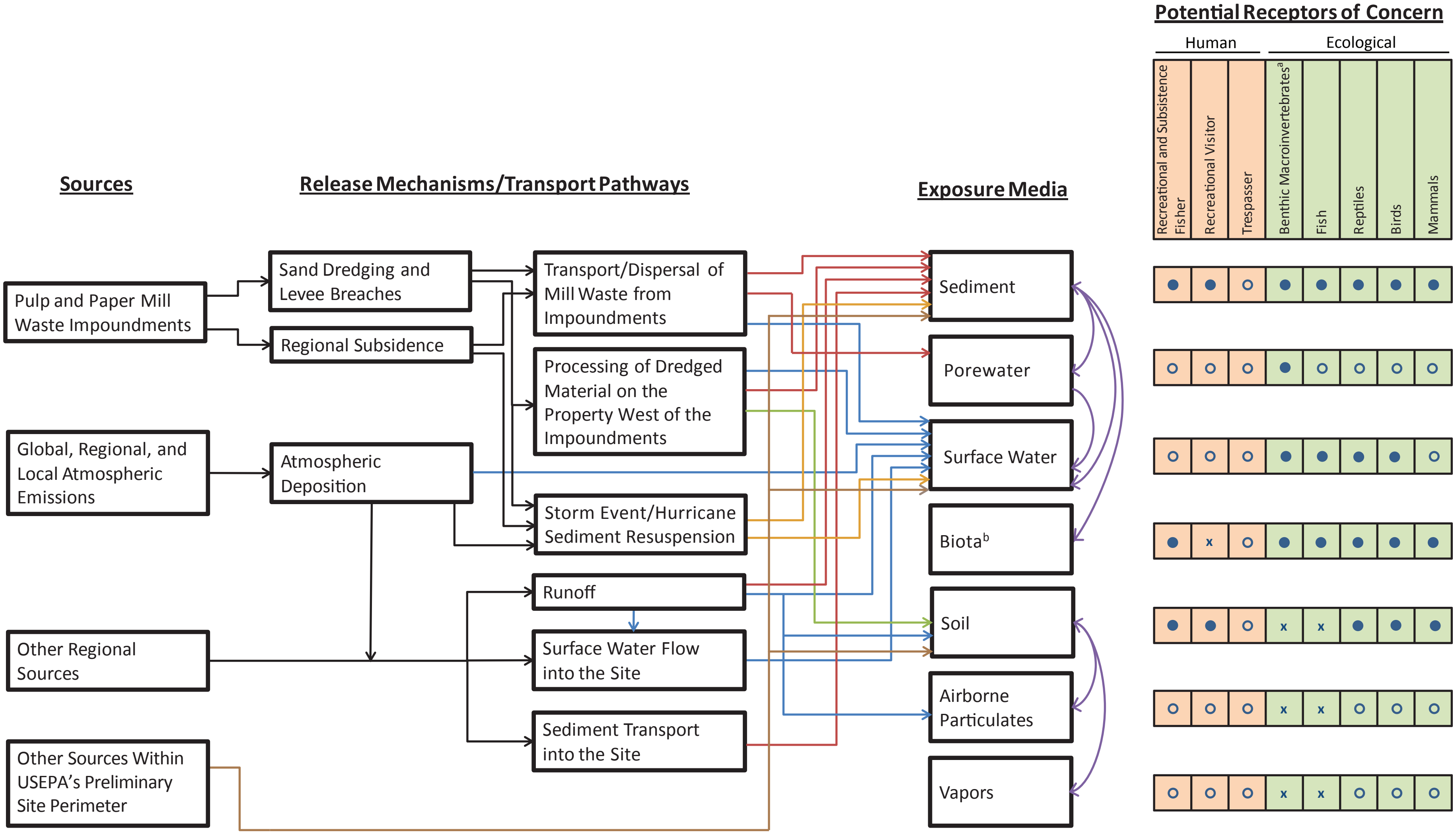
Table 7-4
Hazard Quotients for Site and Background Exposures for Spotted Sandpiper and Killdeer with and without Bioavailability Adjustment for 2,3,7,8-TCDD

Receptor	$COPC_E$	Site Exposures				Background Exposures			
		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based	
		CT	RM	CT	RM	CT	RM	CT	RM
Spotted Sandpiper	RBA[a]:								
	$TEQ_{DF,B}$	**10**	**30**	**1**	**3**	0.1	0.2	< 0.1	< 0.1
	$TEQ_{DFP,B}$	**10**	**30**	**1**	**3**	0.2	0.3	< 0.1	< 0.1
	No RBA[b]:								
	$TEQ_{DF,B}$	**10**	**30**	**1**	**3**	0.1	0.2	< 0.1	< 0.1
	$TEQ_{DFP,B}$	**10**	**30**	**1**	**3**	0.2	0.3	< 0.1	< 0.1
Killdeer	RBA[a]:								
	$TEQ_{DF,B}$	**3**	**9**	0.3	**1**	0.7	**2**	< 0.1	0.2
	$TEQ_{DFP,B}$	**3**	**9**	0.3	**1**	0.7	**2**	< 0.1	0.2
	No RBA[b]:								
	$TEQ_{DF,B}$	**4**	**10**	0.4	**1**	2	**5**	0.2	0.5
	$TEQ_{DFP,B}$	**4**	**10**	0.4	**1**	2	**5**	0.2	0.5

Notes

$COPC_E$ = chemical of potential ecological concern
CT = central tendency
LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
RM = reasonable maximum
Bold values are HQs ≥ 1
a - Including the relative bioavailability factor adjustment for 2,3,7,8-TCDD (results as presented in Table 6-8).
b - Without the relative bioavailability factor adjustment for 2,3,7,8-TCDD.

FIGURES



Exposure Media	Recreational and Subsistence Fisher	Recreational Visitor	Trespasser	Benthic Macroinvertebrates[a]	Fish	Reptiles	Birds	Mammals
Sediment	●	●	○	●	●	●	●	●
Porewater	○	○	○	●	○	○	○	○
Surface Water	○	○	○	●	●	●	●	○
Biota[b]	●	x	○	●	●	●	●	●
Soil	●	●	○	x	x	●	●	●
Airborne Particulates	○	○	○	x	x	○	○	○
Vapors	○	○	○	x	x	○	○	○

● Complete, significant exposure pathway
○ Complete, minor exposure pathway
x Incomplete exposure pathway

Notes:
Other regional sources may include industrial effluents, publicly owned treatment works, and stormwater.
Curved lines indicate potential transport pathways for chemicals of potential concern among exposure media.
[a]Benthic macroinvertebrates include crabs and other crustaceans and shellfish consumed by all receptors, as well as polychaetes and other infauna consumed by fish, other marine life, birds and mammals.
[b]Biota consumed by human receptors are expected to be fish and shellfish.

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Figure 1-1
Conceptual Site Model for the Northern Impoundments and Aquatic Environment
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

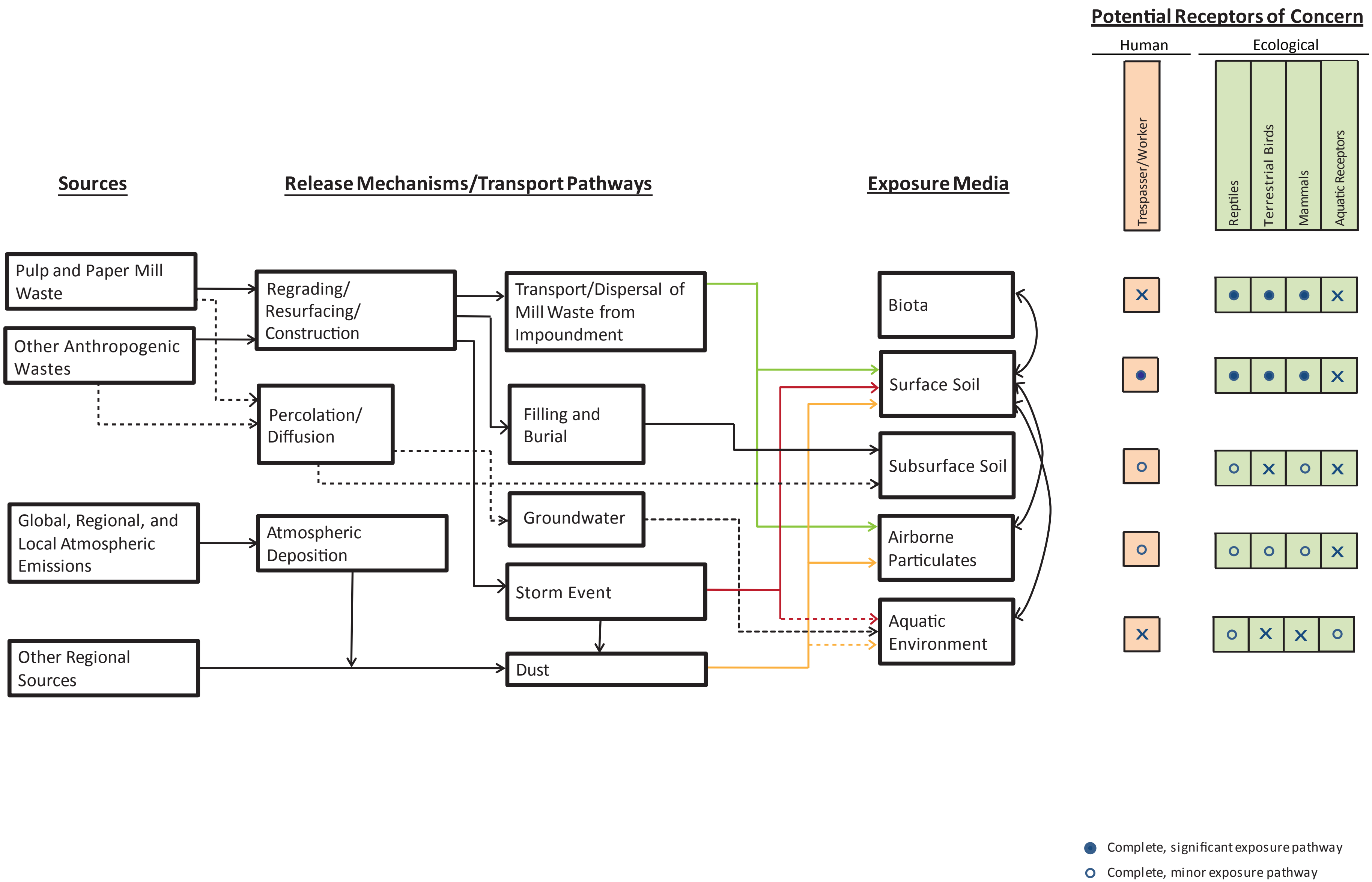
USEPA's Preliminary Site Perimeter
Original (1966) Perimeter of the Northern Impoundments
Area of Soil Investigation South of I-10

0 800
Scale in Feet

Figure 2-1
Overview of Area within USEPA's Preliminary Site Perimeter
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

[a] Designation of the sand separation area is intended to be a general reference to areas in which such activities are believed to have taken place based on visual observations of aerial photography from 1998 through 2002.

FEATURE SOURCES:
Aerial Imagery: 0.5-meter. Photo Date: 01/14/2009
Texas Strategic Mapping Program (StratMap), TNRIS



● Complete, significant exposure pathway
○ Complete, minor exposure pathway
x Incomplete exposure pathway
------ Unknown pathway

Notes:
Local sources may include industrial air emissions, vehicle or machinery fluid leaks, or other releases resulting from ongoing commercial activities on the site.
Curved lines indicate potential transport pathways for chemicals of potential concern among exposure media.

Figure 2-2
Updated Conceptual Site Model Pathways for the Area South of I-10
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

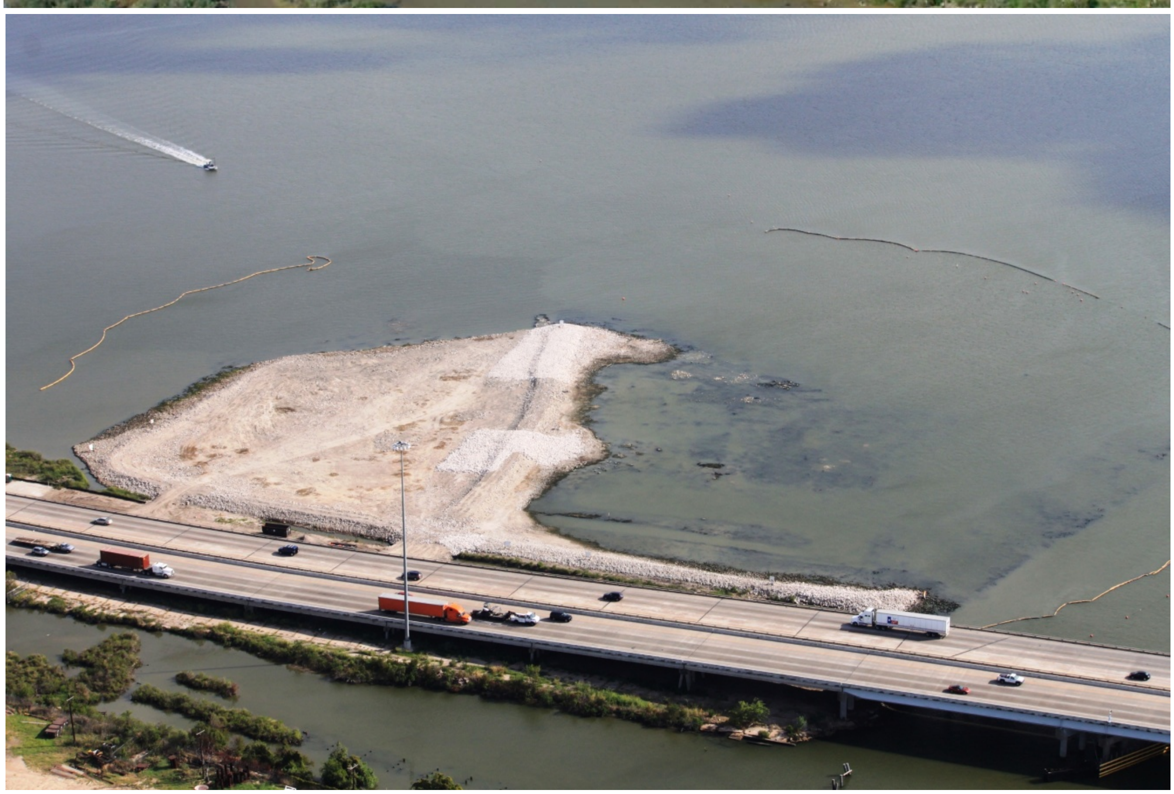


Figure 2-3
Aerial View of TCRA Project Area, Before and After
TCRA Implementation, July 14, 2011
Baseline Ecological Risk Assessment
SJRWP Superfund Site/MIMC and IPC

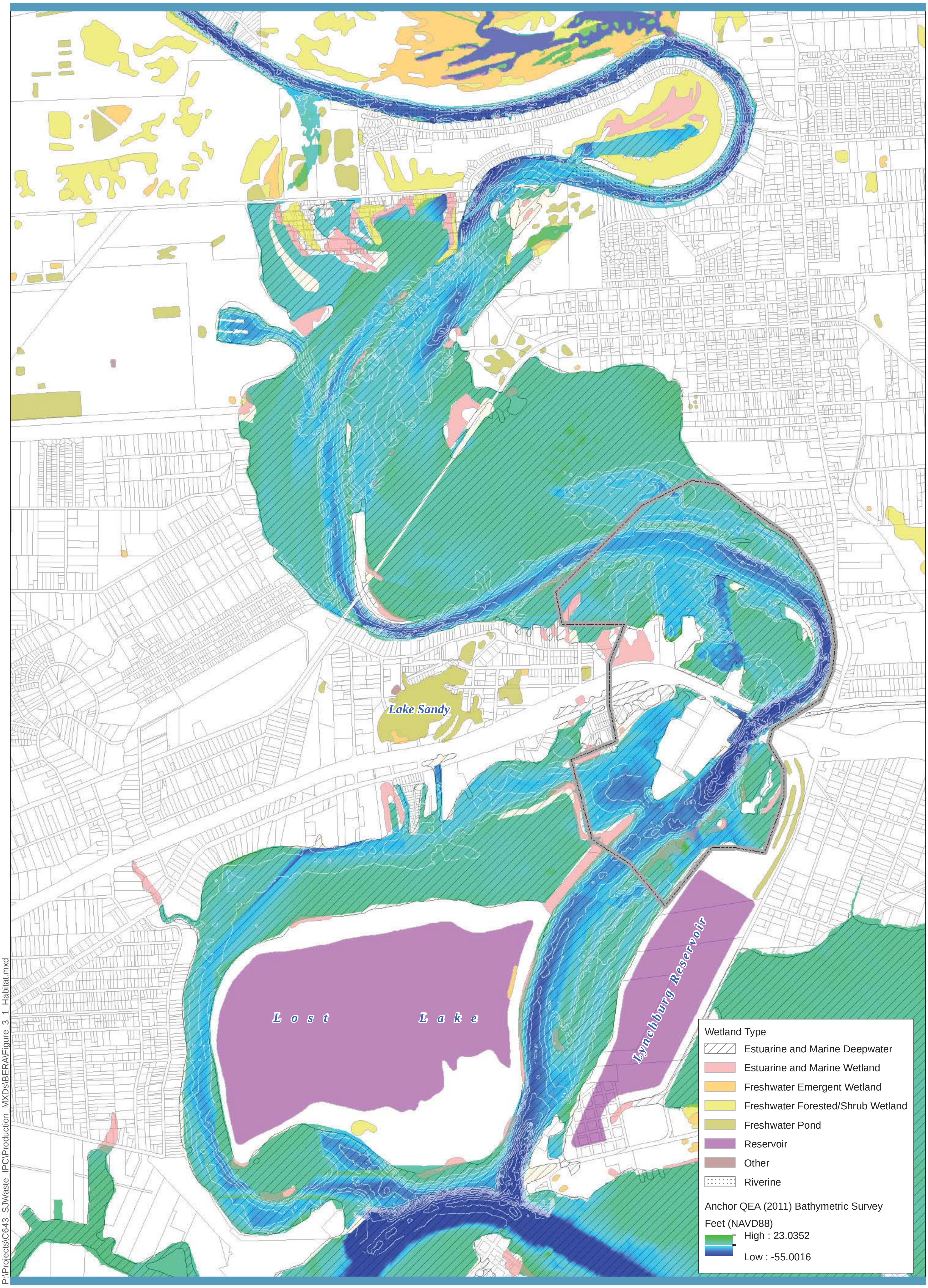


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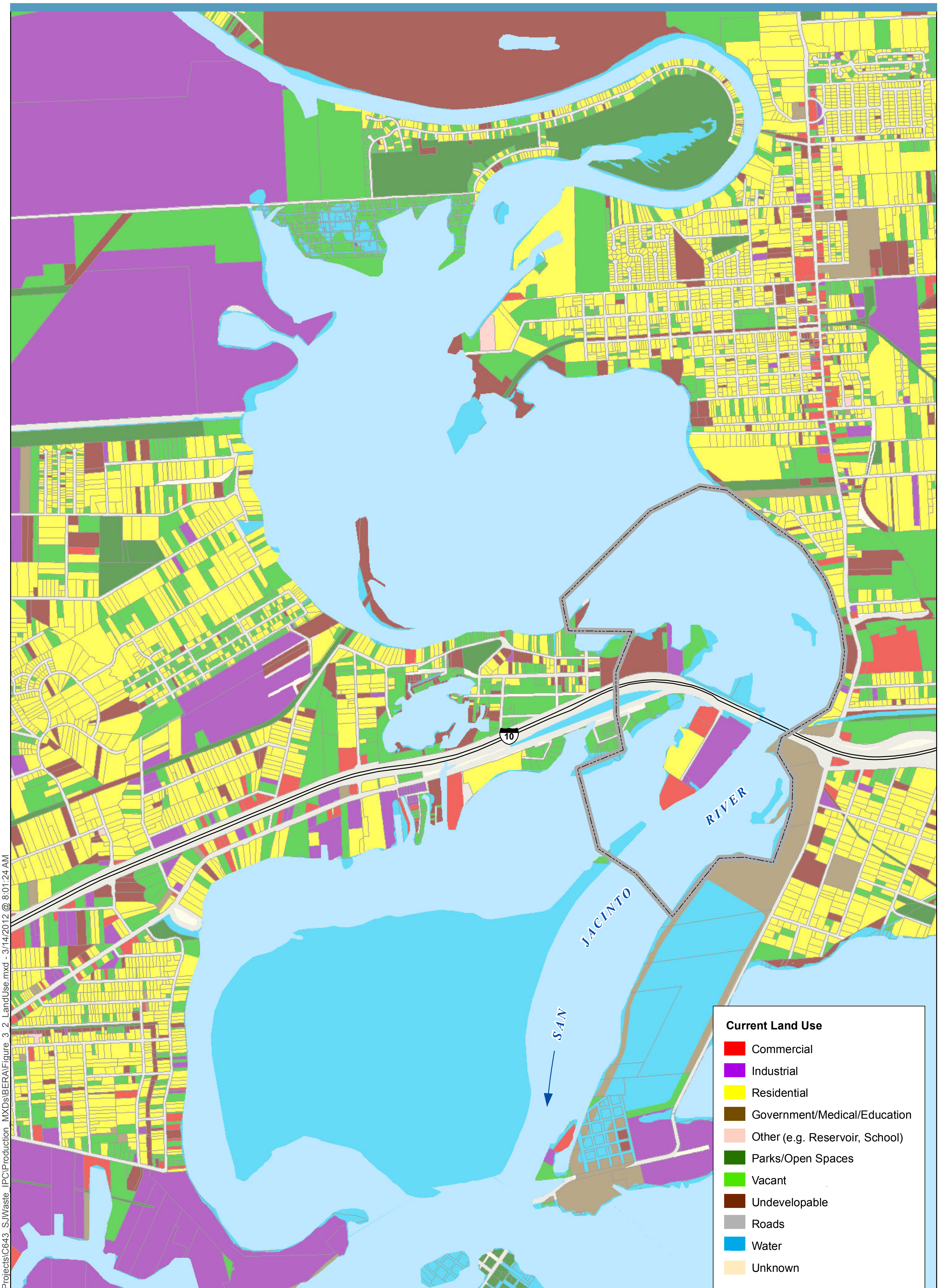
Figure 3-1
Habitats in the Vicinity of the Site
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

FEATURE SOURCES:
Bathymetry and Contours: Anchor QEA 2011
Wetlands: U.S. Fish and Wildlife Service.
Parcel Boundaries: Harris County Appraisal District.



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Preliminary Site Perimeter

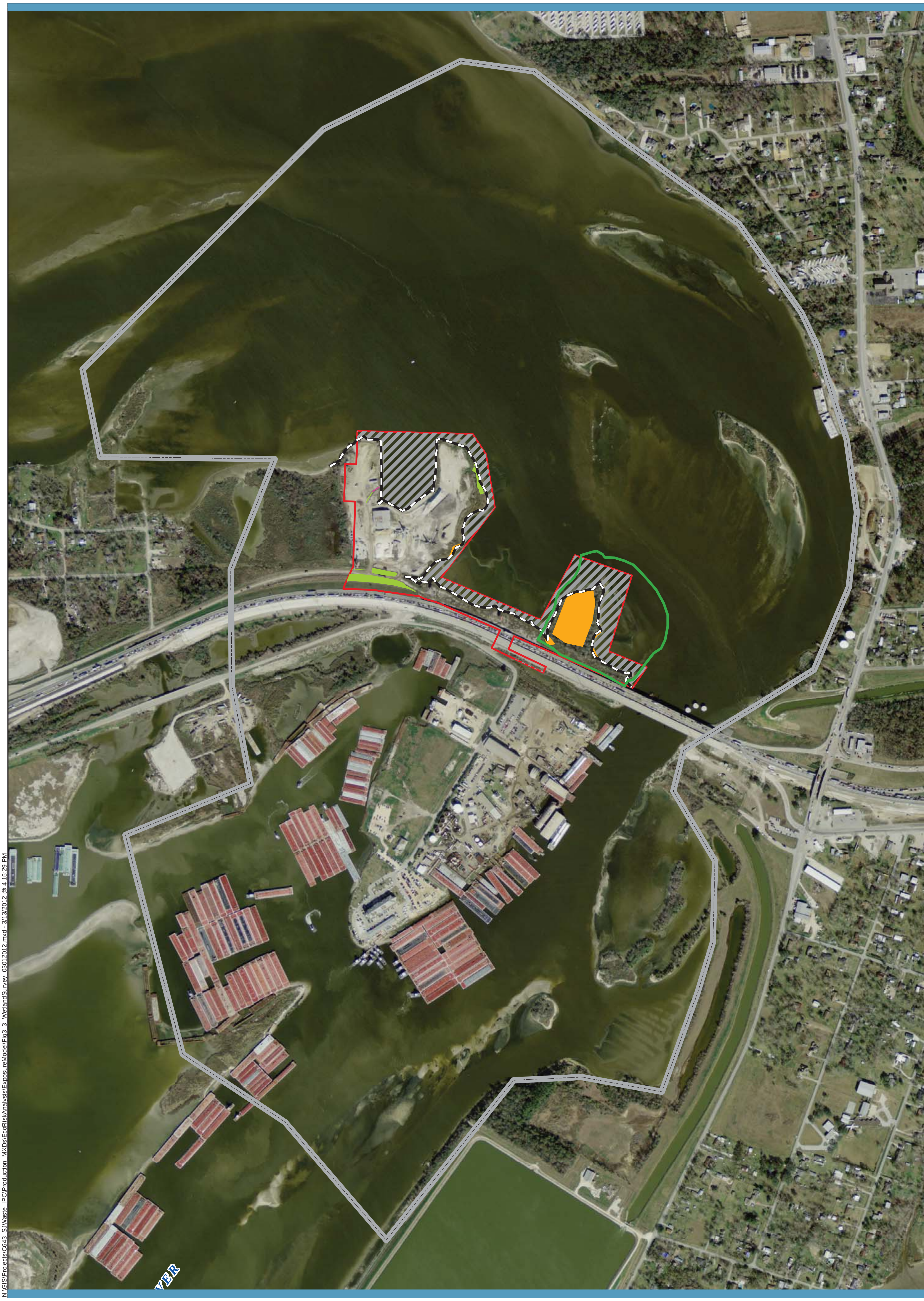
Tax Parcel Boundary

0 0.5

Scale in Miles

FEATURE SOURCES:
Zoning: Houston-Galveston Area Council
Parcel Boundaries: Harris County Appraisal District

Figure 3-2
Land Use in the Vicinity of the Site
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



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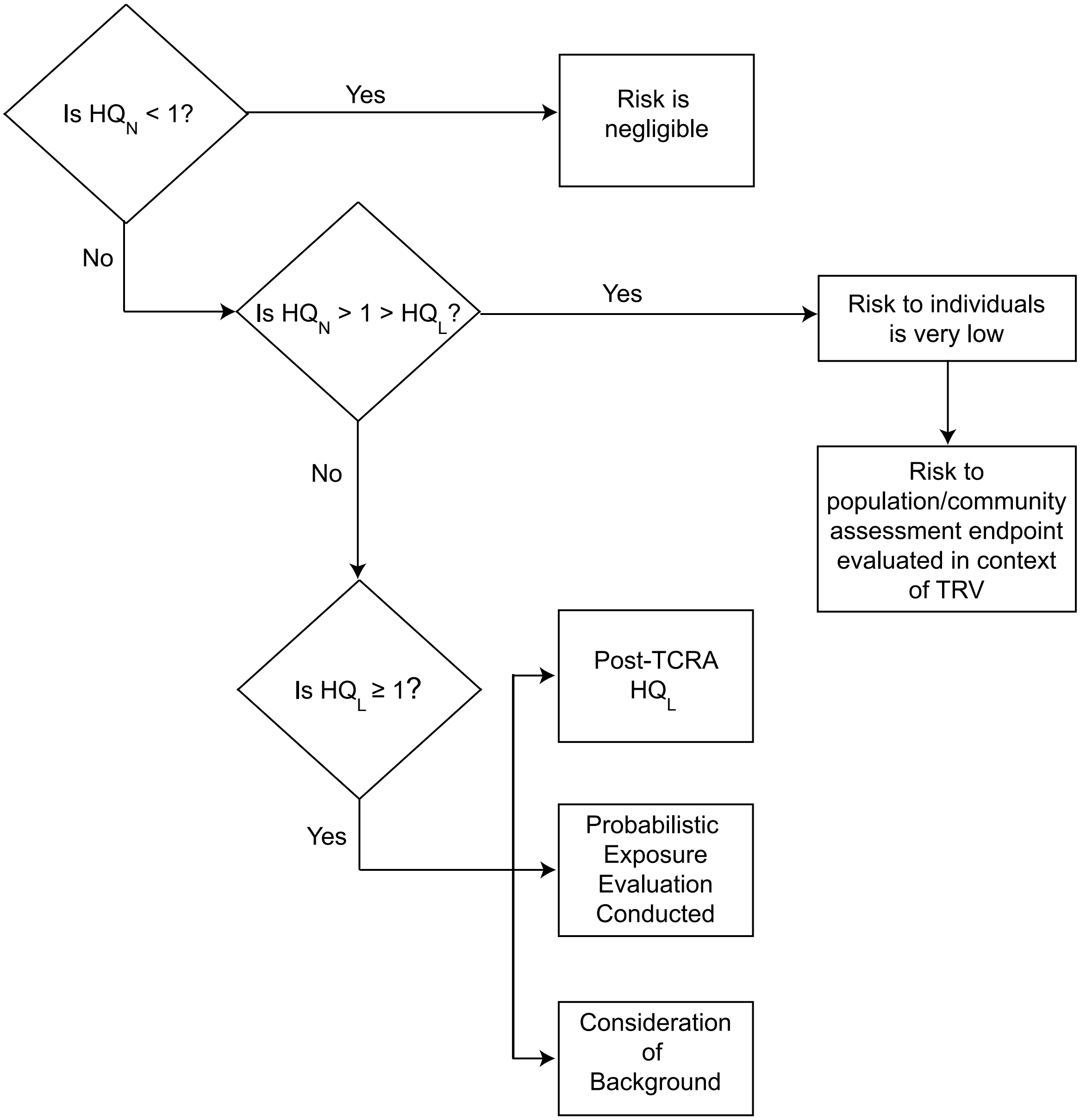
0 500
Feet

Original (1966) Perimeter of the Northern Impoundments
Mean High Water
Wetland Type
Estuarine, Subtidal, Unconsolidated Bottom
Estuarine, Intertidal, Emergent
Palustrine, Emergent
Palustrine, Unconsolidated Bottom
Survey Area
USEPA's Preliminary Site Perimeter

SOURCE: BESI (2010)

Figure 3-3
2010 Site Wetland Delineation
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



Note:
HQ_L = hazard quotient based on comparison to a lowest-observed-adverse-effects level toxicity reference value.

HQ_N = hazard quotient based on comparison to a no-observed-adverse-effects level toxicity reference value.

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USEPA's Preliminary Site Perimeter
Original (1966) Perimeter of the Impoundments
Clams and Small Fish
Small Fish
Clams
Large Fish and Blue Crab Fish Collection Areas

0 800
Scale in Feet

[a] Designation of the sand separation area is intended to be a general reference to areas in which such activities are believed to have taken place based on visual observations of aerial photography from 1998 through 2002.

FEATURE SOURCES:
Aerial Imagery: 0.5-meter January 2009 DOQQs - Texas Strategic Mapping Program (StratMap), TNIS

Figure 4-1
Tissue Sampling Locations
Within USEPA's Preliminary Site Perimeter
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

Study Area
Clams and Small Fish
Potential Blue Crab Collection Area*

0 800
Scale in Feet

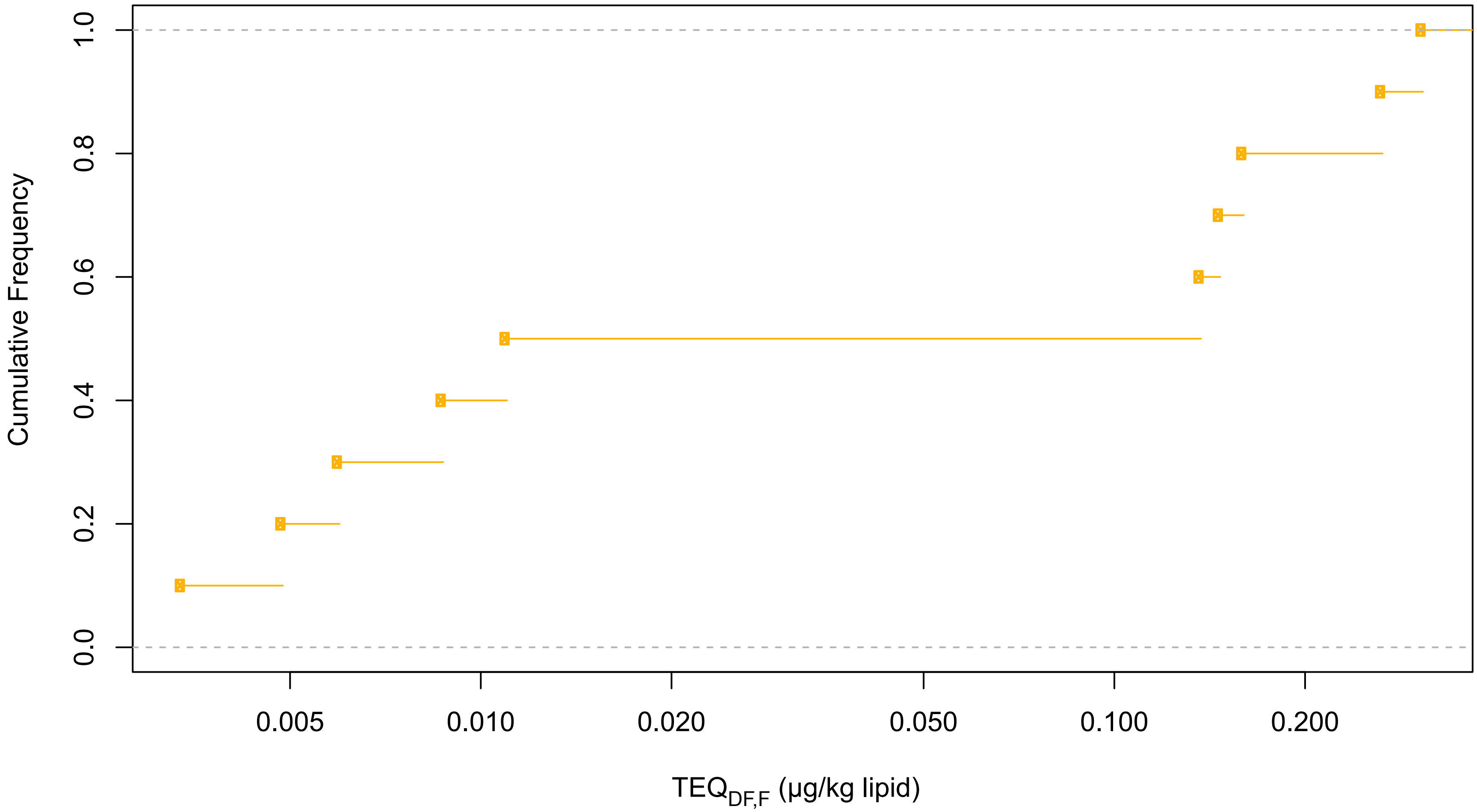
FEATURE SOURCES:
Aerial Imagery: 0.5-meter 2008/2009 DOQQs - Texas Strategic Mapping Program (StratMap) TNRIS;

*Final collection of background blue crab tissues are to be determined in consultation with EPA, and established in an addendum to this SAP.

Figure 4-2
Upstream Background Tissue Sampling Locations
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

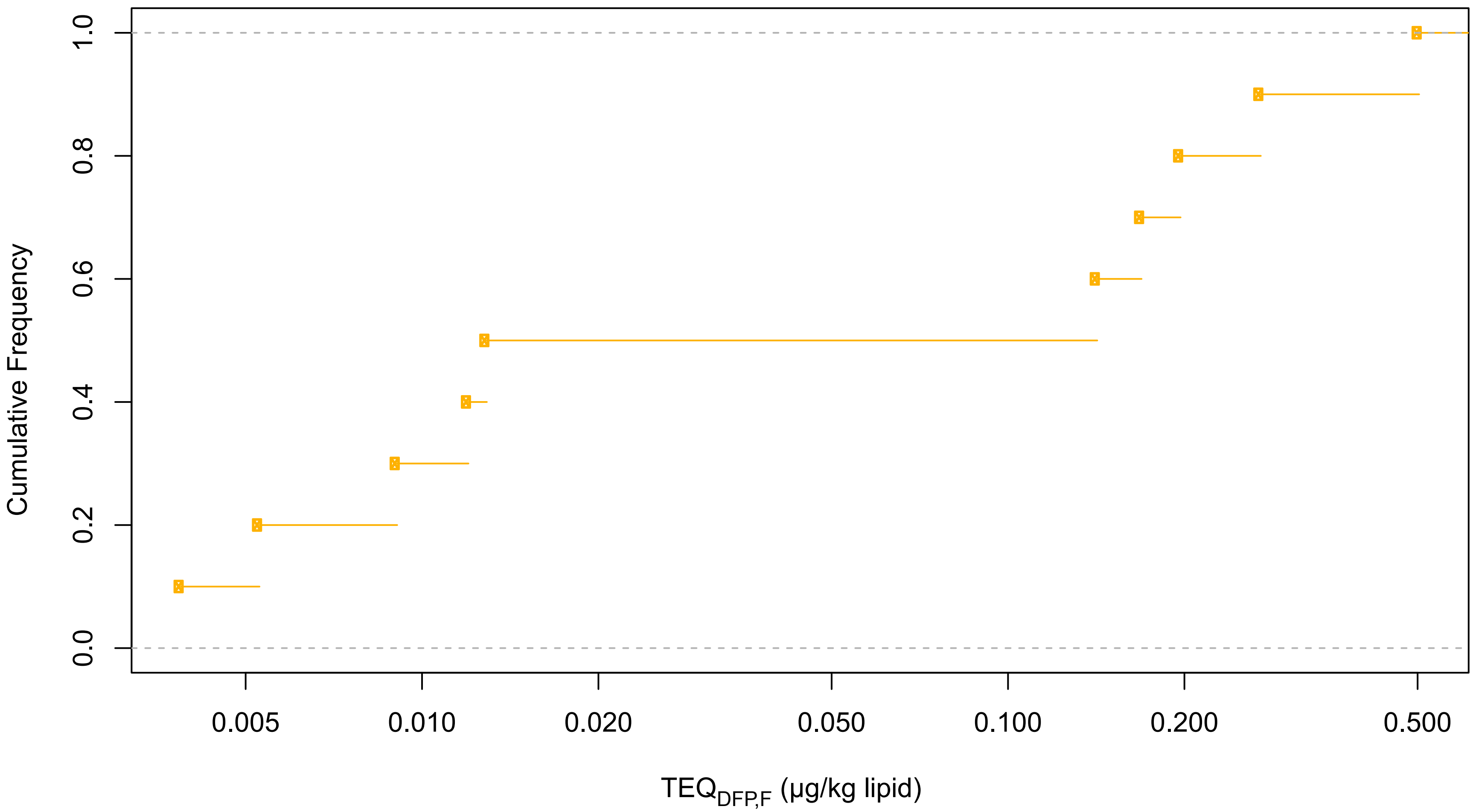
Figure 4-3
Cedar Bayou Background Tissue Sampling Locations
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

FEATURE SOURCES:
Aerial: ESRI USA Prime Imagery, 2008
Transportation Lines: ESRI World Transportation



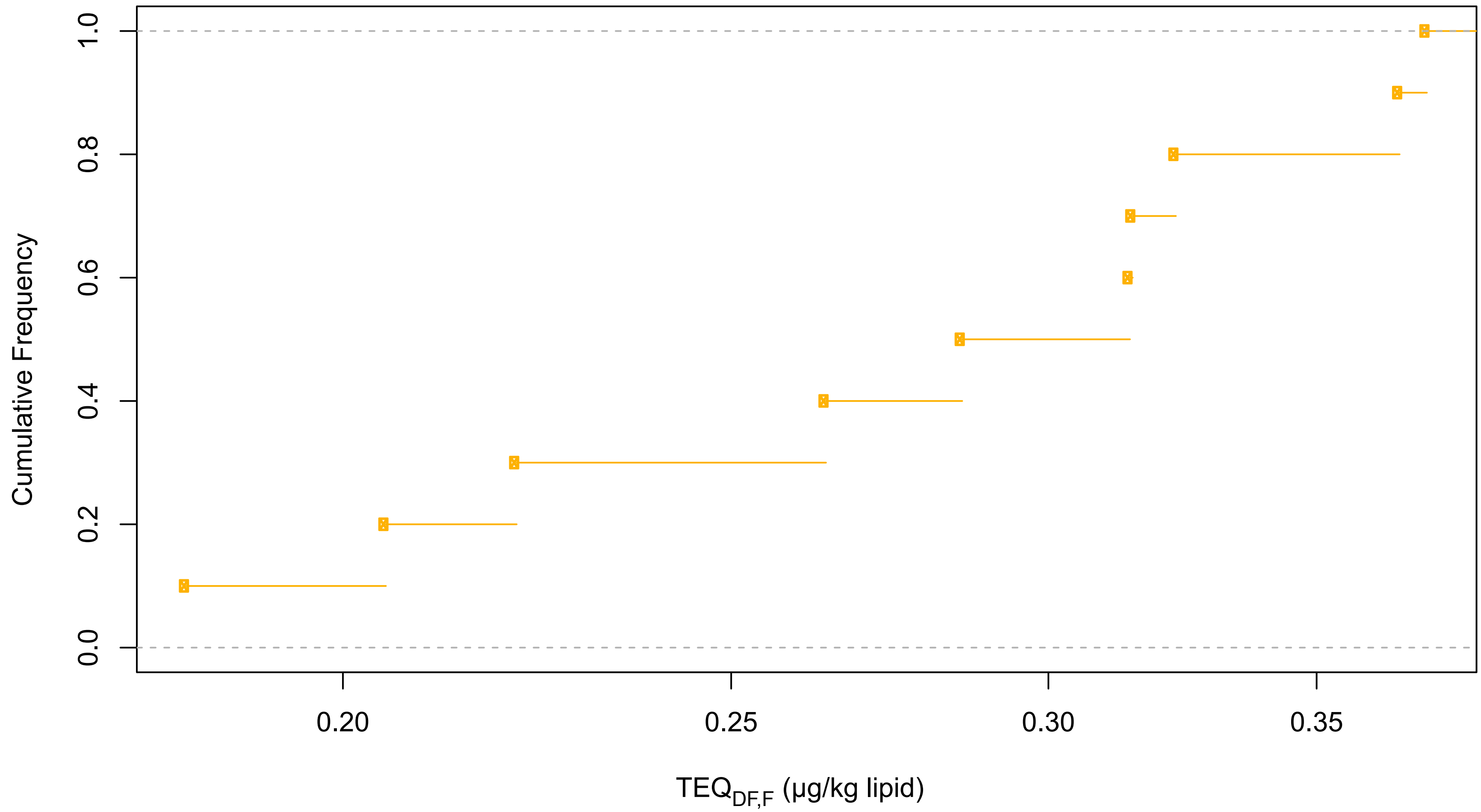
Note:
Lines adjacent to data points indicate distance to the next data point in the distribution.

Figure 4-4
Cumulative Frequency Distribution of Site Gulf Killifish Data (TEQ$_{DF,F}$)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



Note:
Lines adjacent to data points indicate distance to the next data point in the distribution.

Figure 4-5
Cumulative Frequency Distribution of Site Gulf Killifish Data (TEQ$_{DFP,F}$)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



Note:
Lines adjacent to data points indicate distance to the next data point in the distribution.

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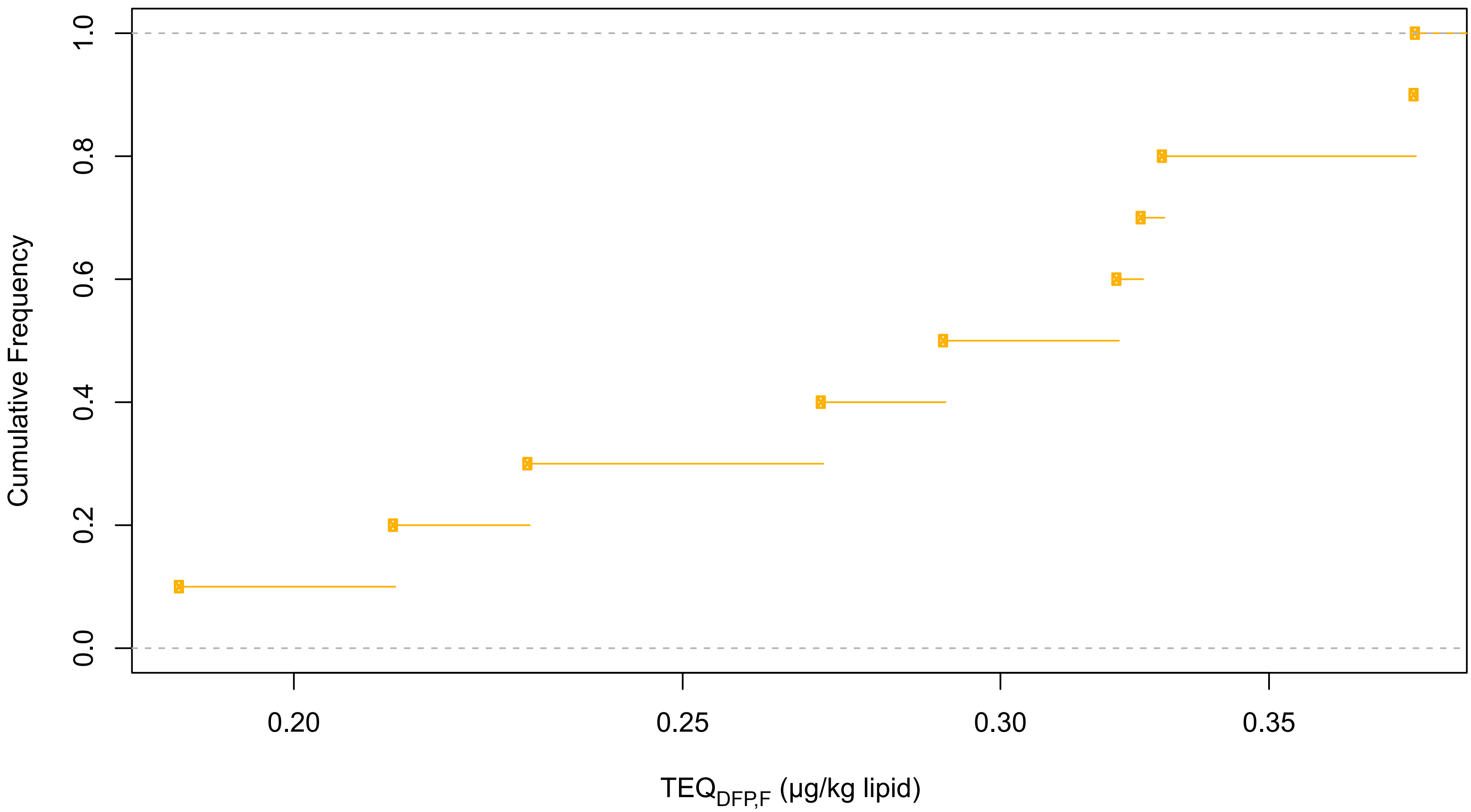
Figure 4-6
Cumulative Frequency Distribution of Site Hardhead Catfish Data ($TEQ_{DF,F}$)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



Note:
Lines adjacent to data points indicate distance to the next data point in the distribution.

Figure 4-7
Cumulative Frequency Distribution of Site Hardhead Catfish Data ($TEQ_{DFP,F}$)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

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Hardhead Catfish Sample Locations
Fish-Crab Sample Area
Preliminary Site Perimeter

0 1,000
Feet

FEATURE SOURCES:
Aerial Imagery: 0.5-meter 2008/2009
DOQQs –Texas Strategic Mapping
Program (StratMap), TNRIS

Figure 4-8
Hardhead Catfish Sample Locations
Within USEPA's Preliminary Site Perimeter
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

Hardhead Catfish Sample Locations
Fish-Crab Sample Area
Preliminary Site Perimeter

0 1,000
Feet

FEATURE SOURCES:
Aerial Imagery: ESRI USA Prime Imagery, 2009

Figure 4-9
Hardhead Catfish Sample Locations
Within Cedar Bayou
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

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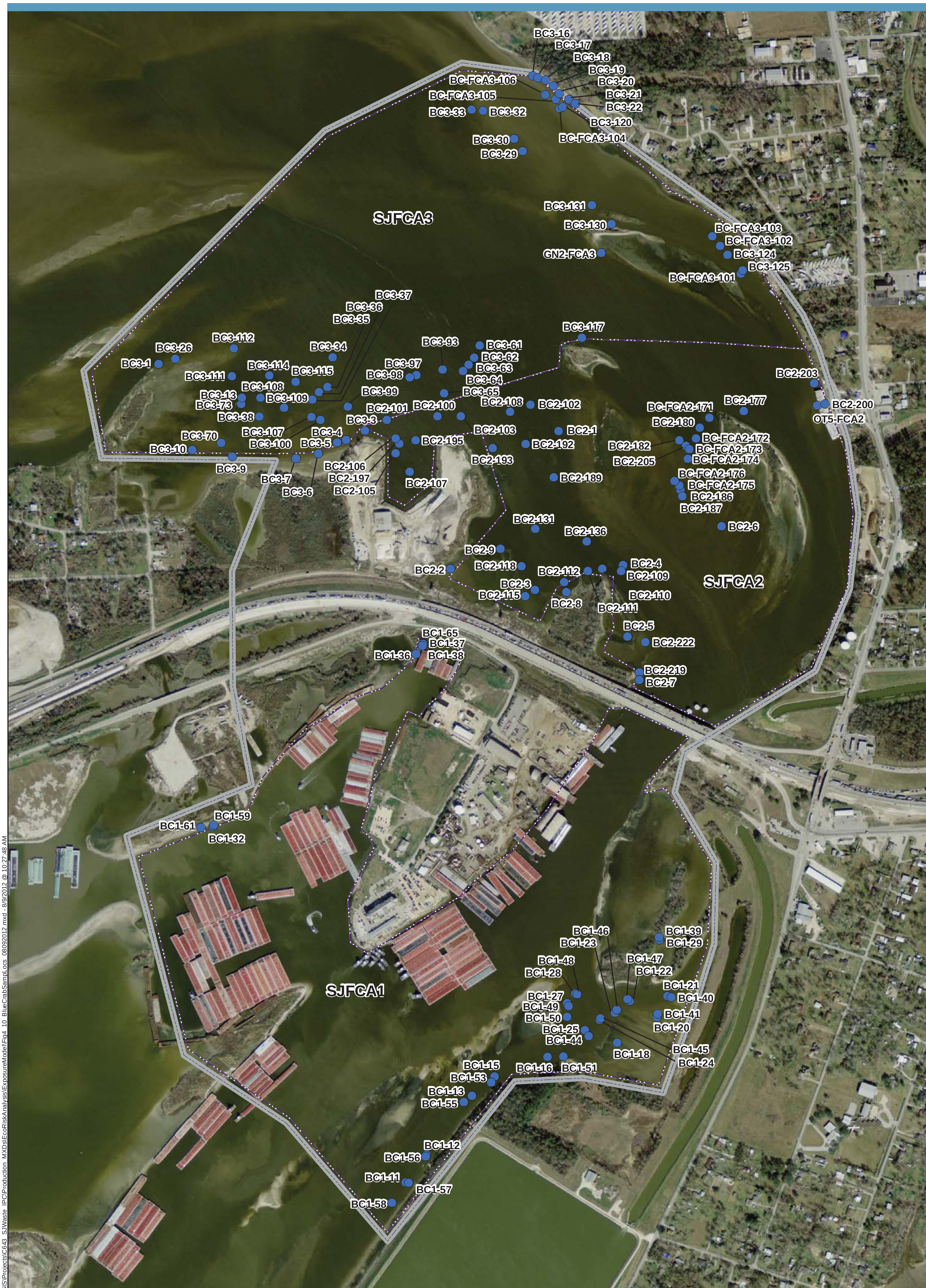

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Blue Crab Sample Stations
Fish-Crab Sample Area
Preliminary Site Perimeter

0 1,000
Feet

FEATURE SOURCES:
Aerial Imagery: 0.5-meter 2008/2009
DOQQs - Texas Strategic Mapping
Program (StratMap), TNRIS

Figure 4-10
Locations of Blue Crab Collections
Within USEPA's Preliminary Site Perimeter
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

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Blue Crab Sample Stations

Fish-Crab Sample Area

Preliminary Site Perimeter

FEATURE SOURCES:
Aerial Imagery: ESRI USA Prime Imagery, 2009

Figure 4-11
Locations of Blue Crab Collections
Within Cedar Bayou
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

ANCHOR QEA

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0 1,000

Feet

Surface Soil Sample Location (0 - 6 inches)

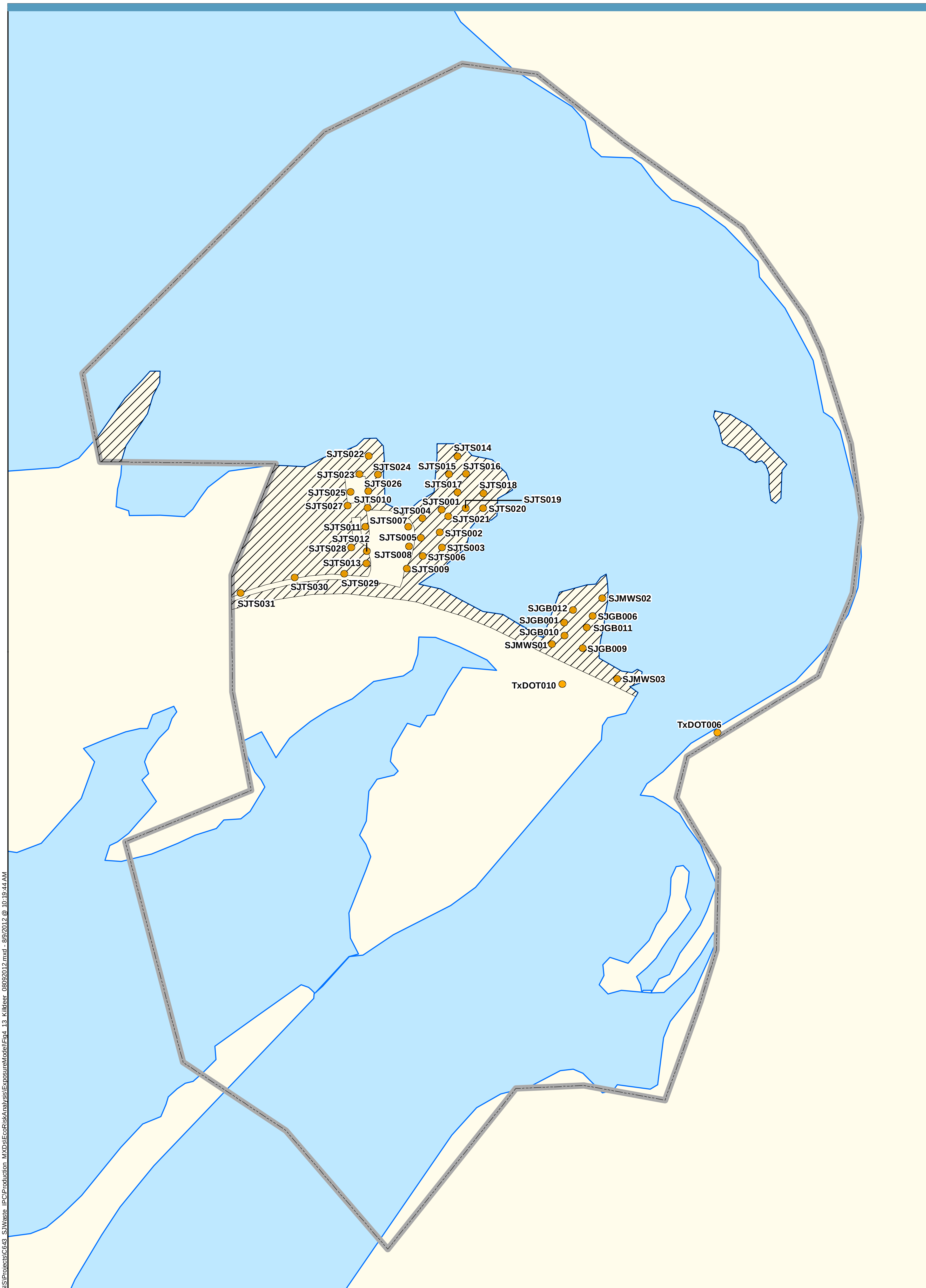
- 0.0005 - 3.2
- 3.2 - 6.5
- 6.5 - 9.7
- 9.7 - 12.9
- Preliminary Site Perimeter
- Area Within the Original (1966) Perimeter of the North Impoundments

Figure 4-12

Concentrations of Mercury (mg/kg) in Surface Soils within the Preliminary Site Perimeter

Baseline Ecological Risk Assessment

SJRWP Superfund/MIMC and IPC



Soil Sample Locations North of I-10 (0 to 6 inches)

Killdeer Estimated Exposure Area

Preliminary Site Perimeter

0 1,000

Feet

Figure 4-13

Exposure Areas and Samples Used for Estimating Exposures to Killdeer

Baseline Ecological Risk Assessment

SJRWP Superfund/MIMC and IPC

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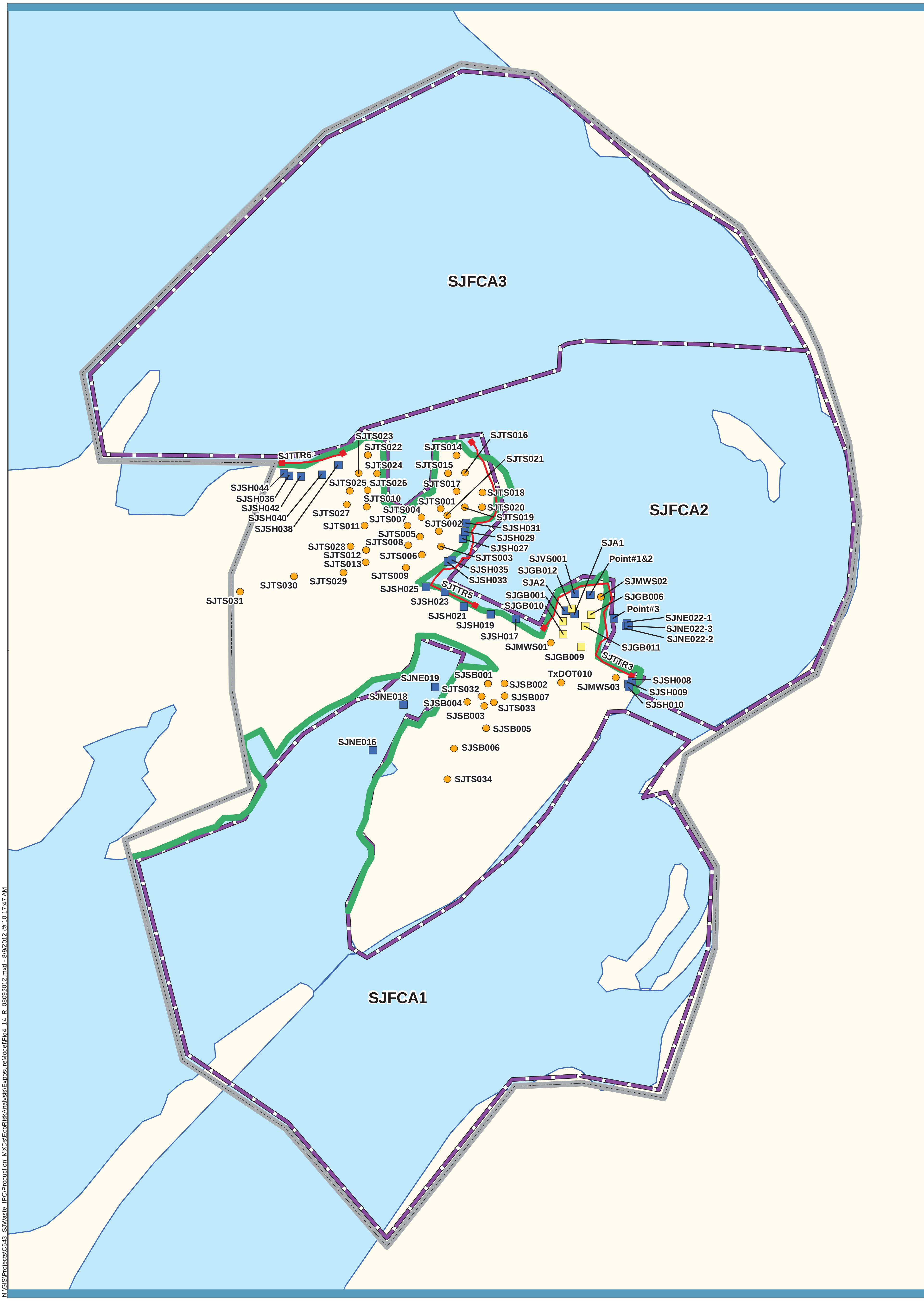

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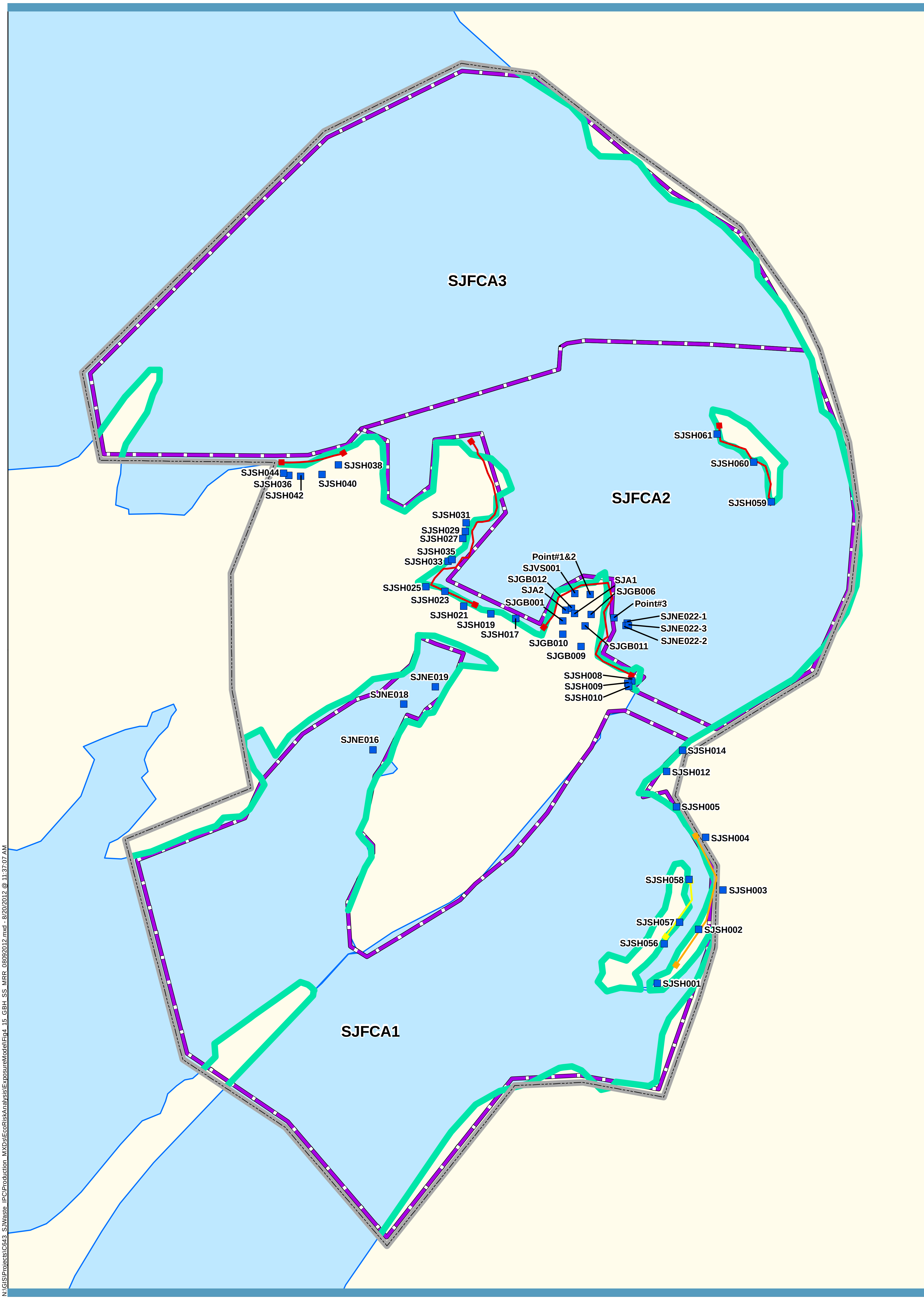
0 1,000
Feet

Shoreline Surface Sediment Sample Locations (0-6 inches)
- Used as Sediment to Evaluate Exposures
- Used as Both Sediment and Soil to Evaluate Exposures

Surface Soil Sample Locations (0 to 6 inches)
- Clams, Seines (Small Fish), and Infauna
- Raccoon Estimated Exposure Area
- Catfish / Blue Crab Sample Area

Preliminary Site Perimeter

Figure 4-14
Exposure Areas and Samples Used
for Estimating Exposures to Raccoons
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



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0 1,000

Feet

Great Blue Heron, Spotted Sandpiper and Marsh Rice Rat Estimated Exposure Areas

Catfish / Blue Crab Sample Area

Sampling Transects

Clams

Clams, Seines (Small Fish), and Infauna

Seines and Infauna

Shoreline Surface Sediment Sample Locations (0-6 inches)

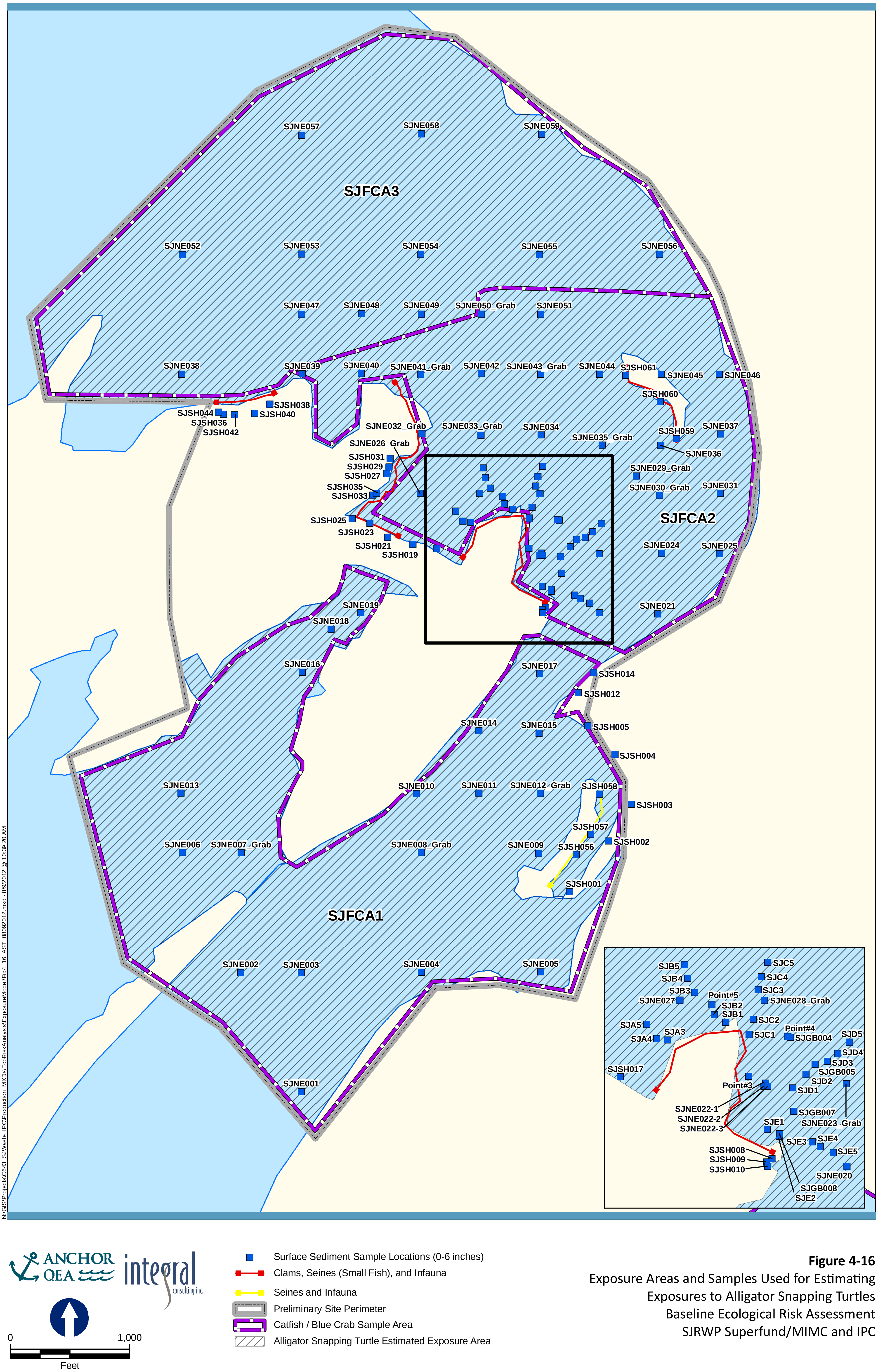
Preliminary Site Perimeter

Figure 4-15

Exposure Areas and Samples Used for Estimating Exposures to Great Blue Herons, Spotted Sandpipers, and Marsh Rice Rats

Baseline Ecological Risk Assessment

SJRWP Superfund/MIMC and IPC



Surface Sediment Sample Locations (0-6 inches)

Clams, Seines (Small Fish), and Infauna

Seines and Infauna

Preliminary Site Perimeter

Catfish / Blue Crab Sample Area

Alligator Snapping Turtle Estimated Exposure Area

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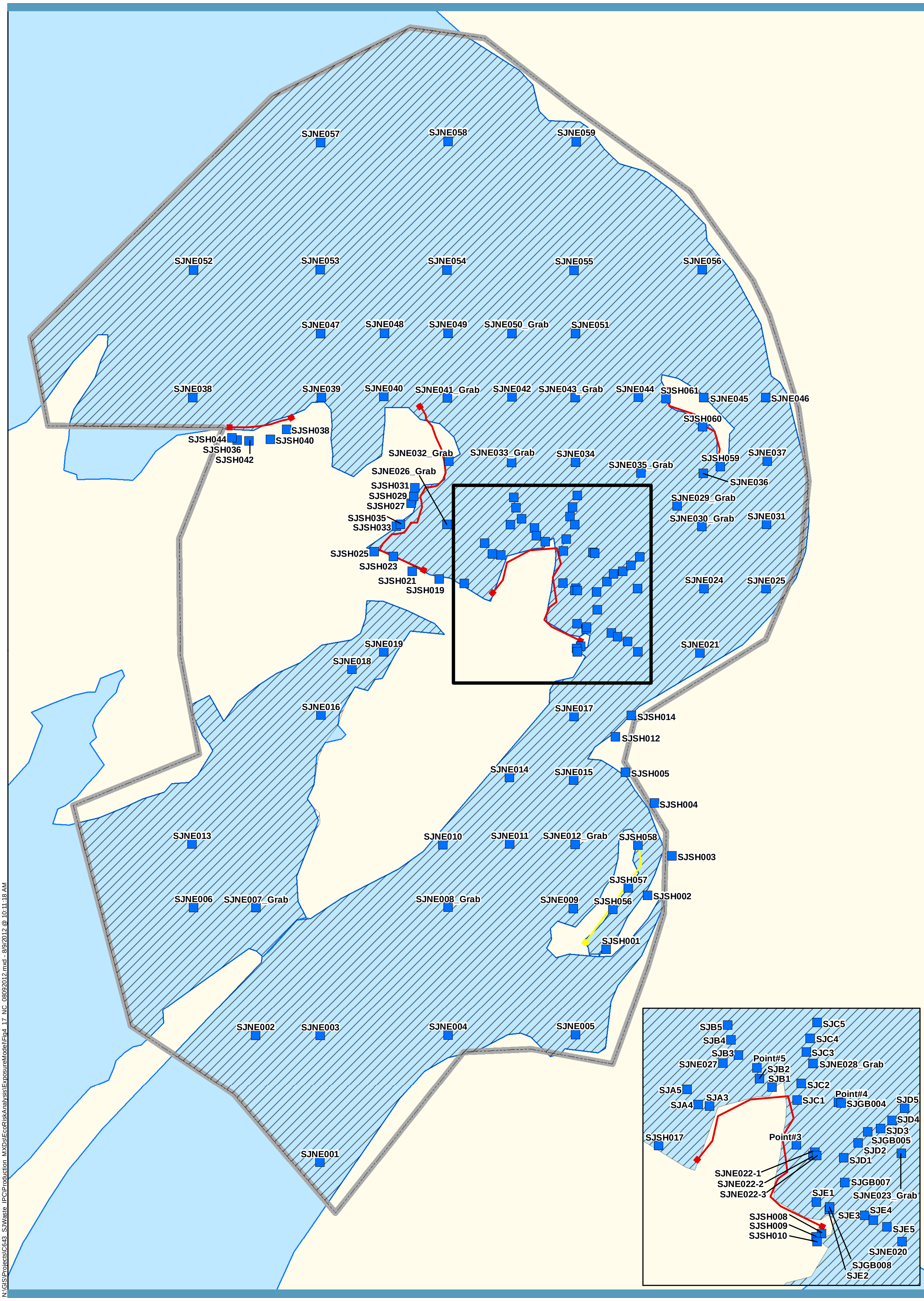
Figure 4-16

Exposure Areas and Samples Used for Estimating Exposures to Alligator Snapping Turtles

Baseline Ecological Risk Assessment

SJRWP Superfund/MIMC and IPC



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Neotropic Cormorant Estimated Exposure Area

Clams, Seines (Small Fish), and Infauna

Seines and Infauna

Surface Sediment Sample Locations (0-6 inches)

Preliminary Site Perimeter

0 1,000

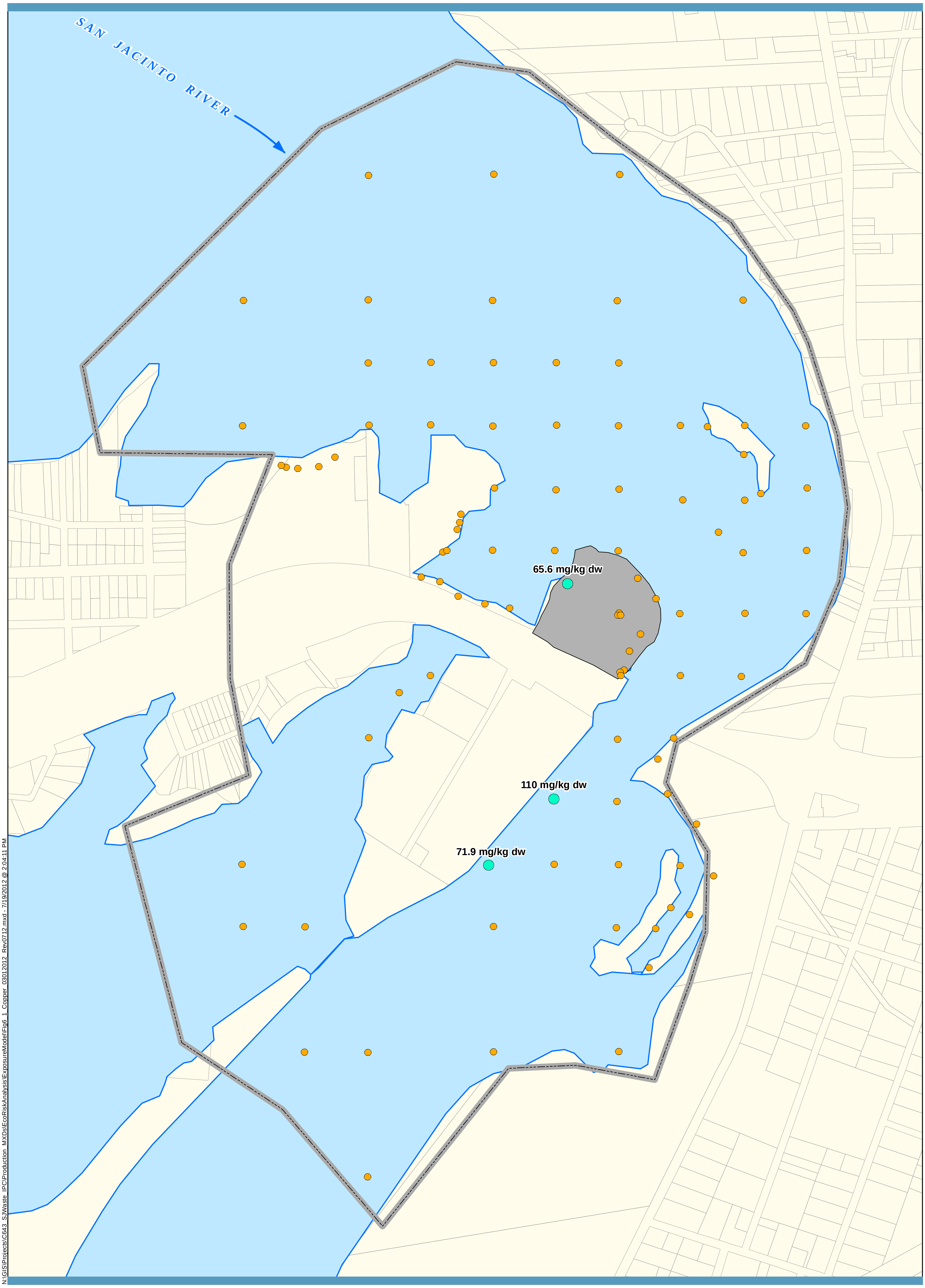
Feet

Figure 4-17

Exposure Areas and Samples Used for Estimating Exposures to Neotropic Cormorants

Baseline Ecological Risk Assessment

SJRWP Superfund/MIMC and IPC



Surface Sediment (0-6 Inches) Sample Location

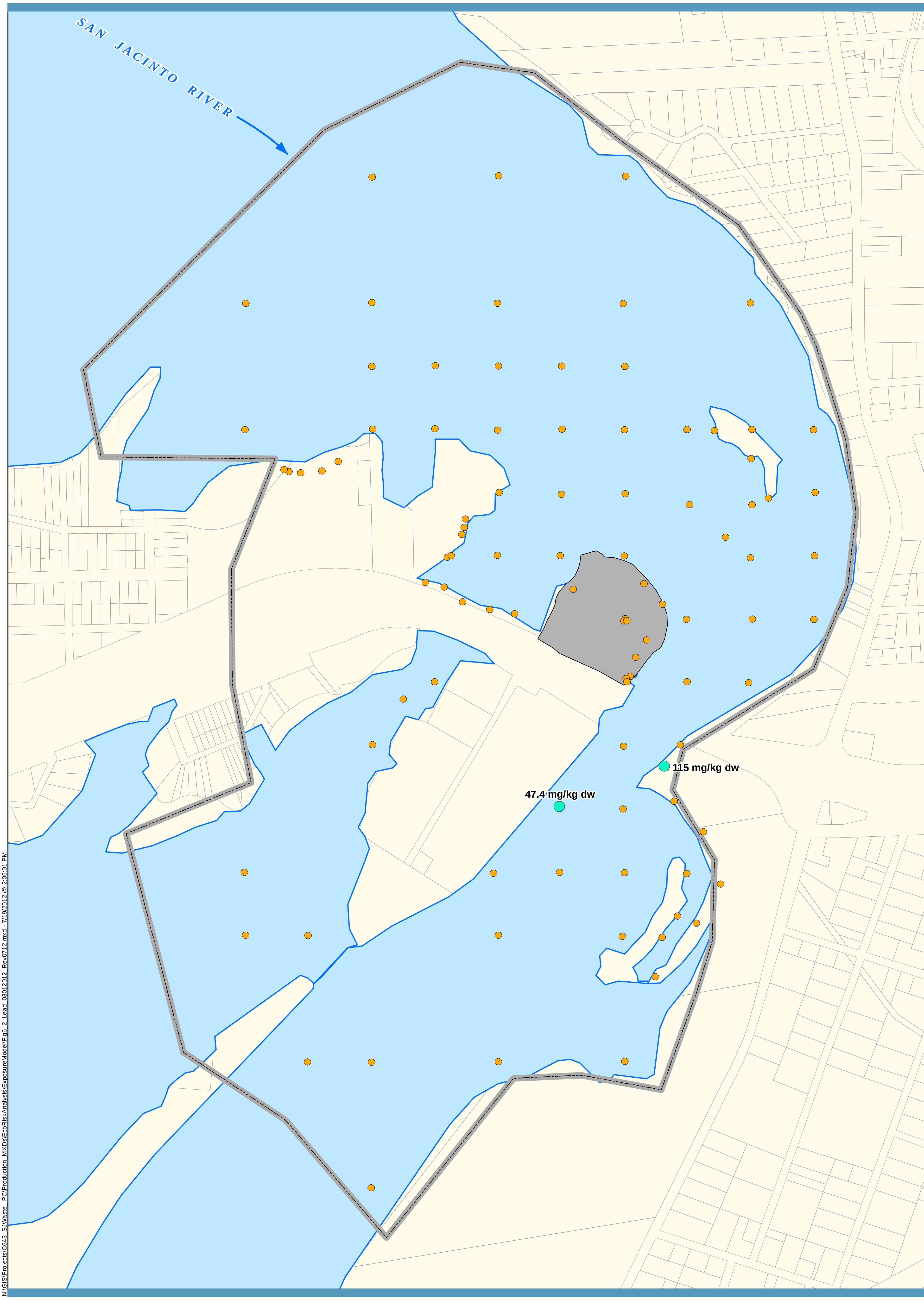
- Does Not Exceed ERL or ERM
- Exceeds ERL (Does Not Exceed ERM)

Copper Effects Range Low (ERL) = 34 mg/kg
Copper Effects Range Medium (ERM) = 270 mg/kg
Concentrations of the chemical are provided if they exceed one or more criteria

- Preliminary Site Perimeter
- Area Within the Original (1966) Perimeter of the North Impoundments

0 1,000
Feet

Figure 6-1
Concentrations of Copper in Sediment Relative to the SQG for Copper (mg/kg)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



ANCHOR QEA
integral consulting inc.

0 1,000
Feet

Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed ERL or ERM

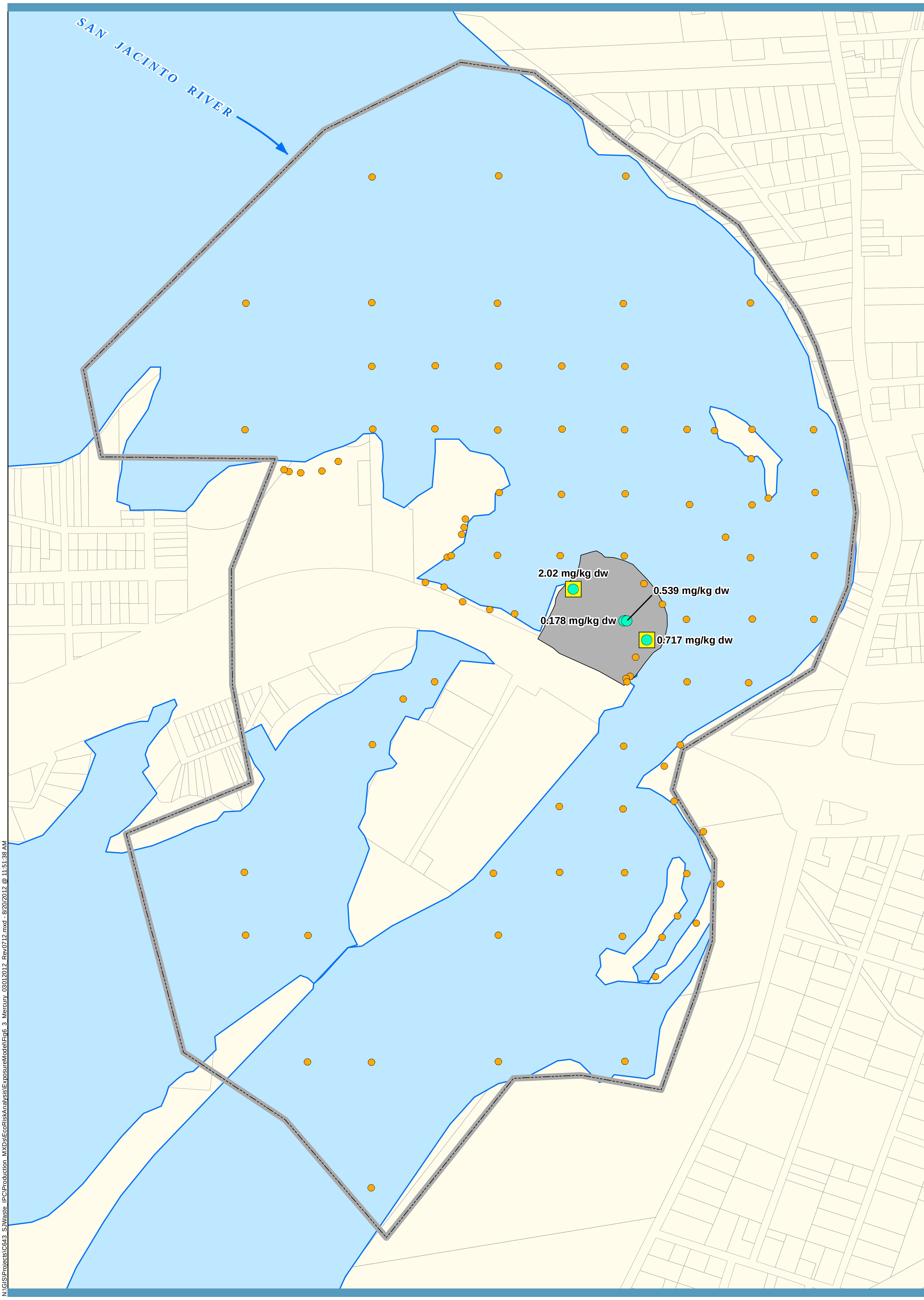
Exceeds ERL (Does Not Exceed ERM)

Lead Effects Range Low (ERL) = 47 mg/kg
Lead Effects Range Medium (ERM) = 218 mg/kg
Concentrations of the chemical are provided if they exceed one or more criteria

Preliminary Site Perimeter

Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-2
Concentrations of Lead in Sediment Relative to the SQG for Lead (mg/kg)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



N:\GIS\Projects\C643_SJWaste_IPC\Production_MXDs\EcoRiskAnalysis\ExposureModel\Fig6_3_Mercury_03012012_Rev0712.mxd - 8/20/2012 @ 11:51:38 AM

ANCHOR QEA

integral consulting inc.

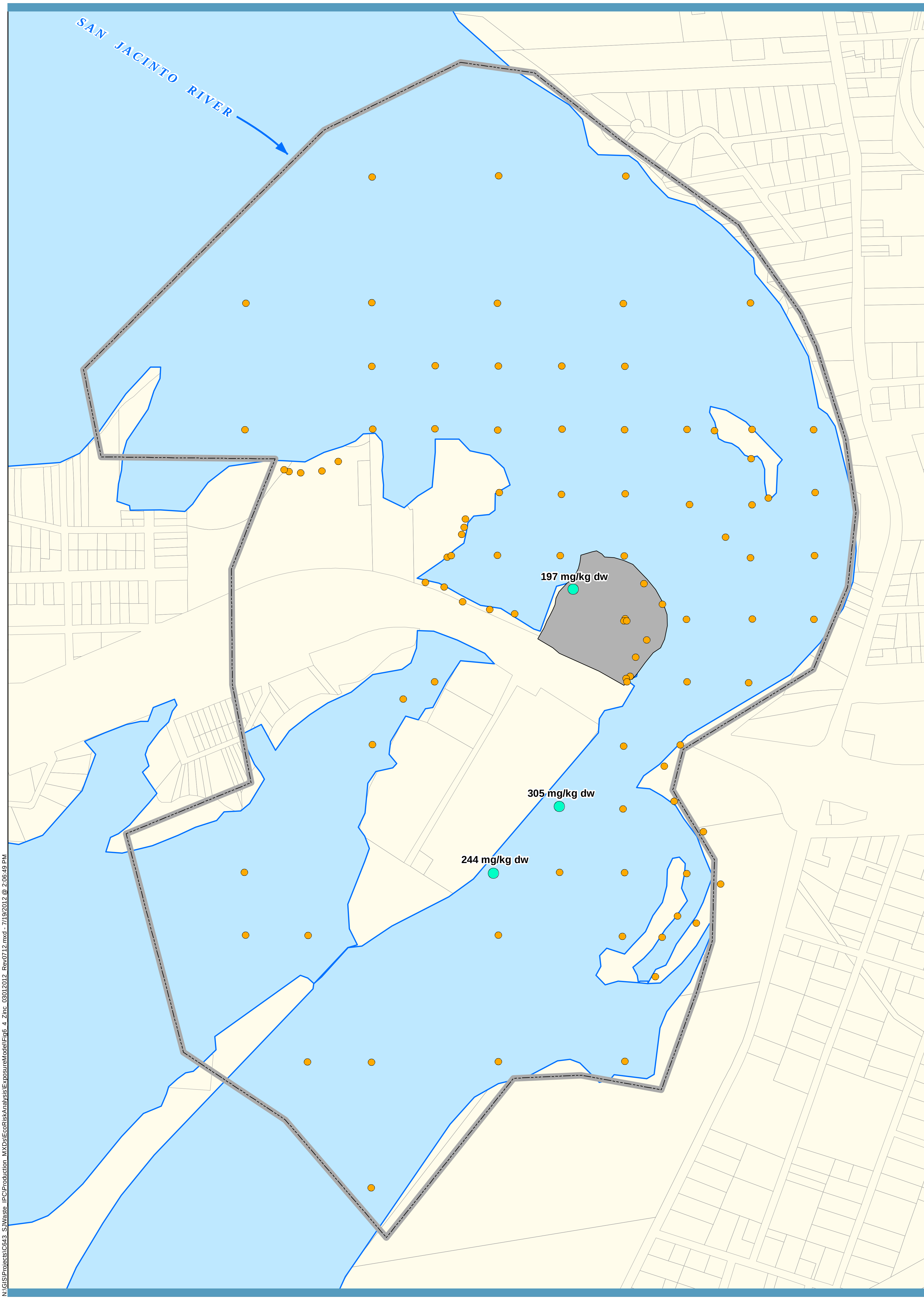
0 1,000
Feet

Surface Sediment (0-6 Inches) Sample Location
- Does Not Exceed ERL or ERM
- Exceeds ERL
- Exceeds ERM

Mercury Effects Range Low (ERL) = 0.15 mg/kg
Mercury Effects Range Median (ERM) = 0.71 mg/kg
Concentrations of the chemical are provided if they exceed one or more criteria

- USEPA's Preliminary Site Perimeter
- Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-3
Concentrations of Mercury in Sediment Relative to the SQG for Mercury (mg/kg)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



ANCHOR QEA
integral consulting inc.

0 1,000
Feet

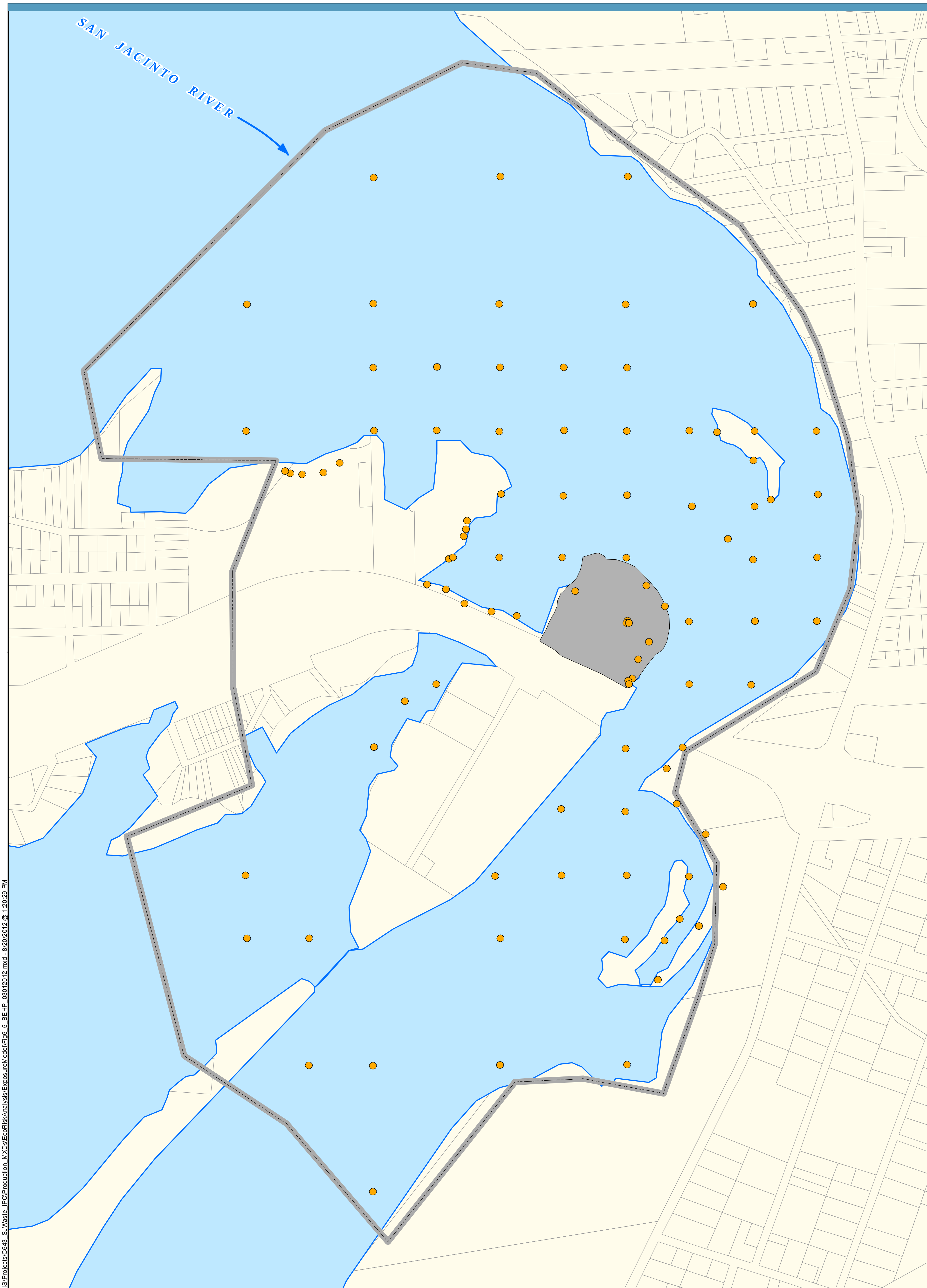
Surface Sediment (0-6 Inches) Sample Location

- Does Not Exceed ERL or ERM
- Exceeds ERL (Does Not Exceed ERM)

Zinc Effects Range Low (ERL) = 150 mg/kg
Zinc Effects Range Medium (ERM) = 410 mg/kg
Concentrations of the chemical are provided if they exceed one or more criteria

- Preliminary Site Perimeter
- Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-4
Concentrations of Zinc in Sediment Relative to the SQG for Zinc (mg/kg)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



ANCHOR QEA

integral consulting inc.

0 1,000

Feet

Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed TRV

Bis(2-ethylhexyl)phthalate Surface Water Toxicity Reference Value (TRV) = 100 µg/L

USEPA's Preliminary Site Perimeter

Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-5

Estimated Porewater Concentrations of Bis(2-ethylhexyl)phthalate (BEHP) Relative to the TRV for BEHP (µg/L)

Baseline Ecological Risk Assessment

SJRWP Superfund/MIMC and IPC

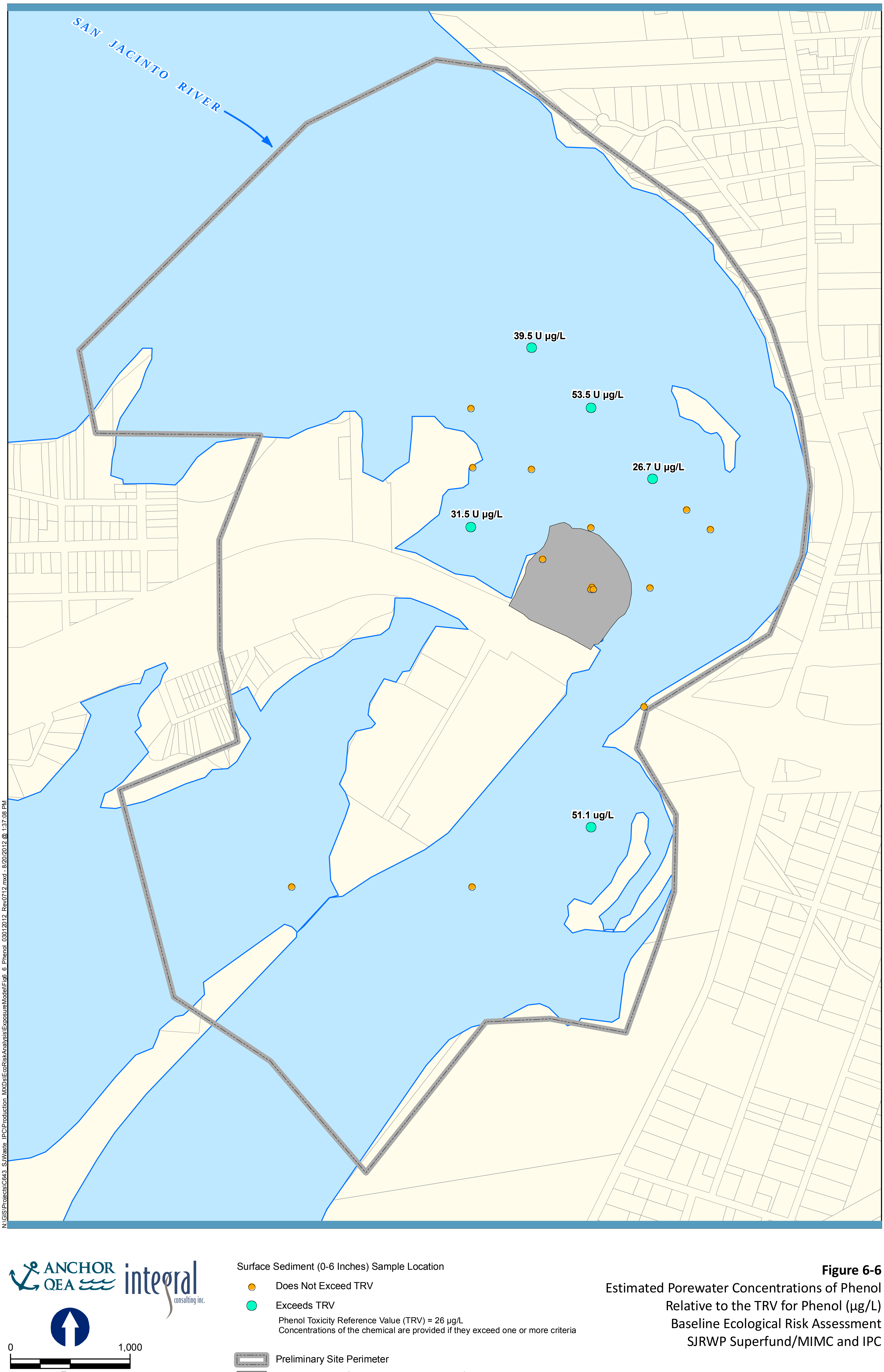
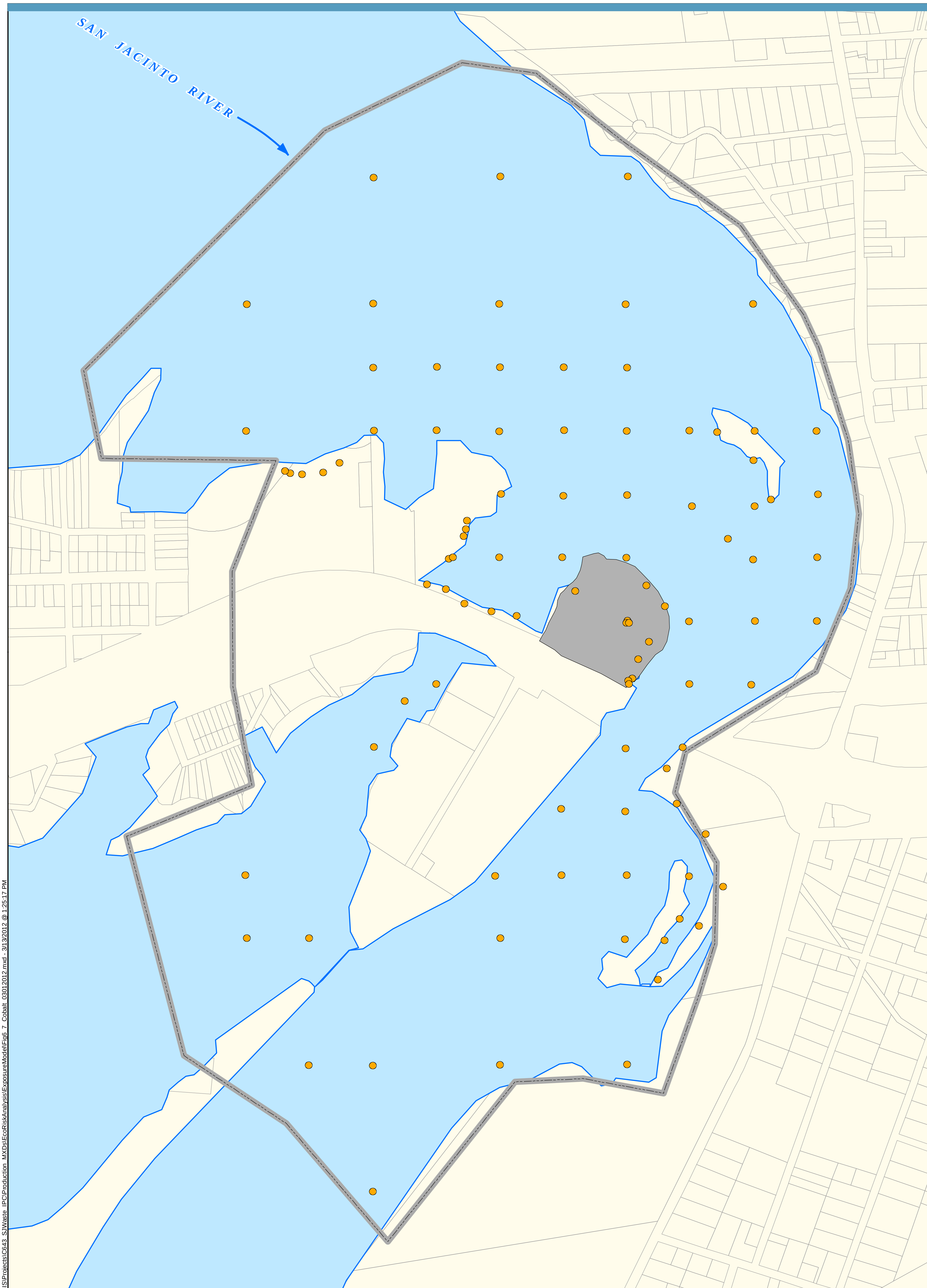


Figure 6-6
Estimated Porewater Concentrations of Phenol Relative to the TRV for Phenol (µg/L)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed TRV

Cobalt Surface Water Toxicity Reference Value (TRV) = 0.45 mg/L

Preliminary Site Perimeter

Area Within the Original (1966) Perimeter of the North Impoundments

0 1,000

Feet

Figure 6-7

Estimated Porewater Concentrations of Cobalt Relative to the TRV for Cobalt (mg/L)

Baseline Ecological Risk Assessment

SJRWP Superfund/MIMC and IPC

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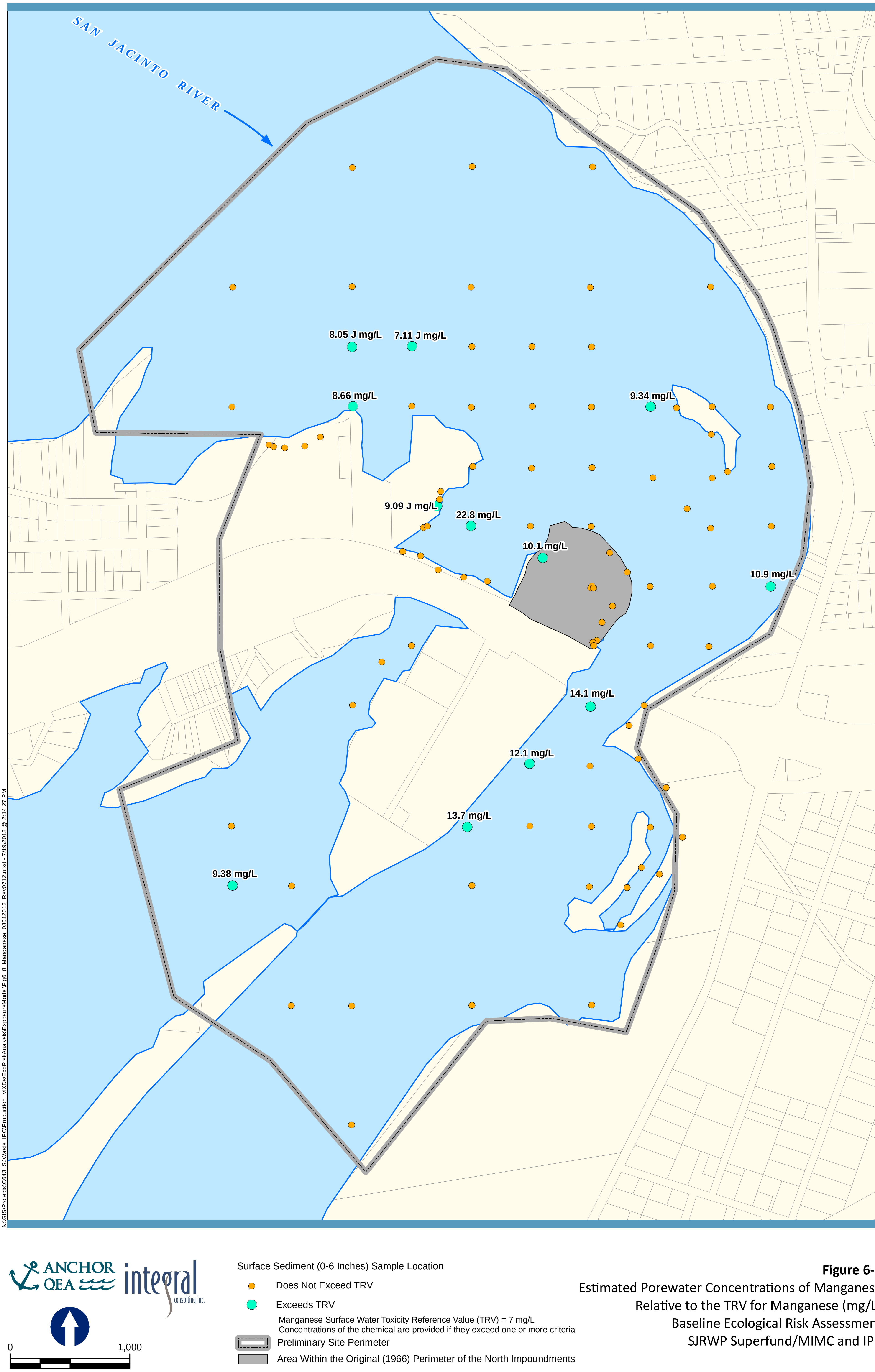

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Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed TRV

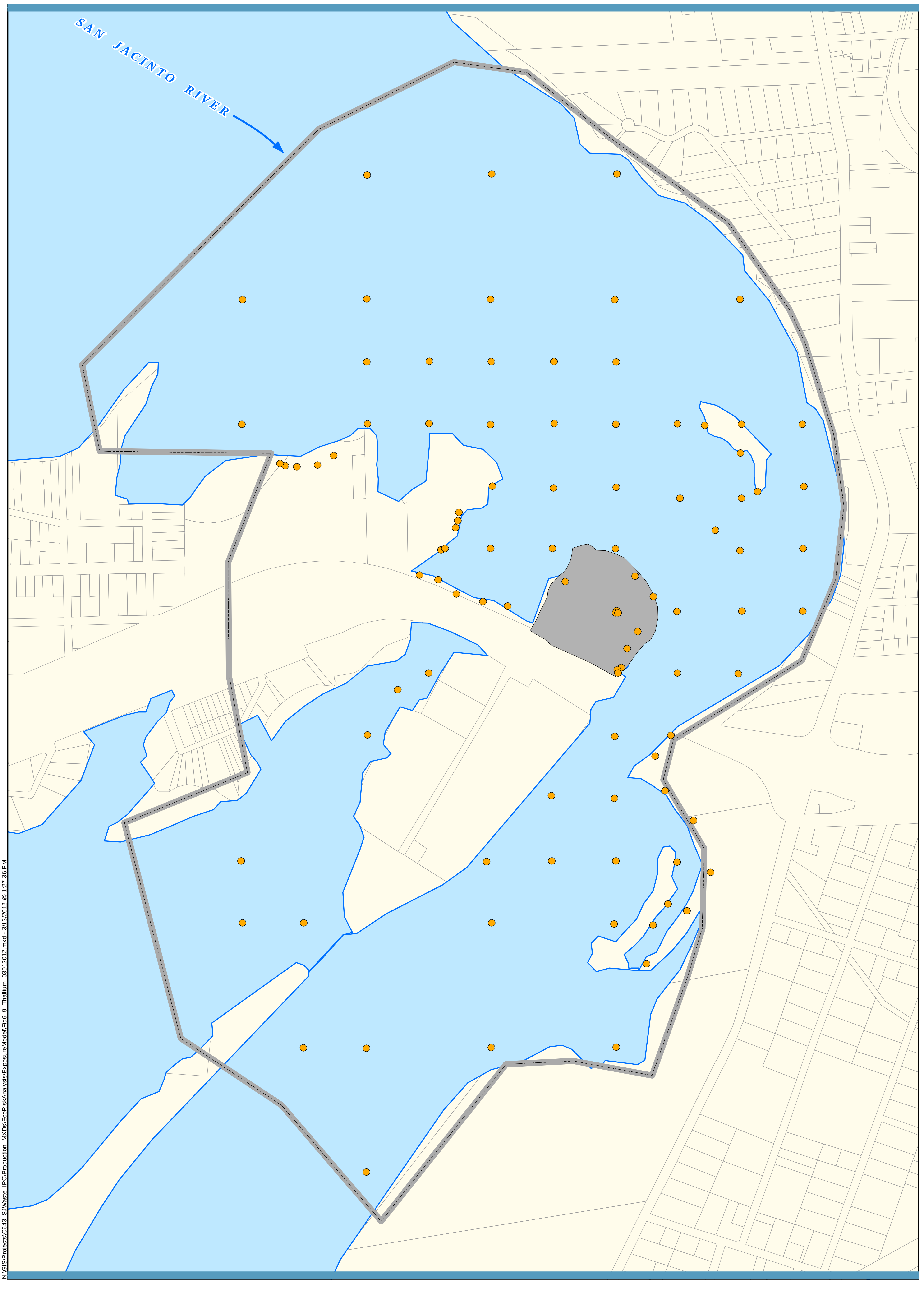
Exceeds TRV

Manganese Surface Water Toxicity Reference Value (TRV) = 7 mg/L
Concentrations of the chemical are provided if they exceed one or more criteria

Preliminary Site Perimeter

Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-8
Estimated Porewater Concentrations of Manganese Relative to the TRV for Manganese (mg/L)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



ANCHOR QEA

integral consulting inc.

0 1,000
Feet

Surface Sediment (0-6 Inches) Sample Location

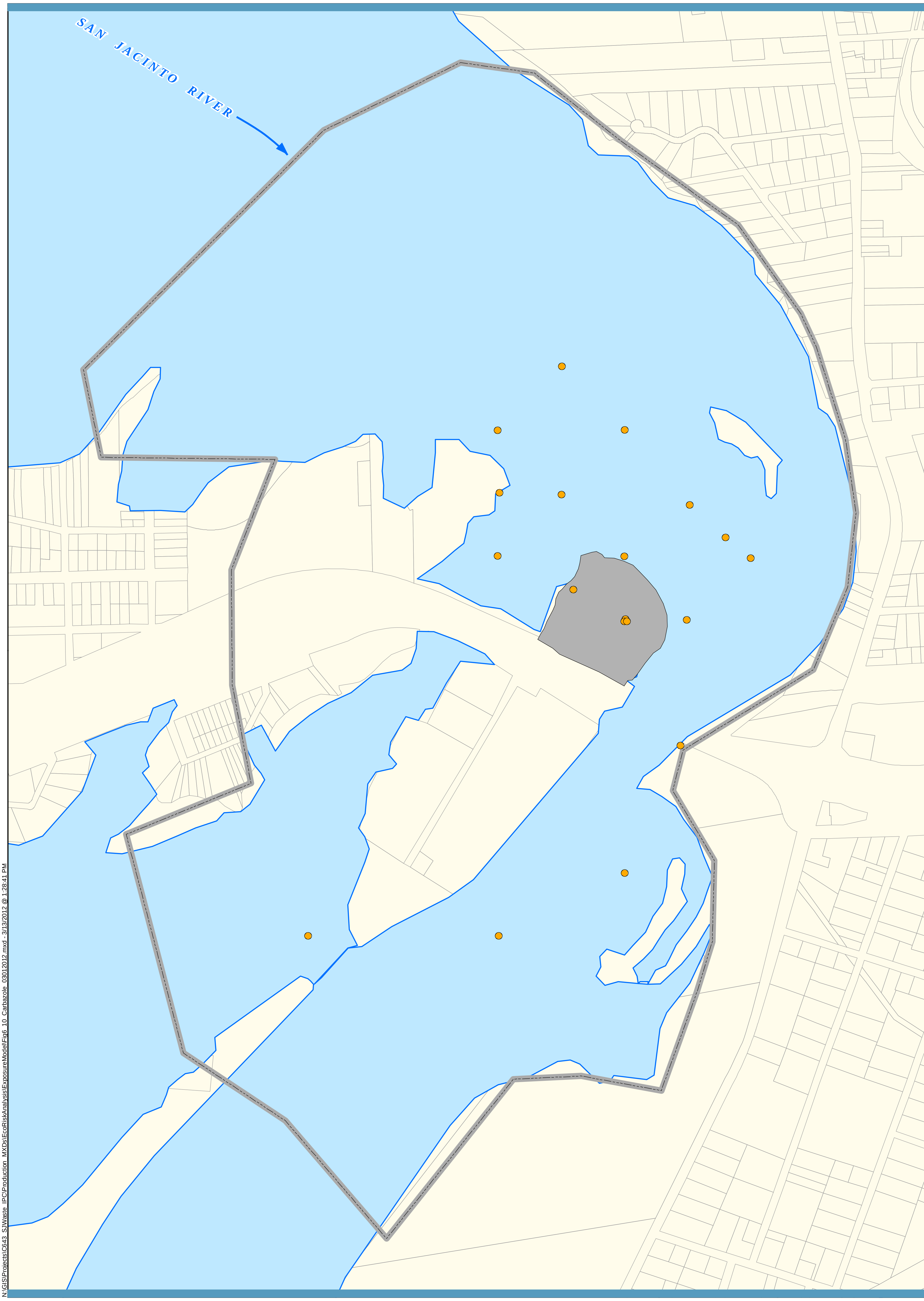
Does Not Exceed TRV

Thallium Surface Water Toxicity Reference Value (TRV) = 0.213 mg/L

Preliminary Site Perimeter

Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-9
Estimated Porewater Concentrations of Thallium Relative to the TRV for Thallium (mg/L)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



ANCHOR QEA

integral consulting inc.

0 1,000
Feet

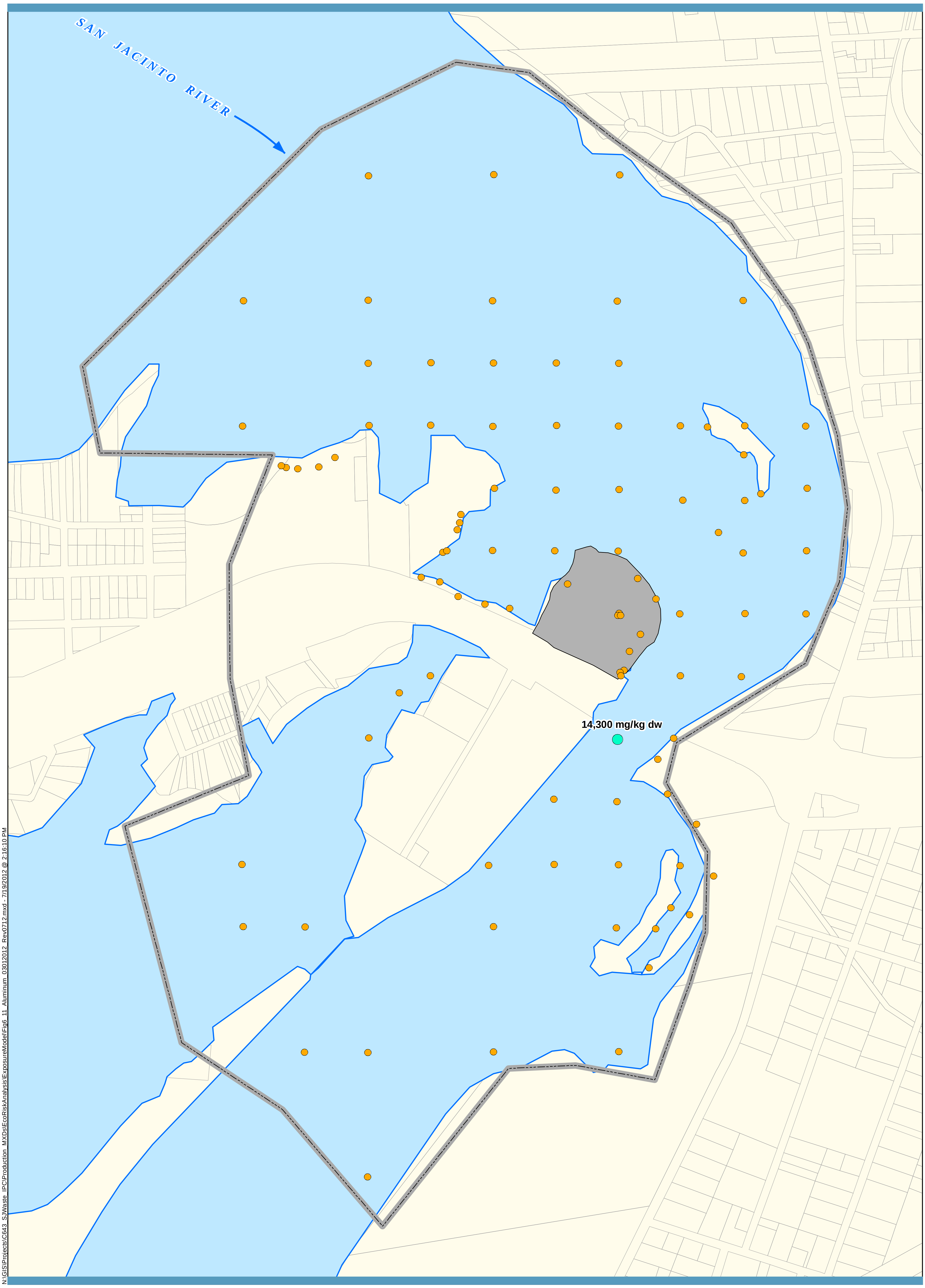
Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed Maximum Upstream Detection Limit
Carbazole Maximum Upstream Detection Limit = 23,300 ug/kg oc

Preliminary Site Perimeter

Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-10
Concentrations of Carbazole in Sediment Relative to the Upstream Maximum Detection Limit for Carbazole
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



ANCHOR QEA

integral consulting inc.

0 1,000

Feet

Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed REV

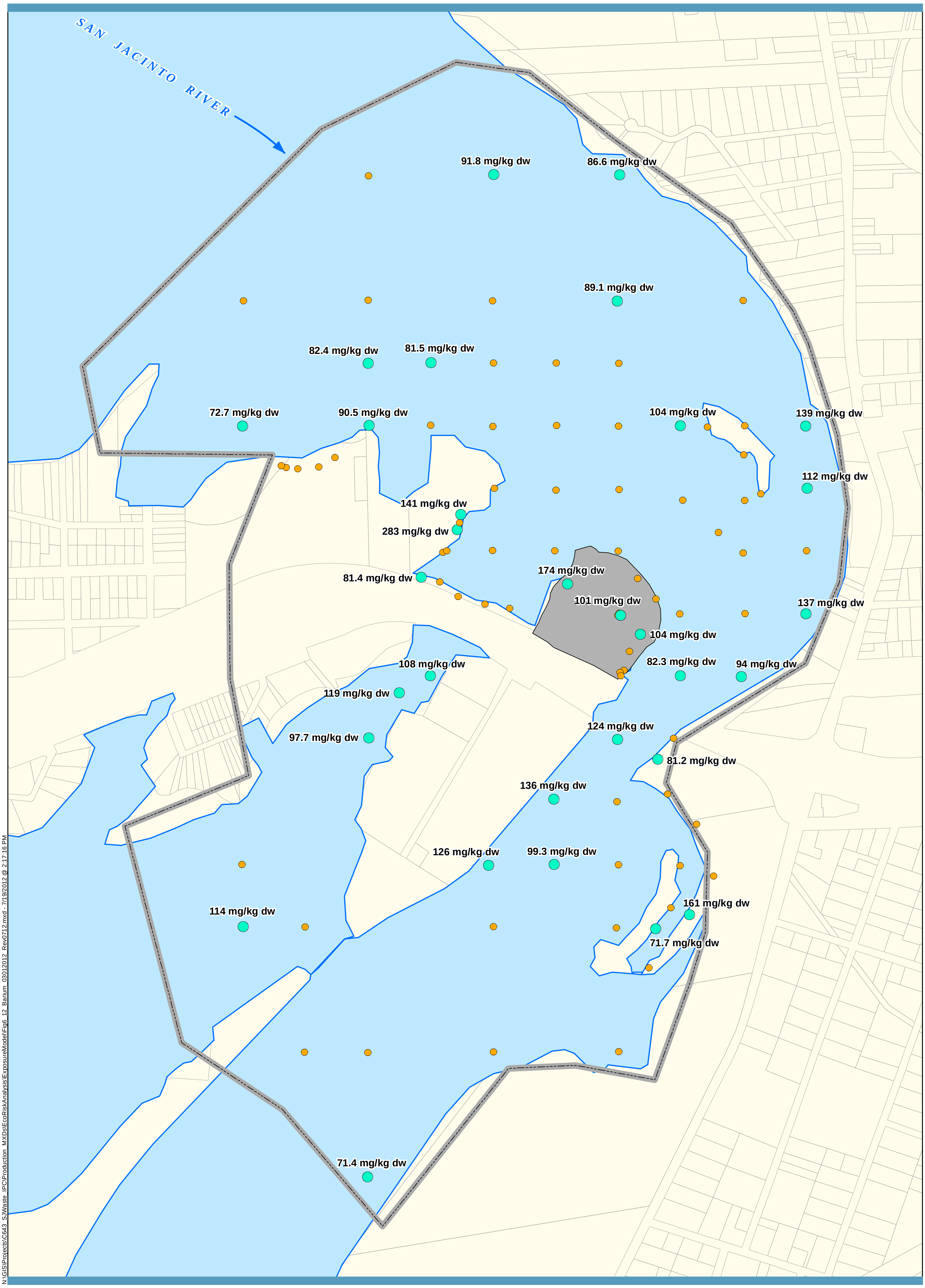
Exceeds REV

Aluminum Reference Envelope Value (REV) = 13,300 mg/kg
Concentrations of the chemical are provided if they exceed one or more criteria

Preliminary Site Perimeter

Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-11
Concentrations of Aluminum in Sediment Relative to the Upstream Reference Envelope Value for Aluminum
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



ANCHOR QEA

integral consulting inc.

0 1,000

Feet

Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed REV

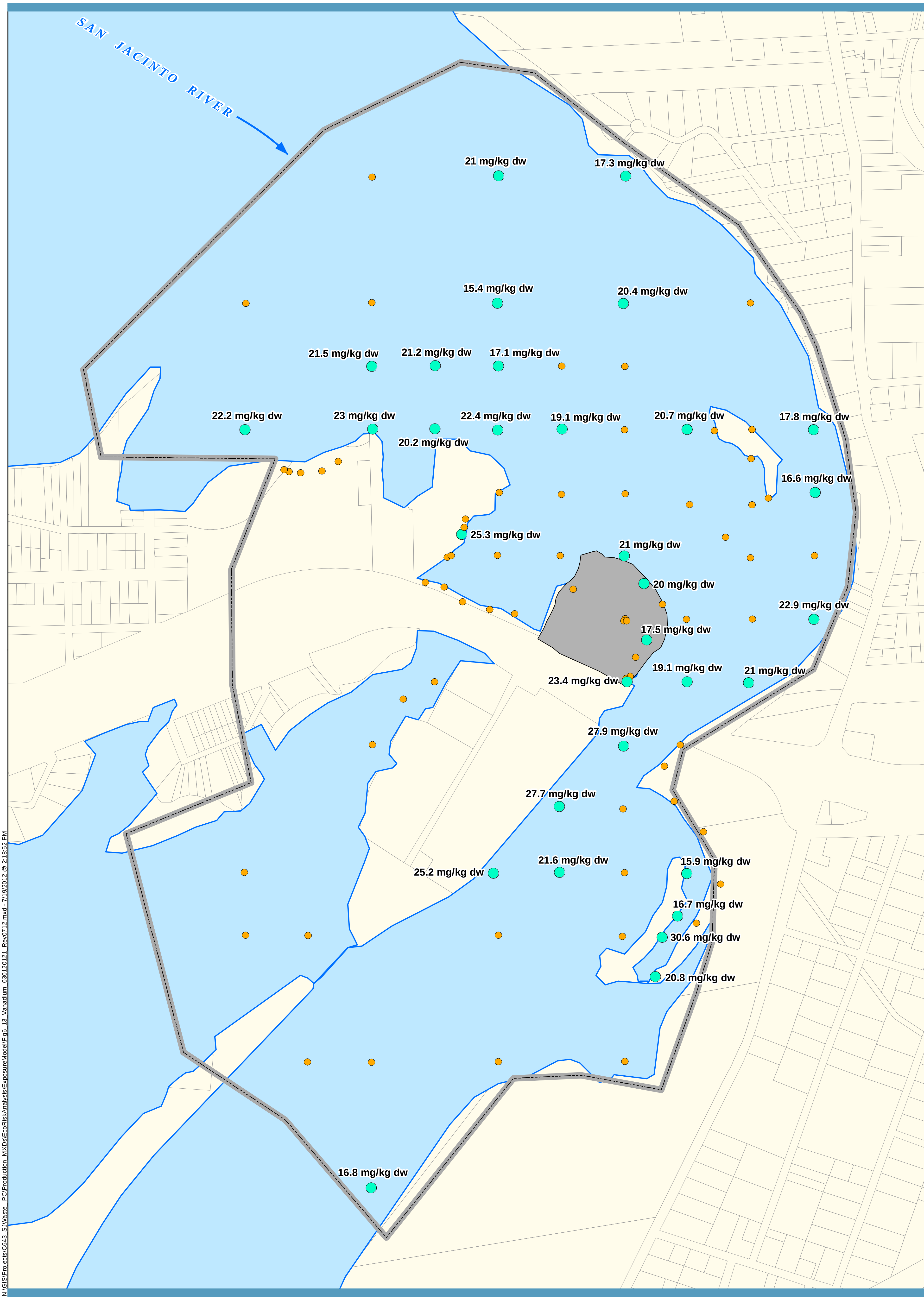
Exceeds REV

Barium Reference Envelope Value (REV) = 69.8 mg/kg
Concentrations of the chemical are provided if they exceed one or more criteria

Preliminary Site Perimeter

Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-12
Concentrations of Barium in Sediment Relative to the Upstream Reference Envelope Value for Barium
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



ANCHOR QEA

integral consulting inc.

0 1,000

Feet

Surface Sediment (0-6 Inches) Sample Location

- Does Not Exceed REV
- Exceeds REV

Vanadium Upstream Reference Envelope Value (REV) = 15.2 mg/kg
Concentrations of the chemical are provided if they exceed one or more criteria

- Preliminary Site Perimeter
- Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-13
Concentrations of Vanadium in Sediment Relative to the Upstream Reference Envelope Value for Vanadium
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

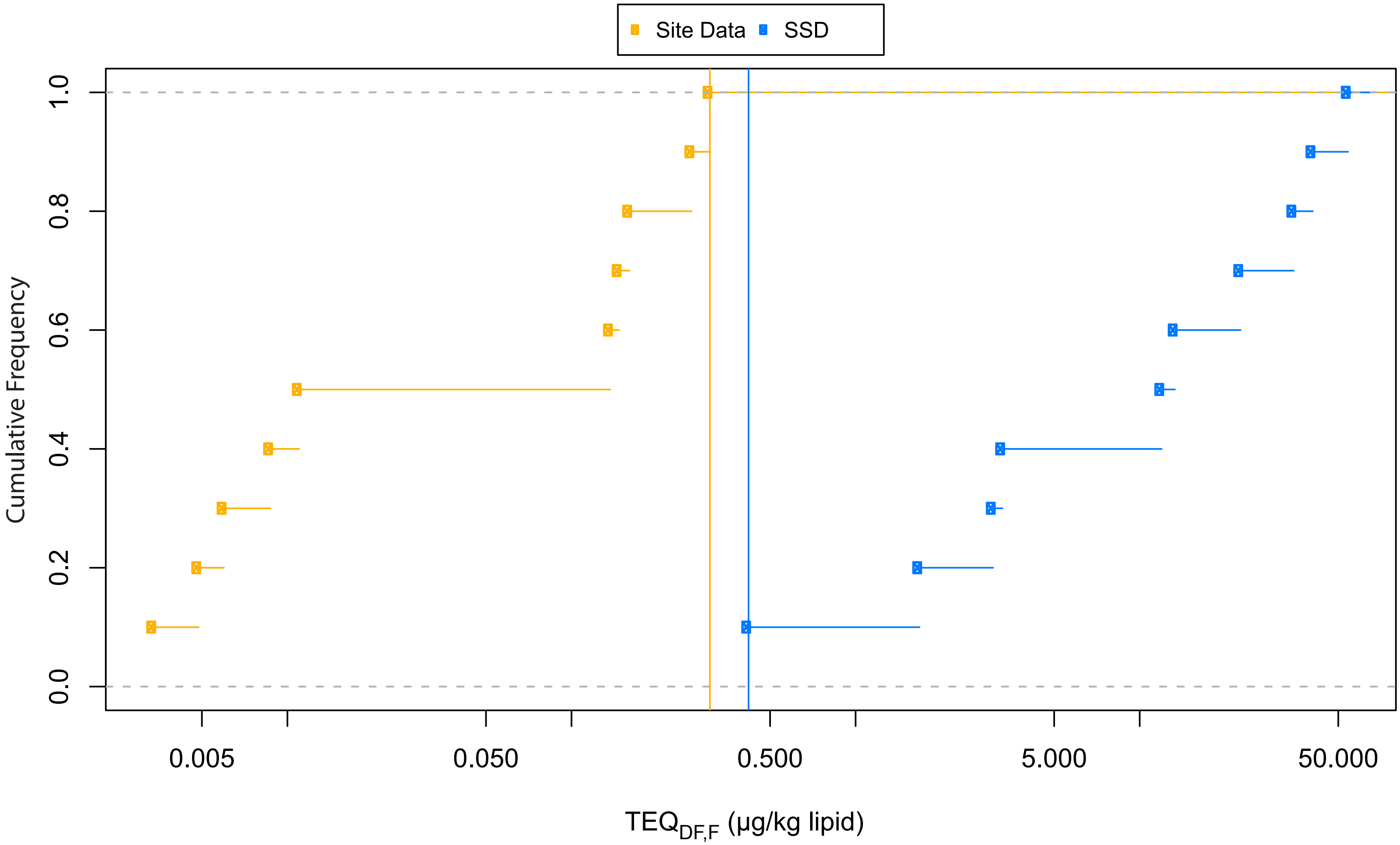
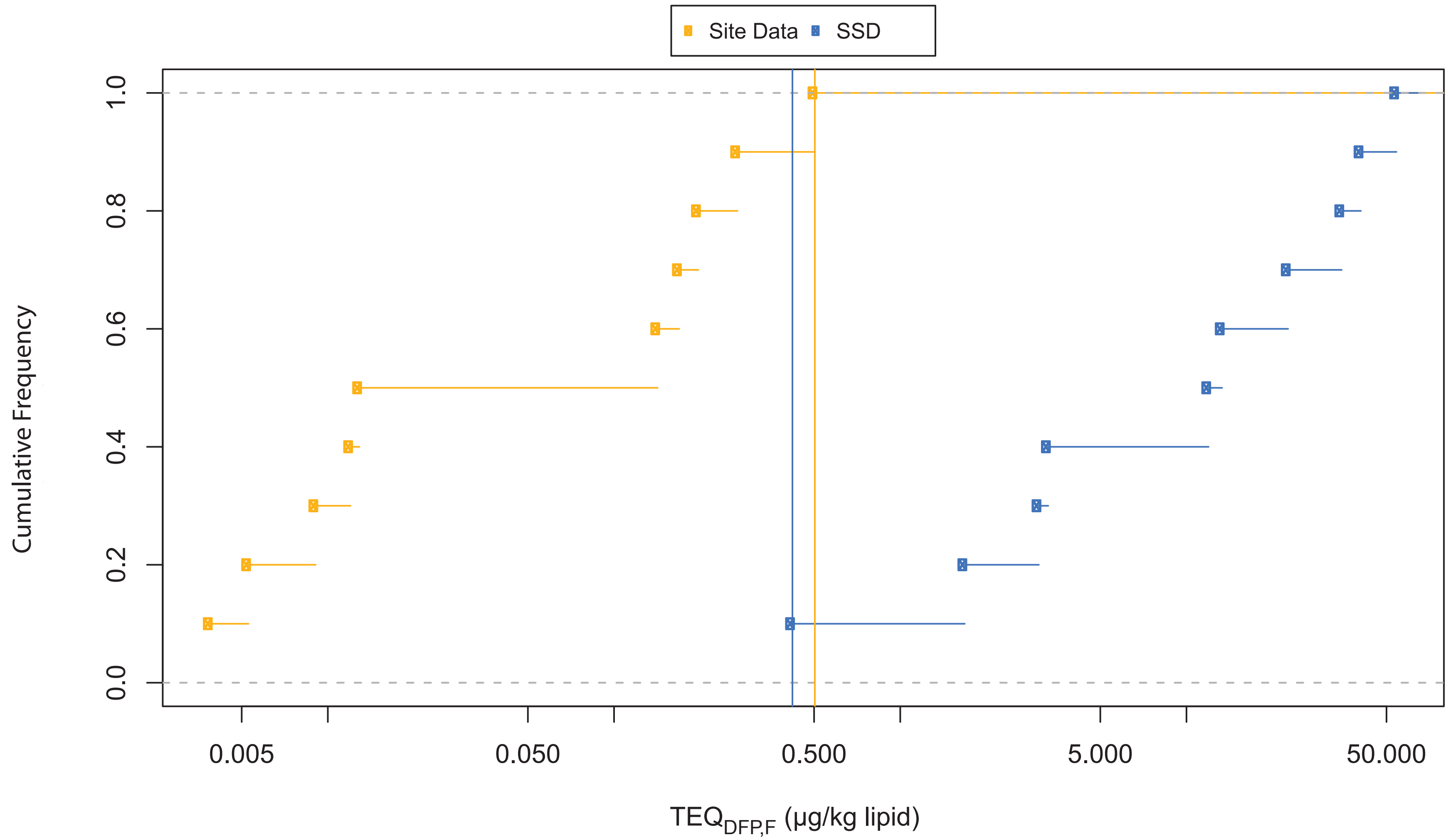
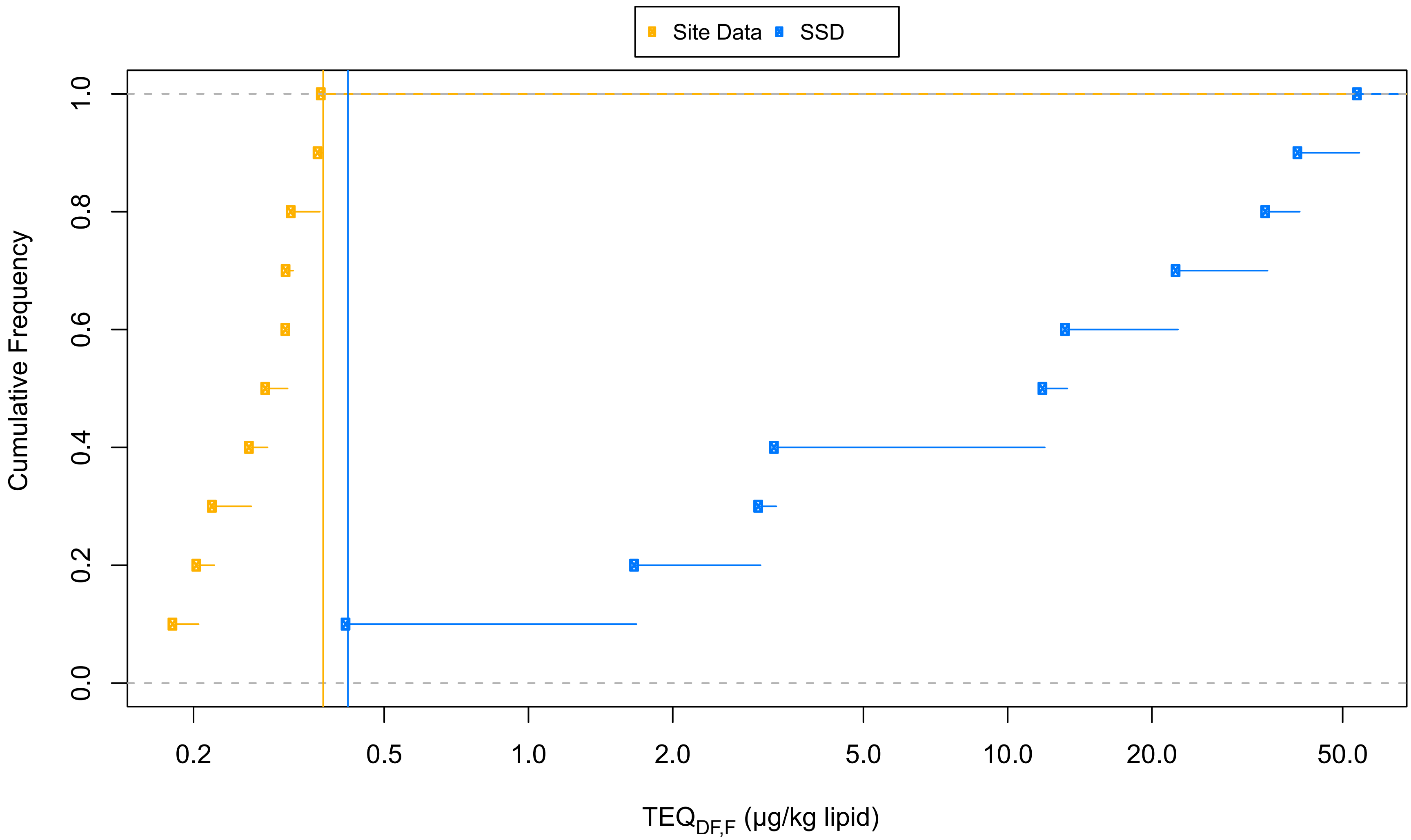


Figure 6-14
Comparison of $TEQ_{DF,F}$ in Gulf Killifish with the Species Sensitivity Distribution
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



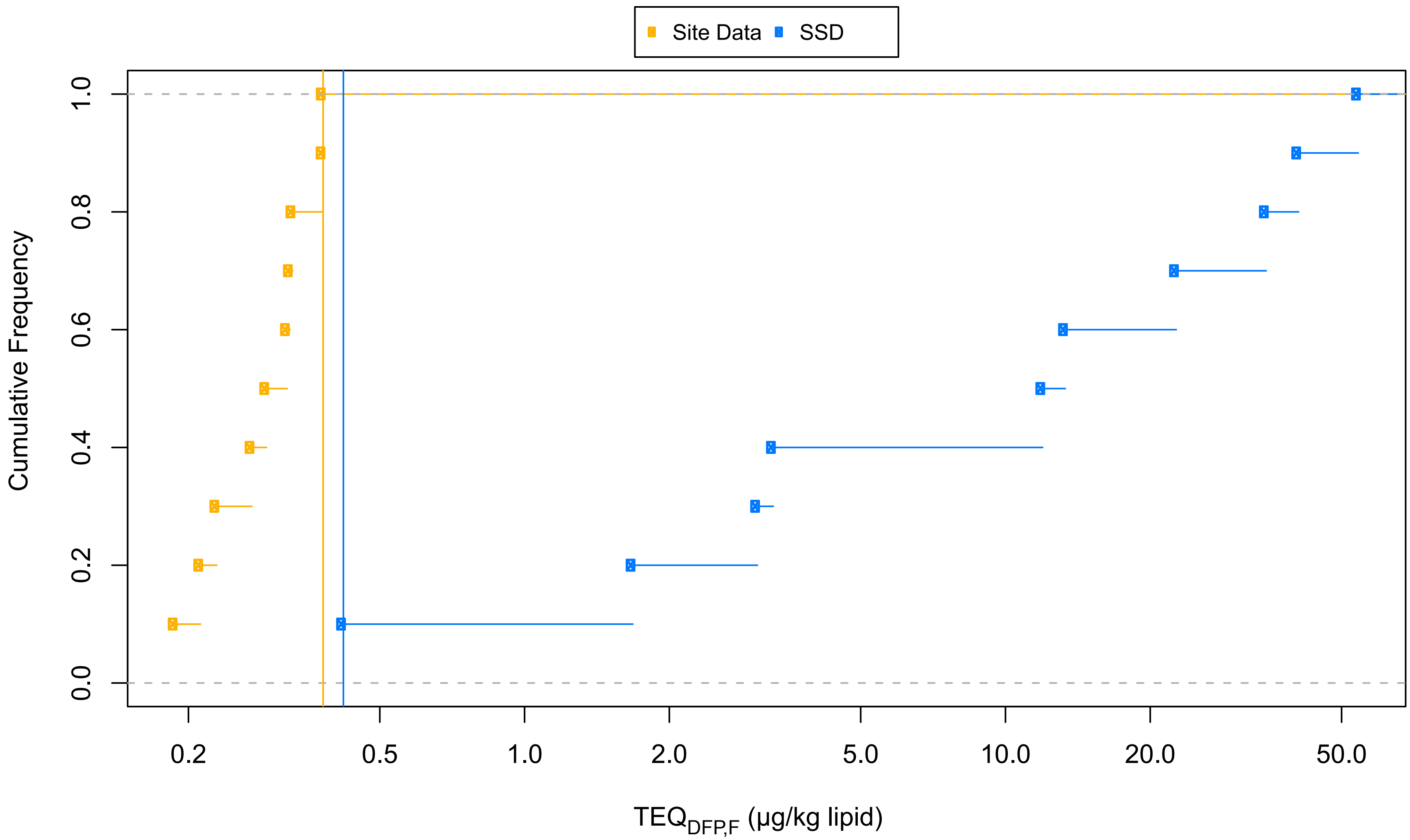
Note:
Lines adjacent to data points indicate distance to the next data point in the distribution.

Figure 6-15
Comparison of $TEQ_{DFP,F}$ in Gulf Killifish with the Species Sensitivity Distribution
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



Note:
Lines adjacent to data points indicate distance to the next data point in the distribution.

Figure 6-16
Comparison of $TEQ_{DF,F}$ in Hardhead Catfish with the Species Sensitivity Distribution
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



Note:
Lines adjacent to data points indicate distance to the next data point in the distribution.

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Figure 6-17
Comparison of TEQ$_{DFP,F}$ in Hardhead Catfish with the Species Sensitivity Distribution
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

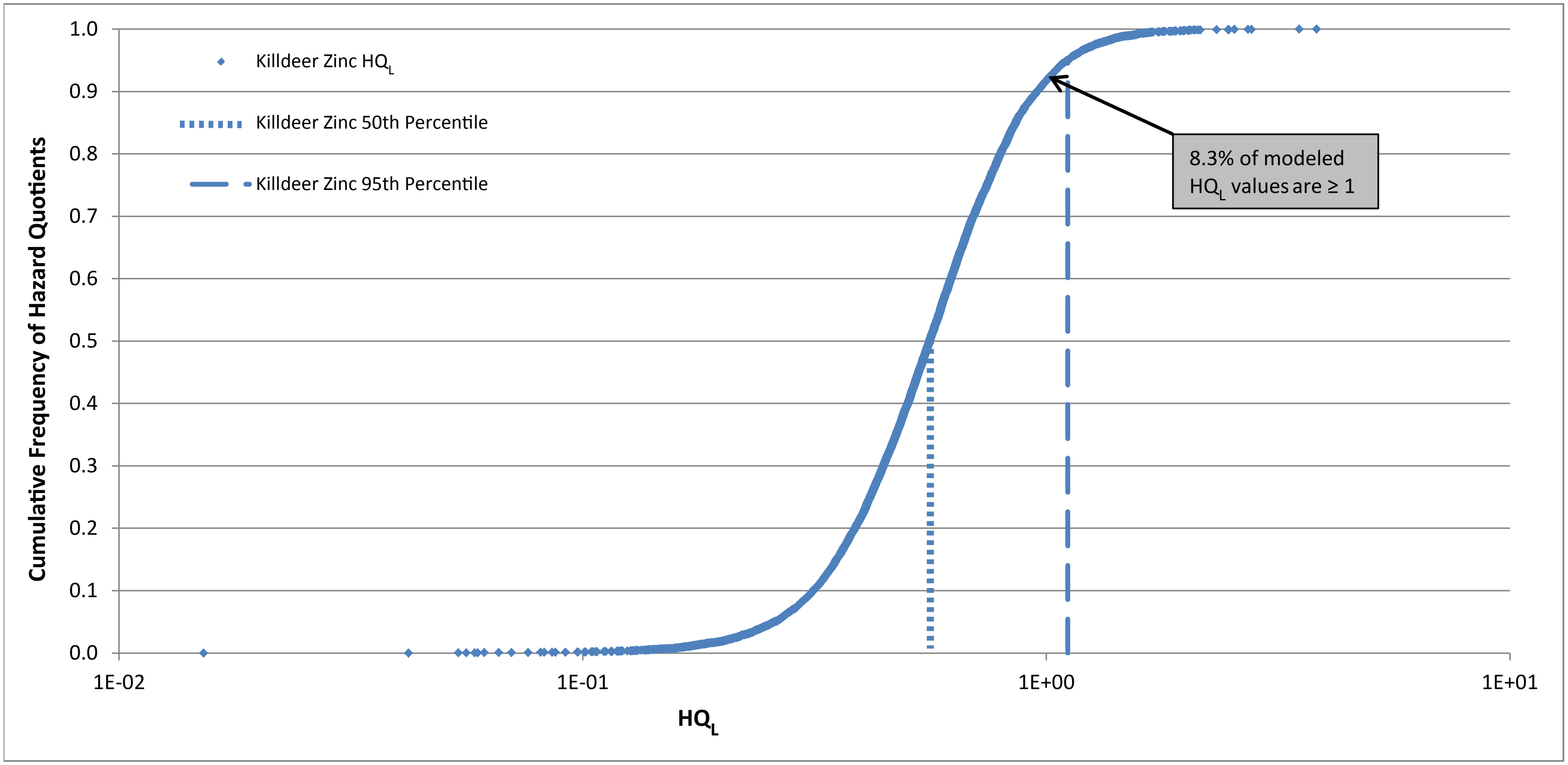

Figure 6-18
Probability Distributions of Zinc Hazard Quotients (HQ_L) for Killdeer
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

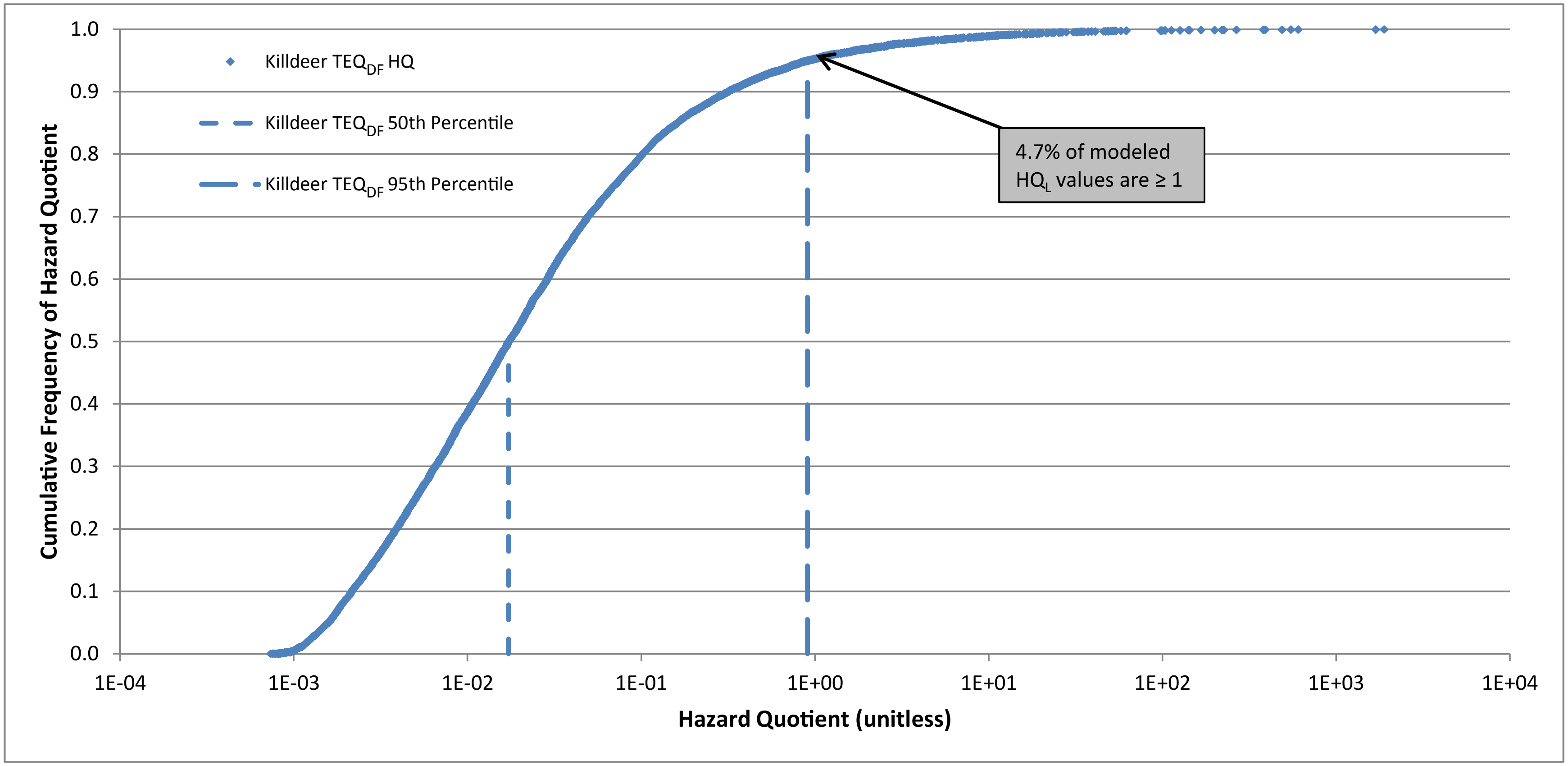

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Figure 6-19
Probability Distributions of TEQ$_{DF,B}$ Hazard Quotients (HQ$_L$) for Killdeer
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

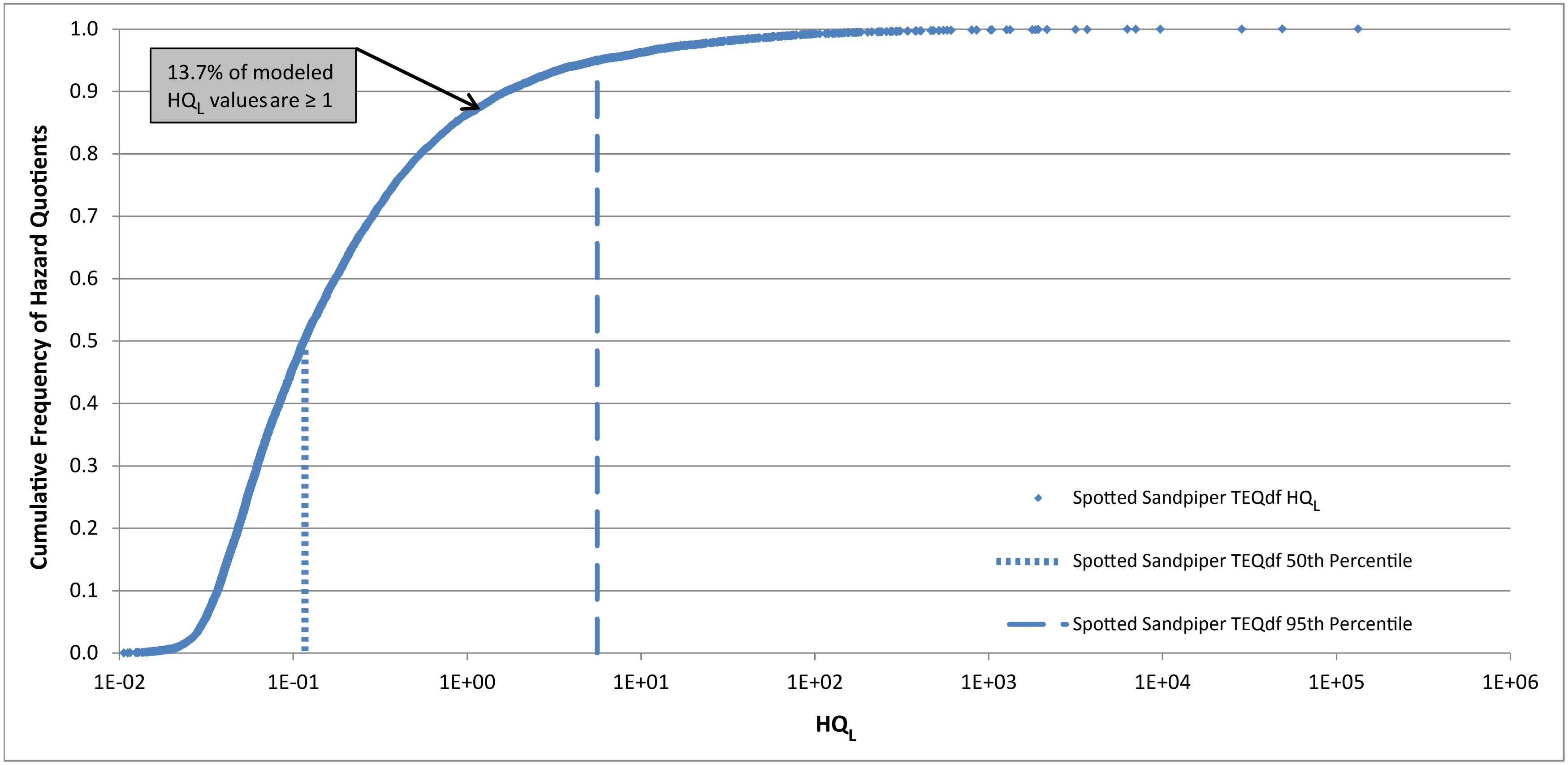

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Figure 6-20
Probability Distributions of $TEQ_{DF,B}$ Hazard Quotients for Spotted Sandpiper
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

ANCHOR QEA
integral consulting inc.

0 1,000
Feet

Surface Soil Sample Location (0 - 6 inches)

- 0.7 - 82.5
- 82.5 - 164.4
- 164.4 - 246.2
- 246.2 - 328
- Preliminary Site Perimeter
- Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-21
Concentrations of Zinc (mg/kg) in
Surface Soils North of I-10
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

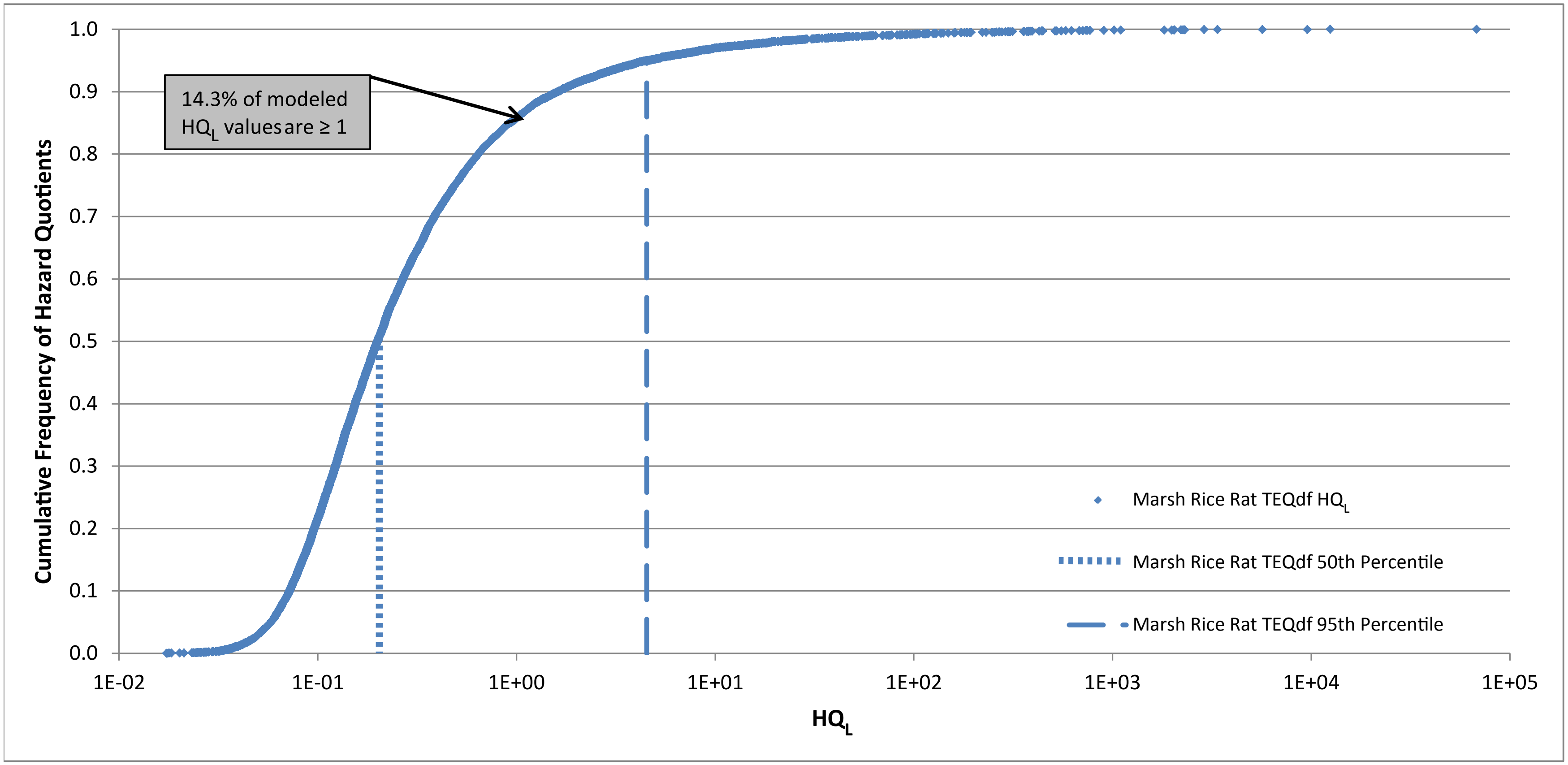

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Figure 6-22
Probability Distributions of $TEQ_{DF,M}$ Hazard Quotients for Marsh Rice Rat
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

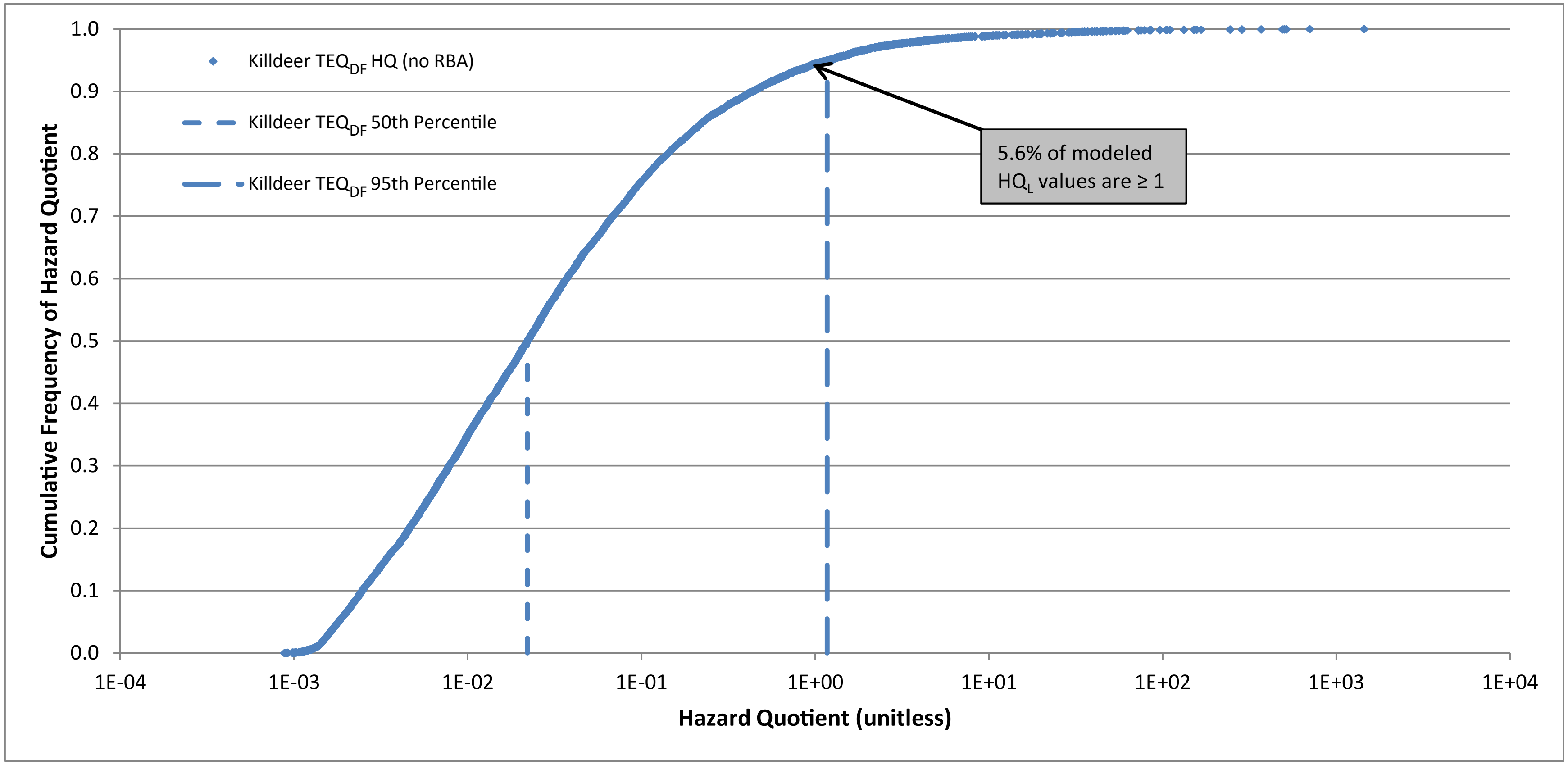

Figure 7-1
Probability Distributions of $TEQ_{DF,B}$ Hazard Quotients (HQ_L) with
No Relative Bioavailability Adjustment for Killdeer
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

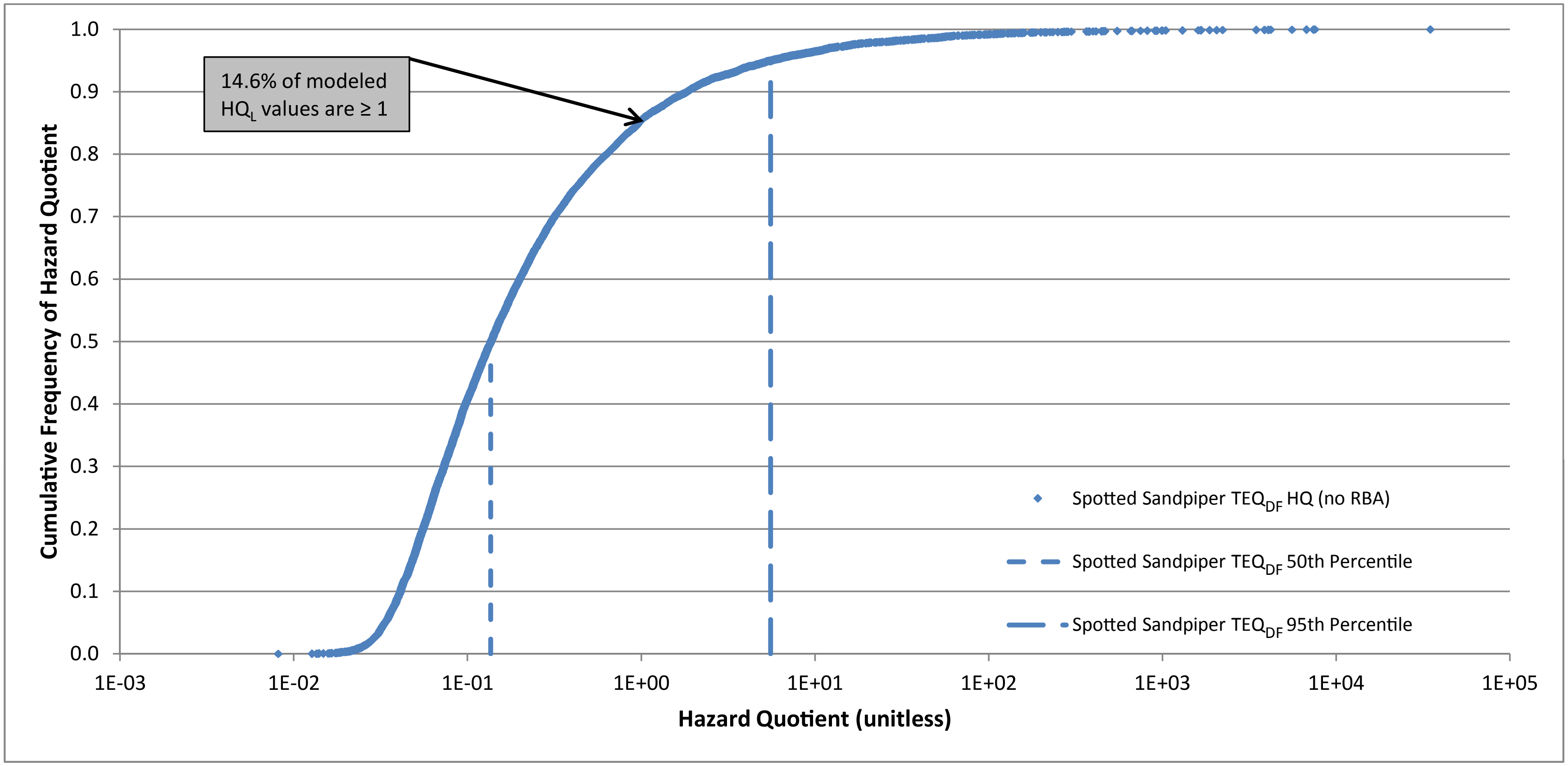

Figure 7-2
Probability Distributions of $TEQ_{DF,B}$ Hazard Quotients (HQ_L) with
No Relative Bioavailability Adjustment for Spotted Sandpiper
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

EXHIBIT 1
FOOD WEB EXPOSURE MODEL AND EXAMPLE OF ITS APPLICATION FOR SPOTTED SANDPIPER EXPOSURE TO DIOXINS AND FURANS

Exhibit 1

Food Web Exposure Model and Example of Its Application for Spotted Sandpiper Exposure to Dioxins and Furans

This exhibit illustrates how estimates of exposure of site receptors to a given $COPC_E$ were calculated. The example provided in the supporting tables is for spotted sandpiper exposure to dioxins and furans.

To estimate the cumulative daily dose for reptiles, mammals, and birds through ingestion of food and water, including incidental soil or sediment ingestion, the following general equation was used, as described in the BERA:

$$\text{Daily Dose} = \left((\text{FIR} \times C_{food} \times RBA_{food}) + (\text{WIR} \times C_{water}) + (\text{SIR} \times C_{sed} \times RBA_{sed})\right) \times \text{AUF} \quad \text{(Eq. 4-5)}$$

Where:

Daily Dose	=	$COPC_E$s ingested per day via food, water, and sediment (mg/kg bw day)
FIR	=	food ingestion rate (kg food dw/kg bw day)
C_{food}	=	concentration in the overall diet (mg/kg food dw)
RBA_{food}	=	bioavailable fraction absorbed from ingested prey items (unitless); set to 1 except as described below
WIR	=	water ingestion rate (L water/kg bw day)
C_{water}	=	concentration in water (mg/L water)
SIR	=	sediment ingestion rate (kg sediment dw/kg bw day)
C_{sed}	=	concentration in sediment (mg/kg dw)
RBA_{sed}	=	bioavailable fraction absorbed from ingested sediment or soil (unitless); set to one except as described below
AUF	=	area use factor (unitless); fraction of time that a receptor spends at the site relative to the entire home range

Table 1 provides an example of the parameters described in the above equation for the spotted sandpiper receptor. These parameters are then combined with site data (Table 2) using the equation above to illustrate how an estimated daily dose is calculated for $TEQ_{DF,B}$ to spotted sandpipers at the site (Table 3).

The relative bioavailability adjustment factors, RBA_s and $RBA_{food,}$ are both set to equal 1, assuming complete bioavailability, except for the dioxin congener 2,3,7,8-TCDD. For this congener, an RBA_{sed} of 0.41 and RBA_{food} of 0.44, for invertebrate food items only, as described in Section 4.3.1.2 of the BERA, are applied to concentrations of 2,3,7,8-TCDD in the database prior to calculation of the $TEQ_{DF,B}$.

Exhibit 1

Food Web Exposure Model and Example of Its Application for Spotted Sandpiper Exposure to Dioxins and Furans

Table 1. Estimated Daily Dose of $TEQ_{DF,B}$[a] to Spotted Sandpiper

	Dose from Food	Dose from Water	Dose from Sediment	Total Dose
Equation	$FIR \times \Sigma C_{food}$[b]	$WIR \times C_{water}$	$FIR \times F_{sed} \times C_{sed}$	$[\ (FIR \times \Sigma C_{food}) + (WIR \times C_{water}) + (FIR \times F_{sed} \times C_{sed})\ ] \times AUF$
Units	mg/kg bw-day	mg/kg bw-day	mg/kg bw-day	mg/kg bw-day
$TEQ_{DF,B}$	1.12E-05	4.40E-09	1.36E-04	1.47E-04

[a]Toxicity equivalent for dioxins and furans calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit.

[b]$\Sigma C_{food} = \Sigma\ (C_{food1} \times F_{food1} + C_{food2} \times F_{food2} ... C_{foodn} \times F_{foodn})$, where F_{food} is the fraction of food in the diet

Table 2. Concentrations in Ingested Media Used to Calculate the Daily Dose to Spotted Sandpiper in Table 1

	Concentrations in Ingested Media			
Exposure Area	Site-wide Aquatic	Shoreline	Site-wide Aquatic	
Equation	C_{water}	C_{sed}	$C_{food\ -\ crustacea}$	$C_{food\ -\ molluscs}$
Units	mg/L	mg/kg	mg/kg	mg/kg
$TEQ_{DF,B}$[a]	2.63E-08	3.43E-03	1.95E-05	8.22E-05

[a]Values shown are for the central tendency exposure.

Table 3. Spotted Sandpiper Life History Parameters Used to Calculate the Daily Dose in Table 1

Component	FIR	$F_{food\text{-}crustacea}$	$F_{food\text{-}molluscs}$	F_{sed}	WIR	AUF
Description	Food Ingestion Rate	Crustacea Fraction of Total Diet	Molluscs Fraction of Total Diet	Fraction of Soil or Sediment in Diet	Water Ingestion Rate	Area Use Factor
Units	kg food/kg bw-day	kg food/kg diet	kg food/kg diet	kg sediment/kg diet	L water/kg bw-day	km habitat/km home range
Value	0.22	0.5	0.5	0.18	0.17	1

EXHIBIT 2A
EXAMPLE CALCULATIONS FOR ESTIMATION OF $TEQ_{DF,B}$ CONCENTRATIONS IN BIRD EGGS

Exhibit 2A
Example Calculations for Estimation of $TEQ_{DF,B}$ Concentrations in Bird Eggs

Each case uses the linear regression equation taken from Elliott et al. 2001,

log(Egg Conc.) = a X log(Ingested Conc.) + b

Where: Egg Conc. = estimated concentration in egg (ng/kg ww)
Ingested Conc. = Concentration (ng/kg ww) of each congener calculated by summation of the concentrations of that congener from each dietary source scaled by its respective fractional contribution to the total mass of ingested media.
a = slope (as determined by Elliott et al. 2001 and applied in this estimate; Table 4-14)
b = intercept (as determined by Elliott et al. 2001 and applied in this estimate; Table 4-14)

Predicted TEQ concentrations were determined for the range of TEFs in each congener group (Table 4-16).

Case 1: Calculation of estimated $TEQ_{DF,B}$ egg concentration in cormorant consuming prey only. Values represent the CT of $TEQ_{DF,B}$ ingested prey, and TEQ was calculated using the maximum TEF.

Formula Used in Cases 1-3: [(Conc. in Media (ng/kg) X Fraction of Diet) = Fraction of Ingested Conc.] Ingested Conc. → [(log(Ingested Conc.) X a) + b] = log(Egg Conc.) → (Estimated Conc. in Egg X TEF) = Predicted TEQ Conc.

Bird Receptor	Ingested item for scenario involving consumption of prey only	Congener	Sample ID	Conc. in Media (ng/kg)	Fraction of Diet	Fraction of Ingested Conc.	Ingested Conc.	log(Ingested Conc.)	a	b	log(Egg Conc.)	Estimated Conc. In Egg	TEF	Predicted TEQ Conc.
Cormorant	Gulf Killifish	TCDF	GK-TTR5-2	0.618	1.00	0.618	0.618	-0.209	0.407	0.333	0.248	1.77	1	1.77
			Table 4-15		Table 3-12				Table 4-14				Table 4-16	Table 4-17

Case 2: Calculation of estimated $TEQ_{DF,B}$ egg concentration for PeCDD in heron consuming three prey types and shoreline sediment. Values used represent the RM of $TEQ_{DF,B}$ in each ingested medium. TEQ concentration is calculated using maximum TEF value for the PeCDD group.

Bird Receptor	Ingested item for scenario involving consumption of prey and sediment	Congener	Sample ID	Conc. in Media (ng/kg)	Fraction of Diet	Fraction of Ingested Conc.	Ingested Conc.	log(Ingested Conc.)	a	b	log(Egg Conc.)	Estimated Conc. In Egg	TEF	Predicted TEQ Conc.
Heron	Blue crab	PeCDD	SJFCA1-CR6	1.16	0.010	0.012	0.470	-0.328	0.647	1.832	1.620	41.7	1	41.7
	Gulf killifish		GK-TTR3-2	0.00995	0.495	0.005								
	Hardhead catfish		SJFCA1-LF1	0.0236	0.495	0.012								
	Shoreline sediment		SJNE022-2	13.4	0.033	0.442								
			Table 4-15		Table 3-12				Table 4-14				Table 4-16	Table 4-17

Case 3: Calculation of estimated $TEQ_{DF,B}$ egg concentration for HxCDF in sandpipers consuming two prey types and shoreline sediment. Values used represent the CT of $TEQ_{DF,B}$ in each ingested medium. TEQ concentrations is calculated using minimum TEF value for the HxCDF group.

Bird Receptor	Ingested item for scenario involving consumption of prey and shoreline sediment	Congener	Sample ID	Conc. in Media (ng/kg)	Fraction of Diet	Fraction of Ingested Conc.	Ingested Conc.	log(Ingested Conc.)	a	b	log(Egg Conc.)	Estimated Conc. In Egg	TEF	Predicted TEQ Conc.
Sandpiper	Blue crab	HxCDF	SJFCA2-CR6	0.087	0.50	0.044	30.650	1.486	0.741	1.400	2.501	317	0.1	31.7
	Common rangia		CL-TTR5-001	0.012	0.50	0.006								
	Shoreline sediment		TCEQ2009_03	170	0.18	30.6								
			Table 4-15		Table 3-12				Table 4-14				Table 4-16	Table 4-17

Notes
CT = central tendency
RM = reasonable maximum
TEF = toxicity equivalence factor
$TEQ_{DF,B}$ = toxicity equivalent for dioxins and furans calculated using avian toxicity equivalence factors with nondetects set at one-half the detection limit

EXHIBIT 2B
EXAMPLE CALCULATIONS FOR ESTIMATION OF PCB CONCENTRATIONS IN BIRD EGGS

Exhibit 2B
Example Calculations for Estimation of PCB Concentrations in Bird Eggs

Egg Conc. = Ingested Conc. X BMF

Where: Egg Conc. = estimated concentration in egg (ng/kg ww)
Ingested Conc. = Concentration (ng/kg ww) of congener calculated by summation of the individual congener concentrations from each dietary source scaled by its respective fractional contribution.
BMF = biomagnification factor for each PCB from ingested medium to egg (Table 4-18)

Predicted TEQ concentrations were determined by application of congener specific TEF (Table 4-18).

Case 1: Calculation of estimated $TEQ_{P,B}$ egg concentration in cormorant consuming prey and sediment for PCB105. Values represent the CT of $TEQ_{P,B}$ in prey and sediment samples.

Formula Used in Cases 1-4: (Conc. in Media (ng/kg) X Fraction of Diet) = Fraction of Ingested Conc. → (Ingested Conc. X BMF) → Estimated Conc. in Egg X TEF = Predicted TEQ Conc.

Bird Receptor	Ingested item for scenario involving consumption of prey only	Congener	Sample ID	Conc. in media (ng/kg)	Fraction of Diet	Fraction of Ingested Conc.	Ingested Conc.	BMF	Estimated Conc. in Egg	TEF	Predicted TEQ Conc.
Cormorant	Gulf killifish	PCB105	GK-TTR5-1	715	1.00	715	819	20	16,372	0.0001	1.64
	Sediment		SJNE022-1	5,180	0.02	104					

Case 2: Calculation of estimated background $TEQ_{P,B}$ egg concentration in cormorant consuming prey and sediment for PCB126. Values represent the RM of $TEQ_{P,B}$ in prey and sediment samples.

Bird Receptor	Ingested item for scenario involving consumption of prey only	Congener	Sample ID	Conc. in media (ng/kg)	Fraction of Diet	Fraction of Ingested Conc.	Ingested Conc.	BMF	Estimated Conc. in Egg	TEF	Predicted TEQ Conc.
Cormorant	Gulf killifish	PCB126	GK-TTR7-1	5.43	1.00	5.430	5.46	18.7	102	0.1	10.2
	Sediment		SJNE065	1.64	0.02	0.033					

Case 3: Calculation of estimated background $TEQ_{P,B}$ egg concentration in heron consuming prey for PCB077. Values represent the RM of $TEQ_{P,B}$ for prey samples.

Bird Receptor	Ingested item for scenario involving consumption of prey and sediment	Congener	Sample ID	Conc. in media (ng/kg)	Fraction of Diet	Fraction of Ingested Conc.	Ingested Conc.	BMF	Estimated Conc. in Egg	TEF	Predicted TEQ Conc.
Heron	Blue crab	PCB077	SJFCACB-CR1	9.06	0.010	0.091	18.7	0.7	13.1	0.05	0.655
	Gulf killifish		GK-TTR7-1	4.42	0.495	2.19					
	Hardhead catfish		SJFCACB-LF6	33.2	0.495	16.4					

Exhibit 2B
Example Calculations for Estimation of PCB Concentrations in Bird Eggs

Case 4: Calculation of estimated $TEQ_{P,B}$ egg concentration in sandpiper consuming prey for PCB118. Values represent the CT of $TEQ_{P,B}$ for prey samples.

Bird Receptor	Ingested item for scenario involving consumption of prey and shoreline sediment	Congener	Sample ID	Conc. in media (ng/kg)	Fraction of Diet	Fraction of Ingested Conc.	Ingested Conc.	BMF	Estimated Conc. in Egg	TEF	Predicted TEQ Conc.
Sandpiper	Blue crab	PCB118	SJFCA2-CR6	2,574	0.50	1,287	2,687	4.61	12,386	0.00001	0.124
	Common rangia		CL-TTR3-002	2,800	0.50	1,400					