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Conducting Ecological Risk Assessments at Remediation Sites in Texas

Remediation Division
TEXAS COMMISSION ON ENVIRONMENTAL QUALITY

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Prepared by
Remediation Division

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Overview

Objective: to instruct users in how to conduct ecological risk assessments at remediation sites in Texas in conjunction with the Ecological Protective Concentration Level Database and the Supporting Documentation for the TCEQ's Ecological Benchmark Tables (see TCEQ publication RG-263b). A case study illustrating how the PCL Database can be incorporated into an ecological risk assessment is also available (RG 263c).

Audience: the regulated community and environmental professionals.

References:

- The regulatory citation for the Texas Risk Reduction Program (TRRP) rule is Title 30, Texas Administrative Code [30 TAC, Chapter 350].
- The TRRP rule, together with conforming changes to related rules, is contained in 30 TAC 350 and was initially published in the September 17, 1999, Texas Register (24 Tex. Reg. 7436–766). The rule was amended in 2003 (effective September 1, 2003; 28 Tex. Reg. 6935–37), in 2007 (effective March 19, 2007; 32 Tex. Reg. 1526–79), and in 2009 (effective March 19, 2009, 34 Tex. Reg. 1866–72).
- Find links for the TRRP rule and preamble, Tier 1 PCL tables, and other TRRP information at <www.tceq.texas.gov/remediation/trrp/>.
- TRRP guidance documents undergo periodic revision and are subject to change. Referenced TRRP documents may be in development. Links to current versions appear at: <www.tceq.texas.gov/goto/trrp-guidance>.
- The TCEQ is committed to accessibility. If you are unable to access the information in any portion of this document, please contact the Technical Program Support Team at the phone number or e-mail address below.

Contact: TCEQ Remediation Division, Division Support Section, 512-239-2200, or <techsup@tceq.texas.gov>. For mailing addresses, refer to <www.tceq.texas.gov/about/directory/>.

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Abbreviations

AhR – aryl hydrocarbon receptor

APAR – affected property assessment report

ATSDR – Agency for Toxic Substances and Disease Registry

AUF – area-use factor

AVS – acid volatile sulfide

BCG – biota concentration guide

BAF – bioaccumulation factor

BCF – bioconcentration factor

BMP – best management practices

BSAF – biota sediment accumulation factor

BTEX – benzene, toluene, ethylbenzene, and xylene

BTF – biotransfer factor

CAS no. – Chemical Abstract Services registry number

CBR – critical body residue

COC – chemical of concern

CSM – conceptual site model

Eco-SSLs – ecological soil screening levels (U.S. EPA)

DW – dry weight

EF – exposure frequency

EMF – exposure modifying factor

EPC – exposure point concentration

EqP – equilibrium partitioning

ERA – ecological risk assessment

ERED – Environmental Residue Effects Database

ESA – ecological services analysis

ESE – expedited stream evaluation

f_{oc} – fraction organic carbon

GIS – geographic information system(s)

GW – groundwater

HEA – habitat equivalency analysis

HI – hazard index

HPAHs – high molecular weight polycyclic aromatic hydrocarbons

HQ – hazard quotient

K_{oc} – organic-carbon partition coefficient

K_{ow} – octanol-water partition coefficient

K_p – partition coefficient

LC_{50} – lethal concentration (50 percent)

LD_{50} – lethal dose (50 percent)

LOAEL – lowest observed adverse effect level

LPAHs – low molecular weight polycyclic aromatic hydrocarbons

MBTA – Migratory Bird Treaty Act

MOU – memorandum of understanding

MTL – maximum tolerable level

MQL – method quantitation limit

MW – molecular weight

NAWQC – national ambient water quality criteria

NOAA – National Oceanic and Atmospheric Administration

NOAEL – no observed adverse effect level

OC – organic carbon

ORNL – Oak Ridge National Laboratory

PAH – polycyclic aromatic hydrocarbon

PCB – polychlorinated biphenyl

PCDD – polychlorinated dibenzo-p-dioxins

PCDF – polychlorinated dibenzofurans

PCL – protective concentration level

PCLE zone – protective concentration level exceedance zone

PFAS – poly-or perfluoroalkyl substances

PFCs – perfluorinated compounds

PHH – planar halogenated hydrocarbons

POE – point of exposure

PQL – practical quantitation limit

QSAR – quantitative structure-activity relationship

RAP – response action plan

RBEL – risk-based exposure limit

RCRA – Resource Conservation and Recovery Act

RIFA – red imported fire ants

RJ – reasoned justification

RRR – Risk Reduction Rule

SEM – simultaneously extracted metals

SETAC– Society of Environmental Toxicology and Chemistry

SLERA – screening-level ecological risk assessment

SSERA – site-specific ecological risk assessment

SQB – sediment quality benchmark

SW – surface water

T&E – threatened and endangered (species)

TAC – Texas Administrative Code

30 TAC 350 – Title 30, Texas Administrative Code, Chapter 350

TCDD – 2,3,7,8-tetrachloro-dibenzo-p-dioxin

TCDF – 2,3,7,8-tetrachloro-dibenzofuran

TCEQ – Texas Commission on Environmental Quality

TDS – total dissolved solids

TEC – threshold effect concentration

TEF – toxic equivalency factor

TEL – threshold effects level

TEQ – toxic equivalency

TL – trophic level

TPAHs – total polycyclic aromatic hydrocarbons

TPH – total petroleum hydrocarbon

TPWD – Texas Parks and Wildlife Department

TRA – tissue residue approach

TRRP – Texas Risk Reduction Program

TRV – toxicity reference value

TSS – total suspended solids

TSWQS – Texas surface water quality standards

UCL – upper confidence limit

UF – uncertainty factor

UPL – upper prediction limit

U.S. ACE – United States Army Corps of Engineers

U.S. DOD – United States Department of Defense

U.S. DOE – United States Department of Energy

U.S. DOI – United States Department of the Interior

U.S. EPA – United States Environmental Protection Agency

U.S. FWS – United States Fish and Wildlife Service

VOC – volatile organic compound

WQB – water quality benchmark

WW – wet weight

1.0 Introduction

This publication outlines the TCEQ's ecological risk assessment (ERA) program and describes the interface between the ERA program and the Texas Risk Reduction Program (TRRP) Rule [30 TAC Chapter 350]. This guidance is also applicable to sites under the Risk Reduction Rule (RRR) [30 TAC Chapter 335].

1.1. Purpose

The purpose of this publication is to promote the development of consistent and technically defensible ERAs to be submitted under TCEQ remediation programs. It also includes technical advice and insight as to how the TCEQ may evaluate ERAs. This guide also discusses interactions of the ERA process with the ecological services analysis (ESA) and the role that the Natural Resource Trustee agencies play in both [see 30 TAC Chapter 7.124]. This guide is not itself a rule or compilation; in the case of any apparent conflict, the rule itself governs.

This guide outlines a three-tiered approach for conducting ERAs with several exit points to allow for preparation and submission of information that is commensurate with the degree of environmental concern an affected property requires. The TCEQ recognizes that other publications specify additional ERA methodologies, and, in fact, this guide is partly patterned after some of them. However, when conducting an ERA under a TCEQ remediation program, the person is strongly encouraged to use this guide, as it has been developed and adapted especially for Texas, vetted by a multi-stakeholder ecological work group, and integrated with the TRRP. Furthermore, nothing set forth herein prevents TCEQ personnel from varying from policies contained herein as dictated by the specific facts and circumstances for an ERA or site.

This document is intended to provide technical guidance for conducting ERAs at TRRP sites. Therefore, terminology specific to the TRRP rule and the science of ERAs is used throughout. Some terms are defined within the context of the relevant discussions and are denoted by italicized text. Others appear in the glossary near the end of this manual. Specifically note the use of "person," which has a special meaning under the TRRP rule. In addition, the words "site" and "affected property" are used interchangeably in this document but both terms are used to denote the entire area of contamination [see 30 TAC 350.4(a)(1)].

This document is divided into numbered chapters, sections, subsections, etc., and numbers in **bold** type reference parts of this publication. For example, "see **3.5.2**" means "see subsection **3.5.2.**" and "is discussed in **14.0**" means "is discussed in chapter **14.0**."

1.2. Definition of an Ecological Risk Assessment

The U.S. Environmental Protection Agency's 1992 Framework for Ecological Risk Assessment sets forth a basic structure and a consistent approach for conducting ERAs but is not intended for program-specific guidance. The Framework*'s* structure and approach have been expounded in this publication to aid in the development of consistent and technically defensible ERAs within the

TCEQ's remediation programs. *Ecological risk assessment* is defined as a process that evaluates the likelihood that adverse ecological effects are occurring or may occur as a result of exposure to one or more stressors. The *Framework* further defines *stressor* as any physical, chemical, or biological entity that can induce an adverse ecological response; this publication only addresses chemical stressors—those subject to risk-management decisions at remediation sites. A risk cannot exist unless the stressor can cause one or more adverse effects and it occurs with, or contacts, an ecological component (*receptor*) long enough and at a sufficient intensity to elicit the identified adverse effect (U.S. EPA, 1992a).

For the purposes of this guide, the primary functions of an ERA are to:

- determine whether actual or potential ecological risk exists at a remediation site;
- screen the chemicals of concern (COCs) present to identify those that might pose an ecological risk, allowing for the focusing of further efforts; and
- if necessary, determine ecologically protective concentration levels (PCLs) to be used in evaluating responses.[1]

Ecological risk assessment is an interdisciplinary field, drawing upon environmental toxicology, ecology, and environmental chemistry, as well as other areas of science and mathematics (U.S. EPA, 1997a). Although this guide attempts to present this information in a straightforward manner, it is important that users understand that ecological risk assessment is a complex process with many parallel activities. Consequently, a basic understanding of ecotoxicology and risk assessment, though not mandatory, will prove useful.

1.3. Changes to the TCEQ's ERA Process

The TCEQ has continuously developed and published ERA guidance and revisions since 1996. The ERA guidance underwent its most significant changes with the January 2017 version including a restructuring of the document based on the 10 required elements as identified in the TRRP rule [30 TAC 350.77(c)]. This current version has been updated to the most recent accessibility standards and a few changes have been made to some of the technical information. For clarity, the discussion of significant changes first presented in the 2017 version are repeated here and additional information or updates presented in this 2018 release are indicated as **"NEW"**.

1.3.1. Data from Ecological Habitat

As was reiterated in the January 2017 version, if the affected property has failed the Tier 1 Exclusion Criteria Checklist for the soil exposure pathway, meaning that the pathways from contaminated soil to ecological receptors are complete and significant, it is important to adequately characterize the nature and extent

[1] *Response*, when used in this guide, is equivalent to *response action* as defined in the TRRP rule (see Definitions, **17.0**).

of contamination as it relates to ecological habitat and in the determination of the appropriate soil exposure point concentrations (EPCs) used in the risk assessment. The TCEQ strongly recommends that, where contamination above assessment levels extends into ecological habitat, enough samples be collected from the habitat to generate an appropriate ecological EPC. Data from ecological habitat is discussed in detail in TRRP-15eco (Determining Representative Concentrations of Chemicals of Concern, TCEQ, 2013a) but is also discussed in **2.4**.

1.3.2. Hot Spot Analysis

As originally described in TRRP-15eco and the 2017 version of this guide, the determination and evaluation of hot spots in the ecological habitat area is now recommended in a Tier 2 or Tier 3 ERA. The presence of hot spots at an affected property can be important in the assessment and management of wildlife and benthic invertebrate risks. The purpose of a hot spot evaluation is to identify any risks to wildlife receptors and benthic invertebrates that would not be identified and mitigated through the standard risk evaluation, which is based on averaging COC concentrations [i.e., using a 95 percent upper confidence limit (UCL) as the EPC] across larger areas. See **10.3** for a discussion on hot spots.

The identification and early treatment of hot spots (e.g., removal) can be useful in addressing risk management objectives for an affected property. For instance, identification and treatment may focus the evaluation on those locations that are most important and effective to remediate. A facility may choose to address a hot spot up front to minimize future investigation or liability. See **14.3** for a discussion of risk management for hot spots.

1.3.3. Ecological PCL Database

The TCEQ and its contractor (West Texas A&M University) have developed an Ecological PCL Database or "PCL Database" that provides default ecological PCLs for soil and sediment for a variety of wildlife receptors and COCs, see <pcl.wtamu.edu/pcl/login.jsp>. The PCL Database was officially released to the public in January 2017.

The PCL Database has undergone extensive peer review by the ecological work group. All inputs from the PCL Database are sourced from the open literature. As new information becomes available, it may be incorporated into the PCL Database following technical peer review. Users may adjust the default PCLs based on site-specific inputs (e.g., home range). In addition, this guidance refers to the PCL Database as a source for assessment levels, toxicity profiles, toxicity reference values (TRVs), life history information, and uptake factors (e.g., bioaccumulation factors). Users are encouraged to choose "Contact Us" to submit additional technical information for consideration.

Note that the PCL Database is considered an extension of this guide; however, the PCL Database does not provide all the necessary components of an ERA (e.g., required element 1, benchmark screening). Therefore, users must still address any remaining required elements as identified in the TRRP rule [30 TAC 350.77(c)].

NEW: Updates and changes to the PCL Database are listed in an Update Page that is posted in the information section of the PCL Database.

> *Specific instructions for use of the PCL Database by subject matter are dispersed throughout this document in relevant locations. Instructional text is set off from the main text in boxes, as shown here. More complete instructions, from logging in to the PCL Database to developing PCLs, appear in the box in **13.3**.*
>
> *A 'help' button on each individual screen is available within the PCL Database.*
>
> *Supporting documentation and additional information is available by clicking on the 'information' button.*

1.3.4. Updates to Benchmark Tables

The screening-level benchmark tables for surface water, sediment, and soil were removed from the 2017 version and are currently posted to the TCEQ's ERA website at <www.tceq.texas.gov/goto/era> as an Excel workbook. Note that statewide soil background values were added in the 2017 version for cadmium, lithium, molybdenum, silver, and uranium. The tables and the supporting documentation are now collectively known as TCEQ publication RG-263b.

NEW: Any updates to the benchmarks are listed in a worksheet labeled "List of Updates" in the Excel workbook. This worksheet, and when necessary, the supporting documentation, will be updated in the future as changes are made to the benchmarks.

1.3.5. Updates to COC Designations as Bioaccumulative

The January 2017 version listed silver as bioaccumulative in soil (**5.1.1**). Although the documented trophic bioaccumulation potential in soil is low, there is a large disparity between the plant benchmark (560 mg/kg, U.S. EPA, 2006) and the wildlife-based PCLs from the PCL Database and the U.S. EPA's Ecological Soil Screening Levels (Eco-SSLs). Therefore, to prevent silver from being screened out through a comparison to its benchmark, the TCEQ is requiring that it be evaluated as if it were bioaccumulative in soil (i.e., a trophic-level assessment).

1.3.6. Body Scaling

The January 2017 version removed the practice of adjusting TRVs to account for the body weight differences between the test species and the wildlife species. Although this extrapolation method is applied in human toxicology and has been used for wildlife risk evaluations, it is no longer recommended for use in ERAs by the U.S. EPA (2007a). Also, the fundamental supporting data have significant limitations. For example, much of the mammalian data are based on anticancer drugs and not on typical environmental contaminants, and secondly, the allometric scaling models for both human and wildlife are based on acute toxicity data with unknown applicability to chronic toxicity (Allard et al., 2010).

1.3.7. Evaluating Soil Intervals for Risk to Burrowing Receptors

The 2017 version addressed the evaluation of burrowing receptors. Burrowing species present at an affected property may be exposed to contaminated soil both at the surface and the subsurface; therefore, it is appropriate that soil exposure concentrations from both intervals be considered. The discussion in **6.6.4** presents a process for collecting and evaluating surface and subsurface soil samples when the presence of a burrowing species is likely and the exposure pathway to impacted subsurface soil is believed to be complete.

1.3.8. Case Study

The TCEQ has developed the publication RG-263c, Case Study for the TCEQ's Ecological Risk Assessment Process or "case study". The case study is an evaluation of a contaminated hypothetical site by using a Tier 1 Exclusion Checklist and a Tier 2 screening-level ecological risk assessment (SLERA). It concludes by developing an ESA as the site remedy. The purpose of this study is to provide the person with an idea of how a site can be ecologically assessed and how outputs from the PCL Database can be incorporated into a SLERA. The case study has been posted on the TCEQ's ERA webpage at <www.tceq.texas.gov/goto/era>.

1.3.9. Evaluating Risk from PAHs

NEW: A major change in this 2018 version is how polycyclic aromatic hydrocarbons (PAHs) are evaluated in soil and sediment. PAHs almost always occur in the environment as mixtures. Therefore, the benchmarks and PCLs provided for total PAHs (TPAHs) are the most relevant for evaluating risk in an ERA. Values for individual, low molecular weight, and high molecular weight PAHs should only be used where there are no benchmarks or PCLs available for TPAHs (e.g., for surface water). The TCEQ has replaced the soil benchmarks for low and high molecular weight PAHs (LPAHs and HPAHs) with a TPAHs benchmark. See **10.5.3** for further discussion on PAHs.

When using the PCL Database to assess wildlife exposure to PAHs, TPAHs should be evaluated – even though values for LPAHs and HPAHs exist – because all individual compounds within this class are included and the TRVs selected for use are those for the most toxic PAH compounds. This methodology ensures both protection against PAH mixtures dominated by the more toxic compounds and consistency between the soil and sediment evaluations.

1.4. Connections to other TCEQ Publications and Rules

The TCEQ has other rules and documents that may help with an ERA:

- Texas Surface Water Quality Standards [30 TAC 307]. The standards (uses and criteria) presented in this rule are uniquely tied to the aquatic life component of the ERA. <www.tceq.texas.gov/goto/tswqs>

- Procedures to Implement the Texas Surface Water Quality Standards (TCEQ, 2010a, publication no. RG-194). This document contains hardness and chloride levels that may be used in determining segment-specific surface water PCLs. <www.tceq.texas.gov/goto/tswqs>

- Determining PCLs for Surface Water and Sediment (TCEQ, 2007a, publication no. RG-366/TRRP-24). Numerous issues are discussed including water-body uses, standards applicability, human-health sediment and surface water PCLs and groundwater to surface water evaluation approaches. <www.tceq.texas.gov/goto/trrp-guidance>

- Texas Integrated Report of Surface Water Quality. This report is a requirement of the federal Clean Water Act, Sections 305(b) and 303(d). If a water body is on the 303(d) listing as impaired, that will affect the determination of groundwater–to–surface water dilution factors and, ultimately, PCLs. <www.tceq.texas.gov/goto/tirswq>

- Determining Representative Concentrations of Chemicals of Concern for Ecological Receptors, TCEQ, 2013a, publication no. RG-366/TRRP-15eco. This document discusses determining the representative concentration (i.e., the EPC) for ecological assessment, including the groundwater–to–surface water pathway. It also discusses the evaluation of hot spots. TRRP-15eco should be considered a primary companion document to this guide. <www.tceq.texas.gov/goto/trrp-guidance>

1.5. Organization of This Document

This document follows the person's perceived path through the ERA process and relies heavily on TRRP-15eco for additional discussion of specific topics. Chapters in this publication:

1. Introduction: Overviews the TRRP rule and how the ERA guidance is incorporated. Also, briefly describes some of the major technical adjustments in this document from the January 2017 version.
2. Affected Property Assessment: Summarizes what the TCEQ expects from an affected property assessment for ecological habitat by media (surface water, sediment, soil, and groundwater).
3. Tier 1: Exclusion Criteria Checklist and Associated Special Circumstances: Discusses the purpose, use, and instructions for completing the Tier 1 Exclusion Criteria Checklist. Also describes special circumstances associated with the Tier 1 Checklist, such as the reasoned justification (RJ) and the expedited stream evaluation (ESE).
4. Tier 2: Screening-Level Ecological Risk Assessment—Introduction: Provides a brief overview of the Tier 2 process.
5. COC-Screening Analysis (Required Element 1): Discusses COC screening, the first step in a Tier 2 SLERA. Defines bioaccumulative

COCs and discusses how to develop benchmarks or use a surrogate chemical.

6. Exposure Pathway Analysis (Required Element 2): Describes the exposure analysis for communities and feeding guilds. Assessment considerations by media.
7. Conceptual Site Model (Required Element 3): Discusses develop and presentation of a conceptual site model (CSM).
8. Fate and Transport, Toxicological Profiles (Required Element 4): Defines the expectations of the TCEQ for the COC fate and transport discussion and the presentation of toxicological profiles.
9. Receptor Effect Levels (Required Element 5): Discusses selection of measurement endpoints, how to characterize ecological effects, and the use of tissue residue data.
10. Exposure Assessment (Required Element 5, continued): Discusses input data and exposure calculations. Specific topics include the use of uptake factors to estimate COC concentrations in food and prey, ingestion rates for wildlife receptors, exposure modifying factors and bioavailability.
11. Hazard Quotient Analyses (Required Elements 6, 7): Describes the procedure for determining the hazard quotient using conservative and less-conservative assumptions.
12. Uncertainty Analysis (Required Element 8): Describes the TCEQ's expectations for the topics to be discussed in the uncertainty analysis, including the hot spot evaluation. The uncertainty analysis is an industry standard that should be presented in every ERA.
13. Ecological PCL Development (Required Element 9): Presents methods for determining the ecological PCL, including the use of the Ecological PCL Database.
14. Ecological Risk Management (Required Element 10): Describes the types of risk-management options under the TRRP rule, including an ESA.
15. Tier 3 Site-Specific Ecological Risk Assessment (SSERA): Provides a general discussion of some of the studies common to Tier 3 SSERAs.
16. References: All the references used in this guide.
17. Definitions: Definitions of terms used in this guide.

The appendices in this publication are:

Appendix A. Food webs of the seven major habitats in Texas. Presents the food webs and a corresponding text description.

Appendix B. Selected Measurement Receptors for Evaluation of Minor Habitats: Lists life history and exposure inputs for common measurement receptors often used in ERAs.

Appendix C. Assessing and Minimizing Impacts to Protected Species: Presents considerations for minimizing impacts to protected species when sampling or remediating in ecological habitat.

1.6. Tiered ERA Process

As prescribed in the TRRP rule and illustrated in Figure 1.1, the TCEQ has developed a three-tiered approach for conducting ERAs. The person may elect to commence the ERA process at any of the tiers:

Tier 1—Exclusion Criteria Checklist

The Tier 1 Checklist sets forth conditions under which an affected property may be excluded from further ecological assessment, based on the absence of any complete or significant ecological exposure pathways. Affected properties that do not meet these exclusion criteria will require further evaluation under Tier 2 or Tier 3 (or both), unless a reasoned justification or an ESE is used to conclude the ERA.

Tier 2—Screening-Level Ecological Risk Assessment

Under Tier 2, non-bioaccumulative COCs may be screened from further evaluation based on comparison to ecological benchmarks. If any COCs are not excluded on this basis, a conceptual site model is developed to characterize complete exposure pathways and representative receptors. Exposures are compared to literature-based effect levels using conservative assumptions that may be later refined with site-specific information. Tier 2 ecological PCLs are derived for any retained COCs.

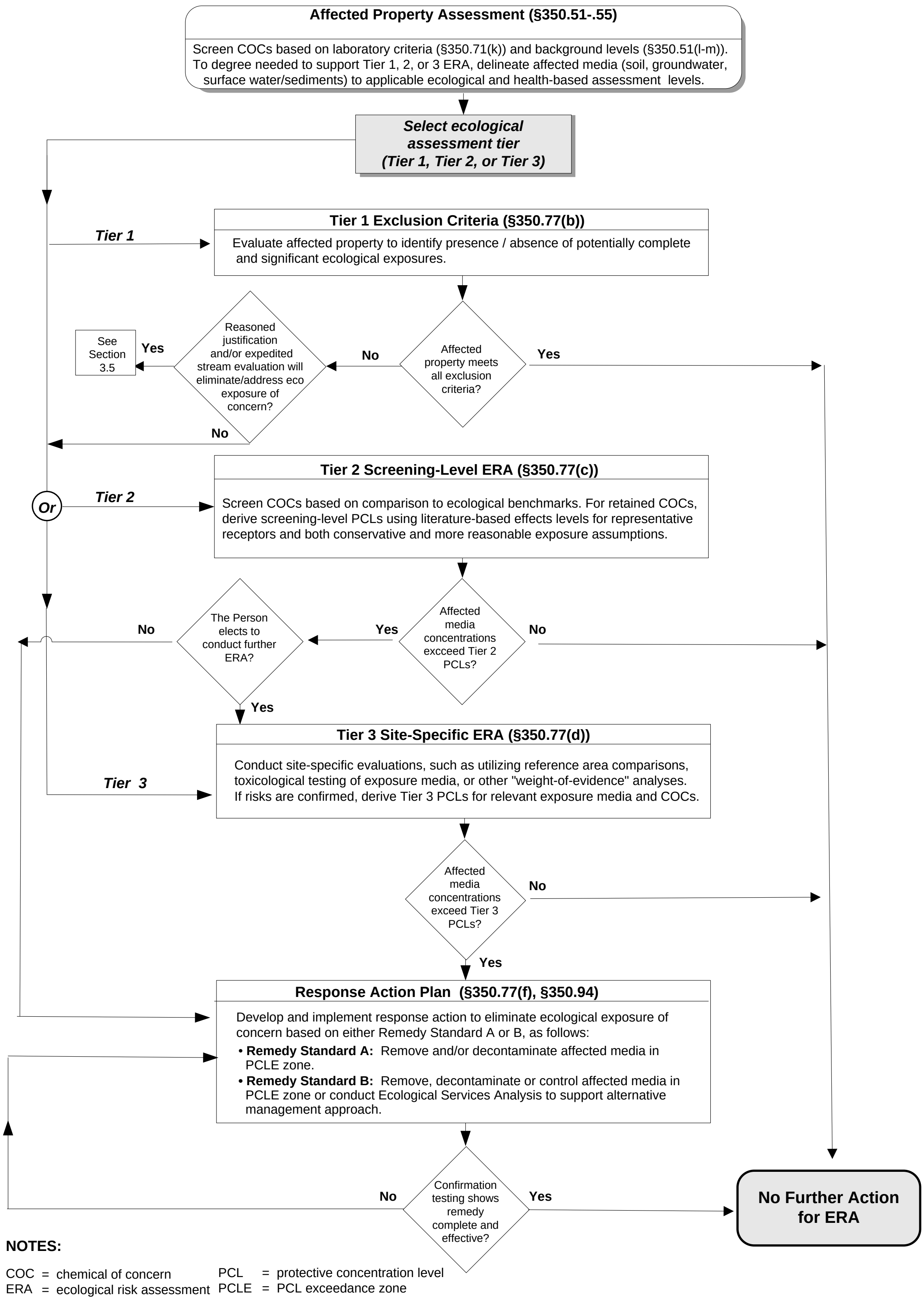

Figure 1.1. Overview of tiered ERA process.

Tier 3—Site-Specific Ecological Risk Assessment

Under optional Tier 3, ecological risk indicated from earlier tiers may be further evaluated using site-specific “weight-of-evidence” information. Such site-specific assessments may include analysis of COCs in tissue, toxicological testing of affected media, comparison of species diversity to reference areas, and other analyses. If ecological risks are confirmed, these site-specific data may be employed to derive Tier 3 ecological PCLs for the relevant receptors and exposure media. Alternatively, if no ecological risks are apparent, the ERA can be concluded without the need for any ecological PCLs.

1.7. Overview of ERA under the TRRP Rule

The ERA process is a key component of the TRRP rule, which establishes a set of consistent, risk-based response actions for applicable sites under the jurisdiction of the TCEQ Remediation Division, although other program areas may also use this rule. The TRRP rule does not obligate corrective action. Rather, it defines the objectives for site-assessment procedures of any corrective action required under applicable program areas of the TCEQ Remediation Division.

An overall flowchart for the Texas Risk Reduction Program appears in Figure 1.2. Under this process, the person must implement a response action as needed to prevent human or ecological exposure to potentially harmful levels of COCs. Following discovery and notification of a COC release that is subject to these response-action requirements, the first step is an affected property assessment [30 TAC 350.51] to define the nature and extent of affected environmental media (i.e., soil, sediment, surface water, and groundwater).

As detailed in Subchapter C of the TRRP rule [30 TAC 350.51 – 350.55)], the affected property assessment must:

- Delineate the lateral and vertical extent of environmental media affected by the release.
- Define the groundwater classification and current land use.
- Characterize the site geology and hydrogeology so that COC fate and transport can be predicted.
- Identify potentially complete exposure pathways and the possible locations of relevant human or ecological receptors.
- Evaluate the effectiveness of existing physical controls.
- Support remedy selection and notify affected landowners.

For each COC associated with the release, affected media must be delineated to an assessment level [fully defined at 350.4(a)(3)]. An assessment level is a critical PCL for a COC where the human-health PCL is established under a Tier 1 evaluation except for the soil-to-groundwater exposure pathway, and where the ecological PCLs are developed, when necessary, under Tier 2 or 3 or via the PCL

Database. The lower of the human health and ecological PCLs is used as the assessment level for each COC. The ecological assessment level may be matched to PCL values developed under either Tier 2 or Tier 3 of the ERA process, or by using the PCL Database. See **2.1** for further discussion on assessment levels.

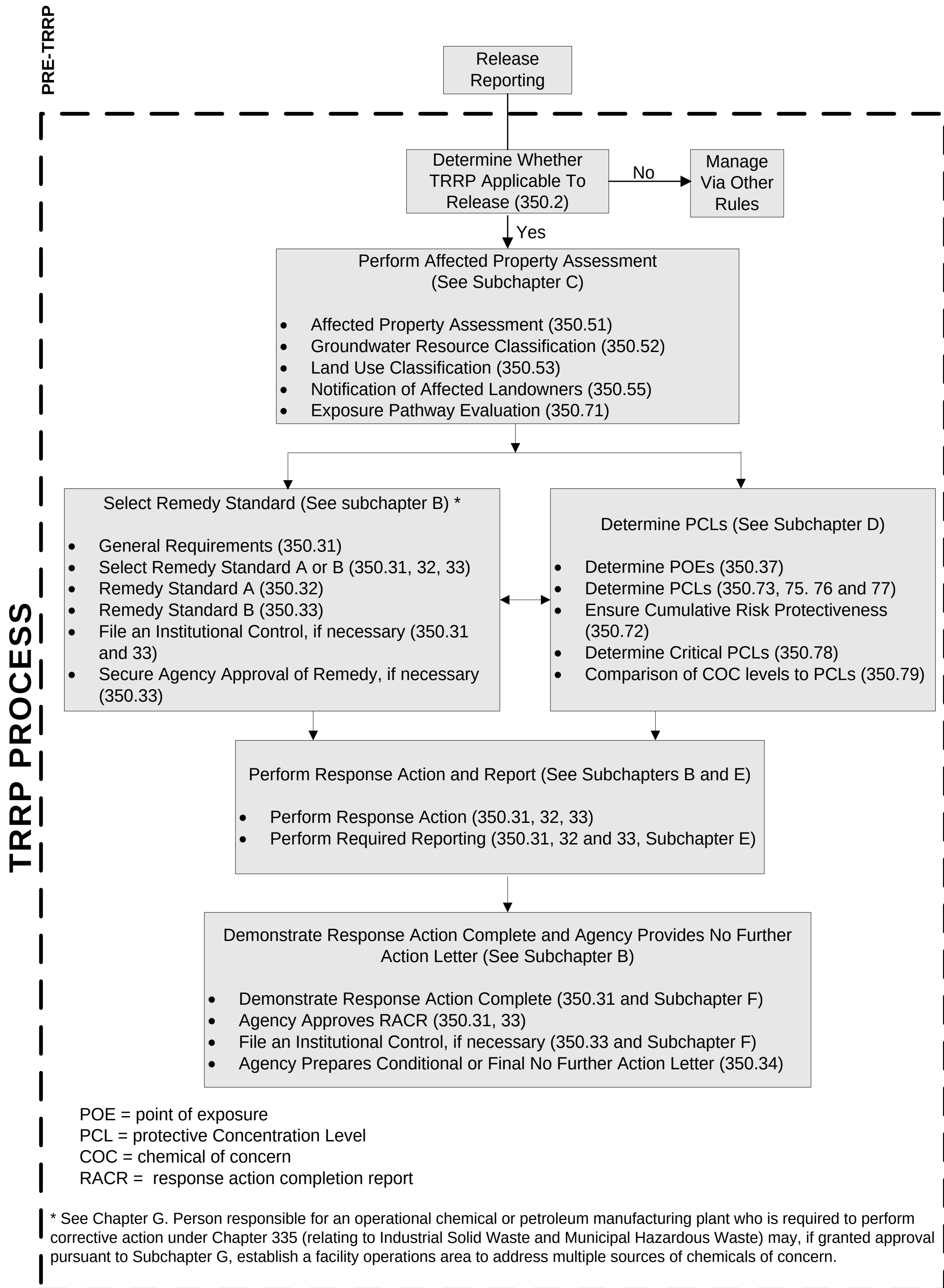

Figure 1.2. General overview of TRRP process.

Before, during, or after the affected property assessment, the person should complete the Tier 1 Exclusion Criteria Checklist to determine whether additional ecological evaluation is necessary. If so, the person may need to conduct a Tier 2 SLERA, a Tier 3 SSERA, or both to determine the applicable PCLs for each affected medium where they are necessary [see 30 TAC 350.77]. The person may use either screening-level or site-specific ecological PCLs, based on a tiered evaluation.

To evaluate the need for a response, COC concentrations are compared to the lower of the human-health PCL or ecological PCL for each COC (the lower of the two is called the *critical PCL*). If COC concentrations exceed the critical PCL for any COC, the person may either refine the PCLs by going to the next tier in the risk analysis—assuming the current tier is 1 or 2 for human health or 2 for ecological—or implement a remedy pursuant to the TRRP requirements. However, if a planned response will eliminate the ecological exposure pathway or render it insignificant, or if human health PCLs will be protective of ecological receptors, then no further ERA is required, provided the person submits a reasoned justification [see Section 350.77(a) and **3.5.1**]. In addition, if an ESE [see Section 350.77(a) and **3.5.2**] demonstrates that the surface water and sediment pathways are insignificant, then no further ERA evaluation is required, provided there are no complete and significant soil exposure pathways.

Responses must conform to one of two options for performance standards, termed Remedy Standards A and B [30 TAC 350.31] Under Remedy:

- Standard A, affected media must be removed or decontaminated to permanently reduce COC concentrations below critical PCLs [30 TAC 350.32].

- Standard B, removal, decontamination, or control measures may be applied to prevent exposure media exceeding critical PCLs [30 TAC 350.33]. Under Remedy Standard B, use of such control measures may entail post-response care and associated financial assurance [30 TAC 350.33(g–n)].

Except for Class 1 groundwater resources (which require decontamination), the person may choose to implement either a Remedy Standard A or B. For a Remedy Standard B response specific to ecological exposure pathways, the person may conduct an ESA to evaluate the net benefit of the response to ecological resources [30 TAC 350.33(a)(3)(B)]. The response action is complete once the applicable Remedy Standard A or B objectives have been satisfied [30 TAC 350.34]. Ecological risk management under Remedy Standards A and B is discussed in **14.0**.

Within the overall TRRP process, the ERA is conducted to develop PCLs that protect against potential ecological exposures. The ERA should be conducted in a manner that results in the protection of ecological receptors subject to management by other state and federal agencies and consistent with these agencies' statutory authority. As defined in the TRRP rule [30 TAC 350.4(a)(27)], the *ecological PCL* is a concentration of a COC within an exposure medium (e.g., soil, sediment, surface water, groundwater) that is protective of: (1) wider-ranging ecological receptors that may frequent the affected property and use less mobile receptors (e.g., plants, soil invertebrates, small rodents) as a food source, and (2) benthic invertebrates within surface waters in the state, where appropriate. See **13.1** for a discussion on ecological PCLs for small-ranging receptors. In general, *small-ranging receptors* are those with a home range less than or equal to 1 hectare (approximately 2.5 acres), and *wide-ranging receptors* are those with a home range greater than 1 hectare.

The focus of this publication is sites under the TRRP rule; however, the TCEQ's ERA process is also applicable to sites under the Texas Risk Reduction Rule [30 TAC 335]. If an ecological assessment is applicable to a site under the RRR, then TCEQ ERA personnel should be contacted for coordination.

2.0 Affected Property Assessment

It is imperative that the affected property assessment includes an adequate site description including land use, site photos and habitat observations pertinent to the ERA. Topography, proximity to water bodies, areas of groundwater seepage, types of vegetation, vegetation density, and vegetation height could be discussed to some degree. The ERA should also discuss any site visit that may have been performed with the goal of surveying the flora and fauna associated with the site. Additionally, general information regarding the operational history of the site would be beneficial, particularly as it relates to exposure areas and COC selection. **The ERA must include figures that diagram the affected property, the surrounding land use, and the sample locations.** The person may reference other documents for detailed information, provided these documents are clearly identified and available to the TCEQ risk assessor. Remember that legal property boundaries are not usually the same as affected property boundaries which, in turn, are not necessarily the same as ecological exposure areas. See **4.3** for additional information that should be included in the ERA. The rest of this chapter describes ecological assessment levels, background concentrations, analytical considerations, and assessment issues as they relate to the various exposure media.

2.1. Assessment Levels and PCLs

For each COC associated with the release, the assessment must delineate affected media to an assessment level [fully defined at 350.4(a)(3)] at the beginning of the TRRP process for all applicable exposure pathways (e.g., surface water, groundwater, sediment, and soil). The lower of the human health and ecological assessment levels is used as the *residential assessment level* for each COC. The required comparison of the COC concentrations against the residential assessment level entails the assumption of residential land use for human-health exposures (e.g., Tier 1 human-health PCLs) **and** ecological exposures when there are complete ecological exposure pathways.

If the affected property has failed the Tier 1 Checklist (see **3.0**) for the soil or surface water and sediment exposure pathways, it is important to select a conservatively appropriate ecological assessment level. Keep in mind that an assessment level is not necessarily a PCL (see additional explanation below by medium). A conservative assessment level is necessary to ensure that ecological concerns for the affected property are addressed, data are available for adequate characterization and evaluation, and to avoid additional data collection to delineate COCs below the ecological assessment level.

Assessment levels for ecological exposures are:

Surface Water and Groundwater: The assessment levels for COCs in surface water, or in groundwater that discharges to surface water or sediment, are the state-adopted surface water criteria and other values as presented in the Surface Water Benchmark Tables at <www.tceq.texas.gov/goto/era>. These tables use the Texas Surface Water Quality Standards (TSWQS) [30 TAC 307] as the primary resource, but also presents surface water screening values from other sources. The use of the acute or chronic criteria is determined by the classification of the water body (see **6.1**) Note that these values are not modified during the TRRP process and the assessment level will become the PCL for surface water if the contaminant remains a COC throughout the process.

Sediment: The assessment levels for COCs in sediment should be protective of the benthic invertebrate community as well as wildlife. The person should review the Sediment Benchmark Table for the freshwater or salt water benchmarks and the PCL protective of benthic invertebrates. Note that the benthic PCL for a specific COC is the midpoint between its benchmark and its second effects level (**13.4**). The person should also review the Ecological PCL Database for determination of the Conservative PCL[2] for wildlife in sediment habitat applicable to the affected property (see **6.2** for a discussion on applicable habitats). The lower of the benchmark (i.e., benthic initial effects level) or the wildlife Conservative PCL for a COC can be used as an assessment level for sediment.[3] The wildlife Conservative PCL values from the PCL Database are based on the no observed adverse effect level (NOAEL), without site-specific modifications, and are appropriately used as sediment assessment levels when there is not a protectable benthic invertebrate community present (e.g., an intermittent stream or ditch; see **6.1**).

The PCL Database also provides two other wildlife PCLs: The Average TRV (toxicity reference value) PCL is based on the average of the NOAEL and lowest observed adverse effect level (LOAEL) TRVs and is without site-specific modifications; and the Refined PCL[4] that includes site-specific modifications made by the user (e.g., using the home range of the receptor). The benthic PCL and the Refined PCL are considered comparative PCLs (see **13.4**), assuming both apply. The lower of these two values is the final ecological PCL for that COC. Once a sediment exposure point concentration (e.g., a 95 percent UCL) has been determined for a COC at the affected property, it should be evaluated against the final PCL. Of course, if the EPC is already below the lowest Conservative PCL, then wildlife at the affected property (including threatened or endangered species), is protected, assuming there are no hot spots, and the person need only evaluate the benthic PCL, if applicable. Similarly, if no protected species are likely present and the EPC is already below the lowest Average TRV PCL, then wildlife is protected.

Soil: The assessment levels for COCs within the ecologically relevant soil interval are protective of the soil invertebrate community, plant community and wildlife. Like the process for sediment, the person should consult the Soil Benchmark Table for the community-based benchmarks protective of soil invertebrates and plants and the PCL Database for the soil Conservative PCLs for wildlife. For each COC, the lower of the soil benchmark or the Conservative PCL may be used as an assessment level. Note that PCLs protective of terrestrial

[2] This PCL is based on the same stipulations as required element 6 (NOAEL-based only), as discussed in **11.2**.

[3] The PCL Database defaults to the lowest conservative PCL for those species from the selected habitat (or from a list of individual species), which may include both sediment-based and soil-based receptors. See **2.9** for a discussion on how to distinguish between these receptors.

[4] This PCL is based on the same stipulations as those in required element 7 (NOAEL- and LOAEL-based and less-conservative assumptions) as discussed in **11.3.**

plants and soil invertebrates are not usually developed (i.e., direct toxicity to plants and soil invertebrates) because the TRRP rule [30 TAC 350.4 (a)(27)] specifically states that PCLs are not intended to be directly protective of receptors with limited mobility or range (e.g., plants and soil invertebrates).

For wildlife receptors, the Average TRV PCL in the PCL Database does not account for site-specific modifications, whereas the Refined PCL includes site-specific modifications made by the user and is the final ecological soil PCL. These two PCLs should not be considered assessment levels. Once a soil EPC has been determined for a COC at the affected property, it should be evaluated against the final PCL. If the EPC is already below the lowest Conservative PCL, then wildlife at the affected property (including threatened or endangered species) is protected. If no protected species are likely present and the EPC is already below the lowest Average TRV PCL, then wildlife is protected.

This instruction box lists the steps to identify soil and sediment assessment levels from the PCL Database:

> *After logging in to the PCL Database (see box in* ***13.3****), the "PCL Calculator" page appears. Select the appropriate habitat (see* ***6.2****) for your site (only one per calculation) from the drop-down list from the "Habitats" radio button or go the "Species" radio button and hold down the control key to select multiple species.*
>
> *Choose the chemical.*
>
> *Click on the "Next" box.*
>
> *Allow the PCL analysis to run.*
>
> *Scroll down the "Analysis" page and look for the outlined number under the "Conservative PCL" column. This value represents the lowest PCL and does not reflect any site-specific adjustments. Use this PCL as the ecological assessment level if it is lower than the soil or sediment benchmark. Note that, if multiple habitats are present at the site, you will need to identify the lowest conservative wildlife PCL among those applicable habitats for the COC to determine the assessment level.*

Method Quantitation Limit: A person wanting to ensure that an assessment level will not require lowering could choose an assessment level equal to the method quantitation limit (MQL) of the COC.[5] Here the TCEQ recommends using the standard available analytical method with the lowest MQL. When the PCL is lower than the MQL, the MQL of the most sensitive available method becomes the assessment level. When the MQL is the assessment level and the COC is detected between the MQL and the method detection limit [30 TAC 350.54(e)(3)], allows the agency to require a demonstration that a lower MQL in not achievable, or is not practicable, using standard available analytical methods. The agency will consider the frequency of detection, the risk scenario, and the available analytical technology to determine if lower levels of quantitation are achievable and warranted.

2.2. Background Concentrations

Risk assessments often fail to provide the history and rationale for the development of background concentrations.

If the TCEQ Remediation Division program area has already approved property-specific background concentrations, the person completing the ERA should submit a reference to each document proposing the background values and include the TCEQ-approval correspondence. The TCEQ ecological risk assessor can then discuss or verify this with the TCEQ project manager. It would also be helpful, but is not a requirement, to include these documents and correspondence as attachments to the risk assessment, along with a map that indicates the sample locations for the background determination.

The risk assessment should always indicate if the background values are site-specific or statewide medians (for metals and inorganics in soils). Use of the risk assessment itself as a vehicle to propose site-specific background concentrations can be difficult without prior coordination with the project manager, the risk assessor, the Natural Resource Trustees, and (in some cases) the TCEQ statistician.

The TCEQ recommends the use of the upper prediction limit (UPL) as the interval estimator for the determination of a background (natural or anthropogenic) concentration for COCs upgradient or upstream of an affected property. This is particularly applicable for small data sets where a single high or low value can greatly influence the variability.

The same Texas statewide median background values for metals and inorganics in soil that are in the TRRP rule [TAC 350.51(m)] are presented in the Soil Benchmark Table and may be used in lieu of site-specific background concentrations.

In addition to the background values in the TRRP rule, background values have been added for several additional metals (**1.3.4**). Normally, the use of soil background concentrations to evaluate sediment constituents is not appropriate since the aquatic and terrestrial sediment and soil environments (chemical and

[5] The TRRP rule [30 TAC 350.54 (e)(3)] requires that the person select an available analytical method with an MQL below the necessary level of required performance for assessment as well as demonstrating conformance with critical PCLs. If it is not possible to achieve an MQL below the necessary level of required performance, and the COC does not meet the conditions of 30 TAC 350.71(k), then the person should select the standard available analytical method that gives the lowest possible MQL for that COC.

biological) are dissimilar and cannot be used interchangeably. For ephemeral streams, however, the use of soil background concentrations may be fruitful where perennial pools do not occur, and there is adequate justification for evaluating the stream bottom as soil only.

2.3. Adequacy and Appropriateness of Data

Fundamental to any soil or sediment assessment is the characterization of the nature and extent of impacts on the media. Sufficient data should be collected to identify sources of contamination, potential migration pathways, and the depth and area of contamination. When evaluating the adequacy of the scope of the soil or sediment assessment, the person should be cognizant of the TCEQ's ecological PCLs, assessment levels, and benchmarks;[6] the TRRP Texas-Specific Soil Background Concentrations [30 TAC 350.51(m)] and those related soil values currently not in the rule; site-specific background concentrations (if applicable); and laboratory MQLs.

It is critically important to adequately characterize the nature and extent of contamination (and, subsequently, the appropriate EPC used in the risk assessment) as it relates to ecological habitat. When planning the soil assessment, the person should consider the location of ecological habitat, the likelihood of ecological receptors being present, and the quality of the habitat at the affected property.

Often, too few soil samples are collected in potentially impacted ecological habitat areas. A lack of samples should not be a problem if the evaluation of nature and extent is complete and shows that COCs are not present above appropriate ecological assessment levels.

On the other hand, if the contamination extends into the habitat above the ecological assessment levels, a subsequent phase of investigation may be necessary to better characterize ecological risks therein.

Furthermore, since soil data collected to define the nature and extent of contamination are not usually the most representative of the exposure area for an ecological receptor, additional characterization or a more focused evaluation may be necessary for sites with higher-quality habitat.

The TCEQ strongly recommends that, where contamination above assessment levels extends into ecological habitat, enough samples be collected from the habitat to generate an ecological EPC.[7]

Communication with the TCEQ ERA staff is recommended when planning to conduct Tier 2 or Tier 3 ERAs. A meaningful discussion up front will help avoid collecting data that do not support the evaluation, are highly uncertain, or

[6] Although not formally recognized as wildlife benchmarks by the TCEQ, the lower of the U.S. EPA's avian and mammalian ecological soil-screening levels can be used as an assessment level, provided the EPA's soil screening level is lower than the value for plants and invertebrates.

[7] TRRP affected properties will vary greatly in size, habitat, receptors, and COC distribution. Given this, the TCEQ is not suggesting a minimum sample number. In determining the number and density of sample locations, the person should consider the foraging habits and relative sensitivity of the receptors in question, and the sample number needed to ensure sufficient statistical power for determining the EPCs used in the risk assessment.

may result in an erroneous conclusion. TRRP-15eco contains additional information on adequacy and appropriateness of data (see 2.1.1 for a discussion on soil and 3.1.1 for a discussion on sediment in TRRP-15eco).

2.4. Analytical Considerations

The accuracy and precision of analytical methodologies play a significant role in determining the suitability of soil, sediment, or water data for use in a risk assessment. Data must meet the specifications in 30 TAC 350.54 and TCEQ (2010b, publication RG-366/TRRP-13). Additionally, analytical data must be generated by a lab that is accredited through the Texas Laboratory Accreditation Program for the most recent standard adopted by the National Environmental Laboratory Accreditation Conference for the matrices, methods, and parameters of analysis. The analytical methods used should have MQLs below the assessment levels. See TRRP-15eco for additional discussion on analytical considerations (see 2.1.4 for discussion of soil, 3.1.4 for discussion of sediment and 4.1.6 for discussion of surface water in TRRP-15eco).

2.5. Surface Water

The TSWQS defines *surface water* in Texas [30 TAC 307.3(a)(70)]:

> Lakes, bays, ponds, impounding reservoirs, springs, rivers, streams, creeks, estuaries, wetlands, marshes, inlets, canals, the Gulf of Mexico inside the territorial limits of the state as defined in the Texas Water Code, §26.001, and all other bodies of surface water, natural or artificial, inland or coastal, fresh or salt, navigable or non-navigable, and including the beds and banks of all water-courses and bodies of surface water, that are wholly or partially inside or bordering the state or subject to the jurisdiction of the state; except that waters in treatment systems that are authorized by state or federal law, regulation, or permit, and that are created for the purpose of waste treatment are not considered to be water in the state.

So, nearly any body of water or ditch could be considered waters in the state absent those that are part of a currently permitted treatment system. For the ERA process in Texas, surface water exposure is characterized by the potential co-occurrence of surface water COCs and ecological receptors that exist or forage in the water column. COCs can be present in surface water in the freely dissolved form or bound to particles and suspended in the water column. Receptors include fish and invertebrate communities and aquatic-dependent or partially aquatic-dependent vertebrate wildlife. Additionally, terrestrial wildlife may be exposed to COCs in surface water if they drink impacted surface waters, although that is not typically a major exposure pathway. As discussed in TRRP-15eco, among the most significant considerations required to assess surface water exposure pathways are:

- The quality of the available surface water data.
- The nature and size of the exposure area.
- Whether the water body is fresh, brackish, or marine.

- Whether a stream or river is perennial, intermittent, intermittent with perennial pools, or ephemeral.
- The physical characteristics of the water body (e.g., temperature, pH, dissolved oxygen, total organic carbon, total suspended solids, conductivity, salinity, biological oxygen demand, chemical oxygen demand, and oxidation reduction potential).
- Whether analytical detection levels are below ecological screening levels protective of aquatic life.
- The statistics used to estimate exposure concentrations.
- The presence and evaluation of elevated concentrations (e.g., hot spots) of COCs.

The primary objective of surface water sampling and analysis is to determine whether site-related COCs have discharged directly into, originated in, or migrated to surface water bodies associated with the site. Other principal objectives of sampling are to delineate and characterize COCs and to evaluate the relationships among impacted surface water, sediments, groundwater, and soil. Ultimately, these data are used to support relevant ERAs and subsequent risk management decisions.

To meet this requirement, the TCEQ encourages early discussion with the TCEQ risk assessors (and Trustees as appropriate) regarding data that are proposed for use in assessments of surface water exposure. The purpose of early dialogue is to ensure that only those data considered relevant and appropriate are used to support the risk assessment. Early dialogue with the TCEQ staff also promotes project efficiencies by minimizing exchange of comments. The dialogue would include a general discussion of how the proposed data are suitable and consistent with the objectives of the evaluation. To facilitate discussions, the person may (for example) develop an optional sampling work plan. The remainder of this discussion centers on key considerations in determining what data may be considered acceptable when assessing ecological exposures to surface water. TRRP-15eco further discusses sampling surface water including—

Routine monitoring parameters: When collecting surface water samples, the sampler should note important characteristics (e.g., general appearance and condition, surrounding vegetation and activities, biological activity, size, depth, and flow conditions) that can be used to characterize the habitat and aquatic life uses associated with the water body. Water quality measurements such as temperature, pH, conductivity, turbidity, salinity, and dissolved oxygen should also be determined in the field. These measurements may be used in the ERA (e.g., characterizing relative habitat conditions, selecting appropriate ecological receptors, or discussions of uncertainty), as well as in the consideration of potential remedial options. Salinity levels are key to determining the applicability of various water quality criteria. Salinity levels, as well as pH, can also influence the solubility of various COCs.

Sampling depth: For most TRRP sites, surface water samples collected 1 foot below the water surface are usually acceptable. Where COC concentrations in surface water may vary with stratification (such as that associated with a salinity gradient or seasonal stratification in lakes), the sampling design should address this possibility. Where impacted groundwater enters a surface water body, there may be reasons to sample and analyze the surface water near its bottom or banks or water in the *hyporheic zone* [i.e., the area of active mixing between surface water and groundwater (Lawrence et al., 2013)] to evaluate potential ecological risks associated with the impacted groundwater. The TRRP is very clear, however, that the monitoring point for the groundwater–to–surface water pathway is normally a groundwater monitoring well placed immediately upgradient of the zone of groundwater discharge to surface water [see 30 TAC 350.51(f)].

Sampling sequence: For flowing water bodies, surface water sampling should proceed from downstream to upstream locations to minimize disturbances on water quality. Where surface water and sediment samples are collected during the same sampling event, they should be collocated (next to each other), and the water samples collected first.

Sample timing, flow considerations and tidal influences: High flow should be avoided unless the intent of sampling is to evaluate surface water quality associated with a runoff event. In fact, lotic surface waters should usually be sampled when flow is low as this is consistent with the approach used for setting wastewater-permit limits (TCEQ, 2010a, as amended). However, exceptionally low flow should be avoided when assessing risks to aquatic life. For tidal water bodies, the sample design should consider that tidal action may cause impacts from site COCs on seemingly upstream areas. Additionally, consider daily and seasonal tidal cycles or groundwater regime when planning a sampling event to ensure that surface water samples are most representative of normal conditions.

Metals in surface water: The aquatic life criteria for most metals (except mercury, selenium, and silver) are expressed in the dissolved form of the metal rather than the total recoverable form. Therefore, when evaluating compliance with the numerical aquatic life criteria (and the equivalent surface water benchmarks), it is most appropriate to analyze surface water samples for dissolved metals. This avoids an apples-and-oranges situation when comparing affected property surface water data with the corresponding screening values.

Consideration of conventional pollutants: Although less common as COCs in surface water for TRRP sites, specific nutrients (e.g., nitrate nitrogen, total phosphate), salinity, chloride, sulfate, total dissolved solids (TDS), and pH must be evaluated at an affected property if they are COCs (or degradation products of parent COCs) for the property. TCEQ (2007a; TRRP-24) and, to a lesser extent, TCEQ (2010b; TRRP-13) discuss selection of the surface water PCLs and risk-based exposure levels for these types of conventional pollutants.

2.6. Groundwater as a Source Medium for Surface Water and Sediment

This section briefly addresses the evaluation of exposure pathways for ecological receptors at the point where groundwater discharges to a surface water body (i.e., at the groundwater–surface water–sediment interface). As discussed in TRRP-15eco, groundwater ecological pathways include discharges of groundwater to:

- surface water bodies;
- ground surface as springs, pooled seeps, etc.; and
- sediments in surface water bodies, where it becomes pore water.

For this guide, groundwater exposure is characterized by the potential co-occurrence of groundwater COCs and ecological receptors (for at least a portion of their life cycle) at the groundwater–to–surface water/sediment interface. Groundwater becomes a source medium for ecological exposure pathways when dissolved or suspended COCs are transported to ecological receptors via groundwater. Exposure pathways and receptors where groundwater is the source medium include:

- Fish, amphibians, and water column invertebrates exposed to water from the groundwater–surface water interface.
- Benthic invertebrate communities living within sediments exposed to pore water.
- Aquatic macrophytes rooted in sediments taking up COCs in pore water.
- Fish and amphibians depositing egg masses in sediments at the groundwater/surface water interface (freshwater only).
- Terrestrial wildlife using groundwater seeps as a source of drinking water.

TRRP-24 discusses determining the groundwater–to–surface water PCL (^{SW}GW) and groundwater-to-sediment PCL (^{Sed}GW). TRRP-15eco provides additional clarity and perspective beyond that appearing in TRRP-24 by specifically discussing groundwater as a source medium for surface water and sediment impacts. Topics discussed in TRRP-15eco include:

- A groundwater monitoring network for groundwater–to–surface water and sediment pathways.
- Temporal and seasonal variations.
- Use of existing groundwater monitoring data.

- Using groundwater data generated in the presence of active remediation systems.
- Multiple groundwater plumes.
- Sampling seeps and other springs.

The TRRP requires investigation for the presence of groundwater beneath a site where a release has occurred. Detailed instructions for performing groundwater investigations and assessments appear in TRRP-8 (Groundwater Classification, TCEQ, 2010c).

Consideration of groundwater as a source medium for ecological exposure pathways requires a complete groundwater assessment, including the delineation of all relevant dissolved COC plumes. If the site is located near a karst area, the person should refer to TCEQ (2007b, publication RG-348). This document presents optional enhanced water quality measures and best management practices for protecting the Edwards Aquifer that will also protect the habitat of certain endangered and candidate karst-dwelling invertebrates.

2.7. Soil

Soil exposure is characterized in the context of the potential co-occurrence of soil COCs and ecological receptors that inhabit the soil or forage there, or both. Receptors include plant and soil invertebrate communities and vertebrate wildlife (i.e., mammals, birds, reptiles, and amphibians). In some cases, livestock should be evaluated as potential receptors (see **6.6.2, 9.2.3.3** and **10.4.6.2**).

As discussed in TRRP-15eco, among the most significant considerations required to assess soil exposure pathways are the quality of the available soil data, the nature and size of the exposure areas within the affected property soils that do not meet the Tier 1 exclusion criteria [30 TAC 350.77(b)], the statistics used to estimate exposure concentrations, and the presence and evaluation of elevated concentrations (hot spots) of COCs. The soil data set used to establish ecological exposure concentrations should be representative and appropriate, such that the data accurately reflect the affected property's potential risks to ecological receptors.

The TRRP rule [30 TAC 350.51(a–b)] requires that relevant and sufficient data be obtained for the assessment of ecological exposures to soils. To meet this requirement, the TCEQ encourages early discussion with agency risk assessors (and Natural Resource Trustees, as appropriate) regarding data collection proposed for use in assessments of soil exposure. Such discussions could result in the development of an optional sampling work plan or address the use and applicability of property-specific data from previous investigations. The intent of early dialogue is to ensure that only those relevant and appropriate data are used to support the risk assessment. Discussions would include how the proposed data are suitable and consistent with the objectives of the evaluation. Early interactions with the TCEQ staff also promotes project efficiencies by minimizing comment exchange. These topics are discussed in detail in TRRP-15eco, but a summary is presented here:

Soil depth: For ecological exposure pathways, the TRRP rule denotes soil in the interval extending from ground surface to 0.5 feet in depth as *surface soil*, and soil in the interval between 0.5 feet and 5 feet in depth as *subsurface soil* [30 TAC 350.4(a)(86, 88)].

Historically, soil data based on samples collected from depths of 0–2 feet or 0–5 feet was presented as "surface soil" data in ERAs submitted to the TCEQ. However, the use of 0–2 or 0–5 feet sample data may dilute out ecologically relevant surface soil concentrations. Because the TRRP rule has been in effect since September 1999, **TCEQ ERA personnel will not accept these data in lieu of surface-soil samples collected in the first half foot of soil.**

Alternatively, if burrowing receptors are evaluated or if food or prey occur at depths greater than 0.5 feet, the soil data used in the exposure calculations should reflect the contribution of subsurface soil to the exposure.

Assessment planning should consider potential ecological exposure areas and potential receptors, rather than attempting to apply data intended to support human health considerations.

Soil sieving: Sieving is physically sorting a soil sample using screens of predetermined size to obtain uniform particle sizes. Heterogeneity of materials in soil can influence COC concentrations, and thereby increase analytical variability. Following collection of a soil sample, vegetation (sticks, roots, leaf litter, and grasses) should be removed. Rocks and gravel should also be removed, as they do not usually retain contaminants nor are amenable to laboratory analysis. Beyond that, the decision whether to perform any sample sieving should be specific to the property, depending on the COCs, soil types, and data-quality objectives for the project.

2.8. Sediment

For the TRRP ERA process, sediment exposure is characterized in the context of the potential co-occurrence of sediment COCs and ecological receptors that inhabit or forage in the sediment. Receptors include benthic invertebrate communities, fish, and aquatic-dependent or semi-dependent vertebrate wildlife (mammals, birds, reptiles, and amphibians).

As discussed in TRRP-15eco, although the benthic invertebrate community will usually be the group of receptors most susceptible to contaminated sediments, wildlife receptors will sometimes be the most at risk. The most common of these instances occurs when the water body will not support a viable benthic community (see **6.1**). In this case, wildlife receptors with a comparatively high proportion of incidental sediment in their diet (e.g., sandpipers, raccoons) may be at risk.

However, even when the benthic community is viable, wildlife could be more sensitive when the COC is highly toxic to wildlife and the evaluated measurement receptors include those with a high proportion of incidental sediment ingestion. Such higher sensitivity has been observed when metals like zinc and copper are COCs and sandpipers are evaluated at the high end of their reported sediment-ingestion range of 7 to 30 percent (Beyer et al., 1994). Also, when the COC is known to biomagnify up the food chain [e.g., dioxins, polychlorinated biphenyls (PCBs)], top wildlife predators could be at greater risk than benthic invertebrates.

Among the most significant considerations in the assessment of sediment exposure pathways are the quality of the available sediment data, the nature and size of the exposure area, the statistics used to estimate exposure concentrations, and the presence and evaluation of elevated concentrations (e.g., hot spots) of COCs. These topics are elaborated upon more fully in the subsequent sections. The reader is further encouraged to obtain additional guidance directly available from the U.S. EPA on ERA methods.

The TRRP rule [30 TAC 350.51(a–b)] requires relevant and sufficient data for the assessment of ecological exposures to sediments. To meet this requirement, the TCEQ encourages early discussion with its risk assessors (and Natural Resource Trustees as appropriate) regarding data proposed for use in assessments of sediment exposure. This could entail, for example, the development of an optional work plan for sampling and analysis. It could also include discussion of other data collected from previous investigations at the affected property. The intent of the early dialogue is to ensure that only data considered relevant and appropriate are used to support the risk assessment. TRRP-15eco provides additional discussion on sampling sediment, including—

Sampling pore water: Benthic invertebrate exposures to sediment COCs may occur through direct contact or ingestion of COCs in bulk sediment, and through exposure to dissolved COCs present in sediment pore water. *Pore water* is generally defined as the water in the spaces between grains of sediment; it can have its origin as various proportions of either surface water or groundwater.

Although bulk sediment samples are typically collected to support ERAs, there are situations where collection of sediment pore water is appropriate, usually in addition to bulk sediment sampling and analysis.

Pore water analysis, in conjunction with bulk sediment analysis, may provide an additional measure of COC bioavailability for some receptors and sediment-associated pollutants (e.g., U.S. EPA 2005a, 2008a). Pore-water data can confirm predictions of equilibrium-partitioning theory and is useful in assessing releases of impacted groundwater to sediment. Also see **10.2.5.3.**

Sediment sampling depth: The TRRP provides for a site-specific determination of the point of exposure (POE) for ecological receptors exposed to sediment. The depth of the sample should target the aerobic layer, which represents more recent deposition and is where most benthic infauna will occur. Ideally, the depth of the biotic zone at a specific location is best derived from sampling the area. The depth of bioturbation and the degree of contact between biota, sediment and pore water is influenced by the life habits of the resident organisms (e.g., degree of motility, creation of temporary versus permanent burrows, whether tubicolous or not) and their local environment.

U.S. EPA (2015) delineates biotic zones for a variety of habitats, such as estuarine intertidal, estuarine subtidal, tidal freshwater, lentic (e.g., lake or pond) and lotic (e.g., stream or river) systems. This document defines the biotic zone (based on benthic abundance) in most estuarine and tidal freshwater environments to be 10–15 centimeters (cm) (4-6 inches). In marine muds (coastal and offshore) and lentic environments, the biotic zone is 15 cm (6 inches). The biotic zone in lotic systems can vary from 15 to 35 cm (6–14 inches) depending upon type of water and habitat (U.S. EPA, 2015).

The person will need to justify the sample depth used in the ERA based on sediment characteristics observed during sampling. Observations of differing color intervals, texture and consistency, and biological inclusions (worm tubes, evidence of movement) may help distinguish between the biologically active layer and deeper layers.[8] Sampling crews should be aware that it is important to make these observations and judgments in the field when they are collecting sediment samples.

Consideration of remedial alternatives and physical mechanisms such as deposition and erosion (e.g., scouring), may dictate sampling at deeper depths. The TRRP defines the sediment POE for human health as the upper 1 foot of sediment [see 30 TAC 350.37(k)]. Therefore, samples collected to evaluate human health pathways may be inappropriate for ERAs unless the biologically active zone extends to that depth.

Sediment vs. soil in intermittent drainages: The TRRP rule [30 TAC 350.4(a)(79)] denotes the non-suspended particulate material lying below surface waters, including intermittent streams, as *sediment*. When a water body is dry, it is reasonable to assume that terrestrial wildlife receptors could forage in the dry streambed. Thus, it is appropriate to evaluate ecological exposure to both the dry streambed (as soil) and the sediment associated with intermittent streams when water is present.

Consider that a terrestrial receptor (e.g., rabbit) may forage along the dry stream bottom during arid times, and that an aquatic-based receptor (e.g., marsh wren) may forage within the stream when it contains water. The exposure duration for a receptor can be adjusted to reflect the usual dry and wet cycles for the water body in question.[9]

The TCEQ allows the evaluation of one scenario or the other based on site-specific considerations; however, a convincing, well-documented argument for not quantitatively evaluating the remaining scenario must be made in the uncertainty analysis.

2.9. Soil- and Sediment-Based Receptors in the PCL Database

The PCL Database:

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[8] Where site-specific circumstances warrant sampling in both the biologically active layer and in the deeper sediments below, vertical compositing of samples should be avoided. Horizontal compositing of localized sediments is fine; however, vertical composites can dilute or otherwise affect the analytical results such that the data do not reflect the true conditions in either the biologically active zone or the sediments at depth.

[9] In an intermittent stream, it would be appropriate to use the "Minor Habitat" (see **6.2.3**) from the Database for evaluation of exposure, as it includes both terrestrial and aquatic receptors characteristic of reduced habitats.

- Provides soil and sediment wildlife-based PCLs for terrestrial receptors (e.g., American robin) and aquatic receptors (e.g., spotted sandpiper), respectively.
- Distinguishes between soil-based PCLs and sediment-based PCLs through the dietary compositions of the species.

For example, the American robin's diet consists of soil invertebrates and fruits and is therefore a terrestrial receptor with soil-based PCLs. However, the spotted sandpiper's diet consists entirely of benthic invertebrates and the sediment it ingests while probing for this prey and is therefore an aquatic receptor with sediment-based PCLs.

When determining an assessment level or PCL from the PCL Database, **it is critical that the person distinguish between sediment-based receptors and those that are soil-based.**

Suppose the Freshwater Systems Habitat most closely represents the habitat at a site and the person wants to determine the sediment assessment level for cadmium from the PCL Database. The PCL Database automatically identifies the most conservative PCL from among those species in this habitat, which in this case, is for the robin, a soil-based receptor that is often found in this habitat. However, that value is significantly less than the value for the spotted sandpiper, which is the sediment-based receptor with the most conservative PCL. Using the conservative PCL for the robin (instead of the sandpiper) as an assessment level for sediment would not only be inappropriate but would likely incur unnecessary remediation costs. To avoid this situation, those species that are considered terrestrial are identified with the two-character field "TR" following their names [e.g., American Robin (TR)] and the aquatic species are designated with "AQ" [e.g., Spotted Sandpiper (AQ)].

3.0 Tier 1 Exclusion Criteria Checklist and Associated Special Circumstances

The Tier 1 Checklist is a standardized form consisting of mostly non-technical questions that could be completed by someone who is familiar with the affected property. It must be completed for all affected properties subject to TRRP [30 TAC 350.77(b)] unless the person decides to begin the ecological evaluation at a higher tier. Also, as stated earlier, sites subject to response actions under Risk Reduction Standards 2 or 3 [30 TAC 335] are required to evaluate and protect ecological receptors and are encouraged to begin their ecological evaluation by completing a Tier 1 Checklist [Figure: 30 TAC 350.77(b)]. The rest of Chapter 3 describes the purpose, use, and major components of the checklist and how it should be interpreted and implemented. The case study (RG-263c) includes an example checklist and is presented in a separate document (see **1.3.8**).

3.1. Purpose

The purposes of the checklist are to characterize the ecological setting of the affected property and to determine the existence of complete and potentially significant ecological exposure pathways using exclusion criteria. As outlined in Figure 3.1, *exclusion criteria* refer to those conditions at an affected property that preclude the need for a formal ERA because ecological exposure pathways are incomplete or insignificant due to the nature of the affected property setting or the condition of media at the affected property. The checklist attempts to make an early, non–resource-intensive determination of the presence of complete and significant ecological exposure pathways using these exclusion criteria.

The completed Tier 1 Checklist should identify any significant ecological exposure pathways that are complete or reasonably anticipated to be complete. If the affected property meets the exclusion criteria, then the person has fulfilled the ERA obligation and is not required to conduct a Tier 2 or Tier 3 ERA unless changing circumstances result in the affected property not meeting the exclusion criteria. If the exclusion criteria cannot be met, then the person may submit a reasoned justification for ending the ERA as discussed in **3.5.1**, conduct an ESE to also conclude the ERA as discussed in **3.5.2**, or perform a Tier 2 SLERA or a Tier 3 SSERA.

3.2. Use

Based on preliminary information regarding conditions of the affected property, the checklist may be applied to confirm the presence or absence of any potentially complete pathways for ecological exposure. Alternatively, if an ecological exposure is already known or suspected, the person may elect to proceed directly to Tier 2 or Tier 3.

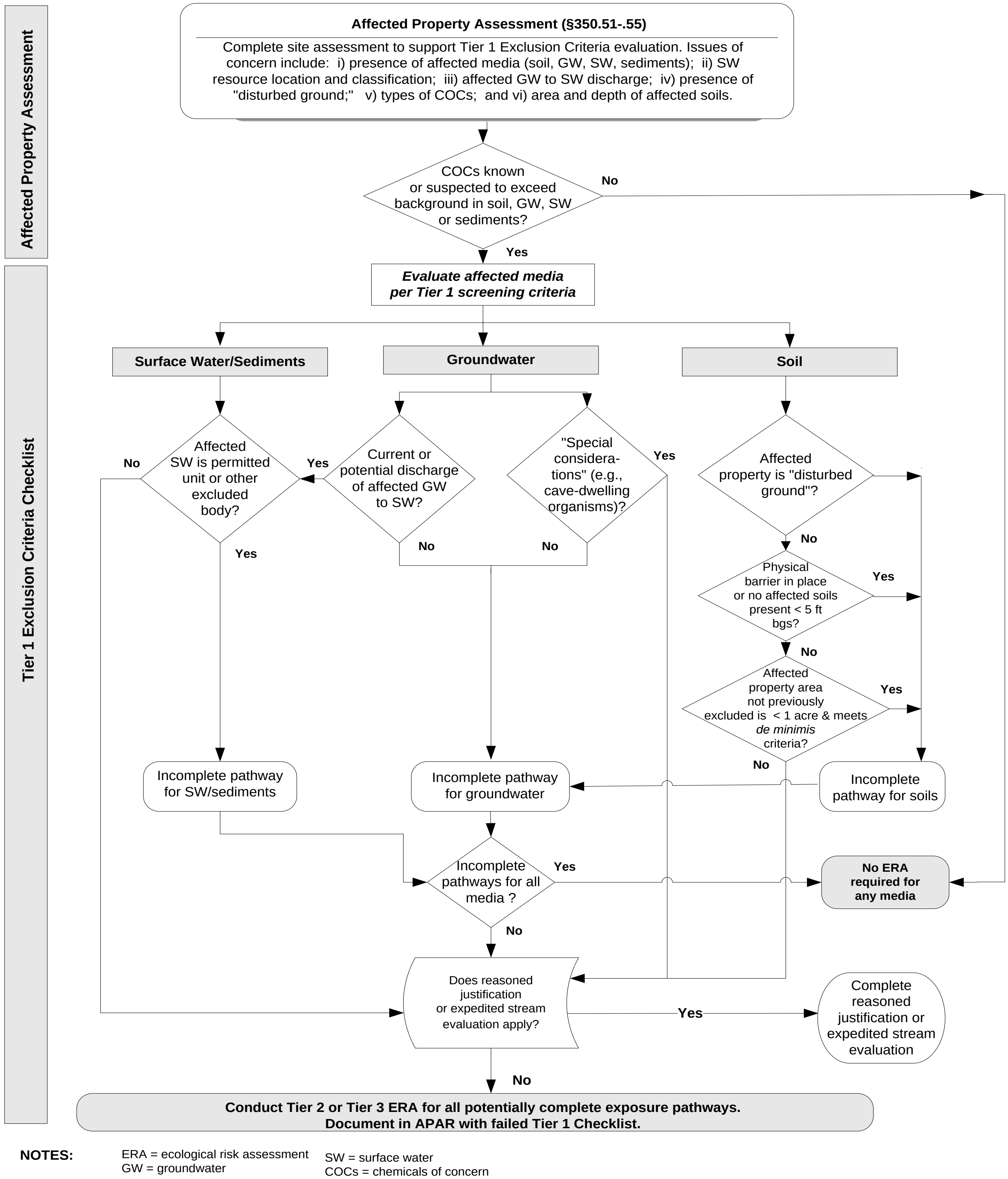


Figure 3.1. Tier 1 evaluation.

However, since the completion of the checklist may eliminate some ecological exposure pathways, it is advisable to begin all ecological evaluations at Tier 1 to better focus the assessment. The Tier 1 Checklist evaluates potential ecological exposure to affected soil, groundwater, or surface water and sediments. The checklist is designed for an early stage of the affected property assessment and, consequently, does not require detailed information on COC concentrations, the precise extent of affected media, or the specific ecological receptors (except for threatened or endangered species). Rather, as shown in Figure 3.1, general site conditions are evaluated to determine whether affected media are present at locations or, in the case of soils, over a sufficient area attractive to ecological receptors such that the receptors face significant exposure.

Except for the *de minimus* exclusion discussed in **3.3.3.4**, the presence of COCs should not be based on a comparison to human-health PCLs. As described earlier (**2.1**), each COC in any affected media must be delineated to an assessment level.

The checklist must be completed with existing information (i.e., any planned remediation may not be considered). However, before beginning a Tier 2 assessment, if the person believes that the implementation of response actions (e.g., covering the affected property with a cap) will effectively eliminate the ecological exposure pathways or render them insignificant, or if remediating to human-health PCLs will protect ecological receptors, then a “reasoned justification” [see 30 TAC 350.77(a) and 350.91(b)(7)] may be developed for ending the ERA by explaining how the planned remediation will eliminate or minimize the ecological exposure pathways (see **3.5.1**).

Accordingly, the person should submit the failed checklist and the reasoned justification in the APAR if the affected property is being addressed under TRRP, or as part of the specific remediation-program requirements when the site is being addressed under the RRR. Furthermore, this response action[10] should be mostly limited to soil; however, an ESE may be used to address releases into certain types of water bodies in lieu of having to conduct a Tier 2 ERA.

3.3. Checklist Instructions and Tips

Although some instructions for completion of the Tier 1 Checklist appear on the form itself, the information presented here may also be helpful.

When including attachments that provide supporting information (e.g., correspondence with wildlife-management agencies) or additional pages when more space is needed to answer a question, the person must ensure that these attachments are clearly identified. Also, supporting information may be referenced from other documents (e.g., an APAR) if the person clearly cites a

[10] There may be times when surface water and sediment are targeted for a response action to protect human health. When a surface water or sediment response action is considered for human-health protection, the person needs to evaluate whether this action would have a significant and highly disproportionate effect on ecological receptors [see 30 TAC 350.33(a)(3)]. Also, in conjunction with this response action, downstream surface water and sediment samples should be collected to evaluate COC fate and transport.

reference (i.e., document name, date, section, and page numbers). The following headings correspond to those within the checklist.

3.3.1. Preliminary Information

Name of facility: Include former names.

Affected property location: Include the county or counties, and the distance and direction from the nearest city.

Mailing address: If the facility is active, give its mailing address. If inactive, give the mailing address of the person.

Tracking, registration, and identification numbers: Include these only as applicable.

Definitions: A subset of the TRRP rule definitions to aid the person in completing the checklist.

3.3.2. Part I. Affected Property Identification and Background Information

These instructions are numbered to correspond with the text in the checklist Part I.

1. The information requested here may be repeated from other documents pertaining to the affected property or supplied as attached photocopies from those documents—or referenced with a clear citation. It is extremely important that the person supply or reference a map, aerial photo, or other photographs of the affected property, as these will greatly help the TCEQ evaluate the responses to the checklist questions.
2. Check all relevant boxes under "Known or Suspected COC Location." Cite any previously submitted information.
3. When completing this section, it is essential that the person have available the most current version of the TSWQS [30 TAC 307], linked at <www.tceq.texas.gov/goto/tswqs>. In addition, the person can locate the nearest surface water body and determine the closest classified segment in the watershed on a stream segment map. Segment maps are available at <www.tceq.texas.gov/goto/swqual-view>. Although unlikely, it is conceivable that the nearest water body may not be the impacted water body and that this portion of the checklist only addresses the nearest one.

The purpose of Part I is to ascertain the nearest surface water body (except for those excluded) that has received, or could receive, releases of COCs from the affected property. This includes waters that may be upgradient of the affected property or not even in the same watershed, but that could have been subject to airborne releases because they are located downwind of the affected property. The primary difference between Part I and the criterion question in Part II, Subpart A is to identify the nearest surface water body, whereas; Part II, Subpart A focuses on the completed pathway to any non-excluded surface water body (that may not be the nearest). In addressing Part I, the person need not be aware of all the potential pathways of exposure from the affected property and should simply identify the surface water body that is physically nearest the affected property.

3.3.3. Part II. Exclusion Criteria and Supportive Information

As indicated, the purposes of the exclusion criteria are to determine the presence of complete and potentially significant ecological exposure pathways at the affected property.

The rest of this subsection discusses each of the four exclusion criteria. The first concerns the surface water and sediment exposure pathway, while the other three address various aspects of the soil exposure pathway. In addition, supportive information—including references, examples, and links to websites—are included to assist the person in responding to the questions within the exclusion criteria.

3.3.3.1. Subpart A. Surface Water and Sediment Exposure

The purpose of this initial question is to identify those surface waters (and underlying sediments) that may be subject to further ecological evaluation because the water bodies have received, or may receive, releases of affected property COCs. Answering this question also identifies waters and sediments that should be excluded from further evaluation.

According to the Texas Surface Water Quality Standards [30 TAC 307.3(70)], surface water in the state is essentially all waters except those in authorized systems for treating waste. These “treatment system waters” are excluded from ERA consideration unless the treatment system is no longer authorized.

However, waste-treatment waters may be considered as sources of COCs if there is an unpermitted release from these waters into other Texas waters. Also, even permitted waste-treatment waters may be subject to other rules and regulations designed to protect ecological receptors—such as the Endangered Species Act or Migratory Bird Treaty Act. Furthermore, when a permitted outfall discharges into a water body, it is the treatment process waters preceding the outfall that are excluded from consideration, not the receiving water itself. For those rare occasions where the receiving water itself is designated as the permitted outfall, COCs regulated by the permit are excluded from evaluation (assuming permit limitations are met). However, impacts on the water body associated with a release of unpermitted COCs from the affected property should be evaluated in the ERA.

Conveyances, decorative ponds, and portions of unpermitted process facilities may be surface waters by definition, but if these are not ultimately in contact with other surface waters in Texas and are not “valuable habitat,” that is, are not used consistently or routinely as a feeding area or sanctuary by wildlife (e.g., migratory waterfowl), these waters may also be excluded from consideration. Obviously, these are judgmental decisions, but some things should be intuitive. For example, occasional observations of a few ducks swimming in a facility’s fire-water pond do not render that pond valuable habitat. On the other hand, continuous or seasonal observations of several species of waterfowl using an unpermitted evaporation pond for weeks at a time should indicate the presence of valuable habitat.

If the affected property has had a release to surface water or sediment, it fails the checklist and will have to undergo additional ecological evaluation. However, that does not necessarily entail a Tier 2 assessment. An ESE may be appropriate for the release of surface water or sediment, depending on the type of water body (**3.5.2**). In any case, the person should complete the remainder of the checklist to determine if there is a complete and significant soil exposure pathway. If the soil pathway is incomplete or insignificant, further evaluations need only focus on the surface water or sediment exposure pathway.

3.3.3.2. Subpart B. Affected Property Setting

Before this portion of the checklist can be addressed, the affected property must be determined not to be attractive to wildlife. Field observations and discussions with others who are also familiar with the affected property should be used to help determine attractiveness. If the affected property is attractive to ecological receptors (including protected species), the person should bypass the question about the setting of the affected property and proceed to Subpart C.

If needed, information on protected species is available from these wildlife-management agencies and, to a lesser extent, their websites:

- Texas Parks and Wildlife Department (TPWD) (Austin) <tpwd.texas.gov/gis/rtest/> and <tpwd.texas.gov/huntwild/wild/wildlife_diversity/nongame/listed-species/>.

- U.S. Fish and Wildlife Service (U.S. FWS) (Austin) <www.fws.gov/endangered/> and <www.fws.gov/southwest/es/AustinTexas/>

"Disturbed ground" primarily refers to a location that is predominantly urban or commercial-industrial in nature (and thus characterized by human presence and activities) where any habitat that may have once existed has been altered, impacted, or reduced to a degree such that it is no longer conducive to use by ecological receptors. Regarding what constitutes "disturbed ground," closed "waste control units" [defined in the TRRP rule at 30 TAC 350.4(a)(91)] with engineered covers are considered disturbed ground, provided they are meeting their design specifications.

Circumstances surrounding the presence of crops, pastureland, or golf courses associated with the affected property require case-by-case evaluation.

- Golf courses are usually considered disturbed ground and are mostly not attractive to ecological receptors. However, if the release or potential release is to a wooded, isolated portion of the golf course, ecological receptors may be present and thus warrant further investigation. Also, if the release or potential release is to waters or sediments within the golf course, the person will be required to conduct further evaluation of those media unless the waters (including water hazards) meet the Subpart A criteria (i.e., the waters qualify as a decorative pond that is not in contact with other surface water in the state and is not valuable habitat).

- Agricultural lands and pasturelands are not considered disturbed ground as they are attractive to wildlife.

Although this criterion is constructed as a “yes or no” question, the TCEQ acknowledges that sometimes a portion of the affected property may qualify for exclusion as disturbed ground. As shown in the case study (RG 263c), a portion of the fictional site was existing office buildings, impervious cover, and maintained landscape and was therefore excluded as disturbed ground. Conversely, the remainder of the site was potentially attractive to wildlife, not excluded and carried forward to the next subpart of the exclusion criteria.

3.3.3.3. Subpart C. Soil Exposure

The first 5 feet beneath ground surface are the zone of active root growth for most plants in the state and therefore the depth to which most burrowing animals will dig. The physical barrier mentioned in this criterion may be either natural (e.g., a geological formation) or of human construction (e.g., an asphalt or concrete parking lot).

3.3.3.4. Subpart D. De Minimus Land Area

The affected property must be able to meet **all four** of the qualifying conditions before the person can consider answering “yes” to the *de minimus* question. When evaluating the qualifying conditions, the person should contact the applicable wildlife-management agencies or consult other sources for information on protected species.

A sensitive environmental area is habitat that may require protection or special consideration because of the presence of certain ecological receptors and natural resources, or because legislative protection (national-monument status) has been conferred (U.S. EPA, 1999). Examples of sensitive environmental areas are listed below. Some local areas not listed below could serve important habitat functions that may require consideration, based on knowledge of local wildlife-management priorities. The person should identify all sensitive environmental areas within one-quarter mile of the affected property.

Examples of Sensitive Environmental Areas (modified from U.S. EPA, 1997a):

- Critical habitat for species designated as endangered or threatened by state or federal government—or any habitat known to be used by such species, or species proposed for such designation, or for which such designation is under review.

- Marine sanctuary.

- State or national park.

- Designated state or federal wilderness area.

- Any area identified under the Coastal Zone Management Act.
- Sensitive area identified under the National Estuary Program or Near Coastal Water Program.
- Critical areas identified under the Clean Lakes Program.
- State or national monument.
- National seashore recreational area.
- National lakeshore recreational area.
- State or national preserve.
- State or national wildlife refuge.
- Unit of Coastal Barrier Resources System.
- Coastal barrier (undeveloped or partially undeveloped).
- State or federal land designated for protection of natural ecosystems.
- Administratively proposed state or federal wilderness area.
- Spawning area critical for the maintenance of fish or shellfish species within river, lake, or coastal tidal waters.
- Migratory pathway or feeding area critical for maintenance of anadromous fish species within river or reaches or areas in lakes or coastal tidal waters in which the fish spend extended periods.
- Terrestrial area used for breeding by large or dense aggregations of animals.
- State or national river reach designated as recreational.
- Scenic or wild river so designated by state or federal government.
- Other state land designated for wildlife or game management
- Other state-designated natural area.
- Any area, relatively small, important to maintenance of aquatic life.
- Wetlands.

The last qualifying condition addresses the potential for the affected property to expand in size in the future. Site conditions, topography and COC fate and transport properties should be considered in evaluating the potential for the affected property to become larger than 1 acre.

If the affected property meets all four of the qualifying conditions, the person should apply human-health PCLs for the applicable land use to determine if the extent of the affected property for each COC is 1 acre or less. The affected property cannot be arbitrarily divided up into 1-acre units. For application of the *de minimus* concept, the assessment level is the lower of the Tier 1 combined soil PCL ($^{Tot}Soil_{Comb}$) and the soil-to-groundwater PCL ($^{GW}Soil$) appropriate for the groundwater classification. The soil-to-groundwater PCL may be established under Tier 1, 2, or 3. The Tier 1 human-health PCLs are available as PDF files or as Excel tables online at <www.tceq.texas.gov/goto/pcl>. The person completing the checklist under the RRR should contact the TCEQ ERA Program for additional information.

3.3.4. Part III. Qualitative Summary and Certification

The person should provide a summary of the information in the checklist, emphasizing why the exclusion criteria were or were not met and recommending the next ecological evaluation action, if appropriate. If the person decides to use a reasoned justification to conclude the ERA process, it may be referred to here as the next action, but it must be submitted separately. The person completing the checklist (e.g., the person's representative or consultant) must be identified in the first set of blank lines and the person must certify the information in the second set.

3.4. Checklist Review and Response

The completed checklist should be submitted to the TCEQ as part of the APAR if the affected property is being addressed under the TRRP or as part of the specific remediation-program requirements when the site is being addressed under the RRR. The person will make the initial decision regarding the need for further ecological evaluation. The project manager will review the checklist and may or may not concur with the person's decision. The project manager may consult with the agency's ERA staff. Completed checklists that indicate the presence of possible ecological exposures accompanied by a separate reasoned justification or ESE (**3.5.2**) will be reviewed by the ERA staff. The project manager will notify the person in writing regarding the approval or disapproval of the checklist and any reasoned justification or ESE (see **3.5**).

In summary, a few important things to remember about the Tier 1 Checklist:

- The checklist can be used for both the RRR and the TRRP rule.
- TCEQ project managers are responsible for review and approval or disapproval.
- Only existing information may be used—planned remediation cannot be considered.

- Knowledge of the ecological receptors present (except for protected species) and the concentrations of COCs that may affect them are not required.
- Agricultural land and pastureland are not considered "disturbed ground."
- For the *de minimus* criterion for exclusion, the extent is based on human-health PCLs (lower of the Tier 1 total soil combined PCL and the soil to groundwater PCL, as applicable),[11] and the affected property cannot be arbitrarily divided up into 1-acre units.
- Completed checklists that indicate the presence of possible ecological exposures, along with a separate reasoned justification for ending the ERA based on planned remediation (**3.5.1**), can be submitted in lieu of proceeding to Tier 2.
- Completed checklists that indicate a complete surface water–sediment exposure pathway, along with a separate ESE (**3.5.2**) for qualifying water bodies, can be submitted in lieu of proceeding to Tier 2 for surface water and sediment exposure pathways.

3.5. Special Circumstances Associated with the Checklist

After completing the Tier 1 Checklist, if it is determined that either the soil or surface water and sediment exposure pathway is complete and significant, further evaluation through a standard Tier 2 SLERA may not be necessary. As described below, if site circumstances are such that a planned response action will eliminate the soil exposure pathway or if the surface water exposure pathway is associated with an intermittent stream, a focused evaluation can be submitted to address the potential ecological risk.

3.5.1. Reasoned Justification

In general, the TCEQ supports the early closure of sites and their exclusion from the ERA process where appropriate. To this end, the "reasoned justification" clause in the TRRP rule [see 30 TAC 350.77(a)] considers a planned response action (for any reason) that addresses ecological exposure, thus allowing the ERA to be concluded without the need for a Tier 2 or 3 ERA. Reasoned justifications (RJs) should be formally incorporated into the APAR [30 TAC 350.77(a) and 350.91(b)(7)] and should contain:

- a clear statement that indicates the document is an RJ proposal;

[11] Alternatively, the person can use any available ecological PCLs, including those in the PCL Database.

- an identification of all potential ecological exposure pathways and, if applicable, how human health PCLs (e.g., total soil combined PCLs) are protective of ecological receptors; and

- a discussion, if applicable, of how the proposed remediation will address ecological risk by eliminating ecological exposure pathways.

Additionally, RJs can be proposed that solely rely on imminent development that will eliminate ecological exposure pathways. Supporting documentation demonstrating a commitment to the project for these RJs should include:

- a general description of the proposed project;

- a schedule, including its start and completion dates and milestone events (e.g., acquisition of any permits);

- maps illustrating the footprint of the proposed development in relation to the affected property that did not meet the Tier 1 exclusion criteria;

- other supporting statements regarding current or anticipated use and occupancy; and

- an affirmation that construction (e.g., ground breaking, clearing) will begin within 1 year of TCEQ approval of the RJ.

If site development is implemented as expected, the TCEQ project manager will ultimately require documentation that confirms the development has eliminated any previous ecological exposure pathways. Since this will be viewed as a response action intended to eliminate an ecological exposure pathway, the development-based RJ (or any other RJ, as appropriate) should be discussed in routine status reports. Documentation could include the status (as in percent complete) of the construction progress, photographs, as-built drawings or figures, and other information detailed in the APAR as requested by the project manager.

For VCP sites, confirmation that ecological exposure pathways have been addressed will be necessary before a conditional or final certificate (whichever comes first) can be approved.

All RJs will be reviewed by the TCEQ's ERA staff.

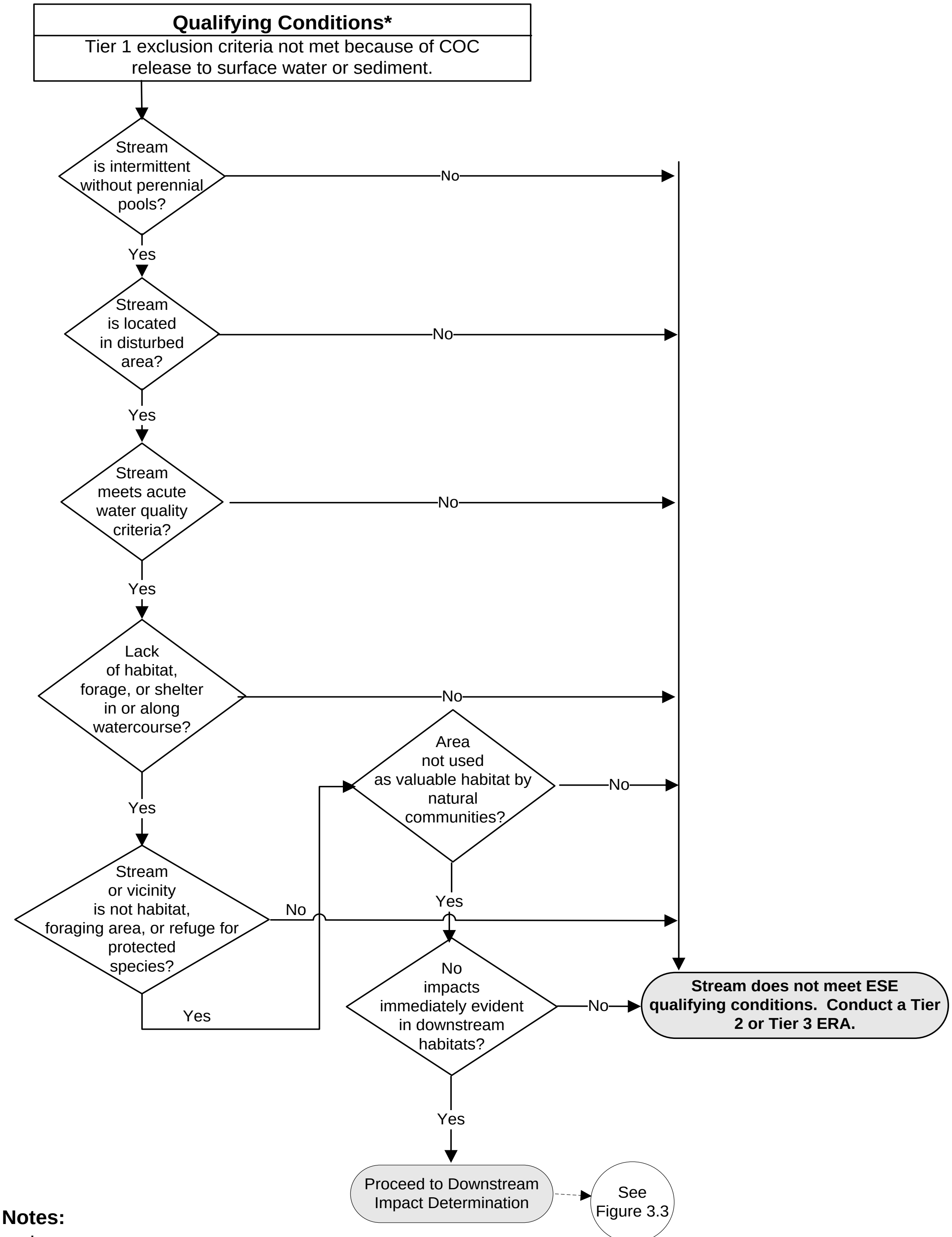
3.5.2. Expedited Stream Evaluation

An "expedited stream evaluation" or ESE [see 30 TAC 350.77(a) and 350.91(b)(7)] can be used to determine that, although a COC release to surface water or associated sediment is a complete pathway, it may not be a significant pathway when the water body and its surroundings meet certain conditions. More specifically, the release must be into an intermittent stream (without perennial pools) that does not support a benthic community requiring a PCL (see the conditions discussed in **3.5.2.1**). In addition, there should be no immediately apparent downstream impacts.

If not meeting the criterion for the surface water–sediment pathway is the only reason the checklist failed (i.e., the soil exposure pathway proved to be incomplete or insignificant), the ESE may be used to conclude the ERA.

Although somewhat similar in rationale, this evaluation is not part of the Tier 1 Checklist contained in the TRRP rule. Also, an ESE cannot be used as the reasoned justification (**3.5.1**) because of the potential for continuing ecological exposure downstream due to transport of COCs.

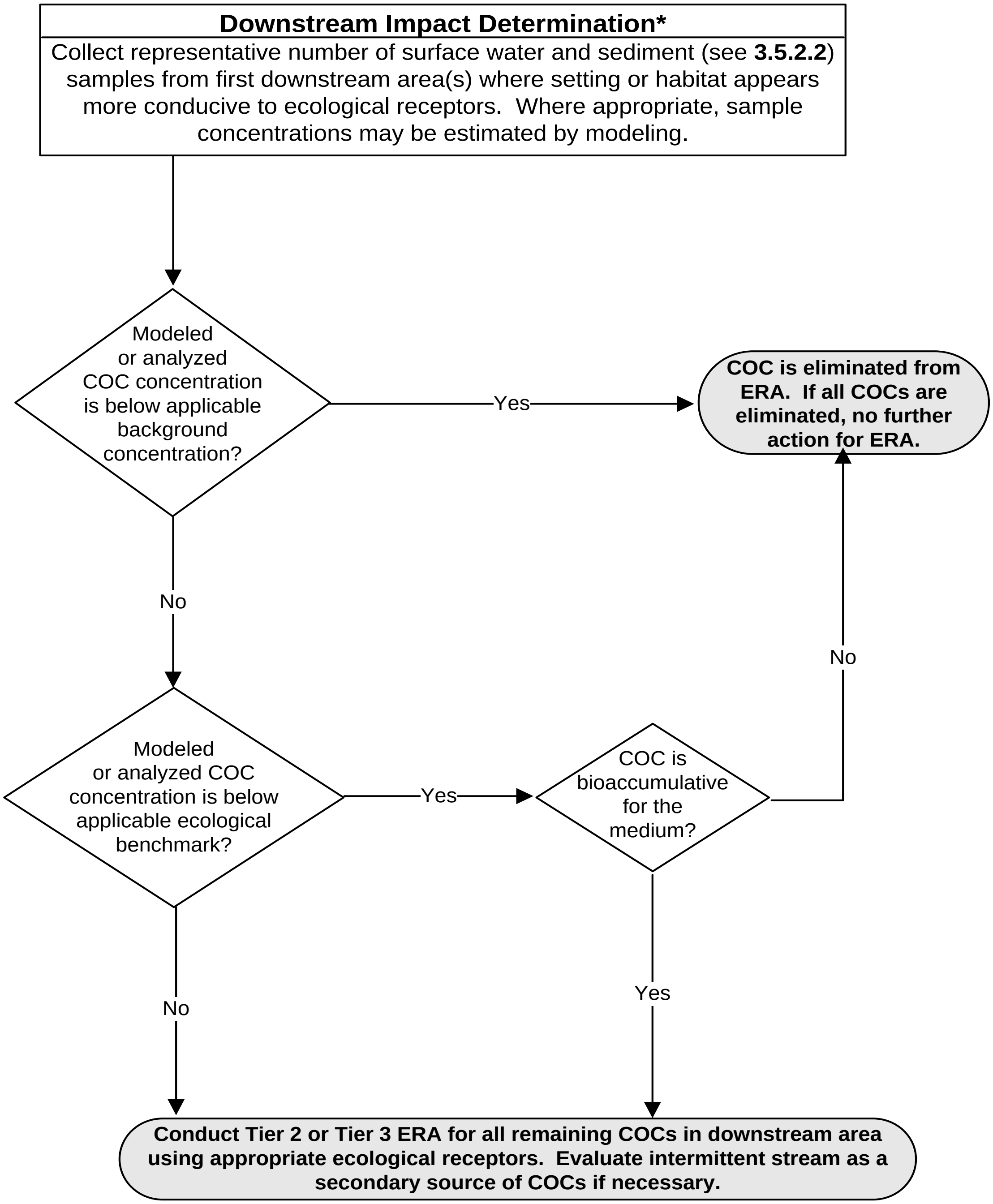
If the water body qualifies for the ESE, then there is no need to perform a Tier 2 ERA on the intermittent section of the stream or ditch. That evaluation moves downstream to an area that is more conducive to aquatic life and wildlife. The ESE process is depicted in Figures 3.2 and 3.3 and is described in **3.5.2.1** and **3.5.2.2**.



Notes:

* = requires supporting photo documentation
COC = chemical of concern
ERA = ecological risk assessment
ESE = expedited stream evaluation

Figure 3.2. Expedited stream evaluation: qualifying conditions.



Notes:

* = requires supporting photo documentation
COC = chemical of concern
ERA = ecological risk assessment

Figure 3.3. Expedited stream evaluation: determining downstream impact.

3.5.2.1. Qualifications for Expedited Stream Evaluation

According to the Procedures to Implement the Texas Surface Water Quality Standards or *"Implementation Procedures"* (TCEQ, 2010a) as amended, an "intermittent" stream is one which has a period of zero flow for at least one week during most years and is considered "intermittent with perennial pools" when adequate pools persist that would be expected to provide habitat for significant aquatic life use. Perennial-pool status is determined case by case, using available data and best professional judgment. As discussed in **6.1**, for intermittent streams without perennial pools associated with the affected property, the person may not be required to develop sediment PCLs for protection of the benthic community. If the intermittent stream in question does not support a viable benthic community, then ecological PCLs for higher-trophic-level receptors are likely not necessary because of a lack of associated aquatic habitat and receptors. As illustrated in Figure 3.2, to determine if an ESE is appropriate, these conditions must be met:

- The stream is intermittent (dries up completely at least one week a year) without perennial pools. [Intermittent streams with perennial pools are defined at 30 TAC 307.3(a)(35) and discussed in the Implementation Procedures].

- The stream is in a disturbed area (generally, a predominantly urban or commercial-industrial setting).

- The stream meets the acute water quality criteria specified in 30 TAC 307.6, Table 1, or appropriate surrogate values if no criterion is specified.

- There is a lack of appreciable in-stream, edge, or riparian habitat, forage, or shelter in or along the watercourse.

- The watercourse or surrounding vicinity is not known to serve as habitat, foraging area, or refuge to protected species.

- The area is not consistently or routinely used as valuable habitat for natural communities including birds, mammals, reptiles, etc.

- No impacts are immediately evident in downstream areas where habitat is more likely to support wildlife.

Compliance with these conditions should be supported by photographic evidence. If these conditions are met, the stream needs no further evaluation (i.e., a Tier 2 SLERA) unless, as discussed in **3.5.2.2**, more thorough downstream analyses reveal impacts. In that case, the stream may need to be evaluated as a potential secondary source of COCs. If any one of these conditions is not met, then the person will need to conduct a Tier 2 SLERA that includes the intermittent stream, as well as any downstream areas that may be impacted.

3.5.2.2. Determining Downstream Impacts

As presented in Figure 3.2, the ESE must include the collection of downstream surface water and sediment samples or modeled downstream concentrations, and comparison of the reported or estimated concentrations to background (upstream) concentrations and the ecological benchmarks listed in the Benchmark Tables. Sampling is needed in the first downstream area (or areas) where the setting or habitat appears more conducive to aquatic life (e.g., a large pool) or wildlife (e.g., riparian vegetation). As discussed below, photo documentation of the sampling locations is recommended. Surface water and sediment samples should be collocated and collected from depositional areas (e.g., pools, point bars on the inside banks of streams). The number of samples will depend on site-specific circumstances, considering spatial scale of the potential depositional area. However, sample numbers should be sufficient for statistical analysis. When the COC release to the surface water is through a groundwater discharge, surface water concentrations should be estimated according to the TRRP rule at 30 TAC 350.37(i) and 350.75(i)(4) and as discussed in TRRP-24.

If the maximum COC concentrations are below the greater of background or benchmarks, then the evaluation may be concluded if no bioaccumulative COCs are present. However, if concentrations are greater than both background and benchmarks, or if COC concentrations exceed background and the COCs are considered bioaccumulative (see Table 5.1), then the person will need to conduct a Tier 2 ERA for the downstream portion of the water body. Although this approach appears like required element 1 of a traditional Tier 2 assessment, it differs in that the COC data will not be based on samples collected from the intermittent stream, but rather from an appropriate downstream location, where COC concentrations are expected to decrease with distance. If a downstream concentration does exceed the appropriate background or benchmark, then the intermittent stream may need to be evaluated as a secondary source of COCs.

3.5.2.3. Determining Risks to Upper Trophic Level Receptors

This guide (see **6.1**) identifies certain water bodies and conditions where the benthic community may be diminished for reasons unrelated to releases of COCs from an affected property subject to the TRRP rule. For these water bodies (e.g., intermittent streams, creeks, or ditches, without perennial pools, or those that are lined with concrete on the bottom and sides), the TCEQ believes it is unnecessary to determine an ecological PCL for sediment that is protective of the benthic invertebrate community. **However, this does not preclude an evaluation of risks to higher trophic level organisms that may forage in these types of water bodies or nearby water bodies (that could become impacted because of sediment COC transport**). The ESE takes a subset of those water bodies identified as not needing a benthic PCL (i.e., intermittent streams without perennial pools) and determines if there is a need to develop PCLs for the higher trophic level receptors, without going through a formal Tier 2 assessment. If the water body qualifies for the ESE, then there is no need to perform a Tier 2 ERA on the intermittent section of the stream or ditch. That evaluation moves downstream to an area that is more conducive to aquatic life and wildlife. To restate, just because a water body is recognized as not needing a benthic PCL, that does not preclude the evaluation of risk to higher trophic level receptors, either through a Tier 2 assessment, or in the case of intermittent streams without perennial pools, through an ESE.

3.5.2.4. Reporting and Review

Any decisions regarding the qualification and appropriate application of the ESE to a water body should be supported by photographs. The person should provide photo documentation of the intermittent stream, the surrounding area, the downstream area, and the sampling locations.

If the results of the ESE indicate that there are no downstream impacts and the Tier 1 Checklist determines that the soil exposure pathway is incomplete or insignificant, the person should submit the failed checklist and the ESE (including analytical data) in the APAR. A summary statement should indicate, based on the checklist and the ESE, that there is no significant ecological risk, and the ERA should be concluded. As with the reasoned justifications, the checklist and ESE will be reviewed by the ERA staff. The TCEQ project manager will notify the person in writing regarding the approval or disapproval of the checklist and the ESE.

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4.0 Tier 2: Screening-Level Ecological Risk Assessment

The purposes of the Tier 2 SLERA are to scientifically eliminate COCs that do not pose an ecological risk and to develop PCLs for those COCs that do pose an unacceptable risk to selected ecological receptors. The Tier 2 SLERA serves to identify COCs, exposure pathways, and ecological receptors of concern based on application of default exposure assumptions and literature-based effect levels. Although the Tier 2 SLERA (Figure 4.1) has been designed to minimize effort, it will probably need to be conducted by an environmental professional. The person who undertakes a Tier 2 evaluation will need to meet several required elements. However, within the required elements, there are four potential exit points from the ERA process before having to develop ecological PCLs. The list of required elements for a Tier 2 SLERA appears in **4.2**; a discussion of each element occurs in **5.0** through **14.0**. The person conducting a SLERA may exit the process, or at least eliminate some COCs or media, if the conditions of required elements 1, 6, 7, or 8 are met. The case study includes an example SLERA and is presented in a separate publication, RG-263c (see **1.3.8**).

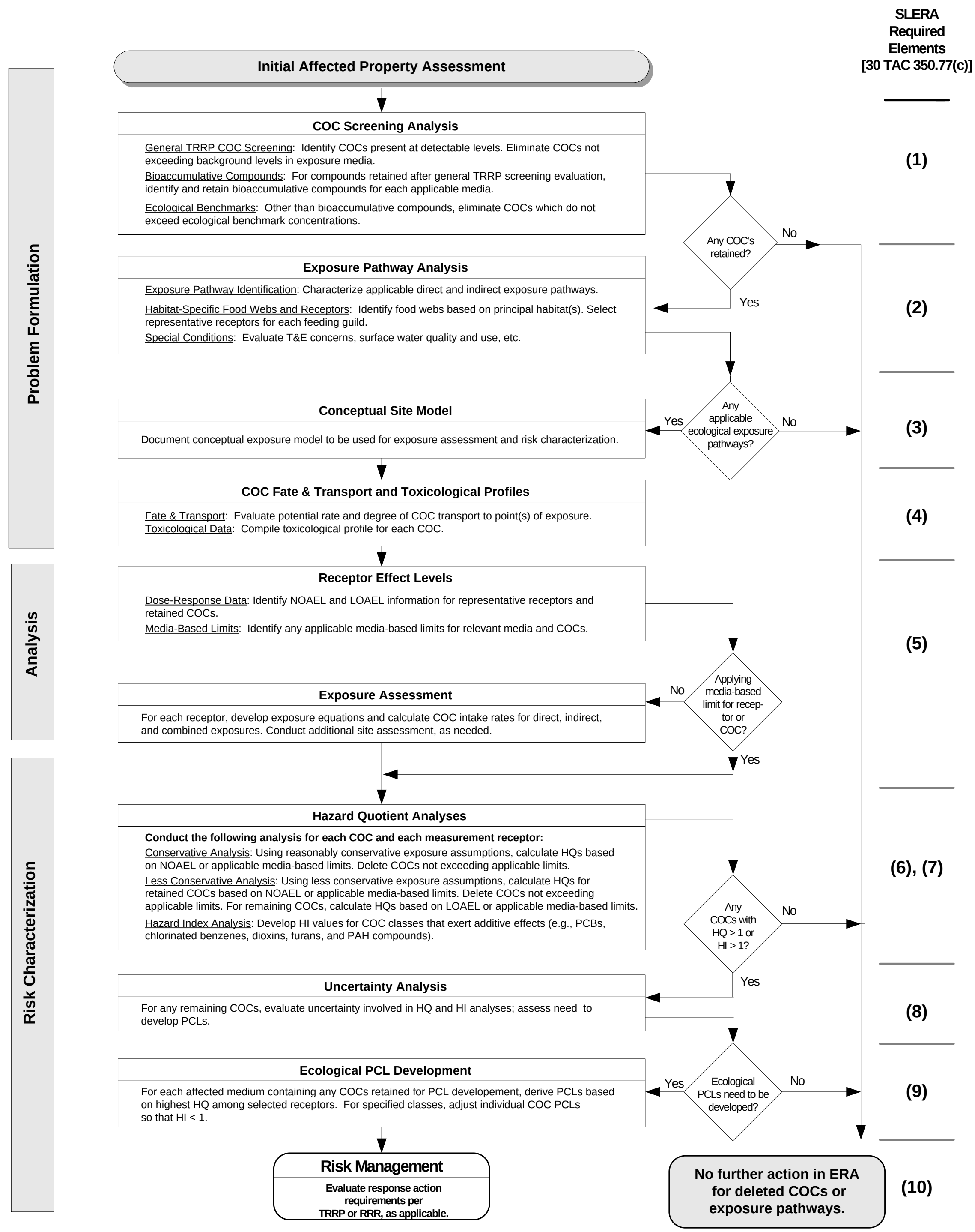
4.1. Phases of a Screening-Level Ecological Risk Assessment

The TRRP rule at 350.77(c) states that the SLERA should contain the three widely acknowledged phases of an ERA. These phases, as described in U.S. EPA (1992a) and as illustrated in Figure 4.2 are:

1. *Problem formulation,* which establishes the goals, breadth, and focus of the assessment.
2. *Analysis,* which consists of the technical evaluation of data on both the exposure of the ecological receptor to a chemical stressor and the potential adverse effects.
3. *Risk characterization,* where the likelihood of adverse effects occurring because of exposure to a chemical stressor is evaluated.

4.1.1. Problem Formulation

Problem formulation is the first phase of the SLERA and establishes the goals, breadth, and focus of the assessment. It is a systematic planning step that identifies the major factors (e.g., size and ecology of the affected property, identity, and distribution of COCs, and potential ecological receptors) to be considered in the assessment. These factors determine the scope of the ERA (U.S. EPA, 1997a).



NOTES:

COC = chemical of concern
ERA = ecological risk assessment
HI = hazard index
HQ = hazard quotient
LOAEL = lowest observed adverse effect level
NOAEL = no observed adverse effect level
PAH = polycyclic aromatic hydrocarbons
PCB = polychlorinated biphenyls
PCL = protective concentration level
RRR = risk reduction rule (30 TAC 335)
SLERA = screening-level ecological risk assessment
T&E = threatened and endangered species
TRRP = Texas Risk Reduction Program (30 TAC 350)

Figure 4.1. Tier 2 screening-level ecological risk assessment.

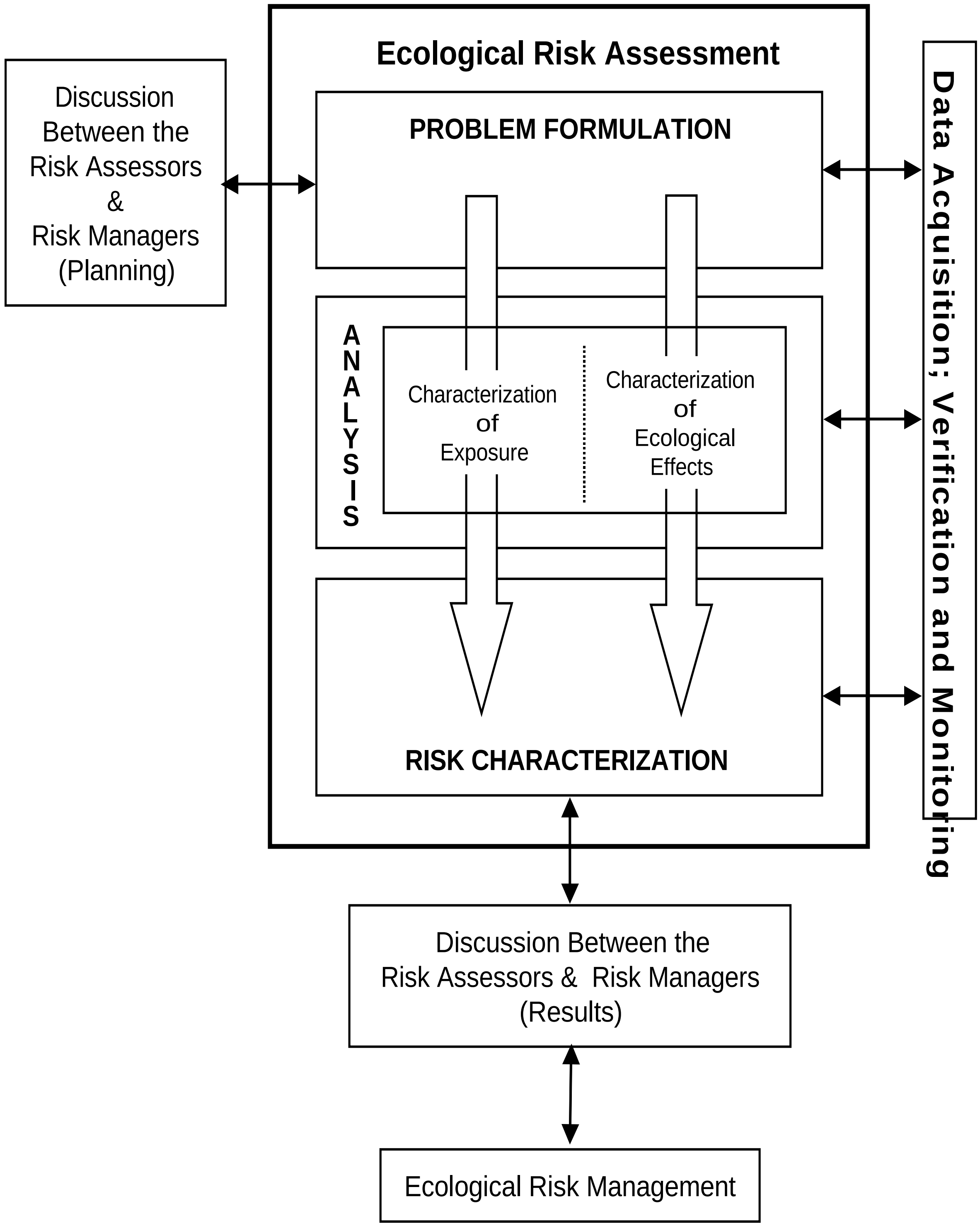


Figure 4.2. Framework for ecological risk assessment.

(modified from U.S. EPA 1992a)

In the context of this guide, the "problem formulation" phase corresponds to required elements 1 through 4. As shown in Figure 4.1, problem formulation includes:

- environmental setting and COC screening (see **5.0**)[12]
- exposure pathway analysis including habitats and receptors (see **6.0**)
- conceptual site model development (see **7.0**)
- COC fate and transport, toxicological profile development (see **8.0**)

The problem formulation phase should also include discussions between risk assessors and risk managers and other stakeholders to identify the stressor characteristics, ecosystems potentially at risk, and ecological effects to be evaluated. During problem formulation, assessment, and measurement endpoints for the SLERA are identified.

A product of problem formulation is a conceptual site model for the SLERA that describes how a given stressor might affect ecological components of the environment. The conceptual model also describes questions about how stressors affect the assessment endpoints, the relationships among the assessment and measurement endpoints, the data required to answer the questions, and the methods that will be used to analyze the data (U.S. EPA, 1997a) (see **7.0**).

4.1.2. Analysis

As illustrated in Figure 4.1, the analysis phase connects problem formulation with risk characterization through an examination of the exposure of a measurement receptor to a COC, and the ecological effects resulting from that exposure. Exposure estimates emphasize contact and uptake into species, while estimates of effects frequently entail extrapolation from test species to the species of interest.

Exposure is the contact of a receptor with a COC (e.g., ingestion). Exposures of ecological receptors to COCs released from affected properties are evaluated through consideration of exposure pathways. As discussed in **7.0**, all exposure pathways that are potentially complete should be evaluated, absent a reasonable justification. The existence of a potentially complete exposure pathway indicates that a receptor may contact a COC; it does not necessarily indicate that a receptor will be adversely affected.

Exposure pathways considered herein include all direct uptake pathways of a COC from affected media (e.g., soil, sediment, and surface water) for lower-trophic-level receptors evaluated at the community level, and ingestion of COC-

[12] When the environmental setting or the list of COCs has been previously discussed in another document, the person need only clearly cite that document.

laden food or prey items or media for higher-trophic-level receptors. Note that exposure pathways currently not addressed[13] in this guide due to limited data on exposure effects include (1) inhalation and dermal exposure for higher-trophic-level organisms, (2) ingestion via grooming and preening, and (3) foliar uptake and direct deposition uptake of COCs by plants.

The ecological effects of a COC are assessed by identifying toxicity reference values (TRVs) specific to a COC and the measurement receptor or receptor class being evaluated. During risk characterization, TRVs are set as the denominator when computing ecological hazard quotients (HQs) The TRVs used in risk characterization for lower-trophic-level communities are typically media specific, whereas TRVs for upper-trophic-level receptors are given in terms of a dose. As used here, the analysis phase corresponds to required element 5 (see **9.0, 10.0**).

4.1.3. Risk Characterization

In risk characterization, data on exposure and effects are integrated into a statement about risk to the assessment endpoints established during problem formulation (U.S. EPA, 1997a). Risk characterization is the final phase of risk assessment and includes two major components: *risk estimation* and *risk description*.

- Risk estimation is an integration of the exposure assessment and the ecological effects or toxicity assessment to determine the potential risk to a community or feeding guild from exposure to a COC. Risk is estimated using the HQ method.

- Risk description depicts the magnitude and nature of potential risk for each community and guild, based on the quantitative results of the risk estimation and calculated HQ values by summarizing the associated uncertainties and identifying a threshold for adverse effects on the assessment endpoints.

To estimate potential ecological risk, an HQ should be calculated specific to each measurement receptor, COC, and exposure-scenario location evaluated in the risk assessment (U.S. EPA, 1992a).

According to the Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities or “Combustion guide” (U.S. EPA 1999), risk description considers the magnitude and nature of potential risk for community and class-specific guild measurement receptors evaluated and supplies information for the risk managers to evaluate the significance of an HQ value. Risk description also discusses the significance of the default assumptions used to assess exposure, because they affect the magnitude and certainty of the calculated HQ value. The resultant risk characterization should consider any major uncertainties and limitations associated with results generated in performing the screening-level risk assessment, because

[13] Should toxicological and exposure data become available for these pathways, the TCEQ may require these evaluations.

uncertainty can be introduced into a risk assessment at every step of the process.

The final outputs of the risk-characterization phase are COC concentrations in each environmental medium that bound the threshold for estimated adverse ecological effects given the uncertainty inherent in the data and models used (U.S. EPA, 1997a). The lower bound of the threshold will be based on reasonably conservative assumptions and toxicity values based on the NOAEL. The upper bound will be based on observed impacts or predictions that ecological impacts could occur. This upper bound will be developed using justified less conservative exposure assumptions, site-specific data, and LOAEL toxicity values. As used in this guide, the risk-characterization phase corresponds to required elements 6–9, which are listed in **4.3** and discussed in **11.0** through **13.0**.

4.2. SLERA: Required Elements

The TRRP rule [30 TAC 350.77(c)] establishes 10 minimum requirements to be satisfied when completing a Tier 2 SLERA. The person shall:

- Compare concentrations of non-bioaccumulative COCs at the affected property against established ecological benchmarks or use approved methodologies to develop benchmarks to determine potential effects and to eliminate COCs that pose no unacceptable ecological risk. If all COCs are eliminated at this point, the assessment ends (see **5.0**).
- Identify communities (e.g., soil invertebrates, benthic invertebrates) and major feeding guilds (e.g., omnivorous mammals, piscivorous birds) and their representative species that are supported by habitats on the affected property for each exposure pathway that is complete or reasonably anticipated to be completed (see **6.0**).
- Develop a conceptual model that graphically depicts the movement of COCs through media to communities and the feeding guilds (see **7.0**).
- Discuss COC fate and transport and toxicological profiles (see **8.0**).
- Prepare a list of input data including values from the literature (e.g., exposure factors, intake equations that account for total exposure, values for the NOAEL and LOAEL, references), any available site-specific data, and reasonably conservative exposure assumptions, then calculate the total exposure to selected ecological receptors from each COC not eliminated according to required element 1. Present these calculations in tables or spreadsheets (see **9.0** and **10.0**).
- Use an ecological HQ methodology to compare exposures to the NOAELs to eliminate COCs that pose no unacceptable risk (i.e., NOAEL HQ ≤ 1); however, when multiple members of a class of COCs that exert additive effects, an ecological hazard index (HI) methodology is also appropriate. If all COCs are eliminated at this point, the assessment ends (see **11.0**).

- Justify the use of less conservative assumptions (e.g., a larger home range) to adjust the exposure and repeat the HQ exercise in required element 6, again eliminating COCs that pose no unacceptable risk based on comparisons to the NOAELs and adding another set of comparisons, this time to the LOAELs for those COCs indicating a potential risk (i.e., a NOAEL HQ > 1); however, when multiple members of a class of COCs are present whose effects are additive effects, an ecological HI methodology is also appropriate. If all COCs are eliminated at this point, the assessment process ends (see **11.0**).
- Analyze the major areas of uncertainty associated with the SLERA, including a justification for not developing PCLs for certain COCs and pathways, if appropriate (e.g., a statement that the NOAEL HQ > 1 > LOAEL HQ, an evaluation of the likelihood of ecological risk, a discussion of the half-life of the COCs). However, when multiple members of a class of COCs with additive effects are present, an ecological HI methodology is also appropriate. If all COCs are eliminated at this point, the ecological risk assessment process ends (see **12.0**).
- Calculate medium-specific PCLs bounded by the NOAEL and the LOAEL used in item 7 for those COCs that are not eliminated because of the HQ exercises or the uncertainty analysis (see **13.0**).
- Make a recommendation for managing ecological risk at the affected property based on the final ecological PCLs, unless proceeding under Tier 3 (this procedure may be included as part of the affected property assessment report, the self-implementation notice, or the response action plan) (see **14.0**).

4.3. Development of a SLERA Work Plan

If the affected property failed the Tier 1 Exclusion Criteria Checklist and a SLERA is needed, or if the person decides to begin the ERA with a Tier 2 SLERA, then it is advisable to prepare a work plan for TCEQ review or discussion. A work plan is not a TRRP requirement; however, development of a work plan with TCEQ involvement can contribute to an efficient and successful assessment.

If submission of a written work plan is not planned, a technical meeting to discuss and agree upon the basic components of the work plan could be beneficial.

When developing a SLERA work plan, the person should be mindful of the three phases of an ERA (see **4.1**), as well as the required elements discussed in **4.2.** In addition, the person should contact the ERA staff as early as possible in the development of the work plan to ensure that the plans for the SLERA are well designed and capable of answering the pertinent questions about the ecological effects of the COCs at an affected property.

The overall purpose of the work plan will be to present a detailed approach for conducting a SLERA. The most effective work plans are those that are tailored to the affected property's specific characteristics and concerns. The work plan should estimate the time frame for submission of the SLERA once the TCEQ (and possibly the Trustees) have commented on the draft SLERA work plan. Any additional site assessment to support the SLERA should be factored into the timetable. Ideally, submitting a work plan will minimize the need for protracted rounds of comment exchange after submission of the draft SLERA. Nevertheless, additional assessment (such as additional sampling and analyses, calculation revisions, evaluation of new toxicity information, and research) may be necessary based on the SLERA findings and the TCEQ review of the document.

4.3.1. Introduction

The work plan should briefly discuss the site history, explain why a TRRP investigation was initiated, and outline the primary objectives of the SLERA. In addition, any existing agreements specific to the affected property (e.g., background concentrations, COCs) should be identified in addition to any existing relevant legal agreements (e.g., administrative or enforcement orders). The work plan should also identify the principal references that will be used to prepare the SLERA.

4.3.2. Problem Formulation

As stated in **4.1.1**, problem formulation establishes the goals, breadth, and focus of the assessment. Thus, the problem, the purpose of the assessment, and the plan for analyzing and characterizing risk are defined. Therefore, this part of the work plan should identify the major factors considered in the assessment, including the affected property size and habitats, the identity and distribution of COCs, and the potential ecological receptors and exposure pathways. The overall objective is to characterize the general ecological setting from which specific assessment and measurement endpoints can be selected and can be linked together in a CSM.

4.3.2.1. Description of the Affected Property and Its Environmental Setting

A summary of the location of the affected property, its size, a history of the facility and any releases associated with the affected property, the known or suspected COCs, and a description of the current and anticipated future land uses of the site should be included. Remember that the size of the affected property is determined by the extent of contamination, not by the property boundaries of the facility [see 350.4(a)(1)]. Submitted information should include:

- For the location and size, maps indicating—
  - o the geographical area (town, county, quadrant, or other appropriate unit) in the vicinity;
  - o the locations of nearby surface waters; and
  - o the locations of potential contaminant sources.
- For the history and COCs, a description of the activities (historical and current) that resulted in a release. Information on chemical-

handling processes, storage locations, and known or potential contaminants and daughter products should be provided.

- For the environmental setting, any available information on: topography, nearby surface waters, ditches, and drainage routes; locations of ecological habitats such as wooded areas, grasslands, floodplains, and wetlands; groundwater–bearing units that may interface with surface water and sediments or discharge at seeps, and adjacent land uses (both current and projected).

4.3.2.2. COC Screening Analysis

The work plan should include a list of known and suspected COCs and describe the COC-selection process. If the program area has already determined and approved site-specific background concentrations, that information should be summarized. If site-specific background concentrations are going to be determined, the overall approach (e.g., sample locations, rationale, and statistics) should be briefly described and coordinated with the TCEQ program area. Bioaccumulative COCs in each medium should be identified. The sample locations and sample data should be tabulated and provided in figures.

The work plan should state that concentrations of non-bioaccumulative COCs will be compared against screening-level benchmarks to determine potential effects and to eliminate those that pose no unacceptable ecological risk (required element 1). Where default TCEQ benchmarks will not be used or no TCEQ benchmark exists, the work plan should discuss potentially applicable or suitably analogous toxicity benchmarks, or methods to derive those benchmarks that will be presented in the SLERA.

4.3.2.3. Exposure Pathway Analysis and Conceptual Site Model

Ecological habitats and likely receptors within and adjacent to the affected property should be described. Additionally, communities such as aquatic life and benthic invertebrates and major feeding guilds and their representative species supported by affected property habitats should be identified (required element 2). The potential presence of protected species that could be supported by these habitats should be discussed, as this will impact the measurement receptors selected for the SLERA. Previous environmental studies may be available for some sites, which could help identify important habitats or species for the assessment to consider. It may be necessary to consult with the TPWD or the U.S. FWS or others familiar with the biology of the area to obtain information about local ecological resources including the potential for protected species to occur at the affected property. This information should be discussed in the work plan.

The work plan should identify the assessment and measurement endpoints and measurement receptors, present a CSM, and should identify proposed sampling locations if additional sampling and analysis is proposed depending on the stage of the overall assessment. The CSM should illustrate the predicted relationships between the COCs, the exposure, and the assessment endpoint responses. The CSM identifies the potential sources and secondary contaminant sources, release mechanisms, transport mechanisms, the potentially complete and incomplete exposure pathways, and the potential ecological receptors (required element 3).

4.3.2.4. COC Fate and Transport and Toxicological Profiles

The work plan should describe how the potential fate and transport mechanisms for each COC will be presented in the SLERA (required element 4) or cite a reference that includes this information. As discussed in **8.0**, it should be determined whether the COCs at the affected property are likely to persist, be degraded, or move beyond the extent of contamination determined in the affected property assessment.

4.3.3. Analysis

As discussed in **4.1.2**, the analysis phase evaluates the exposure of a measurement receptor (or community) to a COC and the ecological effects of that exposure (required element 5). This phase builds off the complete and significant pathways identified in the CSM (**7.0**), as well as the assessment endpoints, the relationships among the assessment and measurement endpoints, the data required to answer the questions, and the methods that will be used to analyze the data. When exposure analysis for wildlife receptors is anticipated for the SLERA, the work plan should present wildlife exposure inputs, including body weights, ingestion rates, dietary compositions, percentage of soil or sediment in the diets, size of exposure areas, and area use factors. Anticipated uptake factors (or methods to estimate uptake) should be discussed. Additionally, the TRVs (and their sources) to be used in the wildlife exposure assessments should be identified. If applicable, any adjustments to these TRVs should be discussed (e.g., use of uncertainty factors).

There should also be a general discussion in the work plan on how receptors such as aquatic life, fish, benthic invertebrates, reptiles, amphibians, and livestock will be evaluated, as appropriate, beyond initial screening. The discussion should briefly explain how the EPCs (i.e., 95 percent UCLs) will be calculated. It should also confirm that the SLERA will evaluate the potential for hot spots.

4.3.4. Risk Characterization

Here the work plan will need to discuss that the SLERA will integrate the exposures and effects analyses and estimate the likelihood of adverse ecological effects occurring. In other words, the estimated dose for a given receptor will be divided by the TRV for each COC to determine an HQ. An HQ of 1 should be identified as the threshold for indicating when an adverse effect may occur. The work plan should state that a conservative HQ will be calculated using a NOAEL-based TRV with no adjustments to the receptors' exposure (required element 6). COC and receptor pairs with an HQ greater than 1 will also need a refined (less-conservative) dose and HQ calculation (required element 7). An uncertainty analysis is also needed (required element 8).

4.3.5. PCL Development and Risk Management Recommendation

The work plan should state that, if necessary, medium-specific ecological PCLs bounded by the NOAEL and the LOAEL will be developed in the SLERA (required element 9) for any remaining COCs that are not screened out because of the HQ calculations or the uncertainty analysis. The work plan should also state that the final ecological PCL (see **13.4**) for each COC will be identified in the SLERA. Finally, the work plan will need to commit the SLERA to making a recommendation for risk management (required element 10) addressing any exceedances of ecological PCLs. This recommendation may include further ecological evaluation or some type of response action. See **14.0** for a discussion of possible risk management recommendations.

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5.0 COC Screening Analysis (Required Element 1)

Comparison of affected property concentrations to ecological benchmarks is the first required element in a Tier 2 SLERA, as specified in the TRRP rule [30 TAC 350.77 (c)(1)]. Benchmarks offer a simple approach for comparing COC concentrations in media at the affected property against concentrations presumed safe to biota likely to be the most exposed (aquatic life, benthic and soil invertebrates, and plants). COCs present in media at sufficiently high concentrations to justify further evaluation in an ERA should be retained, whereas those that present little or no potential risk should be eliminated from further ecological review. If a COC is not bioaccumulative for the media in question **and** the COC is present below benchmark levels, further evaluation of that COC is unnecessary. This text discusses:

- Identification and evaluation of bioaccumulative COCs.
- Application of benchmarks in the Tier 2 SLERA process.
- Use of alternate and proposed benchmarks.
- Use of a surrogate COC where no benchmark is specified for a certain COC.
- Documentation of the COC screening analysis in a Tier 2 SLERA.

5.1. Bioaccumulative COCs

Although not a separate required element under the TRRP rule, the identification and evaluation of bioaccumulative COCs present at the affected property is nonetheless required for properly comparing concentrations at the affected property to ecological screening benchmarks under required element 1.

Bioaccumulative COCs tend to increase in concentration within some organisms relative to their concentration in environmental media and dietary sources due to sequestration in certain body tissues. Biomagnification (bioaccumulation in successive trophic levels of a food chain) can result in concentrations of COCs many times greater than those in environmental media. Bioaccumulation is an important aspect of the ERA process because it can result in increased exposure to multiple trophic levels, compared with COCs that do not bioaccumulate. Also, bioaccumulative COCs can be present at concentrations in environmental media that are protective for community-level receptors, but that can pose indirect risks to higher trophic levels.

The ecological benchmarks presented in the Benchmark Tables evaluate direct exposure to specific media for selected receptors and are not expected to evaluate bioaccumulation concerns.

Bioaccumulative COCs in the environment do not indicate that an ecological risk due to bioaccumulation is occurring or will occur, but that an evaluation of that potential is warranted. Bioaccumulation is an exposure-related parameter that does not equate with inherent risk or toxicity (Feijtel et al., 1997).

A variety of factors determines the site-specific potential for bioaccumulation, including biotic (e.g., feeding strategy, behavior, and physiology) and abiotic (e.g., chemistry of the environment) considerations. For example, polycyclic aromatic hydrocarbons accumulate in most benthic invertebrates because they have a minimal capacity for metabolism of PAHs. Levels of PAHs in fish tissues reflect a dynamic balance between uptake from the water column and sediment, diet, metabolism, gut-assimilation efficiency, partitioning and elimination.

These processes vary greatly by individual PAH and fish species. For example, tissue concentrations of PAHs in bottom prey fish are strongly related to sediment concentrations. Concentrations of 2- and 3-ring PAHs, such as naphthalene and acenaphthylene, likely reflect uptake from water through gills because of greater concentration in water and slow metabolism. Concentrations of high molecular weight PAHs in fish tissue may be low because of gut assimilation efficiency and high metabolism rates (Huang et al., 2014).

5.1.1. Identification of Bioaccumulative COCs

The TCEQ has identified specific COCs (Table 5.1) that may pose substantial risk due to bioaccumulation. In identifying the organic COCs listed in the table, the agency has made use of lists of bioaccumulative COCs by various authorities, including the U.S. EPA (1995, 2000, 2005b), the European Chemicals Agency (2012) through the REACH program, Washington state (2006; Hoffman, 1998, 2003), Alaska (ADEC, 2010), California (California DTSC, 1996), New Mexico (NMED, 2008), New Jersey (NJDEP, 2012), Ohio (OEPA, 2008), the Tri-Service Ecological Risk Assessment Working Group (2003, 2008), Environment Canada (1995), and the Indiana University School of Public and Environmental Affairs (2013). The presence of COCs on multiple lists was used to identify the predominant organic bioaccumulative COCs that appear in Table 5.1. Organic bioaccumulative chemicals were listed for water, sediment, and soil, except for those COCs with a state or federal water quality criterion that underwent additional analysis, as described below.

Except where noted, the list of metals in Table 5.1 is based on media-specific uptake factors (bioaccumulation factors). The listing of metals for soil and sediment exposure is based on the use of uptake factors for soil plants (Bechtel Jacobs Company, 1998a), soil invertebrates (Sample et al., 1998), and benthic invertebrates (Bechtel Jacobs Company, 1998b).

Uptake factors were reported as the ratio of the concentration of a given COC in biota to that in an abiotic medium. Typically, data from those references are based on field studies and the analytical methods were for “total” analysis; thus, it is not possible to determine the actual metal species responsible for the uptake factor. Each of these studies reports the 90th-percentile uptake factor for all data it includes (e.g., the cadmium 90th percentile is based on 120 individual uptake factors). Metals with a reported 90th-percentile uptake factor > 1 for soil plants or invertebrates are listed in Table 5.1 for soil exposure, and metals with a 90th-percentile uptake factor > 1 for benthic invertebrates, for sediment exposure.

The listing of metals for surface water is based on bioconcentration factors obtained from U.S. EPA (1999). Metals with a bioconcentration factor (BCF) greater than 1,000 for aquatic invertebrates or fish were initially listed in Table 5.1 for water exposure. However, those metals identified as bioaccumulative that have a state or federal water quality criterion underwent additional analysis, which is described below. Thallium is the sole metal listed in Table 5.1 for water exposure that does not have a water quality criterion—it is listed based on a BCF > 1,000.

Selenium was included in the previous versions of this guidance based largely on professional judgment, but in 2016 the EPA released the final aquatic life ambient water quality criterion for selenium in freshwater that identifies selenium as bioaccumulative in water and sediment. The proportion of selenium found in particulate matter (algae, detritus, and sediment) is important because it is the primary avenue for selenium entering the aquatic food web. The single largest step in tissue selenium accumulation in aquatic environments occurs at the base of the food web where it accumulates in algae and other microorganisms (U.S. EPA, 2016).

As a precautionary measure, silver is listed as bioaccumulative in soil. Although the documented trophic bioaccumulation potential in soil is reported to be low (Ratte, 1999), there is a large disparity between the plant benchmark (560 mg/kg, U.S. EPA, 2006) and the wildlife-based PCLs from the PCL Database and the EPA's Eco-SSLs. As stated by U.S. EPA (2006) "silver is toxic in laboratory studies to avian and mammalian species with effects including reduced growth and reproduction and increased mortality." Therefore, to prevent silver from being screened out through a comparison to its benchmark, the TCEQ is requiring that it be evaluated as if it were bioaccumulative in soil (i.e., a trophic level assessment). Site silver concentrations in soil may still be compared to bird and mammal PCLs or SSLs for potential ecological risk.

Tributyltin has been listed as bioaccumulative in sediment because it may partition to suspended solids and deposit in sediments. Organotins may be bioconcentrated and then bioaccumulated through aquatic food webs. Tributyltin accumulates in sediment and is relatively persistent and is taken up by benthic organisms such as clams (Sekizawa et al, 2003).

Table 5.1. Bioaccumulative COCs

CAS No.	COC	Chemical Class	Applicable Media
7440-43-9	cadmium	metals	sediment, soil
7440–47-3	chromium	metals	soil
7440-50-8	copper	metals	sediment, soil
7439-92-1	lead	metals	soil
7439-97-6	mercury	metals	water, sediment, soil
744–02-0	nickel	metals	sediment, soil
7782-49-2	selenium	metals	water, sediment, soil
7440-22-4	silver[a]	metals	soil
7440-28-0	thallium	metals	water
688-73-3	tributyltin	metals	sediment
7440-66-6	zinc	metals	sediment, soil
309-00-2	Aldrin	organochloride pesticides	sediment, soil
57-74-9	Chlordane	organochloride pesticides	sediment, soil
72-54-8	DDD[b]	organochloride pesticides	water, sediment, soil
72-55-9	DDE[b]	organochloride pesticides	water, sediment, soil
50-29-3	DDT[b]	organochloride pesticides	water, sediment, soil
60-57-1	Dieldrin	organochloride pesticides	sediment, soil
72-20-8	Endrin	organochloride pesticides	sediment, soil
76-44-8	Heptachlor	organochloride pesticides	sediment, soil
1024-57-3	Heptachlor epoxide	organochloride pesticides	sediment, soil
8001-35-2	Toxaphene	organochloride pesticides	sediment, soil
2385-85-5	Mirex	other pesticides	sediment, soil
3980-114-4	Photomirex	other pesticides	sediment, soil

CAS No.	COC	Chemical Class	Applicable Media
1336-36-3	PCBs	PCBs	water, sediment, soil
not applicable	dioxins	semivolatiles	water, sediment, soil
not applicable	furans	semivolatiles	water, sediment, soil
118-74-1	hexachlorobenzene	semivolatiles	water, sediment, soil
608-73-1	hexachlorocyclohexane	semivolatiles	sediment, soil
29082-74-4	octachlorostyrene	semivolatiles	water, sediment, soil
87-86-5	pentachlorophenol[c]	semivolatiles	sediment, soil

[a] Silver is not bioaccumulative in soil but is being listed here to address sensitivity in birds and mammals not captured in the soil benchmark.

[b] DDT and its metabolites (DDD and DDE) need to be evaluated cumulatively (i.e., use the HI approach) and are therefore all listed for water, sediment, and soil.

[c] Pentachlorophenol is listed for sediment and soil based on its log K_{ow} of 4.74, and U.S. EPA (2007b), indicating the potential for risk to birds and mammals at soil concentrations significantly below levels protective of plants and soil invertebrates.

Bioaccumulative COCs listed in Table 5.1 also have ecological screening benchmarks to aid in the understanding of the potential for varying impacts—or lack thereof—at different ecosystem trophic levels. The benchmarks do not directly address the food-chain transfer of bioaccumulative COCs. When the concentration of a bioaccumulative COC falls below a corresponding ecological screening benchmark, and the COC for that medium has been recognized as being of concern for bioaccumulation to higher trophic levels, then the person may be required to further evaluate the possible risks from exposure to that COC for that medium through the food chain. For example, a bioaccumulative COC may be evaluated for upper trophic level wildlife receptors (e.g., heron, raccoon, mink), but the screening-level benchmark may indicate that the sediment concentrations are not likely to adversely affect the benthic community.

The listing of bioaccumulative COCs for water in Table 5.1 does not impose or suggest a new or different water quality criterion at the affected property that would apply to the water column or to sediments. The evaluation of COCs is intended to address site-specific exposure pathways that might not be addressed by the statewide water quality criteria. This approach is analogous to that taken by other regulatory programs of the TCEQ. For example, wastewater permits sometimes require additional treatment of pollutants—beyond that needed to meet numerical water quality criteria—when this additional treatment is needed to address site-specific water quality concerns.

The TCEQ retains the ability to identify and require evaluation of additional compounds, case by case, that may pose a risk due to bioaccumulation.

Even though Table 5.1 is the primary tool for identifying bioaccumulative COCs, it is not expected to identify all COCs that tend to bioaccumulate. The agency reserves the right to require evaluation of additional compounds that may pose a risk due to bioaccumulation under site-specific conditions (e.g., COC distribution in environmental media, food-chain dynamics, and receptors) and considering COC-specific characteristics. Connell (1990) includes a general discussion of the commonly interrelated characteristics that allow bioaccumulation to occur. Characteristics of importance include chemical structure, molecular weight, molecular dimensions, stability, log K_{ow}, water solubility, and degree of ionization.

Organic COCs in sediment or surface water with log K_{ow} values between 3.8 (Feijtel et al., 1997) and 8.0 (Thomann, 1989) may be identified by the TCEQ as warranting evaluation for bioaccumulation. Similar criteria are used by agencies such as the U.S. EPA (2000, log K_{ow} > 3.5), the Ohio EPA (2008, log K_{ow} > 3.0), the European Chemicals Agency (2012, log K_{ow} > 4.2) and Washington State (Hoffman, 2003; log K_{ow} > 3.5). Furthermore, COCs with a molecular weight > 700 are considered to have a reduced potential to bioaccumulate, regardless of their log K_{ow}.

In developing this guidance, the work group was not able to identify any specific mechanism or threshold to trigger the TCEQ's evaluation of potentially bioaccumulative COCs in soil not listed in Table 5.1. As stated earlier, the TCEQ will determine whether to require evaluation of COCs not listed in Table 5.1 case by case. There is no default requirement for the person to evaluate bioaccumulation potential for COCs not listed in Table 5.1. However, the agency prefers that any voluntary evaluations (quantitative or qualitative) be included in the ERA.

5.1.2. Evaluation of Bioaccumulative COCs for Risk

All COCs listed in Table 5.1, as well as those that are retained through the benchmark screening process, are subject to further evaluation assuming they are detected at concentrations greater than background. Identification of COCs that bioaccumulate precedes the application of ecological benchmarks and is used to retain COCs for food-chain analysis regardless of their concentration relative to ecological benchmarks. Bioaccumulative COCs that are present in media at a concentration below applicable benchmarks will not be evaluated further for direct exposure to those media. For COCs to be retained for evaluation of risk to higher trophic levels, the metals listed in Table 5.1 must be present above background concentrations. The bioaccumulation evaluation, including any site-specific considerations (e.g., metallic species present) should be conducted during the food-chain analysis (see **10.4**). Justification is required to eliminate from food chain analysis those COCs listed in Table 5.1.

An important component of bioaccumulation potential relates to the metal species present. Many metals are only bioaccumulative in specific forms (e.g., oxidation state, elemental or organic compounds) and are more likely to bioaccumulate than other species of the same metal (e.g., lead acetate tends to bioaccumulate as opposed to elemental lead). Metals are assumed to be in a bioaccumulative form unless sufficient data are available to identify the species present so that their individual potential to bioaccumulate can be evaluated. Such evaluations need to consider environmental and biological transformation between metallic species in relation to bioavailability, including uptake and elimination rates. For instance, some COCs may have a very low media concentration of the bioavailable form because it is rapidly accumulated by organisms, which can give a false impression that it is not present at levels of concern.

5.2. Ecological Screening Benchmarks

As discussed in **1.3.4**, the Benchmark Tables and the associated supporting documentation are now contained in companion publication RG-263b and appear on the TCEQ's ERA webpage: <www.tceq.texas.gov/goto/era>. As needed, the TCEQ will update the benchmark values and supporting documentation if it deems the newer value or derivation process superior in quality and accuracy, and in response to rule and policy changes such as a revision to the Texas Surface Water Quality Standards or federal water quality criteria.

Benchmarks are intended to be conservative, and generally should not be used as triggers for remediation, or as cleanup goals. Similarly, the media benchmarks are not intended to be used in HQ calculations. For the Tier 2 SLERA, the TCEQ recommends that, for initial screening in required element 1, the maximum measured COC concentration in the exposure medium be compared to the medium-specific ecological benchmark values, unless the maximum value can be demonstrably considered an extreme outlier for the data set of the exposure medium, in which case the next highest value (that is not an extreme outlier) would be used. (See Appendix B of TRRP-15eco for descriptions of the preferred methods for defining and identifying outliers.) However, for all subsequent iterations of benchmark comparisons in the Tier 2 SLERA (or Tier 3 SSERA), the person should use the EPC (i.e., 95 percent UCL).

5.2.1. Considerations for Hardness, Total, and Dissolved Criteria for Surface Water Screening

For some metals, the freshwater criteria are a function of hardness. The benchmark values are based on a default hardness value of 50 mg/L. The person has several options for using an alternate hardness value to calculate the benchmark value. The person may use the segment-specific 15th percentile hardness value (for the nearest downstream segment) or property-specific hardness data using site-sample results in accordance with the Implementation Procedures (TCEQ, 2010a, or latest revision).

Specific numerical aquatic-life criteria for metals and metalloids apply to dissolved concentrations where noted.[14] Dissolved concentrations can be estimated by filtration of samples prior to analysis, or by converting from total recoverable measurements in accordance with the latest revision of the Implementation Procedures. The TCEQ usually prefers dissolved-metals data for surface waters rather than the mathematical conversion. If the conversion

[14] Aquatic life criteria for aluminum, arsenic, cadmium, chromium, copper, lead, manganese, molybdenum, nickel, silver, uranium, and zinc are applicable to dissolved values.

method is used, the person must use either the concentration of total suspended solids (TSS) for the nearest classified downstream or downgradient segment (as listed in the Implementation Procedures), or property-specific TSS data from site sample results (in accordance with the Implementation Procedures). The person should also be aware that the TSWQS define site-specific criteria for aquatic-life protection for selected water bodies (30 TAC 307, Appendix E). As these values are higher (less conservative) than those in the Benchmark Tables, the person should determine if there is a site-specific criterion for the surface water (and COC) in question.

An example calculation for copper:

Example conversion of dissolved copper to total for Segment 0806

Segment 0806: West Fork of the Trinity River

Hardness = 136 mg/L (from Implementation Procedures)

TSS = 10 mg/L (from Implementation Procedures)

Freshwater chronic criterion for segment-specific dissolved copper:

$$0.960e^{0.8545(\ln(\text{hardness})) - 1.6463} = 12.31\ \mu g/L$$

Determination of Partition Coefficient:

$$K_p = 10^b \times TSS^m$$

where:

K_p = partition coefficient (L/kg)

b = intercept and m = slope (both found in Table 6 of Implementation Procedures)

Therefore:

$$K_p = 10^{6.02} \times 10^{-0.74}$$

$$K_p = 1.9055 \times 10^5$$

Determination of Dissolved-to-Total Ratio:

$$\frac{C_d}{C_T} = \frac{1}{1 + (K_p \times TSS \times 10^{-6})} = \frac{1}{1 + (1.9055 \times 10^5)(10)(10^{-6})} = 0.344$$

Total Segment-Specific Chronic Copper Criterion:

In this example, the total segment-specific chronic copper criterion would be the predetermined freshwater chronic criterion of 12.31 µg/L divided by the dissolved-to-total ratio of 0.344 (unitless).

$$\frac{12.31\ \mu g/L}{0.344} = 35.78\ \mu g/L$$

Example conversion of dissolved silver, as free ion, to total for Segment 0604

The Texas surface water criterion for silver is expressed in terms of a free ionic form, which is the most biologically toxic component of dissolved silver. When analytical data are reported in terms of total concentrations, for comparison purposes, the person should convert the dissolved criterion for silver to a total silver value.

Data collected from a variety of water bodies throughout the United States show that a correlation exists between the dissolved chloride concentration and the percentage of free ionic silver (U.S. EPA, 1985a). Therefore, the segment-specific chloride value should also be incorporated into the calculation of the total silver criterion. Alternatively, the person may use the federal benchmark for freshwater silver from the Surface Water Inorganic Benchmark Table.

A discussion of the conversion from a dissolved criterion to total is presented in the Implementation Procedures and an example is provided below:

Segment 0604: Neches Below Lake Palestine

TSS = 10 mg/L (from Implementation Procedures)

Chlorides = 24 mg/L (from Implementation Procedures)

Freshwater Chronic Standard for free ion form of dissolved silver = 0.08 µg/L.

Determination of percent dissolved silver in free ionic form (Y):

$$Y = e^{e^{\left(\frac{1}{0.6559 + 0.0044\ Cl}\right)}}$$

where:

Cl = dissolved chloride concentration (mg/L)

= 41.18 percent dissolved silver.

Determination of Partition Coefficient (Kp):

$$K_p = 10^b \times TSS^m$$

where:

b = intercept and m = slope (both from Table 6 of Implementation Procedures)

K_p = partition coefficient (L/kg)

Therefore:

$$K_p = 10^{6.38} \times 10^{-1.03}$$

$$K_p = 2.239 \times 10^5$$

Determination of Dissolved-to-Total Ratio:

The ratio of the dissolved concentration to the total recoverable concentration incorporates the K_p and TSS concentration.

$$\frac{C_d}{C_T} = \frac{1}{1 + (K_p \times TSS \times 10^{-6})} = \frac{1}{1 + (2.239 \times 10^5)(10)(10^{-6})} = 0.31$$

Total Segment-Specific Silver Chronic Criterion:

In this example, the freshwater chronic standard for free ion form of dissolved silver of 0.08 µg/L is divided by the product of the dissolved-to-total ratio of 0.31(unitless) and the fraction dissolved silver of 0.41(unitless). The resulting criterion is 0.63 µg/L.

$$\frac{0.08\ \mu g/L}{(0.31)(0.41)} = 0.63\ \mu g/L$$

5.2.2. Alternate and Proposed Benchmarks

An *alternate* benchmark is one developed in lieu of the benchmark specified for a certain COC or medium in the Benchmark Tables. Where no benchmark is specified in those tables, the COC should be carried forward in the ERA unless a *proposed* benchmark can be developed, or an appropriate surrogate chemical is used. With one exception, alternate ecological-screening benchmarks for water, sediment, and soil may be developed if the person adequately justifies their use. The exception is surface water benchmarks based on existing state or federal criteria, for which alternate benchmarks may not be substituted.

Alternate and proposed benchmarks that are developed or cited by the user should be fully explained. The person should justify why there is a need for the alternate benchmarks, including why the guidance-specified benchmarks are not suitable, if applicable. The literature sources and derivation steps used in obtaining the alternate or proposed benchmarks should be cited. The alternate benchmarks should be comparable or more appropriately suited to site conditions than the guidance-specified benchmarks in their data quality, statistical power, toxicological effects level, and mode of exposure. Where multiple alternate or proposed benchmarks of similar nature are available, the person may calculate a geometric mean for the COC. Durda and Preziosi (2000) have outlined a two-phased approach for evaluating the quality of ecotoxicological data to be used in benchmark development that may be useful in selecting alternate or proposed benchmarks. Similarly, Clark et al. (1999) discuss benchmark development and application. Information detailing the level of comparability (with the guidance-specified benchmark) should be included for each alternate media benchmark.

The use of freshwater benchmarks for marine waters or marine benchmarks for inland waters, including highly saline waters in West Texas, is unacceptable and is also not recommended for sediment COCs. The person should make every effort to use benchmarks that are appropriate for the surface water conditions in question, rather than immediately defaulting to other benchmarks.

5.2.2.1. Derivation of Surface Water Benchmarks Using LC50 Data

The TSWQS [30 TAC 307.6(c)(7)] provide a mechanism for deriving numerical criteria where there are no standards **and** available data are insufficient to allow the use of EPA guidelines (e.g., U.S. EPA, 1985b, 1992b). Depending on the persistence and bioaccumulative nature of the COC in question, the TSWQS specify that a multiple of the LC_{50} for the most sensitive aquatic organism may be used:

- for all COCs, acute criteria = $LC_{50} \times 0.30$,
- for non-persistent COCs, chronic criteria = $LC_{50} \times 0.10$,
- for persistent COCs that do not bioaccumulate, chronic criteria = $LC_{50} \times 0.05$, and
- for COCs that bioaccumulate, chronic criteria = $LC_{50} \times 0.01$.

In selecting toxicity data to calculate a value, LC_{50} test results are preferred, and marine or freshwater (depending on the nature of the receiving water) species indigenous to Texas should be used whenever possible. Results from aquatic plant and algae toxicity tests will not usually be accepted (TCEQ, 2010a). Additionally, flow-through tests with exposure times of 48 hours (for invertebrates) or 96 hours (for vertebrates) are preferred, but static test results can be used, particularly where the data indicate a higher sensitivity. If the toxicity-test data do not meet these conditions, the person should justify the rationale behind the selection of those data. Generally, the most conservative LC_{50} should be used that meets the selection preferences indicated. If more than one LC_{50} data point is available for a species, the geometric mean should be calculated and used. If the LC_{50} approach is used, the person should justify the selection of each of the LC_{50} values. The EPA ECOTOX database, available online at <https://cfpub.epa.gov/ecotox/>, may be used as a source of aquatic (and terrestrial) toxicity data, although it is certainly not the only source. When possible, the user should obtain the original paper associated with the ECOTOX reference and should review the documentation and control codes assigned to the reference. The source paper should be indicated (and preferably reviewed), rather than simply citing the ECOTOX reference number alone.

Selection of the appropriate multiplier for the calculation is a function of persistence and the tendency for that COC to bioaccumulate. For calculating alternate or proposed criteria, COCs are considered persistent if the half-life in water or sediment is 60 days or greater, and COCs are considered bioaccumulative if the bioconcentration factor or bioaccumulation factor for the constituent (measured or estimated using regression analysis) is 1,000 or greater (TCEQ, 2010a). BCFs or BAFs determined from laboratory or field studies using water-column invertebrates or fish are preferred over estimated values. A range of BCF or BAF values may be presented. The person should use half-lives or environmental-fate rate constants (e.g., volatilization, photolysis, hydrolysis, biodegradation) that are most appropriate for the surface water or sediment exposure pathway in question. A range of half-life values may be presented. The person should briefly justify the selection of the BCF or BAF and half-life information used in these decisions and cite the source.

5.2.2.2. Derivation of Sediment Benchmarks Using Equilibrium Partitioning

The ecological work group has discussed the utility of the approach based on equilibrium partitioning (EqP) for developing sediment benchmarks, particularly for chemicals where other preferred benchmarks were unavailable from common and generally accepted sources. The EqP approach may be desirable for this purpose because the chemical-specific partitioning coefficients necessary for the application of EqP are generally either known or easily estimated, based on literature values.

The EqP theory states that a nonionic chemical partitions between sediment organic carbon, interstitial (pore) water, and benthic organisms (U.S EPA, 2008b). The EqP method is designed to address direct toxicity to benthic organisms exposed to contaminated sediments and does not address ingestion of sediments by benthic invertebrates. Benchmarks derived using the EqP method are also not designed to address risks that may occur through bioaccumulation and subsequent exposure to pelagic aquatic organisms (e.g., predatory fish), or wildlife (U.S EPA, 2008b). The EqP approach does generate defensible, national, numeric chemical-specific benchmarks applicable across a broad range of sediment types.

The three principal observations that underlie the EqP approach are:

- The concentrations of nonionic organic chemicals in sediments (expressed on an organic-carbon basis) and in interstitial waters correlate to observed biological effects on sediment-dwelling organisms across a range of sediments.
- Partitioning models can relate sediment concentrations for nonionic organic chemicals on an organic-carbon basis to freely dissolved concentrations in interstitial water.
- The distribution of sensitivities of benthic organisms to chemicals is like that of water column organisms; thus, water quality criteria can be used to define acceptable effects concentrations of chemicals freely dissolved in interstitial water.

The EqP approach assumes that (1) the partitioning of the chemical between sediment organic carbon and interstitial water is at or near equilibrium; (2) the concentration in either phase can be predicted using appropriate partition coefficients and the measured concentration in the other phase; (3) organisms receive equivalent exposure from water-only exposures or from any equilibrated phase; (4) for nonionic chemicals, effects concentrations in sediments on an organic-carbon basis can be predicted using the organic-carbon partition coefficient (K_{oc}) and effects concentrations in water; (5) the water quality criterion is an appropriate effects concentration for a freely-dissolved chemical in interstitial water; and (6) sediment criteria derived as the product of the K_{oc} and water quality criteria are protective of benthic organisms (U.S. EPA, 2008b).

The theory assumes that the amount of organic carbon in a system generally determines the extent of COC partitioning between the sediment particles, pore water, and dissolved organic carbon. The theory predicts that, if all phases are at equilibrium, the bioavailability of a constituent should be directly proportional to COC activity in interstitial water, and inversely proportional to the organic-carbon content in the sediment since organic carbon largely controls the sorption of sediment particles. Thus, the sediment pore-water concentration and the bulk sediment COC concentration are related by the carbon-normalized sediment water partition coefficient (K_p), which depends on the sediment-particle organic-carbon partition coefficient (K_{oc}), and the mass fraction of organic carbon in sediment (kilogram of organic carbon per kilogram of sediment):

$$K_p = f_{oc} \times K_{oc}$$

K_{oc} is the partitioning coefficient of a COC to organic carbon, and is used to describe the distribution of COC between the organic fraction of sediment and the interstitial water. The sediment quality benchmark (SQB), in mg COC/kg sediment, dry weight, can be determined using the partitioning coefficient K_p, in L/kg, between sediment and interstitial water:

$$\text{SQB (in mg/L)} = K_p \times \text{WQB}$$

where:

WQB (mg/L) is the water quality benchmark acute or chronic value.

The person should use the log K_{ow} and log K_{oc} values specified in the TRRP rule [30 TAC 350.73(f)]. Finally,

$$\text{SQB} = f_{oc} \times K_{oc} \times \text{WQB}$$

The principal advantages of this approach are that it allows for the derivation of COC-specific sediment benchmarks and it can be adapted to site conditions by adjusting the organic carbon parameter. This EqP methodology is applicable for sediments with a f_{oc} value of 0.2–12 percent (U.S. EPA, 1992c) and has been tested on nonionic organic compounds with log K_{ow} between 3.8 and 5.3 (U.S. EPA, 1997b).

The TCEQ utilized an EqP approach to develop sediment benchmarks for VOCs and munitions listed in the sediment benchmark table. The EqP approach, as described by Fuchsman (2003), U.S. EPA (2008b), and Pascoe et al. (2010), was used to develop freshwater and marine sediment benchmarks and second effects levels for volatile COCs and munitions where values were unavailable from preferred sources. This method can be used to develop benchmarks for COCs not listed in the sediment benchmark table:

$$SQB\ (mg/kg) = WQB\ (mg/L) \times \left[(f_{oc} \times K_{oc}) + \frac{1 - f_{solids}}{f_{solids}} \right]$$

where:

SQB = sediment quality benchmark (mg/kg)

WQB = acute or chronic water quality benchmark (mg/L)

K_{oc} = organic carbon partition coefficient (unitless, taken from the TRRP table Chemical and Physical Properties)

f_{oc} = fraction of organic carbon (0.01 kg organic carbon/kg sediment, TRRP-24 default value)

f_{solids} = fraction of solids (= 1 – porosity; porosity = 0.37, TRRP-24 default value)

If applied to a well-characterized sediment bed and COC, the EqP method can more realistically evaluate site-specific relevant receptors and measurement endpoints. The method is conservative for most COCs (i.e., those with moderate K_{ow} and K_{oc}) and may overestimate risks especially with high-K_{oc} chemicals.

Although the agency understands that EqP-based sediment benchmarks for protecting the benthic community do not address the sediment-ingestion exposure pathway, it does recognize that EqP may be more applicable to more water-soluble classes of COCs. Therefore, the TCEQ used the Fuchsman (2003) EqP model to derive sediment benchmarks for VOCs and munitions. When deriving sediment benchmarks using the EqP method, the person should explain and justify the water quality criterion, fraction of organic carbon and which EqP equation was used and why.[15]

5.2.2.3. Use of Surrogate Chemical Data

Where a benchmark is not specified for a certain COC and there is insufficient information to derive a benchmark, the person may use toxicity data or available benchmarks for surrogate compounds, under the assumption that the surrogate compound and the COC in question will have similar toxicity. In support of this assumption, the person should evaluate:

- Similarity in chemical structure.
- Chemical substitutions.
- Molecular properties (e.g., lipophilicity, polarity).
- Metabolites or breakdown products.
- Mechanisms of action.

[15] If there is a source of COCs directly impacting a water body (primarily through the groundwater–to–surface water pathway), the agency may request pore-water sampling for decision making in lieu of whole-sediment testing.

- Similarity in effect at different exposure durations or test endpoints, metabolic pathways, and physicochemical properties (e.g., octanol-water partition coefficient, water solubility).

COCs in the same class of chemicals, groups of isomers, congeners, and close homologues are often used as surrogates. Proper surrogate selection (including TRV endpoints) is critical, particularly where sensitive species may be present.

5.3. Documentation of COC Screening Analysis

As part of this screening process, the person should present tables that identify any bioaccumulative COCs, show the benchmarks being used, the source of the benchmarks, and corresponding background values where appropriate. These tables should compare the COC maximum concentration side-by-side with the appropriate benchmark, and there should be a column for each medium that indicates if the COC was screened out based on a benchmark comparison, background concentration comparison, or consideration of bioaccumulation. The tables should also reflect detection levels where the chemical was below detection, rather than simply including a dash or "NA."

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6.0 Exposure Pathway Analysis (Required Element 2)

Under the TRRP rule [350.77(c)(2)], the second required element of the Tier 2 SLERA is the identification of communities, feeding guilds, and representative species that might be supported by habitats on the affected property. Each of these ecological groupings is discussed in the following sections. In addition, the roles that these groups play in the selection of assessment endpoints are discussed, as well as what ecological receptors the agency is trying to protect under the definition of *ecological PCL* [see 350.4(a)(27)]. The TCEQ recognizes several taxa of organisms (reptiles, amphibians, livestock, and cave-dwelling receptors) that have not been traditionally evaluated in ERAs. However, the TCEQ believes that assessment of these receptors is a developing area in the field of ERAs and the person should determine their potential presence at the affected property and evaluate their potential risk to the extent possible based on the information available. These taxa have been identified as "species with exposure and toxicity data gaps" (**6.6**).

6.1. Communities

Ecological communities are collections of plant and animal populations occupying the same habitat in which the various species interact with one another. However, **in this guidance, communities refer to those groups whose exposure to COCs can be evaluated in terms of the media in which they reside**. These communities consist of soil invertebrates, terrestrial vegetation, benthic invertebrates, water-column invertebrates, fish, algae, and rooted aquatic vegetation.

COCs that exceed ecological (community-level) benchmarks, but do not subsequently prove to be a risk to higher-trophic-level receptors, may still harm these community-level receptors. Depending on site-specific circumstances, the person may be required to demonstrate that impacts to these communities will not result in unacceptable consequences for the mobile or wide-ranging receptors. Generally, it is not anticipated that exceedances of benchmarks will result in the development of COC-specific PCLs for communities, except for benthic invertebrates, as discussed below.

Ecological PCLs are primarily intended to be protective of more mobile or wide-ranging ecological receptors and, where appropriate, benthic invertebrate communities within Texas waters. Although benthic invertebrates play a critical role in the aquatic food chain as a critical pathway for the transfer of energy and nutrients to higher trophic level organisms, the TCEQ recognizes that the benthic community in some water bodies may be diminished for reasons unrelated to releases of COCs from property subject to the TRRP regulation.

The bullet points that follow indicate where the agency believes it unnecessary to determine an ecological PCL for sediment that is protective of the benthic invertebrate community. This list does not preclude an evaluation of risks to higher-trophic-level organisms that may forage in these types of water bodies or nearby bodies (that could be impacted from sediment COC transport). Nor does it preclude the TCEQ from requiring additional evaluations at these types of locations, case by case, where significant exposure conditions warrant (e.g., acutely toxic concentrations, or the presence of free product).

- Routinely dredged water bodies. This applies to the portion of the channel that is dredged every three years or more often. Risks to benthic communities potentially exposed to COCs in sediments not routinely dredged (such as significant areas of shallow waters near the banks that are not used for shipping) should be evaluated where the exposure pathway is complete (see discussion below).

- Intermittent streams (that dry up completely at least one week a year) without perennial pools [See the definition at 30 TAC 307.3(a)(34)].

- Water bodies with concrete-lined channels (bottom and sides).

- Segments 1006 and 1007 of the Houston Ship Channel [see 30 TAC 307.10, Appendix C], excluding their tidal tributaries.

Benthic recovery in dredged areas is rapid, requiring as little as eight months (Lokkeborg, 2005). Hence, three years was selected as a reasonable estimate of benthic recovery. For federal projects, the district offices of the U.S. Army Corps of Engineers can provide the frequency of dredging for a given reach of a water body.

Where appropriate, the person should evaluate the possibility of COC transport and potential impact on benthic communities downstream or downgradient of the types of water bodies indicated above.

For water bodies not discussed above, or those documented to be scheduled for dredging within three years of APAR submission, the person may evaluate the suitability of a sediment PCL protective of benthic invertebrates as part of the uncertainty analysis discussed in **12.0**. This would only be required for sediment COCs that have not been eliminated from a Tier 2 SLERA.

6.2. Feeding Guilds, Food Webs, and Habitats

The term *feeding guilds* refers to broad groups of related ecological receptors (e.g., piscivorous birds) that represent the variety of species potentially exposed to COCs at the affected property. Feeding guilds are based on a shared feeding strategy, similar potential for exposure, and physiological or taxonomic similarity. Identification of these feeding guilds collectively defines the food webs specific to potentially affected habitats for evaluation in the risk assessment.

For assessing wildlife populations, the TCEQ recommends a feeding-guild approach, in which species sharing a similar feeding strategy (e.g., piscivores, carnivores, insectivores) are grouped together and assessed as a single unit. For each feeding guild, a representative species (often the most sensitive) is selected for exposure assessment. The results of the risk assessment for this indicator species (the measurement receptor) are intended to be descriptive (and protective) of all populations of species contained in that guild.

Food webs are interlocking patterns of food chains, representing the straight-line transfer of energy from a food source (e.g., plants) to organisms feeding on the source or on other organisms feeding on the source (Odum, 1971). The importance of a food chain as an exposure pathway primarily depends on the receptors in the food chain and what they eat, and other factors, including bioavailability and depuration of the COC evaluated.

Habitat is defined as any physical area whose resources and conditions allow, or may allow, wildlife and communities to live, forage, and reproduce for extended periods of time (i.e., be able to support long-term populations). Habitat-specific food webs are developed for use in the ERA to:

- define direct and indirect exposure pathways
- formulate assessment endpoints
- develop mathematical relationships among guilds for estimating exposure
- enable quantitative exposure analysis for ecological receptors

Food webs for the seven major habitats found in Texas are presented in **Appendix A** and can also be found in the PCL Database. These habitats are discussed in **6.2.1**. Each food web identifies its associated communities and feeding guilds. Users may rely upon these food webs to help identify the appropriate communities and feeding guilds associated with a habitat, or they may develop their own, as discussed in **6.2.2**.

Often, not all feeding guilds supported by the habitats on an affected property are proposed for evaluation. After the types of habitat (e.g., shortgrass prairie) that can be supported by the affected property have been determined, the feeding guilds within the food web for each of those habitats need to be identified. Not all the feeding guilds need to be mathematically evaluated in the ERA, but they all need to be somehow addressed (e.g., a justifiable rationale could be presented stating that the protection of one guild will protect another as well). The ERA could discuss the selected receptor's likelihood of exposure and sensitivity to COCs, as compared to those guilds that were not quantitatively addressed. The discussion could also emphasize habitat availability and the likelihood that any of the guilds could or would use the affected area for foraging. By not addressing all potentially impacted feeding guilds, a cascading effect is created throughout the ERA that affects the assessment endpoints, the measurement endpoints, the measurement receptors, the food webs, and the conceptual site model.

6.2.1 Major Habitats in Texas

A brief discussion of the seven major habitats follows:

1. **Upland Forest:** Generally characterized by deciduous or evergreen trees mostly greater than 30 ft. tall with 71–100 percent canopy cover. Wildlife can include deer mouse, raccoon, red fox, American robin, and red-tailed hawk. Example: pineywoods of East Texas.

2. **Tallgrass Prairie:** Characterized by tall grasses, such as little bluestem, big bluestem, switchgrass, and indiangrass, and which may include a large percentage of forbs and invading brush species. Wildlife can include eastern cottontail, nine-banded armadillo, bobcat, mourning dove, American woodcock, and barn owl, Example: east and west cross timbers and prairies of North and Central Texas.
3. **Shortgrass Prairie:** Native shortgrass prairie features blue grama, buffalograss, and fringed sage, and mixed grass areas; also includes sandsage prairies and shinnery sands. One of the most remarkable ecological features in this habitat is playas – ephemeral freshwater shallow circular wetlands, most more than 15 acres in area that are primarily filled by rainfall. Characteristic wildlife includes black-tailed prairie dog, least shrew, pronghorn, swift fox, burrowing owl, and lesser prairie-chicken. Example: Texas high plains.
4. **Shrub-Scrub:** Characterized by individual woody plants generally less than 9ft tall scattered throughout semiarid regions with less than 30 percent woody canopy cover. The expansion of ashe juniper (cedar) has had a tremendous impact on the ecosystem, causing a decrease in diversity of plant species and an increase in soil erosion. Wildlife includes white-tailed deer, feral hog, turkey, quail, dove, American kestrel, and reptiles. Example: Texas Hill Country.
5. **Desert-Arid:** Vegetative cover is predominantly semi-desert grassland and arid shrubland, except for high elevation islands of oak, juniper, and pinyon-pine woodland. Wildlife includes desert shrew, mule deer, desert bighorn sheep, coyote, javelina, turkey, and quail. Example: Trans-Pecos area.
6. **Freshwater Systems:** Encompasses a wide variety of aquatic habitats including rivers, creeks, swamps, marshes, bogs, and flood plains. Many protected species use wetland habitat, and most species of amphibians are dependent on sources of water (such as wetlands) for reproductive success. Wildlife includes red-winged blackbird, heron, egret, rail, bittern, moorhen, duck, geese, muskrat, mink, otter, raccoon, opossum, frogs, turtles, snakes, salamanders, and a variety of benthic and aquatic invertebrates. Example: Riparian areas throughout the state.
7. **Estuarine Systems:** Saline and brackish wetlands are complex and highly productive ecosystems, containing a variety of plant and animal species that are specially adapted to fluctuations in salinity, water levels, and seasonal temperatures and can include saltwater marshes, sand flats, sandy sea shores, mangrove swamps, and barrier islands. Wildlife includes swamp rabbit, marsh rice rat, mink, otter, mallard, marsh wren, spotted sandpiper, great blue heron, osprey, and

a variety of benthic and aquatic invertebrates. Example: Gulf Coast region.

These habitats and the food webs (**Appendix A**) are based on information describing the flora and fauna of Texas (see references under "General Literature" in **6.4**). Supplemental information was also collected from the EPA's Wildlife Exposure Factors Handbook (1993a) or "*Handbook"*.

Although the food webs in **Appendix A** look complex, users need not evaluate risk to all feeding guilds within a web if there is a logical justification. For example, PCBs tend to bioaccumulate and biomagnify in food chains and are not taken up by most plants but are accumulated by soil invertebrates. Thus, mammalian herbivores would not be exposed to PCBs in their diet but may take in PCBs through incidental ingestion of soil attached to plants. The herbivore exposure would be far less than that of mammalian insectivores that would be exposed through their diet and through incidental soil ingestion. This rationale emphasizes the importance of evaluating the mammalian-insectivore guild for PCB effects and serves as a reasonable justification for not evaluating the herbivorous-mammal guild (U.S. EPA, 1997a). Completed ecological exposure pathways should be illustrated in a conceptual site model (CSM; see **7.0**).

In the PCL Database, receptors that derive their food (and any incidental medium ingestion) from soil are identified with the two-character field "TR" to indicate Terrestrial, that appears after the receptor's name [e.g., American Robin (TR)]. Similarly, if the receptor's food (and medium ingestion) is based on sediment, an "AQ" to indicate Aquatic, appears after its name [e.g., Spotted Sandpiper (AQ)].

Using the descriptions above, the person should determine which of these habitats exist on the affected property. The PCL Database instructions for accessing the habitats, the individual wildlife species within these habitats, and the food webs are shown below. More complete instructions for evaluating wildlife risk using the PCL Database appear in the box in **13.3**.

On the "PCL Calculator" page, click on the rectangular tab labeled "Habitat" toward the top of the page, not the "Habitat" radio button. The "Habitat List" page appears and the habitat names and descriptions are provided. To view all the species within a habitat, click on the "Assoc. Species" arrow (on the left). To view the entire food web for a habitat, click on the underlined name of the habitat. To view all the habitats that support a certain species, click on the rectangular "Species" tab toward the top of the page, not the "Species" radio button, to bring up the "Species List" page, then hover over the globe icon to view all the habitats supporting that species.

6.2.2 Development of a Site-Specific Food Web

Information obtained during the characterization of the exposure setting of the affected property should be used to develop one or more habitat-specific food webs representing communities and guilds of receptors potentially exposed to COCs from the affected property. Food webs can be developed using the community approach, which includes (1) identifying potential receptors in a habitat for grouping into feeding guilds by class and communities, (2) organizing food-web structure by trophic level (e.g., primary producer, primary and secondary consumer), and (3) defining dietary relationships between guilds and communities. The result is a relatively complete food web for a defined habitat, one of which should be developed for each habitat in the affected property to be evaluated.

The first step in developing a habitat-specific food web is to identify—based on the dietary habits and feeding strategies of receptors—the major feeding guilds for birds, mammals, reptiles, amphibians, and fish. Invertebrates and plants are not assigned to guilds, but rather grouped into their respective community by the environmental media they inhabit, because the risk to these groups is characterized differently (see **10.2**).

Once the major feeding guilds are identified (e.g., herbivore, omnivore, carnivore, invertivore), receptors should be grouped by class (e.g., mammals, birds, amphibians, and reptiles). As noted, fish, invertebrates, and plants are grouped into their respective community by the media they inhabit. As for the major habitats, these site-specific classes and communities and their potential exposure pathways need to be graphically presented as shown in the example CSM in Figure 7.1.

The structure of a food web is usually organized by *trophic level* (TL)—one of the successive levels of nourishment and energy transfer. The first TL contains the primary producers and includes algae, grasses, and other green plants. Members produce their own food from nutrients, sunlight, carbon dioxide, and water. These primary producers are also the source of food for members of the second TL, whose members—often referred to as the primary consumers—are animals that eat plants (herbivores) and animals that subsist on detritus (decaying organic matter) found in sediment and soil (detritivores). The third TL contains omnivores—animals that eat plant and animal matter. The fourth TL contains only carnivores—sometimes referred to as the dominant carnivores: animals at the top of the food chain (e.g., raptors). Carnivores eat primarily animal matter. Some species can occupy more than one TL at a time; thus, professional judgment should be used to categorize receptors without making the food-web model unduly complex for the risk assessment.

The recommended information for inclusion in site-specific food webs includes:

- Affected media (soil, sediment, water).
- Trophic levels that include at a minimum: producers (TL 1), primary consumers (TL 2), secondary consumers (TL 3), and carnivores (TL 4).
- Feeding guilds divided into classes (e.g., herbivorous mammals, omnivorous birds, carnivorous reptiles) and communities.
- Major dietary interactions.

6.2.3 Minor Habitat

Occasionally, the affected property consists of fragmented ecological habitat or isolated island-like areas that cannot easily be categorized among the seven major habitats (e.g., an unmaintained grassy area adjacent to a laydown yard or a small, manufactured stock pond).

Such minor habitats may still be productive but support only a limited food web. In these circumstances, the person should identify the affected media and the TLs associated with the habitat and determine a reduced number of representative species to evaluate in the ERA. A subset of receptors from the various major habitats appears as **Appendix B.** These species are commonly found in Texas and are routinely evaluated in ERAs. These species are also incorporated into the PCL Database collectively, appearing as members of "Minor Habitat" and as individuals in the seven major habitats.

Additionally, sub-habitats of Minor Habitat occur, identified as "Minor Habitat—Terrestrial" and "Minor Habitat—Aquatic." When such reduced habitat occurs at the affected property and needs to be assessed, it is recommended that the person select Minor Habitat or its sub-habitats, based on the type of affected media.

The instructions for using the PCL Database to make this evaluation occur below and more complete instructions appear in the box in **13.3.**

*Information for Minor Habitat and its sub-habitats, as discussed above, is also incorporated into the PCL Database and can be accessed using the instructions in **6.2.1**.*

To populate a site-specific habitat for species from major and minor habitats, under Step 1 choose the "Species" radio button and highlight a receptor from the list, hold down the control key, and select the next desired receptor.

It is incumbent upon the user to justify the use of individual species instead of a major habitat, as protected species occur in all major habitats.

6.3. Selecting and Evaluating Assessment Endpoints

An *assessment endpoint* is an explicit expression of the environmental value (i.e., a desirable ecological attribute) to be protected (U.S. EPA, 1997a). A critical ecological attribute of a feeding guild or community is a characteristic that is essential to ecosystem structure and function. Protection of the critical ecological attributes of each feeding guild and community is assumed to also ensure the protectiveness of habitat-specific food-web structure and function. Therefore, assessment endpoints should be identified for each **evaluated**, class-specific feeding guild (and community, where appropriate) within each TL of the habitat-specific food webs. Selection of endpoints is a significant decision in the assessment. Since risk characterization and subsequent risk-management decisions depend on the selection of assessment endpoints, they should be developed with input from risk managers and other stakeholders.

Generally, assessment endpoints (and measurement endpoints and measurement receptors) should be developed or selected considering the definition of ecological PCL (i.e., limited to more mobile or wide-ranging species and benthic invertebrates, where appropriate). Several small-ranging receptors that are not excluded under the definition of ecological PCL (e.g., robin, shrews, marsh wren, spotted sandpiper) are among the most potentially exposed and sensitive species at an affected property and should always be included in the ERA. These receptors may not actually be present at the affected property, but they serve as representatives for others from the same feeding guilds that could be present.

Chemical toxicity resulting from direct exposure to a COC through media contact, or from indirect exposure through ingestion of plants or prey that have accumulated the COC, is the first and most common basis for developing an ecological PCL. In a Tier 2 SLERA, the potential for chemical toxicity is evaluated by estimating the total direct and indirect exposure of each measurement receptor to the COC and deriving an HQ (based on a NOAEL or a LOAEL). The goal is to determine if the ecological risk requires further evaluation (i.e., HQ > 1) and to develop ecological PCLs for the affected medium that will protect against chemical toxicity.

However, depending on the circumstances, the person may be required to consider impacts to receptors with limited mobility or range (e.g., plants, soil invertebrates, and small rodents) by evaluating whether loss of habitat or reduced energy transfer might affect the more mobile or wide-ranging receptors that may depend on the lower TLs for habitat and food. In general, *small-ranging* receptors are those with a home range less than or equal to one hectare (approximately 2.5 acres), and *wide-ranging* receptors are those with a home range greater than 1 hectare. In these cases, protection of the viability and presence of the more mobile or wide-ranging receptor remains the assessment endpoint, but these lower-trophic-level species would be the measurement receptors, and chronic NOAELs or LOAELs for the populations of the lower-TL species would be used as the measurement endpoints. For example, an acetone spill is unlikely to cause chemical toxicity in a hawk population. The residual acetone could, however, harm vegetation and rodents, and so reduce the habitat or food available to the local hawks. Whether that could lead to any significant adverse impact on the hawk population depends on several factors, particularly the:

- Size of the affected property.
- Overall area used by the hawk population (e.g., its home range).
- Response of the habitat and food organisms to the chemical at a population or community level.
- Duration and availability of toxic residuals.

Examples of potential ecological attributes appear below to illustrate the interdependent nature of various components of the ecosystem. Numerous components of a habitat-specific food web provide critical food sources and shelters for the wildlife that constitute the assessment endpoints used to determine ecological PCLs. As indicated, if terrestrial plants, soil invertebrates, or small mammals are significantly harmed, then there may be a resultant impact on the wildlife community. Consequently, elevated COC concentrations over substantial areas, particularly in regions that are undeveloped or particularly attractive to wildlife, could result in a disruption of this ecosystem or unacceptable consequences to wider-ranging receptors. Under these circumstances, development of PCLs derived to protect terrestrial plants, soil invertebrates, or small rodents could be warranted.

Examples of critical ecological attributes for feeding guilds include:

- seed dispersal (by birds and mammals)[16]
- major source of food (for predators)
- natural selection (removal of the weak and abnormal from the gene pool)
- pollination (cross-fertilization of plants by animals)
- regulation of prey populations (e.g., small rodents)

Critical ecological attributes for communities include:

- diversity or species richness
- community composition
- productivity
- decomposition
- major source of food (for consumer)
- habitat for wildlife

Examples of critical ecological attributes to be protected (i.e., assessment endpoints) associated with several feeding guilds and communities in a productive terrestrial ecosystem in an undeveloped or off-property area include:

[16] Many birds (e.g., robins, geese, and ducks) and mammals (e.g., mice, rabbits, and raccoons) disperse seeds. However, the person is not expected to develop HQs or otherwise measure ecological effects to the receptor's ability to disperse seeds or, similarly, function as a pollinating agent. By selecting a TRV for the measurement receptor of an evaluated feeding guild that addresses population impacts (e.g., growth, reproduction, mortality), its ecological attributes are inherently protected through the development of any warranted PCLs. These ecological attributes (and the interconnecting food webs) emphasize that ecosystems are composed of many interdependent functional groups.

Herbivore productivity: herbivores incorporate energy and nutrients from plants and transfer it to higher TLs, such as first- and second-order carnivores (e.g., snakes and owls, respectively). Herbivores also are integral to the success of terrestrial plants, through such attributes as seed dispersal.

Omnivore productivity: omnivores incorporate energy and nutrients from lower TLs and transfer it to higher levels, such as first- and second-order carnivores.

First-order-carnivore productivity: these carnivores are food for other carnivores (both first- and second-order), omnivores, scavengers, and microbial decomposers. They also affect the abundance, reproduction, and recruitment of receptors at lower TLs, such as vertebrate herbivores and omnivores, through predation.

Second-order-carnivore productivity: these carnivores affect the abundance, reproduction, and recruitment of species in lower TLs in the food web.

The viability and presence of upper-TL receptors: this includes instances where these critical attributes are significantly affected through the impairment or loss of habitats, food sources, and energy transfers associated with lower-TL receptors. Herbaceous plants provide an important pathway for energy and nutrient transfer from soil to herbivorous (e.g., rabbit) and omnivorous (e.g., mouse) receptors and provide critically important habitat for small animals. Woody plants provide an important pathway for energy and nutrient transfer from soil to herbivorous and omnivorous vertebrates (e.g., white-tailed deer, red-cockaded woodpecker) and provide critically important habitat for terrestrial wildlife. Terrestrial invertebrates provide a mechanism for the physical breakdown of detritus for microbial decomposition and are also a major food source for omnivores.

While there is agreement that limitations in habitat and energy transfer can conceptually affect the higher-TL receptors, little ecotoxicological research is available to quantify what level of change constitutes a significant difference, given the large areas used and the variable diets common in these species.

6.4. Representative Species

The TRRP rule defines "selected ecological receptors" as species that are to be carried through the ERA as representatives of the different feeding guilds and communities that are being evaluated. These species may not actually be present at the affected property but may be used to represent those within the feeding guild or community that may feed on the affected property.

Representative ecological receptors should be chosen to ensure that the potentially complete exposure pathways to the associated feeding guilds are included in the CSM (see **7.0**).

As discussed in greater detail in **9.1,** the selection of representative species should be based on several factors, including:

- ecological relevance
- potential for COC exposure
- sensitivity to the COCs
- social or economic importance
- species known or expected to be present
- availability of natural-history information

Sources and general information available for selection of site-specific ecological receptors include:

Government Organizations: Texas Biological and Conservation Data Systems (TPWD) and the U.S. FWS (National Wetland Inventory Maps at <www.fws.gov/wetlands>) provide maps or lists of species based on geographic location and are helpful in identifying threatened or endangered species or areas of special concern.

General Literature (field guides): These guides describe the flora and fauna of Texas that are useful in the development of habitat-specific food webs (see **6.2.2**). Examples include:

- The Vegetation Types of Texas Including Cropland by C.A. McMahan, R.G. Frye, and K.L. Brown (1984). This book is available online download from the TPWD website: <www.tpwd.state.tx.us/publications/pwdpubs/pwd_bn_w7000_0120/download_book/>.
- Mammals of Texas by W.B. Davis and D.J Schmidly (1994), also available online: <www.nsrl.ttu.edu/tmot1/>. The printed version of The Mammals of Texas is in its seventh edition (Schmidly and Bradley, 2016).
- The Patuxent bird identification information center (<www.mbr-pwrc.usgs.gov/id/framlst/framlst.html>) provides life history, food preferences and range maps for a variety of birds.
- A Field Guide to Reptiles and Amphibians of Texas (Garrett and Barker, 1987) and A Field Guide to Texas Snakes (Tennant, 1998).

Private or Local Organizations: These include the National Audubon Society; the National Geographic Society; local naturalist, wildlife, and birding clubs; state and national park systems; and universities.

6.5. Protected Species

An ERA must conservatively evaluate potential risks to protected species if they could occur at the affected property. ERAs often do not clearly present the evidence used to conclude that threatened and endangered species are not potentially exposed to COCs at the affected property, or they fail to discuss protected species at all.

Both federally-listed **and** state-listed species should be addressed. The preferred method for eliminating a protected species as being potentially present is by providing supporting documentation from a wildlife management agency to confirm the absence of that species on the affected property. Where input is sought from a wildlife management agency, it is preferable to initially consult with the TPWD rather than the U.S. FWS, since there are more state-protected species than federally-protected species, and the county lists provided by the TPWD reflect both state and federal species. However, if the species is federally listed and known to or have a significant potential to occur on the affected property, then the person may need to get a biological opinion from the U.S. FWS. County lists of rare, threatened, and endangered species of Texas are available at <tpwd.texas.gov/gis/rtest/>.

The TPWD has made great efforts to reduce the time needed for a consultation on protected species to 30–45 days. Therefore, there is little room for excuses to not pursue a formal consultation on this issue. To initiate the consultation, the person should contact the TPWD Wildlife Habitat Assessment Program described online at <tpwd.texas.gov/huntwild/wild/wildlife_diversity/habitat_assessment/>.

Although ERA Program personnel will be looking for this consultation during the technical review, the program recognizes that there may be occasional site-specific circumstances where such consultation is not feasible or warranted. In those instances, the person should provide a convincing discussion of the lack of suitable habitat by comparing the available habitat with the habitat needs of threatened and endangered species that could possibly occur in the county. It is not enough to simply state that no protected species are known to occur at an affected property; which is different from a supported statement that none are expected to occur based on the available habitat and the needs of a protected species. Any discussion of a lack of suitable habitat must be by a qualified individual (e.g., a local expert such as an academician, or a senior staff ecologist).

The TCEQ may request substantiation of this individual's qualifications (e.g., a résumé or descriptions of his or her wildlife-related projects in areas local to the site or in similar habitat). Additionally, the lack of observation of a species is not a good indicator of absence, as other variables (e.g., time of day, weather conditions, population densities, preferred habitat, and methods of observation) will be influential. A species' absence can only be determined through repeated negative observations that consider all the variable factors that can contribute to the absence. Also, the lack of a "critical habitat" designation is an insufficient justification alone. A copy of the county list of protected species should be included to support this discussion.

If the presence or absence of a protected species cannot be determined, then the species should be considered as being present and potentially impacted. The ERA must then demonstrate through exposure or PCL calculations that the species will either not be impacted or that protective PCLs will be developed, usually by calculating the exposure and evaluating the risk to the protected species or to a surrogate (a receptor from the same feeding guild) for the protected species. The ERA should also explain why the receptor chosen is a suitable surrogate for the sensitive species. When evaluating potential risks to the surrogate, life-history information (e.g., body weight, diet composition, home range) should be like that of the protected species and should be used in conjunction with TRVs, allometric equations for food or water ingestion and any appropriate uncertainty factors to estimate risk. It is inappropriate to eliminate the surrogate or the protected species from evaluation based on a lack of data or uncertainty in the available data.

Some protected species appear in the PCL Database. These species can be found by choosing the "Species" radio button and selecting the individual species from the drop-down list. Protected species (i.e., threatened and endangered species) appear in red lettering. Protected species also appear, as appropriate, in the major habitats.

6.6. Considerations for Species with Exposure and Toxicity Data Gaps

As previously mentioned, the TCEQ recognizes all species of reptiles, amphibians, livestock, and cave-dwelling receptors as species that have not been traditionally evaluated in ERAs. Nevertheless, the person should determine their potential presence at the affected property and corresponding ecological risks. The TCEQ acknowledges that a risk evaluation for these species is currently limited by the availability of exposure information and toxicity data. The following text focuses on the exposure pathway analysis for these general groups. Toxicity data applicable to these species appear in **9.2.3** and exposure evaluation procedures appear in **10.4.6.**

6.6.1. Reptiles and Amphibians

Reptiles and amphibians should be included as receptors in ERAs, as they are commonly found in Texas and may include sensitive or protected species. A qualitative or quantitative evaluation of amphibians and reptiles may be completed depending on available information on toxicology and life history for each species. A more rigorous evaluation is required where a threatened or endangered reptile or amphibian species may occur at the affected property.

The life histories for amphibian and reptile species indicate they are potentially sensitive receptors. In general, they are not as mobile as birds and mammals, and their home ranges are smaller, which could prolong exposure. Many are in constant, or at least frequent, contact with surface water or sediments (or soils).

Their feeding strategies can change during their lifetime, exposing them to a wider range of prey or forage items than birds or mammals. For instance, the larvae of some amphibian species feed on plant material and detritus on stream bottoms, whereas they are completely carnivorous as adults. Many reptiles and amphibians are upper TL predators, which makes them potentially sensitive to bioaccumulative COCs. Dermal exposure to COCs for amphibians is expected to be more significant than for reptiles, although dermal exposure could potentially be an important pathway for reptiles, as indicated by limited empirical data (Weir et al., 2010). Analysis of exposure pathways for reptiles and amphibians is specifically discussed below.

6.6.1.1. Reptiles

Reptiles are a diverse class including turtles, snakes, lizards, and the American alligator. Reptiles are linked by many traits (e.g., ectothermia, pulmonary respiration, epidermal scales, and internal fertilization), yet possess a diverse array of life-history characteristics and differences among species (e.g., population distributions, migration patterns, diets, and metabolic processes) (Gardner and Oberdörster, 2006).

Reptiles are declining globally. Hypotheses for reptile decline include habitat loss and degradation, impacts from invasive species, disease, parasites, global climate change, and environmental pollution (Gibbons et al., 2000). Detecting population declines in reptile populations is inherently difficult because of their cryptic or secretive nature, various home range sizes, low population densities and lack or rarity of congregational behavior (Irwin and Irwin, 2006). Turtles present a challenge for assessing risk from COCs because they are not only cryptic but extremely long-lived compared to other reptiles and many birds and mammals (Salice et al., 2014).

All reptiles are cold blooded or, more appropriately, *ectothermic* (i.e., their body temperature varies with that of the external environment). Unlike warm-blooded birds and mammals, reptiles are unable to regulate the temperature of the body internally. Reptiles can, however, absorb heat from the ground, from the surrounding air, and from objects next to them. Because they are ectothermic, reptiles do not need to maintain high levels of metabolism and consequently do not have to eat frequently. Additionally, growth does not have to be constant and can be discontinuous with long pauses between spurts. Snakes can fast between meals, sometimes for as long as three years (TPWD, 2016a). Therefore, exposure to COCs in prey items can be sporadic and acute.

Being ectothermic makes reptiles sensitive to both very cold and very hot temperatures; they can only survive within a narrow range of temperatures of about 30 degrees. Reptiles will behaviorally adapt to the temperature ranges of an area. In habitats where daytime temperatures are lethally hot, they will confine their activities to the night. In an area with excessively cold winter temperatures, a sudden drop in temperature will induce long periods of wintertime hibernation, with activities resuming in the spring with the return of warm temperatures (TPWD, 2016a). The natural fluctuations of temperatures and the seasons will influence exposures of reptiles to COCs in media and prey.

The wide variety of reptile species and their associated ecology, body weights, and preferred food, influence how reptiles are exposed to COCs in media or prey. Snakes (e.g., the timber canebrake rattlesnake, *Crotalus horridus*) represent top predators that could amass bioaccumulative COCs. There is also a wide variety of lizards and turtles in Texas. These include protected species of turtles (e.g., the alligator snapping turtle, *Macrochelys temminckii*) and lizards (e.g., the Texas horned lizard (*Phrynosoma cornutum*), which should be evaluated in the ERA if these species are likely to occur on the affected property.

Snakes: Texas hosts four of the five U.S. families of snakes: the slender blind snakes (Leptotyphlopidae), the advanced snakes (Colubridae, the largest group), the Old World fixed-front-fang snakes (Elapidae) and the New World pit vipers, the hinge-fanged snakes, which include the subfamilies Azemiopinae, Causinae, Crotalinae, and Viperidae (TPWD, 2016a). All species from the subfamily Crotalinae are venomous and the females are viviparous (give live birth). These snakes include the copperhead (*Agkistrodon contortrix*), the cottonmouth (*Agkistrodon piscivorus*), the western diamondback rattlesnake (*Crotalus atrox*) and the timber canebrake rattlesnake (*Crotalus horridus*), classified as threatened in Texas. Species from the family Colubridae include both egg-laying snakes, such as the eastern ratsnake (*Pantherophis obsoletus*) and viviparous snakes, such as the western ribbon snake (*Thamnophis proximus).*

No snakes are herbivores; all are confirmed carnivores. Depending on the species, prey consists of eggs, slugs, worms, insects of all kinds, crustaceans, fish, amphibians, other reptiles, and birds and mammals. Most snakes have restricted food preferences, while some will take anything that they can effectively subdue and engulf. Many snakes specialize in taking certain kinds of prey and have special adaptations to do so. Slender blind snakes (*Leptotyphlops dulcis dulcis*) specialize in eating ants and termites. Hognose snakes are toad specialists. The crayfish snake (*Regina rigida sinicola*), as its name implies, targets crayfish or similar crustaceans. Some snakes are fish eaters; others are bird-egg specialists. King and indigo snakes are snake specialists; smooth and rough green snakes (*Opheodrys vernalis* and *O. aestivus*) are insect eaters with a special liking for crickets, grasshoppers, and caterpillars. The Texas rat snake (*Elaphe obsoleta lindheimerii)* and gopher snake (*Pituophis catenifer*), are great friends of the farmer, eating mice, voles, and rats (TPWD, 2016a). The rough earthsnake (*Virginia striatula*) consumes only earthworms and is frequently found in backyards and open lots of eastern Texas, often when overturning logs and stones.

In general, snakes can live 10–25 years in the wild. ERAs should evaluate upper trophic level top predators, such as the threatened timber canebrake rattlesnake. Although snakes could be exposed to COCs in soils via dermal exposure and incidental soil ingestion, exposure is assumed to be primarily via ingestion of prey.

Lizards: The Texas horned lizard (*Phrynosoma cornutum* or horny toad) is one of four state threatened species. It is found in arid and semiarid habitats in open areas with sparse plant cover. Because horned lizards dig for hibernation, nesting, and insulation, they commonly are found in loose sand or loamy soils. Within the U.S., declines in population may be related to the spread of fire ants and use of insecticides for their control, heavy agricultural use of land and other habitat alterations, and over-collecting for the pet and curio trade. This species is extremely vulnerable to changes in habitat, especially the loss of harvester ants (Carpenter et al., 1993). Harvester ants comprise up to 69 percent of the diet (Pianka and Parker, 1975), and fire ants are thought to out-compete native harvester ants for food and space (Henke and Fair, 1998). The widespread use of broadcast insecticides is also thought to contribute to declines. Insecticides can be detrimental by directly causing illness or death or indirectly by severely reducing or eliminating harvester ants (Henke and Fair, 1998). Mortality from road traffic is also a threat.

Turtles: A variety of turtles are also present in Texas. Many of the species have especially long lifespans of several decades or more. The alligator snapping turtle (*Macrochelys temminckii*) is one of three state-listed species other than sea turtles. If potentially present, it should be evaluated for risks associated with surface water and sediment COCs. The alligator snapping turtle is the largest freshwater turtle in North America, reaching carapace lengths of 80 cm (32 in), and has a very long lifespan. It is most at home in deep rivers, lakes, and large streams with muddy bottoms. This is a mostly carnivorous species, feeding on many live or dead vertebrate or invertebrate animals; however, roots and fruit are found to be important components of the diet of those living in smaller streams and rivers. Other turtle species are included on their menu.

6.6.1.2. Amphibians

Amphibians are ectothermic vertebrates and many species have complex life cycles in which they change from an aquatic, water-breathing, limbless larva (or tadpole) to a terrestrial or partially terrestrial, air-breathing, legged adult (or, in the case of some salamanders, legless or with only hind legs). Amphibians include frogs and toads, salamanders and newts, and caecilians (limbless amphibians).

The natural histories for amphibian species indicate that they are potentially exposed to contaminants in multiple media. Many species of amphibians lay their eggs in water, and the larvae live immersed in water until metamorphosis to the adult stage. Some species of salamanders may remain aquatic throughout their adult stage. The larvae of frogs and toads (tadpoles), as well as some species of salamanders, have gills during early development or into the adult stage (i.e., certain species of salamanders), and so absorption of water across the gill membrane is a potential exposure route. COC transport across the skin, that may or may not be used for respiration in different species, could be the most significant route of exposure overall (Smith et al., 2007) for amphibians. Thus, exposure to surface water and sediment should be assessed for amphibians, particularly where sediment COCs may likely partition to surface water.

Many listed frogs and salamanders (TPWD, 2016b) could occur in many Texas counties, particularly along the Texas-Mexico border and in association with springs and karst-cave features (Gunnar, 2002). Potential exposure areas should be evaluated carefully where listed amphibians may be present at an affected property. The evaluation should consider, as appropriate, the potential for these receptors to be exposed to COCs as the amount of available habitat in temporary wetlands or pools diminishes with fluctuating water levels. COCs that may slow development or growth could reduce larval survival and adult fitness. A shorter larval stage is especially important for amphibians breeding in ephemeral pools or temporary ponds, since anything that lengthens the time to metamorphosis, including COCs in sediment or water, could lead to indirect mortality (see, e.g., Bridges and Semlitsch, 2005) if the water body dries up before metamorphosis is complete. Keep in mind that some amphibians can be exposed to multiple media, for example, Moriarty (2013) reports that frogs are exposed to arsenic via soil, food (invertebrates and plants), and water. Dermal exposure can also be significant but is currently an understudied exposure route. For example, amphibians indirectly exposed to pesticides (e.g., Atrazine) through contact with contaminated soil had measurable body burdens after eight hours of exposure (Van Meter et al., 2014).

6.6.2. Livestock

Potential risks to livestock receptors such as cattle, horses, goats, and sheep should be evaluated where livestock are known or expected to use a site. Although uptake of COCs by livestock may result in risks to humans (e.g., from consumption of meat or milk), SLERAs should evaluate the potential health risks to the livestock animals themselves because of exposure to site COCs. As a commodity, livestock health can be a public concern at affected properties. As ecological receptors, livestock animals are unique in that institutional controls can be used to limit their exposure to COCs. Livestock can be exposed to site COCs by feeding on plants that have accumulated COCs in their tissues, by ingesting impacted soil or sediment that has adhered to food matter, by deliberately or incidentally ingesting impacted soil or sediment, or by ingesting impacted water.

In two instances, the TRRP rule explicitly mentions livestock as potential receptors. First, the Tier 1 Checklist (**3.0**) characterizes livestock as potential ecological receptors in Subpart B. Affected Property Setting, as evidenced by the text prefacing the exclusion-criterion question: “In answering ‘Yes’ to the following question, it is understood that the affected property is not attractive to wildlife or livestock, including threatened or endangered species (i.e., the affected property does not serve as valuable habitat, foraging area, or refuge for ecological communities).” Additionally, the rule is specific that the surface water RBEL should preclude toxicity to livestock [30 TAC 350.74 (h)(7)(B)].

6.6.3. Cave-Dwelling Receptors

Texas has a rich but not well-known cave-dwelling fauna consisting of mammals, a bird, fish, amphibians, reptiles, insects, and other invertebrates. According to Reddell (1994), approximately 1,040 terrestrial and 150 aquatic species have been recorded from the state. Although these numbers have undoubtedly increased over the last 20+ years, it is estimated that over 50 percent of these known aquatic species and at least 15 percent of these known terrestrial species are *troglobites*—animals that are specially adapted to subterranean existence and spend their entire lives underground (e.g., endangered salamanders, cave beetles, cave spiders). Troglobites usually have small eyes (or no eyes), long appendages, reduced pigmentation, and other adaptations to a subterranean environment. In the last few years, several additional troglobitic species have been discovered in Travis, Williamson, Hays, and Bexar counties, which are among the most intensively studied counties in the state.

6.6.3.1. Habitat Requirements

The habitat of these species includes karst limestone caves and *mesocaverns*, which are humanly impassable voids. *Karst* is a type of terrain formed when calcium carbonate from limestone bedrock is slowly dissolved by mildly acidic groundwater (Veni and Associates, 2008). This process creates numerous caves, sinkholes, fractures, and interconnections so that in places the bedrock resembles a honeycomb. Within this habitat, karst animals depend on high humidity—typically near 100 percent for caves supporting troglobitic invertebrates (TPWD, 2010)—stable temperatures, and surface-derived nutrients including leaf litter, animal droppings, and animal carcasses. While these species spend their entire lives underground, their ecosystem is dependent on the overlying surface habitat (U.S. FWS, 2011a). The life span of troglobites is typically long relative to that of related surface species. Average life spans of the listed troglobitic invertebrates in central Texas are unknown, but are likely multiple years for some species, based on observations of juveniles kept in captivity (Veni and Associates, 2008).

Mesocaverns provide important sheltering habitat for karst invertebrates. During temperature extremes, small mesocavernous spaces may provide more favorable humidity and temperature levels than the larger cave passages (Howarth, 1983). Troglobites may spend most of their time in such retreats, only leaving them during temporary forays into the larger cave passages to forage (Howarth, 1987). Human access to mesocaverns is limited; therefore, data about troglobitic use of mesocaverns is limited. Scientists have hypothesized that most of the nutrients are in humanly accessible portions of terrestrial caves with open entrances (Culver and Pipan, 2009), and for that reason they are believed to be the foci of troglobitic populations that may occur in low densities throughout the karst. However, because metabolic rates of troglobites are typically low, they may be able to sustain periods ranging from months to years in mesocaverns with limited or no food (Howarth, 1983).

6.6.3.2. Importance of Surface Communities

Because there is little light and limited capacity for photosynthesis by plants, karst ecosystems depend almost entirely on surface plant and animal communities for nutrients and energy (Campbell, 2003). Caves receive nutrients from the surface in the form of leaf mulch, plant roots, and other organic debris that washes or falls into the cave. Cave crickets are especially important as a nutrient source because many invertebrates are known to feed on their feces, eggs, or on the nymphs and adults directly (U.S. FWS, 2011b). Cave crickets roost and lay eggs in caves during the day and leave the cave at night to feed on the surface. Raccoons and other small mammals are also important in many cave communities because their feces provide a rich medium for the growth of fungi and, subsequently, tiny insects that become prey for troglobites.

Surface plant communities surrounding karst features range from pastureland to mature oak-juniper woodland. In addition to providing nutrients to the karst system, maintaining adequate plant cover is important in minimizing temperature fluctuations and drying within the cave, and filtering pollutants before they enter the karst ecosystem as groundwater contamination (U.S. FWS, 2011a). Surface plant communities provide nutrients for *trogloxenes* (species that spend part of their life underground and part on the surface) on the surface and for karst invertebrates through leaf litter and roots that either wash or grow into caves.

Surface vertebrates and invertebrates are important components of a functioning karst ecosystem. Surface invertebrates that enter or are washed into caves provide food for some karst invertebrates and for trogloxenes, such as cave crickets, bats, toads, and frogs. Many of the vertebrate species that occasionally use caves bring in a significant amount of energy in the form of scat, nesting material, and carcasses (U.S. FWS, 2011a). Also, a healthy native arthropod community at the surface may better fend off red imported fire ants (*Solenopsis invicta*), a threat to the karst ecosystem (Porter et al., 1988, 1991; Taylor et al., 2003).

6.6.3.3. Drainage Basins

Water primarily enters the karst ecosystem through surface and subsurface drainage basins but can also percolate through the soil and mesocaverns (Cowan et al., 2007; Veni and Associates, 2008; Hauwert, 2009). Well-developed pathways, such as cave openings and fractures, rapidly transport water through the karst with little or no purification (White, 1988). Therefore, caves and karst are susceptible to pollution from contaminated water entering the ground (Drew and Hötzl, 1999). The surface drainage basin is dependent on topography and slope. It typically includes the cave entrance, adjacent sinkholes, and the adjacent soil (Cowan et al., 2007; Hauwert, 2009). The subsurface drainage basin includes mesocaverns, subterranean streams, bedding planes, buried joints, and sinkholes that have a connection to the surface that is not always observable from the surface (Veni and Associates, 2002). It is critical to have drainage basins with a natural quantity and quality of water because cave fauna require high humidity and materials brought in from the surface (U.S. FWS, 2011a).

6.6.3.4. Nutrients

Nutrients in most karst ecosystems are derived from the surface (Howarth, 1983; Culver, 1986), either from organic material washed in or brought in by animals or by feeding on the animals themselves. Habitat changes that affect nutrient sources can affect listed karst invertebrates because they are at the top of their food chain (Culver et al., 2000). Primary sources of nutrient input include leaf litter, root masses, and trogloxenes, such as bats, raccoons, snakes, and skunks.

For predatory troglobites, accidental species of invertebrates (those that wander in or are trapped in a cave) may be an important nutrient source in addition to other troglobites and *troglophiles* (species that may complete their life cycle underground but may also be found in dark, moist environments on the surface—e.g., earthworms, crickets, beetles, spiders, frogs, salamanders) (U.S. FWS, 2011a). In some cases, the most important source of nutrients for a karst invertebrate may be any of the fungi, microbes, or smaller troglophiles and troglobites found on the leaves or feces left inside a cave (Elliott, 1994; Gounot, 1994). In deeper cave reaches, nutrients enter through water containing dissolved organic matter percolating vertically through karst fissures and solution features (Howarth, 1983; Holsinger, 1988).

In some instances, eutrophication (excessive nutrients) of the surrounding surface environment may attract trogloxenes, which often take shelter inside caves. This can result in the trogloxenes bringing excess nutrients into a cave. For example, observations of decreased troglobitic diversity have been made in some caves that have excessive raccoon scat (Balcones Canyonlands Preserve, 2005, 2006, 2007).

6.6.4. Burrowing Receptors

Contaminated sites may support habitats for potential ecological receptors that dig burrows (or use existing ones) in various soil types. *Burrows* are holes or tunnels excavated into the ground for habitation or refuge. Most species primarily use burrows to provide shelter from predation or from extremes in weather and for protection while giving birth and raising offspring. Other species dig burrows to procure food and may live in them all year round (e.g., moles). Burrowing species may be key components of terrestrial habitats, particularly in arid environments where burrowing is an especially important life history strategy and a means to conserve body water.

As discussed in TRRP-15eco, normally only surface soil data should be used in the ERA, unless there is a site-specific reason for considering deeper soil. For sites where burrowing animals and those that occupy others' burrows are the measurement receptors, deeper soils (at depths of up to five feet below ground surface) may need to be considered, depending upon the assessment endpoints selected and the nature of the conceptual site model.

As burrowing species may be exposed to both impacted surface and subsurface soil on-site, it is appropriate that soil exposure concentrations from both intervals be considered. Commonly-found, non-rodent burrowing species include armadillos, moles, foxes, skunks, and rabbits. The determination of the presence of these species and the collection of subsurface soil is based on the following conditions:

- if burrows greater than three inches wide and deeper than six inches are found on-site; and
- there is evidence of subsurface contamination from samples collected for human health purposes or there is knowledge of historical subsurface contamination; then
- samples from 0.5–5.0 feet should be collected, and burrowing receptors should be evaluated for subsurface soil exposure.[17]

[17] An exception would be where human health samples indicate that all COC concentrations are higher in the surface soil, but the sampling intervals would need to be comparable to those for ecological subsurface soil.

If both conditions cannot be met, but a burrowing species has been observed on-site or there are indications of its presence (e.g., scat, tracks), this species should be included as a measurement receptor, but only for exposure to surface soil.

According to TRRP-15eco, except for incremental sampling, surface soil samples to support ERAs should generally not be composited. However, for subsurface soil (0.5–5.0 feet), a depth-integrated composite sample (from a single core sample) may be used to analyze COCs, the rationale being that a burrowing animal is likely to be exposed across this depth interval. Also, food items (i.e., invertebrates and plant roots) may take up COCs across this depth interval. Samples submitted for volatile COC analyses should not be composited due to the potential for COC loss during mixing.

Sufficient subsurface samples should be collected to allow development of a 95 percent UCL. In the initial exposure evaluation (required element 6), this subsurface UCL should be compared to the surface soil 95 percent UCL. The higher of these UCLs should be applied to all aspects of the burrowing receptor's exposure (i.e., food ingestion and incidental soil ingestion). In the refined assessment (required element 7), the person may propose to use justified percentages of the two UCLs to evaluate risk.

Alternatively, the person may collect subsurface samples from various depths (e.g., 0.5–2.0 feet, 2.0–3.0 feet, 3.0–5.0 feet) and use statistics to develop a representative concentration; however, obtaining prior approval from the TCEQ is strongly suggested and will depend on site conditions (e.g., receptor to be evaluated, plant type, root depth). Other methods to evaluate subsurface soil exposure may be proposed and will be reviewed on a case-by-case basis.

Inhalation of VOCs is a recognized and an important exposure pathway for burrowing animals, particularly at sites where VOCs are present in the groundwater and subsurface soil. Here, soil vapor could concentrate within burrows because the potential for atmospheric dilution is limited. Much progress on evaluating these pathways has been made over the last decade. The development of inhalation-based ecological soil screening levels has greatly facilitated this evaluation (Gallegos et. al., 2007). However, most of the studies conducted at air force bases and Superfund sites have shown no significant risk to burrowing receptors. The TCEQ will consider the appropriateness of evaluating this pathway on a site-specific basis.

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7.0 Conceptual Site Model (Required Element 3)

The purpose of the conceptual site model (CSM) is to illustrate the complete (or reasonably anticipated to be complete) ecological exposure pathways in the ERA. A CSM is required element 3. The steps to building the CSM include:

- Determine the types of habitat supported by or on the affected property.
- Relate habitats on the affected property to: one or more of the seven major habitats and their associated food webs (see **Appendix A**); or to a site-specific food web (see **6.2.2**); or to Minor Habitat and its sub-habitats (see **6.2.3** and **Appendix B**).
- Include the direct (media) exposure pathways and define the routes of exposure to each community, feeding guild or representative species.
- Refine exposure pathways for applicability of COCs to any feeding guild.
- Build the CSM for the food web and COCs by illustrating the potential contaminant sources, release mechanisms, transport pathways, exposure media, and receptors considered for the SLERA.

The CSM graphically depicts the movement of COCs from sources through media to the community and feeding guilds or to the selected ecological receptors of those guilds (i.e., measurement receptors). The format of the CSM can be graphic or a line-and-box drawing and can be simple to complex. Figure 7.1 is an example CSM and uses the following exposure categories:

- The exposure pathway is complete and significant and therefore a quantitative evaluation will be presented in the ERA. This notation represents the primary complete exposure pathways evaluated quantitatively in the ERA. Examples include soil and food ingestion by mammals and birds.
- The exposure pathway is complete or potentially complete, but quantitative evaluation is not possible, and a qualitative evaluation will be completed. An example of this pathway is dermal exposure of soil to reptiles. The pathway is recognized by the person and the limitations of the evaluation are also recognized.
- The exposure pathway is minor or insignificant; no evaluation is presented in the ERA. An example of this would be dermal exposures to birds and mammals, which is not expected to be significant.
- The exposure pathway is incomplete. An example of this is water ingestion for birds in a coastal area.

Although not incorporated into the example CSM, a ‘stop’ sign can be used to signify when a pathway is not complete for physical reasons. An example would be a site whose hydrogeology does not indicate that the groundwater discharges to surface water.

Development of the CSM can be an important tool in communicating the exposure pathways and potentially any unique characteristics of a site. The CSM is a living document and should be updated as additional site information is developed during the investigation process. For instance, for the groundwater–to–surface water pathway, the site investigation may determine that the groundwater only discharges from the uppermost groundwater-bearing unit and only in a portion of the site. This information should be presented within the text of the SLERA, and graphically on the CSM.

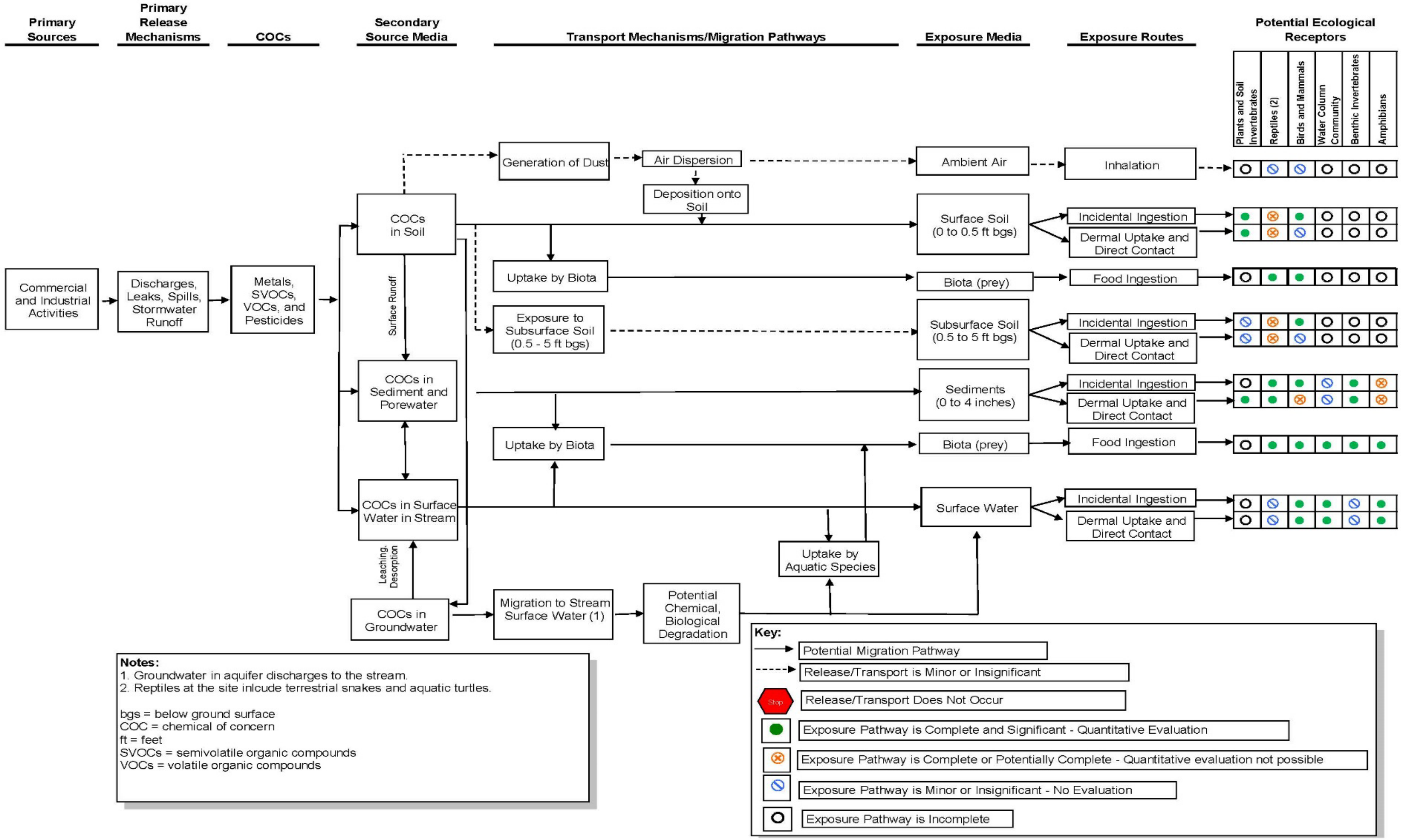

Figure 7.1. Ecological conceptual site model.

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8.0 Fate and Transport, Toxicological Profiles (Required Element 4)

According to the TRRP rule [350.77(c)(4)], a discussion of COC fate and transport and associated toxicological profiles is required element 4 of the Tier 2 SLERA. A determination should be made as to whether the COCs at the affected property are likely to persist, be degraded, or move beyond the extent of contamination initially determined in the affected property assessment. During the assessment, the person characterizes the nature, extent, and potential fate and transport of COCs. This characterization includes physical, chemical, and biological processes and their influence on the movement, persistence, form, toxicity, and availability of COCs to the degree necessary to understand and characterize risk.

If a COC in an aquatic ecosystem is highly lipophilic (i.e., essentially insoluble in water), it is likely to partition primarily into sediments and not into the water column. Factors such as sediment particle size and organic carbon influence COC partitioning and should be characterized when sampling sediments. Similar considerations regarding partitioning apply to COCs in soils (U.S. EPA, 1997a).

COCs can undergo any of several chemical processes in the environment, including:

- degradation (e.g., photooxidation)
- complexation
- ionization
- precipitation
- adsorption
- radioactive decay

Physically, COCs can move through the environment by one or more means, such as:

- volatilization
- erosion
- deposition (COC sinks)
- weathering of parent material with subsequent transport
- water transport:
- in solution
- as suspended material in the water

- bulk transport of solid material

Several biological processes also affect COC fate and transport in the environment, including:

- bioaccumulation
- biodegradation
- biological transformation
- food-chain transfers
- excretion

Information should be gathered on past and current mechanisms of COC release from source areas at the affected property. The mechanisms of release, along with the chemical and physical form of a COC, can affect its fate, transport, and potential for reaching ecological receptors. Any chemical or physical parameters (e.g., vapor pressure, solubility, log K_{ow}) used in the Tier 2 SLERA should be sourced from the TCEQ's table of chemical and physical parameters included in the human health PCL tables.

The toxicity profiles presented in the Tier 2 SLERA should describe the toxic mechanisms of action, to the degree known or available, and the exposure routes being evaluated. Understanding the toxic mechanism of a COC helps to evaluate the importance of potential exposure pathways and focuses the selection of assessment endpoints. Toxicological profiles should focus on the information needs relevant to the site-specific situation, depending on the site. For instance, discussions of aquatic toxicity are not relevant to a terrestrial site. Further, discussion of human health toxicity is not necessarily relevant to a SLERA unless mammalian toxicity is pertinent to both.

Professional judgment should be used to determine which COCs (or groups of COCs) should be described, but relevant considerations should include COC toxicity, bioaccumulation, and site concentrations. Toxicological profiles available in various databases (e.g., HSDB, Medline, Toxline, ECOTOX) can be used to evaluate the likelihood of toxic effects in different groups of organisms.

The PCL Database incorporates toxicological profiles from the sources mentioned above, as well as others, for all the COCs it currently contains. These profiles have been compiled to reflect only the most meaningful ecological effects of the COCs and therefore are very amenable to ERA development; and thus, the TCEQ encourages their use.

In the PCL Database on the "PCL Calculator" page, choose the "Chemicals" tab toward the top of the page (not the drop-down menu under Step 2). Find the COC under the "Chemical Name" column on the left side of the screen. Click on the CAS number of the COC of interest and wait for the PDF chemical profile to appear. The PDF contains fate and transport information, the toxicity profile, water quality criteria, bioaccumulation factors, TRVs, and all associated references.

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9.0 Receptor Effect Levels (Required Element 5)

Required element 5 of the TRRP rule initially concerns the collection and preparation of the input data and their use to calculate the exposure (dose) for the receptor, whereas **10.0** addresses exposure considerations under required element 5. The discussion here is of the various types of effect levels as inputs and their use in the PCL Database. Effect levels can be single-point media values (e.g., mg/kg, mg/L) or doses used for direct comparison to:

- concentrations in site-specific media (surface water, sediment, or soil);
- dose-based TRVs (mg/kg of body weight per day); or
- concentrations of COCs measured in tissues that are correlated to an effect.

This text also discusses species with toxicity data gaps.

9.1. Selecting Measures of Effect (Measurement Endpoints)

Measures of effect are used to evaluate the response of the assessment endpoint when exposed to a COC. Measures of effect are selected as:

8. toxicity values developed or adopted by state or federal agencies (e.g., TCEQ surface water quality criteria) for protection of media-specific communities; or
9. receptor-specific chronic dose-based toxicity values (e.g., NOAELs or LOAELs or Effect Concentrations) for ecologically relevant endpoints.

The evaluation of the measure of effect to the assessment endpoint requires identification of a measurement receptor representative of the assessment endpoint. A species, population, community, or assemblage of communities may be selected as a measurement receptor, specific to each feeding guild. **Appendix B** discusses selection of measurement receptors for Minor Habitat, as presented in **6.2.3** of this document**.**

9.1.1. Measurement Receptors for Communities

When a community that occupies surface water, sediment, or soil or an assemblage of such communities is selected as the measurement receptor, no specific species are selected (unless there are protected-species concerns). Therefore, it may be inferred that critical ecological attributes of these communities are not adversely affected if a COC concentration in the respective medium does not exceed the toxicity benchmark specific for that community (see the benchmark tables). Remember that this exercise in comparative concentrations is conducted as required element 1 (see **5.0**). COC concentrations that exceed these benchmarks but do not subsequently prove a risk to upper trophic level receptors may still have an impact on these community-level receptors. As indicated during the discussion of the definition of *ecological PCL*, depending on site-specific circumstances, the person may be required to demonstrate that impacts on these communities will not result in unacceptable consequences for the more mobile or wide-ranging receptors.

Representative measurement receptors for communities include:

- *surface water*—phytoplankton, water-column invertebrates, fish
- *sediment*—benthic invertebrates
- *soil*—soil invertebrates and terrestrial plants

Fish, benthic macroinvertebrates, soil invertebrates, and plants are frequently considered community-level assessment endpoints in that the species richness and abundance of the communities are viewed as the endpoint properties, rather than properties of the component populations (Suter, 1996a). In the evaluation of community-level receptors, indirect exposure and effects should be considered. For example, toxicity data for aquatic invertebrates (an important food source for fish) as well as fish should be considered for evaluation of the fish community—an effect on aquatic invertebrates could result in indirect effects on fish. In addition, some fish species may experience toxic effects at concentrations that affect aquatic invertebrates (Suter, 1996b) due to similar sensitivities.

9.1.2. Measurement Receptors for Feeding Guilds

A measurement receptor should be selected for each **evaluated** class-specific feeding guild to model the COC ingested dose from media and the COC concentration in food. The selected measurement receptor should be representative of other species in the guild with respect to its feeding niche in the ecosystem. The risk assessment should demonstrate that using the measurement receptor ensures that risk to other species in the guild is not underestimated, particularly if protected species are a concern.

These factors should be evaluated when selecting a representative measurement receptor:

Ecological relevance. Highly relevant receptors fulfill an important function or form an important structure in the ecosystem. Attributes of these receptors typically fall under the categories of food, habitat, production, seed dispersal, pollination, and decomposition. Critical attributes include those that affect or determine the function or survival of another population in the same community. For example, a population of forage fish might be critical to the sustainability of a population of carnivorous game fish.

Exposure potential. These receptors tend to have higher potentials for exposure than other receptors, due to their metabolism, feeding habits, location, or reproductive strategy. For example, the metabolic rates of small receptors are generally higher than those for large animals, resulting in a higher ingestion per body weight—i.e., increased exposure potential. Receptors with significantly higher incidental ingestion of exposure media (e.g., sandpiper species and sediment) often have the highest exposure potential within a feeding guild.

Sensitivity. Highly susceptible receptors include those with low tolerances to a COC as well as receptors with enhanced COC susceptibility due to other concomitant stressors that may not be related to a COC, such as reduced habitat availability. For example, raptors are highly sensitive to the effects of chlorinated pesticides that bioaccumulate through the food chain.

Social or economic importance. An assessment endpoint may also be based on the social importance (e.g., songbirds) or economic importance (e.g., big game) of the receptor. For these receptors, critical attributes include those that affect growth, reproduction, and survival.

Presence of known, expected, or protected species. The receptor need not always be present or even occur at the affected property to be selected as a measurement receptor. A species that is expected to occur, based upon its range and the availability of suitable habitat, but does not occur because of its susceptibility to COCs from the affected property, may be a good measurement receptor. If a protected species is potentially present at the affected property, a measurement receptor may be selected that will not underestimate risk to the protected species.

Consideration of small-ranging and wide-ranging receptors. The selection of measurement receptors should consider site conditions regarding use by small-ranging receptors (i.e., home range ≤ 1 hectare) and wide-ranging receptors (i.e., home range > 1 hectare). These receptors will have different exposures, with small-ranging receptors usually receiving more exposure than wide-ranging; however, the person should also consider the applicability of ecological PCLs to small-ranging receptors in required element 9 (see **13.1** for discussion of ecological PCLs for small-ranging receptors).

Availability of information on natural history and toxicology. Information on natural history and toxicology is essential to the quantitative evaluation of risk to measurement receptors. If this information (e.g., body weight, food, and media ingestion rates, and COC-specific TRVs) is unavailable for the desired measurement receptor, information for one or more of these exposure variables should be obtained from a closely-related species. Consultation with a TCEQ ecological risk assessor on this issue is recommended, particularly for amphibians and reptiles. Uncertainty associated with using such surrogate information should be discussed (see **12.0**).

Note that more than one measurement receptor can be selected per assessment endpoint. Also, although each of these factors should be evaluated when selecting the measurement receptor, one of the measurement receptors selected to represent a class-specific feeding guild should have the highest exposure potential (i.e., highest ingestion rate per unit body weight) and the greatest sensitivity (i.e., lowest TRV). Identification of a receptor that is both the most exposed and the most sensitive to a COC and its use as the measurement receptor ensures that risk to other species in the guild is not underestimated. Users will not have to evaluate all feeding guilds for a food web provided there is a logical justification.

The U.S. EPA's *Wildlife Exposure Factors Handbook* (1993a) is an excellent source of information on diet and other natural history, although other sources have been previously identified. However, receptor information obtained from any source should be verified and documented.

9.2. Characterization of Ecological Effects

From the affected property assessment and remediation perspective, one of the key tools used to gauge the potential for risk to wildlife is the TRV. For each COC with a complete exposure pathway that is not excluded by a comparison to non-bioaccumulative ecological benchmarks, a TRV should be identified or developed. For wildlife, a TRV is generally defined as a dose above which ecologically relevant effects might occur following chronic dietary exposure and below which it is reasonably expected that such effects will not occur. TRVs are typically derived from laboratory or field toxicity studies that are evaluated for population scale, or relevant responses (such as growth, reproductive success, fecundity, offspring impacts, and mortality).

Significant tasks in the Tier 2 SLERA effects characterization are:

1. collecting toxicity data from the literature
    - critical evaluation of the data
    - selection of appropriate endpoint values
    - extrapolation between tested systems and the major feeding guilds that are the focus of the SLERA

The TCEQ advocates selection of the most relevant study or studies available for determination of TRVs. Non-preferred studies may be used to corroborate conclusions or to contribute to a weight-of-evidence case, but the endpoint should be derived from a preferred study. The study selected (for species and endpoint) should reflect its relation to the Texas feeding guilds supported by the habitat at the affected property and the endpoints most likely to affect populations (e.g., growth, reproduction, and mortality). Dermal and inhalation exposure routes for wildlife are typically not addressed in ERAs due to limited toxicity information and, generally, the lesser significance of these exposure routes relative to oral exposure. Circumstances do exist, however, where dermal and inhalation exposure may be significant, such as for amphibians, burrowing wildlife, and species that inhabit the burrows of others.

9.2.1. Selecting TRV Data from the Literature

In the ideal case, a TRV from the literature will be available for a COC based on a study specific to the species selected to be representative of the feeding guilds that are evaluated. In such a case, the overall quality of the study should be evaluated and, if the study is acceptable, the NOAEL and LOAEL endpoints should be determined from the study and used as TRVs in the risk characterization. Studies with paired NOAELs and LOAELs based on a chronic duration and ecologically relevant endpoint (e.g., growth, reproduction, or survival) are preferred.

One concern with use of documents summarizing data or presenting reference values is the nature of the data collection and screening used to derive the values. Compendia of TRVs may in fact be tertiary collections of values derived by others. The original papers summarized in these compendia should be evaluated whenever possible to ensure acceptable data quality and relevance to the ecological PCLs defined in the rule (e.g., impairment of microbial processes would not be useful information), and to ensure that the TRV was calculated correctly. Original literature is preferred, particularly where critical endpoint values are derived. However, if using a compendium document that contains multiple NOAELs and LOAELs for growth and reproduction of acceptable quality (e.g., EPA Eco-SSL documents), the person may propose a geometric mean of the NOAELs and LOAELs as the TRVs.

9.2.1.1. Uncertainty Factors

When toxicity information for a COC is incomplete, uncertainty factors (UFs) can reduce the likelihood of underestimated risk. Historically, UFs have been used for various extrapolations, and their applications reflect policy to derive conservative estimates of risk (Chapman et al., 1998). Although UFs may reduce the probability of underestimating ecological risk from exposures to releases of COCs, the use of too many UFs may result in an overestimation of risk.

UFs should be used to convert a toxicity value to a chronic NOAEL-based TRV (TRV_{NOAEL}). In most cases, the UFs discussed below should be applicable to available toxicity values. In some cases, however, irregular toxicity data (such as, a subchronic LC_{50}) may be the only available information. In these cases, the toxicity data should be thoroughly reviewed, and professional judgment should be used to identify appropriate UFs that are consistent with those listed below. Special attention should be taken with toxicity values from single oral-dose and intraperitoneal tests. Specifically, UFs should be used to account for extrapolation uncertainty due to differences in test endpoint and exposure duration:

- Test endpoint uncertainty—extrapolation from a non-NOAEL endpoint (e.g., LOAEL, LD_{50}) to a NOAEL endpoint
- Duration uncertainty—extrapolation from an acute or subchronic duration to a chronic duration
- Except as noted above for irregular toxicity data, these UFs may be used to convert a toxicity test endpoint to a TRV that is equivalent to a chronic NOAEL:
    - A chronic LOAEL should be multiplied by an uncertainty factor of 0.1 to convert it to a chronic NOAEL.
    - A subchronic NOAEL should be multiplied by an uncertainty factor of 0.2 to convert it to a chronic NOAEL.
    - An acute lethal value (such as an LC_{50} or LD_{50}) should be multiplied by a UF of 0.01 to convert it to a chronic NOAEL.

Although the UFs listed above are preferred, alternative UFs may also be obtained from the open literature. The SLERA should present the TRVs used and list any UFs applied to the TRVs.

9.2.1.2. Extrapolation to Texas-Specific Ecological Receptors

It is preferred that TRV data be for test species comparable to species in the Texas feeding guilds, as taxonomic similarity is a widely used criterion. Data from species more closely related to those occurring on the affected property are preferred to data from more distant species. The mode of exposure (e.g., dietary or media) for the test species should reflect the pathway analysis for the affected property. The measured effect should correspond to growth, reproduction, or mortality endpoints. Ideally, all TRVs should be derived with a thorough understanding of the underlying mechanism of toxicity and physiological differences between species; additionally, it is preferable that there be enough toxicological data available to fit species-specific dose-response curves (Allard et al., 2010).

9.2.1.3. Benchmark Dose Analysis

The development of wildlife TRVs is often limited by the available data. Advances in wildlife ecotoxicology and dose-response assessment can provide an opportunity to reduce uncertainties (Filipsson et al., 2003; Davis et al., 2011 and Mayfield et al., 2014). One of these developments is the use of benchmark dose (BMD) software which involves modeling all tested dose levels and variability of responses. It allows for estimation of the entire dose-response curve, specified levels of response (e.g., ED_{10}, ED_{20}, ED_{30}) and confidence intervals (Mayfield and Skall, 2014). Use of BMD methods involve fitting mathematical models to dose-response data and using the different results to select a BMD that is associated with a predetermined benchmark response. EPA's Benchmark Dose Software facilitates these operations by providing simple data-management tools and an easy-to-use interface to run multiple models on the same dose-response data (<www.epa.gov/bmds>).

9.2.2. Derivation of TRVs in the Ecological PCL Database

TRVs are available from the PCL Database for use in Tier 2 and Tier 3 ERAs and follow a standard methodology for development. Instructions for accessing TRVs in the PCL Database appear at the end of this section. The selection criteria for determination of TRVs included:

- The primary literature was published in either a peer-reviewed journal or document from a government agency [e.g., the U.S. EPA, the U.S. Department of Defense (U.S. DOD), Agency for Toxic Substances and Disease Registry ATSDR]. Supporting information was presented in the primary reference, including exposure and effect information such as the chemical form (e.g., salt or oxidation state for metals), test species, age, sex, test endpoint and effect type, method and frequency of dosing, number of doses used, exposure duration, and whether statistics were used to identify the TRV.

- The exposure preference was oral, specifically through food or drinking water. TRVs based on intravenous or intraperitoneal exposure were generally not considered because of irrelevance to environmental exposure. TRVs based on inhalation were not considered for oral exposure.

- Both NOAEL and LOAEL endpoints were defined by the original study.
- The TRV was reported as a dose (mg/kg-day) rather than a concentration (mg/kg or mg/L).
- If the TRV was reported as a concentration, a dose was estimated using the body weights and ingestion rates.
- The TRV was based on a **measured** dose or concentration.
- The TRV was based on exposure to organisms during a critical life stage (e.g., juveniles or reproduction).
- An appropriate range and number of doses were tested.
- The exposure duration, in order of preference, was: chronic, subchronic, acute.
- The effect type measured was relevant to the sustainability of the population (e.g., for reproduction endpoint: percent of surviving progeny vs. weight of eggs).
- The TRV was based on a test organism taxonomically like a receptor species, when possible.
- TRVs based on cattle and other ruminants were not considered for wildlife due to (a) the difference in body weight between these animals and most receptors, such as shrews and mice, and (b) the differences in the digestive systems of ruminants vs. most receptors in the PCL Database (See **9.2.3.3** for a discussion of livestock).

Secondary literature sources containing large amounts of peer-reviewed TRVs were often used to locate primary studies. The main secondary literature sources included U.S. EPA's Eco-SSL documents and Sample et al. (1998). When TRVs could not be obtained from these sources, various online databases were searched using the chemical's name, synonyms, and Chemical Abstract Services Registry Number (CAS No.).

Theoretically, TRVs based on growth and reproduction should be lower than TRVs for mortality since growth and reproduction are typically more sensitive endpoints. However, that was not always the case, because TRVs for sublethal endpoints could be based on exposure (a) to a less-toxic form of the COC, (b) to adult organisms (whereas TRVs for mortality were based on exposure to neonates or juveniles), (c) to a less-sensitive test species, or (d) over a short (subacute or subchronic) duration. Some COCs simply did not appear to cause adverse effects on growth or reproduction below lethal levels due to differing mechanisms or modes of action.

If TRVs for growth or reproduction exceeded TRVs for mortality, critical TRVs were chosen by selecting values that would be critical to the survival or sustainability of the population. If critical TRVs were chosen, they were included in the toxicological profile along with an explanation of their selection.

TRVs were not adjusted for body weight using allometric scaling. Allometric equations for adjusting TRVs from test species to wildlife species developed by Sample and Arenal (1999) were developed for acute endpoints and were not considered appropriate for extrapolating chronic TRVs across body sizes.

In the PCL Database on the "PCL Calculator" page, choose the "Chemicals" tab toward the top of the page (not the drop-down menu under Step 2). Find the COC under the "Chemical Name" column on the left side of the screen. Click on the CAS number of the COC of interest and wait for the PDF chemical profile to appear. The PDF contains the TRVs used in the development of the PCLs.

9.2.3. Considerations for Species with Exposure and Toxicity Data Gaps

The TCEQ recognizes that health-effects data specifically for reptiles, amphibians, livestock, and cave-dwelling receptors are sparse for many COCs. The rest of Chapter 9 presents a current assessment of effects data for these receptors, along with some recommendations on application.

9.2.3.1. Toxicity Data for Reptiles

During the past decades, reptilian toxicology has made up a disproportionately small percentage of toxicological studies of vertebrates. Characteristics of some reptile species make them difficult to study, including long life span and generation time, low fecundity, and incompatibility with laboratory handling techniques.

Currently, much less is known about the accumulation and effects of COCs in reptiles than in any other vertebrate class, making prediction of COC impacts on reptiles difficult. Risk predictions based on toxicity thresholds established for other vertebrates (e.g., birds and fish) may be inappropriate for many reptiles (Weir et al., 2010) because of their unique combination of physiological and life history characteristics (e.g., long life span, relatively small home ranges, high trophic position, and ectothermic physiology) (Hopkins et al., 2002). Reptiles may respond differently from birds and mammals to some environmental contaminants because their metabolic rates may slow the elimination and detoxification of toxic substances. Reptiles may maintain higher body burdens of COCs. Many reptile species are known to store significant amounts of body fat, which may serve to bioaccumulate lipophilic COCs. Many reptiles are predators or scavengers that occupy high positions in trophic food chains, potentially resulting in an increased exposure to persistent contaminants because of biomagnification (Selcer, 2006).

Reptiles can be exposed to COCs by several routes, including ingesting contaminated food or soil, contact with skin, maternal transfer into eggs and embryos, and uptake from the nest materials by incubating eggs. Although ingestion of contaminated food is probably one of the most important routes for COCs to enter reptiles, ingestion of soil could also be an important route for the uptake of soil COCs (Rich and Talent, 2009).

Of the types of reptiles, turtles have been studied more frequently than others; most of the studies have focused on organic contaminants (Hopkins et al., 2002). In general, past reptile studies have focused on measuring body burdens of various pollutants from samples collected in the field. While those data are useful for understanding historical exposures of given populations, the actual risks, and population-level effects of pollution on reptiles are still largely unknown and generally under-studied (Weir et al., 2010).

Relatively few laboratory studies have been conducted on the dose-response of toxicants and no standardized tests involving reptile models are in use (Talent et al., 2002). Campbell and Campbell (2001, 2002) reviewed the open literature for metals data for reptiles and reported one study using snakes in their 2001 publication, and three effects studies for lizards and five for snakes in their 2002 publication. Campbell and Campbell (2001) states: “The available data on reptiles were too scanty to allow for meaningful analysis of levels or effects.” Fryday and Thompson (2009) collated chemical toxicity data to reptiles available in the scientific literature. Few effects values were found, and most studies did not calculate LD_{50} or LC_{50} values, but only reported mortality or symptoms. The lack of standard dose-response toxicity testing makes determining a TRV virtually impossible or very imprecise.

Researchers are considering various reptile species as environmental indicators (Heinz et al., 1980; Clark et al., 2000); although there is not a consensus on test species or testing protocols. Standardized methodology for reptile toxicity testing is important for future toxicity testing. Weir et al. (2015) recommended the use of gelatin capsules for oral dosing and a variety of aqueous solutions for dermal testing. Maintenance of breeding populations of most reptile species under laboratory conditions is not practical because of their size and maturation rate.

Talent et al. (2002) proposed the western fence lizard (*Sceloporus occidentalis*) as an excellent candidate for a laboratory reptile model because a complete life cycle can be completed under laboratory conditions in less than a year and each stage in the life cycle can provide several endpoints for evaluating the effects on environmental toxicants. For example, they bury their eggs in moist substrate, and water-soluble contaminants could be transported into the egg. Selcer (2006) evaluated candidate test species for turtles, lizards, snakes, and crocodylia. For snakes, the genus *Thamnophis*, which includes garter and ribbon snakes, was proposed because of their broad range from Canada to Mexico, common distribution including urban settings, use of a variety of habitats, and opportunistic diet. The garter snake has been studied extensively from the standpoint of reproductive ecology and physiology, although no toxicity studies on metals were found in the literature. Selcer (2006) also recommended water snakes (*Nerodia* spp.) as a good toxicology model because they are primarily aquatic, widely distributed, and reasonably abundant.

The TCEQ recommends that the person search the open literature for studies relevant to the site COCs, type of reptile (snake, turtle, or lizard), method of exposure, and endpoint. For example, if lead is a site COC and the Texas horned lizard is present, a study by Salice et al. (2009) could be used to determine a toxicity endpoint for lead. The authors studied the toxicity of lead acetate to the western fence lizard. Acute-lethal-dose and subacute (14-day) toxicity studies were used to narrow exposure concentrations for a subchronic (60-day) study. In the subchronic study, adult and juvenile male lizards were dosed via gavage with 0, 1, 10, and 20 mg/kg-day. Mortality was limited and occurred only at the highest dose. There were statistically significant sublethal effects at 10 and 20 mg/kg-day on body weight, cricket consumption, organ weight, hematological parameters, and post-dose behaviors. Of these, lead-induced changes in body weight are most useful for ERAs, because they link to fitness in wild lizard populations. Applying a UF of 0.2 based on the duration of subchronic exposure, the PCL Database uses a NOAEL of 0.2 mg/kg-day and LOAEL of 2 mg/kg-day for the growth endpoint derived from this study for lead exposure to reptiles.

Lacking COC-specific toxicity data for reptiles, a bird TRV can be used along with information on reptile life history (e.g., body weight, food ingestion rate) to calculate a dose and an HQ. Although the TCEQ does not normally encourage extrapolations across classes, this is the one occasion where it is allowable. In fact, this is preferred where a protected species may occur at a site. If this approach is used, the TCEQ recommends a UF of 0.1 for the extrapolation. See the case study for an example of this approach using the PCL Database. All assumptions will need to be discussed in the uncertainty analysis. See **10.4.6.1** for recommendations on the exposure dose equation for reptiles.

9.2.3.2. Toxicity Data for Amphibians

Toxicology information for amphibians for COCs may be available from Linder et al. (2010), Sparling and Krest (2010), ENSR (2004), or an online literature search from a database such as ECOTOX <cfpub.epa.gov/ecotox/> or TOXNET <toxnet.nlm.nih.gov>. A database called the Reptile and Amphibian Toxicological Literature (RATL) from the Canadian Wildlife Service (Pauli et al., 2000) should be reviewed for applicable toxicological data; available online at <publications.gc.ca/collections/Collection/CW69-5-357E.pdf>. The open literature should also be reviewed as more research is being conducted on amphibians. For example, Sparling et al. (2006) exposed larval southern leopard frogs (*Rana sphenocephala*) to lead-contaminated sediments to determine lethal and sublethal effects. Where toxicity data are available, amphibians can be evaluated based on media-specific effects concentrations, or in some cases based on an ingested dose.

Exposure to surface water COCs can be much more pronounced for amphibians than for reptiles. As discussed in Rowe et al. (2003), the entire integument of larval amphibians and some species of adult salamanders is very thin and highly vascularized, and functions as a respiratory surface in many species (in addition to the gills). Additionally, cutaneous respiration and water exchange are important mechanisms of gas exchange and osmotic regulation in juveniles and many adults. The concentrations of COCs in eggs and larvae may be equal to ambient surface water concentrations (Birge et al., 2000). For an ERA, the numeric water quality criteria specified in the TSWQS and ecological benchmarks are assumed to be protective of amphibians. This assumption is supported by the derivation of numeric criteria protective of aquatic organisms (for freshwater), which includes the requirement for a third family in the phylum Chordata, including amphibians (Stephan et al., 1985). For amphibians, significant effects data (e.g., LC_{50} endpoints) are available for evaluating exposure to toxicants in surface water.

If a protected amphibian species could be exposed to a COC that **does not** have a state-adopted or federal criterion, the person should further evaluate potential risk to that species through effects data. Some effects data (e.g., LC_{50} endpoints) are available for evaluating amphibian exposure to COCs in surface water, but if non-amphibian (e.g., fish) data are used, the person should evaluate the sensitivities between amphibians and the test species. As mentioned above, Weltje et al. (2013) found that fish and amphibian toxicity data are often correlated; however, there are exceptions such as diazinon and nonylphenol. If the sensitivities between the test species and amphibians are unknown, a UF of 0.1 should be applied to a chosen concentration endpoint for the protected species. The person should also note that amphibians can be exposed to sediment depending on site conditions and species.

9.2.3.3. Toxicity Data for Livestock

As with wildlife receptors, TRVs can be selected to calculate HQs for livestock exposure to COCs and the toxicological information should be based on studies where the routes and duration of exposure are like those expected for the livestock receptor. Chronic-exposure studies with reproductive or growth endpoints are preferable. Because livestock sensitivity to various COCs may differ from sensitivity in smaller mammals used in toxicity tests, livestock-specific toxicity thresholds should be used where possible to evaluate risks. For example, the National Research Council's maximum tolerable levels (MTLs) can be used to evaluate risks in lieu of dose-based TRVs extrapolated from laboratory studies. The MTL of a mineral is defined as "the dietary level that, when fed for a defined period of time, will not impair animal health or performance" (NRC, 2005). MTLs are available for horses, cattle, and sheep (and poultry and swine). If used, MTLs will need to be converted from mg/kg diet to mg/kg body weight per day (i.e., multiply by a food ingestion rate and divide by a body weight).

9.2.3.4. Toxicity Data for Cave-Dwelling Receptors

There are several limitations to assessing potential risks to the cave-dwelling community. Presumably the primary exposure route associated with remediation sites is from COCs in impacted groundwater or surface water. Most of this community consists of invertebrate species, many of which are protected. Although plenty of data on invertebrate toxicity are available in ECOTOX, cave species tend to reside in inaccessible mesocaverns, so determining species-specific impacts is difficult. As discussed in **6.6.3**, contaminated groundwater (or runoff) can enter karst areas with very little filtering; however, most karst invertebrates are not in direct contact with water. Groundwater monitoring-well concentrations at remediation sites near karst areas can be evaluated for compliance with aquatic life criteria or other surface water screening levels. There is no unobtrusive way to determine if the pathway to karst receptors is complete, much less if a dilution factor is applicable. Nevertheless, the evaluation should be conservative if impacts to a protected species are possible.

9.3. Tissue Concentrations and Effects

The tissue residue approach (TRA) is a method that relates toxicological responses of an organism to concentrations of a chemical that are measured or predicted in that organism's tissue. The TRA should not be viewed as a replacement for conventional exposure or dose-based methods for toxicity assessment but can complement conventional approaches to toxicity assessment. The methods by which TRA can be incorporated into ERAs vary widely. Some methods are empirical in scope (i.e., relying on measured tissue residue-response relationships); others are rooted in a more predictive basis (i.e., relying on model-predicted or implied relationships between toxicity and tissue residues) (Sappington et al., 2010). Meador et al. (2014) discusses the use of tissue residue concentrations in development of environmental quality standards. There are advantages to using tissue concentrations over exposure concentrations (water, sediment, soil, or diet) when deriving environmental quality standards. The main advantage is the implicit consideration of bioavailability and toxicokinetics of a COC. Limitations to using an approach based on tissue residue include the evaluation of metabolized COCs (e.g., PAHs in vertebrates), irreversible toxicants (e.g., organophosphates), and reactive toxicants. In the TCEQ ERA process, tissue concentrations, either estimated or measured, are used in the assessment of the sediment-to-fish pathway (**9.3.3**).

A conceptual advantage of the TRA is that tissue residue toxicity metrics are likely to be less variable among species and environmental conditions compared to those responses expressed as a function of an ambient exposure concentration (in water, sediment, soil, or prey). When toxicity is defined in terms of tissue concentrations, the variability is often reduced substantially because the toxicokinetics and bioavailability characteristics for that compound are incorporated in the tissue residue determinations (Meador et al., 2010). The TRA can be an effective tool in a weight-of-evidence evaluation; however, the majority of TRA is applicable to aquatic systems with significantly less information available on terrestrial systems. Two fundamental approaches form the basis of TRA:

- measured residue-effect relationship (empirical)
- calculated residue-effect relationship ("critical body residue" approach)

9.3.1. Measured Residue-Effect Relationships

The empirical approach is based on monitoring data where concentrations of COCs are measured in field-collected or laboratory-exposed organisms or organs. In practice, it is difficult to prove that tissue residues in field-collected organisms are linked to adverse effects. Additionally, this approach is limited by the quantity and quality of residue data (Steevens et al., 2005; Beckvar et al., 2005; Hendriks et al., 2005). In general, tissue-based toxicity evaluations have been carried out under conditions that are less standardized than water-based toxicity tests. Comprehensive compilations of tissue residue data can be found in publicly-available databases such as:

- The U.S. Army Corps of Engineers Environmental Residue Effects Database (ERED). This database contains 2399 studies, 484 species, 419 analytes, 15 effects and 74 endpoints. It contains both laboratory- and field-based data including body burdens and effects for aquatic invertebrates, fish, and amphibians. <https://ered.el.erdc.dren.mil/>

- U.S. EPA Toxicity/Residue database. Adapted from Jarvinen and Ankley (1999), this database contains more than 3,000 effect and no-effect endpoints for survival, growth, and reproductive parameters for invertebrates, fish, and the aquatic life stage of amphibians. Data were abstracted from approximately 500 literature references on approximately 200 chemicals and 190 freshwater and marine test species. Survival endpoints account for about 74 percent of the total data, with growth and reproduction accounting for 19 and 7 percent, respectively. <archive.epa.gov/med/med_archive_03/web/html/tox_residue.html>

- U.S. EPA PCBRes database focuses on dioxin-like PCB congeners, dioxins, and furans. The purpose of this database was to develop PCB critical residue values for fish, mammals, and birds, especially as they relate to aquatic and aquatic-dependent species. This database also expresses critical residue values based upon PCB Aroclors and total PCB-based congeners, because PCBs occur as complex mixtures. Because PCB toxicity occurs via the aryl hydrocarbon receptor, it has also been expressed using the sum of the dioxin-like PCBs after adjustment using toxic equivalency factors (TEFs). <archive.epa.gov/med/med_archive_03/web/html/pcbres.html>

The open literature can also be searched for relevant articles and information. Although there are abundant data on body burdens, there are fewer studies where body burden is related to an ecologically relevant effect.

9.3.2. Calculated Residue-Effect Relationships

The emphasis in this area of work is whether a constant body burden (on a molar basis) can be associated with an effect, such as lethality (i.e., critical body residue, CBR). Most of this research involves nonpolar organics that exert toxicity on aquatic animals through a common physiological pathway, narcosis (e.g., McCarty, 1991; McCarty et al., 1992; Connell and Markwell, 1992; Van Wezel et al., 1995a; McCarty et al., 2013), although some polar organics are also amenable to this approach (Smith et al., 1990; van den Heuvel, 1991; Van Wezel et al., 1995b).

The CBR postulate assumes that a given residue is always associated with a given toxicological response, e.g., lethality. CBRs associated with acutely lethal baseline neutral narcosis in small aquatic organisms typically range between approximately 2 nmol/kg and 8 nmol/kg (wet weight approximately 5 percent lipid). Different CBR ranges could be associated with standard mode-of-toxic-action categories for fish, from narcosis to dioxin-like toxicity for both acute (lethal) and chronic responses (McCarty et al., 2013). The emphasis for this approach has been on prediction rather than diagnosis or monitoring.

Bioaccumulation data can be combined with media-based effects concentrations (e.g., LC_{50}) to estimate toxic residues. CBRs can be estimated by multiplying the media-based toxicity metrics with the bioconcentration or bioaccumulation factor. For organic substances acting through nonspecific modes of action, quantitative structure-activity relationships (QSARs) indicate that both the bioaccumulation factor and LC_{50} are related to the octanol-water partition coefficient (K_{ow}) (Sappington et al., 2010). QSARs are mathematical models that are used to predict measures of toxicity from physical characteristics of the structure of chemicals. The QSAR has the additional advantages of (a) allowing prediction of residues for any nonpolar organic chemical and (b) linking those residues to an adverse effect. In turn, the predicted residue can be used in conjunction with empirical or derived biota-sediment or water accumulation factors to estimate whether field-collected water or sediments are likely to have adverse effects.

It appears that CBR is unlikely to be useful for predicting the effects of metals on aquatic organisms for various reasons, including:

1. Different organisms have different residues of naturally occurring micronutrients (e.g., copper, zinc, chromium) and these concentrations fluctuate over time and with reproductive and nutritional status.
2. Metals, such as cadmium and mercury, bioaccumulate naturally over time, so that older organisms will have higher concentrations.
3. If exposure is low enough, the metal residue will be compensated for without measurable adverse effect (e.g., mercury in swordfish).
4. Exposure to sublethal levels of metals results in the production of metal-binding proteins (i.e., metallothioneins) that can alter toxicity.

Many models that explicitly include residues can be reformulated to apply toxicity data without actual knowledge of toxic residues, although they are still an implicit component of the model. One example is the biotic ligand model for metal bioavailability discussed by Adams et al. (2011). This model is premised on toxic effects being associated with specific levels of metal accumulation on a receptor's gills. This TRA-based model is often implemented without actual measurement or calculation of this critical accumulation; rather, the model is used to define a relationship between toxicity and environmental conditions that can address effects without explicit values for accumulation.

There is a general bias towards using sediment and soil concentrations over CBR or measured residues in biota, primarily due to simplicity of measurement: it is much easier (and generally less expensive) to collect soil, sediment, and water than to collect biological tissue samples. But use of the TRA can provide another useful line of evidence in screening and monitoring programs because of its applicability to:

- Alternate receptors associated with sediment or soil contamination (e.g., fish, herbivorous insects).
- Integration of multiple exposure pathways and evaluation of irregular exposure concentrations and duration of exposures.
- Site-specific field validation of ERA assumptions.

9.3.3. Evaluation of the Sediment-to-Fish Exposure Pathway

As mentioned previously, one direct use of TRA in the TCEQ ERA process is in the assessment of the sediment-to-fish exposure pathway. Limited sediment guidelines are available for evaluating the potential risks of COCs in sediment to fish. The sediment-to-fish pathway is typically evaluated using estimated tissue residue concentrations based on sediment concentrations coupled with BAFs or biota-sediment accumulation factors. BSAFs are a simple tool used to predict the bioaccumulation of hydrophobic organic compounds in fish and other aquatic organisms from measured concentrations in sediment (Wong et al. 2001; Burkhard, 2009).

The TCEQ prefers an evaluation of potential risks associated with whole-fish COC concentrations. That said, the TCEQ acknowledges that it is unlikely that uptake factors will be available that are specific to different tissue types (e.g., whole body, organs, eggs, larvae) and many of the effects endpoints may be associated with specific tissue types as opposed to whole-body concentrations. For Tier 2 SLERAs, modeled fish-tissue concentrations are compared with effects concentrations to evaluate potential risks to the fish as receptors, rather than their predators.

Additionally, COC-specific thresholds for fish-tissue residue have been developed for copper and cadmium (Meador, 2015), mercury and DDT (Beckvar et al., 2005), PCBs for juvenile salmonids (Meador et al., 2002), selenium for freshwater fish (U.S. EPA, 2016), and dioxins (Steevens et al., 2005). Depending on the COC, these effects databases may contain information on a variety of fish species and life stages, reflecting an array of test conditions, exposure types, tissue types, and effects. Certainly, effects endpoints that are directly related to the survival, growth, and reproduction of fish are preferable. The person should evaluate the utility and appropriateness of the varied effects information case by case and consult with the TCEQ ecological risk assessors as necessary.

There are uncertainties and limitations associated with using modeled tissue concentrations coupled with effects data that include the:

- Lack of available data that link toxicity responses to tissue residues.
- Variability and uncertainties associated with the use of BAFs and BSAFs
- Fact that the exposure of fish in the laboratory studies summarized in the databases is often based on water, diet, or injection, and not sediment exposure, and a comparison of modeled whole-body concentrations to laboratory- or field-based effects data where only a specific organ of the fish (e.g., the liver) or fillet was analyzed.

Additionally, the existing residue-effect studies are associated with a varying degree of exposure-effect causality, depending on whether the data were derived from a single-purpose, controlled laboratory experiment, or from incidental observations during a field survey.

Given the uncertainties associated with this approach, the SLERA discussion should also consider fish age, species sensitivity, species home range, and applicability to the affected property's habitat. A more detailed discussion of the TRA as a tool for risk assessment tool appears in several papers (e.g., Barron et al., 2002; Meador et al., 2008; McCarty et al., 2011; McElroy et al., 2011). Despite the uncertainties in using information on effects based on concentrations of residue in tissue, this information remains the primary tool available for evaluating the sediment-to-fish pathway for Tier 2 SLERAs.

9.4. Documentation of TRVs

The text and the tables in an ERA should justify, with references, the TRVs for each COC and receptor pair. TRVs used in the ERA should be thoroughly documented and discussed. Documentation should include: a reference for the TRV study, study species, study endpoints, duration of tests, type of TRV (e.g., chronic NOAEL, LD_{50}, subchronic LOAEL), application of uncertainty factors, and the basis for selection of each TRV. Risk assessments often fail to discuss why a TRV is selected; they simply indicate the TRV and the effect, which is inadequate. If a literature compilation (e.g., the Oak Ridge National Laboratory or a U.S. EPA Eco-SSL document) is used as a source of toxicity values, the original literature source listed in the compilation document should be cited and reviewed for applicability.

In addition to compilation references, the person should consult the open literature to obtain toxicity values for the COCs, or for suitable surrogate compounds. Relevant toxicological endpoints associated with the surrogate selection should be reviewed to evaluate whether the candidate surrogate is appropriate given the selected receptors and food web. If such review is not possible, the person should strive to qualitatively evaluate potential risks in the uncertainty analysis. This evaluation could include a discussion of:

- the relative toxicity associated with similarly structured chemicals or the chemical class in general;

- available information on toxicity (that may not reflect a preferred effect endpoint);
- fate and transport characteristics relative to ecological exposure;
- the expected bioavailability of the COC at the affected property; or
- relative distribution of the COC.

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10.0 Exposure Assessment (Required Element 5, continued)

This section addresses the exposure assessment part of required element 5. Effect levels (e.g., NOAEL TRV, tissue-residue effect concentrations) are also addressed in required element 5 and are discussed in **9.** This text summarizes concepts presented in TRRP-15eco on exposure point concentrations and exposure considerations by media, including hot spots. This text also discusses exposure variables such as uptake factors of COCs into food items, ingestion rates, and exposure modifying factors. Exposure information for reptiles, amphibians, livestock, and cave-dwelling receptors is presented. How bioavailability can be incorporated into a SLERA is discussed, as well as specific information on special COC classes (e.g., metals, dioxins, PAHs, TPH, explosives, radionuclides, and emerging contaminants).

Measures of exposure are defined as the COC concentrations in media or dose (e.g., ingestion of media or tissue), as discussed in **10.4**. They measure how exposure may be occurring, including how a stressor may co-occur with the assessment endpoint. *Measures of effect*, in conjunction with measures of exposure, are used to make inferences about potential changes in the assessment endpoint (U.S. EPA, 1997a).

10.1. Exposure Point Concentrations

The generation of EPCs and considerations regarding the data set they are based on are discussed in TRRP-15eco (by media) and are only briefly discussed here. EPCs for evaluating risk to soil community receptors, benthic invertebrates in sediment, aquatic life, fish, and wildlife are discussed in detail in TRRP-15eco.

The 95 percent UCL of the arithmetic mean is the preferred value for the EPC. The 95 percent UCL is defined as a value that, when calculated repeatedly for randomly drawn subsets of site data, equals or exceeds the true mean 95 percent of the time. In other words, the 95 percent UCL is a conservative estimate of the true mean of the data set. Since the Tier 2 SLERA is a conservative exercise in risk estimation, a conservative EPC is appropriate and consistent with the TCEQ’s regulatory approach. The 95 percent UCL accounts for uncertainty in COC concentrations throughout the exposure area via its conservative nature.

The TCEQ has selected the 95 percent UCL as the preferred EPC, since the goal is to protect receptors at a population scale, and not individually (except for threatened and endangered or rare species). Exposure for fish, benthic macroinvertebrates, soil invertebrates, and plants is generally expressed in terms of media concentration (mg/L or mg/kg), while exposure for terrestrial wildlife such as birds and mammals is generally expressed in terms of an ingested dose (mg/kg body weight-day). In both the initial and refined exposure assessments the 95 percent UCL of the arithmetic mean may be used to compute EPCs from COC concentrations in the exposure medium. An arithmetic or geometric mean should **not** be used in lieu of the 95 percent UCL.

The 95 percent UCL is easily computed using readily available software. The U.S. EPA's ProUCL will compute a variety of statistics from a single data set, including UCLs <www.epa.gov/land-research/proucl-software>. The appropriateness of a given UCL statistic should be carefully considered, given that factors such as sample size, data variance, site features, level of conservatism considered appropriate, and receptor type may all have some bearing on the determination. The ProUCL output should be included with the ERA report.

If most of the computed 95 percent UCL concentrations exceed the highest measured concentration (particularly true for small data sets with high standard deviations), then the person may need to evaluate the appropriateness of the data set for calculating an EPC. The person may also need to consider collecting additional samples from the exposure area to minimize variability and improve the quality of the data set, thus allowing the computation of a reliable 95 percent UCL. Alternatively, the highest measured COC concentration can be used to represent the EPC, although this should be done with caution. A separate hot-spot analysis (see **10.3** and TRRP-15eco) should be performed to identify unusually high COC concentrations relative to other sample locations.

Comparisons with a PCL on a point-to-point concentration basis[18] are relevant when the sample size is too small to use statistical methods to estimate an EPC. When the 95 percent UCL is selected as the EPC for wildlife exposure (as opposed to a point-to-point comparison), the SLERA must also consider if COC hot spots are present in the exposure areas.

10.2. Exposure Considerations by Media

Exposure is characterized in the context of the potential co-occurrence of COCs in media (i.e., soil, sediment, sediment, surface water and groundwater) and ecological receptors that inhabit the media of interest and potentially forage there. The text presented here addresses the primary media of interest: soil (**10.2.1**), sediment (**10.2.2**), surface water (**10.2.4**) and groundwater (**10.2.5**). Additional information can be found in TRRP-15eco.

10.2.1. Soil

Soil is a key medium in terrestrial ecosystems because it directly and indirectly supports plants, soil invertebrates, and wildlife. Being one of the primary exposure media, soil serves as a principal depository and carrier of anthropogenic COCs released into the environment to which wildlife may be exposed via direct contact, ingestion, and food chain transfer. Assessment considerations and EPCs for plants, soil invertebrates, and wildlife in relation to the soil medium are discussed below briefly, and in detail in TRRP-15eco.

[18] In general, a point-to-point comparison is a comparison of the COC concentration at each sample location with a PCL or screening value. Response actions or further evaluation are triggered if the COC concentration at the single sample location (as opposed to an average or 95 percent UCL concentration for multiple sample locations) exceeds the PCL or screening value.

10.2.1.1. Plant and Invertebrate Exposures to Soil

Populations of plant and soil invertebrate communities are important ecosystem components in that they are an energy and nutrient link between soil and higher trophic level receptors. In addition, plants (e.g., grasses, shrubs, and trees) provide protection and cover for wildlife. Soil invertebrates help to break down plant matter and detritus for microbial decomposition. Therefore, plant and soil invertebrate communities play key roles that must be maintained to ensure the viability of the entire ecosystem.

However, potential risks (direct toxicity) to terrestrial plants and soil invertebrates are not usually evaluated in a Tier 2 SLERA because the TRRP rule [30 TAC 350.4 (a)(27)] specifically states that PCLs are not intended to be directly protective of receptors with limited mobility or range (e.g., plants, soil invertebrates, and small rodents). Additionally, plants and invertebrates are not directly evaluated for risks associated with soil COCs because habitat and foraging areas of wildlife that depend on them are frequently large enough to compensate for any localized losses of food and shelter. However, there are some exceptions, which include sites that demonstrate major soil impacts over a substantial area, and sites on non-private land where protected plant or soil invertebrates occur. The person will be required to assess potential impacts to plants and soil invertebrates if soil COC concentrations are at levels where these organisms no longer support the upper trophic level receptors in terms of habitat, shelter, and forage. Also, if protected terrestrial plants and soil invertebrates can be potentially exposed to COCs on public (government-owned) land, these organisms will need to be evaluated for potential ecological risks, and wildlife-management agencies should be contacted. Although there are several listed plant species, the American burying beetle is the only listed terrestrial invertebrate in Texas not associated with caves or karst features (see **10.4.6.3**).

An important consideration for plant and soil invertebrate exposure is the depth of the biologically active zone where plant roots and invertebrates may occur. As already discussed in **2.7**, the TRRP rule defines soil in the interval extending from ground surface to 0.5 feet in depth as *surface soil*, and soil in the interval between 0.5 feet and 5 feet in depth as *subsurface soil* [30 TAC 350.4(a)(86, 88)]. Averaging exposure across the entire depth interval (i.e., surface to 5 feet) is not appropriate as it may underestimate the actual exposure.

10.2.1.2. Wildlife Exposures to Soil

In developing technically-defensible approaches for the evaluation of risk to wildlife from impacted soil, it is necessary to consider the appropriate receptors and ecological scale (i.e., at the organism, population, or community level). Discussion with the TCEQ ERA staff prior to field activities may be needed to prevent unnecessary expenditure of time and resources. Unless the affected property is used by threatened or endangered species, a primary goal of the soil investigation, assessment, and remediation stipulated by the TCEQ is the protection of wildlife populations. As such, methods and measures employed should benefit this ecological scale, unlike protected species, which require individual protection under federal and state law.

Typically, birds and mammals dominate risk assessments for terrestrial biota; however, reptiles and amphibians (e.g., frogs) may also be included as they are commonly found in Texas. A qualitative or quantitative evaluation of amphibians and reptiles, depending on available toxicological and life history information, should also be included if they are expected to occur at the affected property. It is not acceptable, as has been common in previously submitted ERA reports, to state that the PCLs computed for wildlife are protective of reptiles and amphibians without justification—particularly where a protected species may occur at the affected property. When appropriate, livestock should be considered as receptors. Additional discussion on species with exposure and toxicity data gaps appears in **6.6, 9.2.3** and **10.4.6**.

For this guide, the *exposure area for soil* is defined as the *ecological habitat* [that portion of the affected-property soils that does not meet the Tier 1 exclusion criteria at 30 TAC 350.77(b)] within the affected property. An ecological receptor may use only portions of the ecological habitat within the affected property, as dictated by that receptor's specific life-history needs (e.g., foraging habits and nesting requirements). This generic approach assumes the entire ecological habitat within the affected property represents a receptor's exposure area *and* should be used in the determination of the EPC. A subdivision of the ecological habitats within the affected property according to the property-specific characteristics would be the exception rather than the norm. Once the exposure area has been defined, the information and assumptions that support its designation should be included in the risk assessment.

10.2.2. Sediment

Sediment is a key medium in aquatic ecosystems because it directly and indirectly supports aquatic communities (e.g., benthic invertebrates and fish) and wildlife. Sediment serves as a principal depository and carrier of anthropogenic contaminants released into the environment, to which aquatic communities and wildlife may be exposed via direct contact, ingestion, and food chain transfer. Assessment considerations and EPCs for benthic invertebrates, fish, and wildlife in relation to the sediment medium are briefly discussed below and in detail in TRRP-15eco.

10.2.2.1. Exposure of the Benthic Invertebrate Community to Sediment

The overall goal of any benthic assessment and resulting risk management action is to be protective of the benthic macroinvertebrate community as opposed to individual organisms. Species-specific benthic invertebrate evaluations are not typically performed, except under special circumstances, such as when protected species are present. As discussed in **6.1**, an *ecological community* is defined as a group of numerous species with similar geographical and physical requirements, such as temperature, media composition, and light regime. The TCEQ requires protection at the community level, and so a spatial area must be defined to constitute both the community and exposure area under evaluation. Benthic invertebrates spend much or all their life cycle residing within or immediately on top of sediments. Since these organisms are sessile, they are likely to reside within relatively small confines for significant periods.

Like other ecological pathways, there may be reasons to divide the affected property sediments into smaller exposure areas if impacted sediments occur over a large area. Variations in exposure caused by anthropogenic effects (e.g., releases or discharges from locations not part of the affected property assessment) and variations in habitat should govern the selection of these benthic exposure areas. Physical features, spatial distribution, and sediment chemistry should be evaluated in determining if subdivision of sediment exposure areas is warranted. Two example scenarios (freshwater creek and estuarine bay) that describe data groupings based on circumstances appear in Appendix C of TRRP-15eco.

The TCEQ recognizes that the benthic invertebrate community may be diminished for reasons unrelated to COCs. For these water bodies (e.g., intermittent water bodies without perennial pools, or those that are lined with concrete on the bottom or sides), the TCEQ believes it is unnecessary to determine an ecological PCL for sediment that is protective of the benthic invertebrate community. However, that does not eliminate the need for an evaluation of risks to wildlife that may forage in these or nearby water bodies. See **6.1** for more discussion of the benthic PCL exclusion.

10.2.2.2. Exposure of Wildlife to Sediment

A primary goal of sediment investigation, assessment, and remediation is the protection of wildlife populations, and individuals of threatened and endangered species.

ERAs sometimes fail to select a receptor that will conservatively reflect sediment exposure. For example, the TCEQ often recommends that selection of the spotted sandpiper as an avian measurement receptor. Its low body weight and high sediment ingestion rate make the sandpiper a good representative species for determining avian risk from sediment. If sandpipers or other small shorebirds are not present because of lack of habitat, a small wading bird (e.g., green heron, yellow or black-crowned night heron) is preferred over a larger bird like the great blue heron. In some risk assessments, a heron or kingfisher is designated as the only avian predator, under the assumption that fish will dominate the bird’s diet. In general, the TCEQ prefers that the person model receptors that are more likely exposed to COCs due to their feeding strategy and food type. Birds with lower body weights and higher percentages of invertebrates in their diet are generally preferred to maximize the exposure from impacted sediment. Where PCBs or other bioaccumulative or biomagnifying COCs are present in sediment, species that consume a high percentage of fish, such as the American mink, should also be evaluated.

The *exposure area for sediment* is defined as the area within the affected property throughout which a measurement receptor may reasonably be assumed to move, and where direct or indirect contact with sediment is likely. *Indirect exposure* refers to exposure of the wildlife receptor via ingestion of food or prey that contains COCs originating from the affected sediments. A wildlife receptor may choose portions of the affected property sediments, as dictated by that receptor's specific natural history needs (e.g., foraging habits, water depth for wading birds, substrate type, vegetation present, nesting requirements). In these cases, the exposure area for a wildlife receptor will be smaller than the entire area of the affected property sediments. The generic approach presented herein, however, is to assume that all the affected property sediments make up a receptor's exposure area, and that entire area should be used when determining the EPC. A key challenge to resolve up front is a clear delineation of the affected property sediments. The affected property is defined by the assessment level that corresponds to the critical PCL for an exposure pathway. Assessment levels and ecological PCLs protective of wildlife are available in the PCL Database and discussed in **2.1**.

10.2.2.3. Exposure of Fish to Sediment

Potential risks to fish as receptors can be an important element of an ERA for impacted sediments. Fish are important components of aquatic food webs because they process energy from aquatic plants (i.e., primary producers), zooplankton, and benthic macroinvertebrates (i.e., primary consumers) or detritivores. Fish are also important prey for piscivorous wildlife and are certainly the key to the state's recreational and commercial fisheries. Protected fish species can occur throughout the state and should be conservatively evaluated at the individual level where they may be present at an affected property.

Many fish species have relatively low direct contact with sediment, and concern over this pathway is generally minor compared with that for benthic invertebrates, which are generally more sensitive indicators of sediment contamination. However, benthic and pelagic fish species can be exposed to COCs in sediment to varying degrees through several exposure routes, including direct contact with contaminated sediments (for benthic species), or contact with contaminated pore water (for those species that burrow into the sediments or spawn in or on the bottom substrates), and diet. Direct exposure can occur from foraging, nest or redd building or resting, or spawning, and through incidental ingestion while feeding. Consumption of contaminated prey is an important indirect exposure route for species that consume infaunal invertebrate or that forage-fish. Diet is likely the most important route of exposure for carnivorous fish for bioaccumulative substances in sediment such as PCBs, dioxins and furans, selenium, mercury, and organochlorine pesticides.

Risk assessments presented to the TCEQ often assume that exposure to COCs in the water column is the only route of exposure to fish or is the predominant route of exposure to COCs on the affected property. Undoubtedly, water can be the prevailing exposure route for many fish and can be the risk driver in some cases. Nonetheless, epibenthic fish species, upper trophic level fish, and sensitive life stages (e.g., eggs and larvae) of many fish may be more highly exposed to sediment COCs than water-column COCs.

In the ERA, sediment concentration data can be coupled with BAFs or BSAFs to estimate a tissue-residue concentration. Additional information on tissue concentrations and effects as it relates to the sediment-to-fish pathway is presented in **9.3.3**.

The *exposure area for fish* is defined as the area within the affected property sediments over which any life stage of the fish community may reasonably be assumed to move throughout, and where direct or indirect contact (from ingestion of food or prey) with sediment is likely. When impacted sediments occur over a larger area of the affected property sediments, there may be reasons to divide them into smaller exposure areas for the fish community. Once the exposure area has been defined, the information and assumptions that support the identification of the exposure area should be provided in the risk assessment discussion.

An initial screen for evaluating the sediment-to-fish pathway is the use of the midpoint PCL value between the benchmark and second effects level for benthic invertebrates. As in the screening process for the sediment-to-benthic invertebrate pathway, bioaccumulative COCs should be retained for further evaluation, whereas non-bioaccumulative COCs detected below the midpoint PCL for benthos can be removed from further consideration for the sediment-to-fish pathway. Like the process for benthic invertebrates, all COCs without a midpoint PCL specified in the sediment benchmark table should be retained for further evaluation. Because of the uncertainty associated with using screening values developed for benthos to evaluate the sediment-to-fish pathway, the midpoint PCL should not be used as a PCL for the sediment-to-fish pathway, but only as a tool to determine which COCs warrant additional evaluation. For PAHs in sediment, only the total PAH midpoint PCL should be used for this screening step, rather than the individual PAH midpoint PCLs.

For non-bioaccumulative COCs, it is assumed that the sensitivities of sediment-dwelling organisms to COCs are like those of water-column species (i.e., fish; Di Toro et al., 1991). To support this approach, note that the derivation of some empirically-based sediment quality guidelines protective of benthos included data on the effects on fish exposed to contaminated sediments (e.g., Long and Morgan, 1990; Long et al., 1995). The TCEQ believes that the sediment benchmarks for non-bioaccumulative COCs are generally protective of the sediment-to-fish pathway (even sensitive life stages such as eggs and larvae) for both marine and freshwater fish.

Therefore, for nonbioaccumulative COCs, the TCEQ will generally accept this assumption in lieu of a specific sediment-to-fish evaluation, unless the highest measured COC concentration exceeds the applicable midpoint sediment PCL, or a more specific evaluation is needed where a protected fish species is expected or known to be present.

10.2.2.4. Evaluating Indirect Sediment Exposures

The aquatic portion of an ERA should consider exposure to bioaccumulative COCs in sediments using uptake factors for that portion of the measurement receptor's diet that is sediment based. Often, sediment uptake through the food chain is not represented in the ERA calculations. Instead, prey tissue concentrations for aquatic invertebrates and fish are estimated using BCFs based on surface water concentrations. In these cases, sediment food-chain transfer to invertebrates and fish is not modeled for any wildlife receptor. Certainly, it is appropriate to evaluate exposure from COC transport through the water column to aquatic biota; however, the TCEQ believes that the evaluation of COC uptake from the water column alone will greatly underestimate or overlook the potential for exposure to sediment COCs in the food chain.

For bioaccumulative COCs, it may be appropriate to assume that a generic fish prey species receives an equal proportion of uptake from the water column and sediments by using a water-based uptake factor for 50 percent of the diet, and a sediment-based uptake factor for the remainder of the uptake, to predict fish-tissue concentrations that reflect exposure from both water and sediments. Similarly, the person can apply the same approach to receptors that may ingest water-column or benthic invertebrates depending on the feeding habits of the measurement receptor. Fish mobility and the extent of sediment contamination should be considered, as well.

10.2.3. Biased Data for Soil and Sediment

Soil and sediment assessments evaluated for sites under the TRRP typically employ judgmental (biased) sampling as opposed to a random geospatial sampling regime. The TRRP rule allows judgmental samples, provided the resulting estimated representative concentration is demonstrably not biased low [30 TAC 350.51(l)(1)]. Typically, environmental sampling is biased high, given the initial objective to identify known or potential source areas. When sampling sediments, depositional areas dominated by fine-grained sediments should be targeted. Professional judgment is needed to ensure collection of data in a manner that most appropriately represents the true statistical population of soil concentrations relative to potential ecological exposure conditions. Any identified biases (high or low) should be discussed in the uncertainty analysis within the ERA.

Soil or sediment sample locations outside the boundaries of the affected property or the habitat or foraging area for a particular receptor, guild, or community should generally not be included in the calculation of the soil or sediment EPC.[19] The primary point is that sediment data collected to define the nature and extent of contamination are not necessarily equivalent with the exposure area for a receptor or the affected property by definition.[20] It may be inappropriate to include sediment-sample locations that do not appear to be affected by the release in question, such as locations at the fringe of the sampled area. Additionally, if sediment samples are being collected to assess

[19] See the discussion in TRRP-15eco regarding the evaluation of potential hot spots for sediment-associated wildlife that may forage within an area smaller than that used to determine the sediment exposure point concentration.

[20] Since the affected property represents the entire area that contains releases of COCs at concentrations greater than or equal to the assessment level, some sediment-sample locations (such as some included in the nature and extent evaluation) may not meet the definition of *affected property*. Sediment-concentration data from these locations should not be used in the ERA. Additionally, *affected property* should not be confused with the physical or legal boundary of a facility.

the groundwater-to-sediment pathway, areas of groundwater discharge should be targeted.

When considering soil, the key concept is that the affected property boundaries are determined by the assessment level for a given COC, which is the lower of the human health and ecological PCLs (see discussion of assessment levels in **2.1**). In keeping with the TRRP rule at 30 TAC 350.51(l)(1), an EPC based on soil samples collected outside an ecological exposure area may be acceptable if these data are at least representative of, or higher in concentration than, the soil concentrations that an ecological receptor may experience within a given exposure area. This must be demonstrated with data from the affected property, or with qualitative use of historical knowledge of affected property operations or historical data (or both). Avoid using high-biased data to generate an ecological EPC that results in apparent risk, because such risks cannot be explained away in the uncertainty analysis as simply being too conservative without further justification or data collection.

10.2.4. Surface Water

Before beginning a discussion of the ecological exposure pathways associated with surface water, the person should have a clear understanding of what is meant by "surface water." The TRRP rule [30 TAC 350.4(a)(89)] defers to the TSWQS for the definition of *surface water* in Texas [30 TAC 307.3(a)(70)]. Nearly any body of water or ditch could be considered waters in the state absent those that are part of a currently permitted treatment system or otherwise authorized discharge. The surface water environments in Texas are varied, complex, and dynamic. Surface water as an exposure medium can be found in numerous settings:

- flowing rivers, creeks, streams, and ditches
- ponds and lakes
- wetlands or low-lying areas that are permanently or intermittently flooded
- tidal bays, estuaries, rivers, bayous, and channels
- ephemeral waters (arroyos, wetlands, ditches, pools, playa lakes)

Surface water exposure is characterized by the potential co-occurrence of COCs and ecological receptors that exist or forage in the water column. Assessment considerations and EPCs for water-column receptors and wildlife in relation to the surface water medium are discussed below briefly, and in detail in TRRP-15eco. The Texas Surface Water Quality Standards [30 TAC 307] and supporting documentation should also be referenced when evaluating ecological risks from surface water exposure.

COCs can be present in surface water in the freely dissolved form or bound to particles and suspended in the water column. Receptors include fish and invertebrate communities and aquatic-dependent or partially aquatic-dependent vertebrate wildlife. Terrestrial wildlife may be exposed to COCs in surface water if impacted surface waters are used for drinking, although that is not typically a major exposure pathway.

The U.S. EPA (1994) defines *aquatic community* as an association of interacting populations of aquatic organisms in a given water body or habitat. Aquatic life receptors include water-column organisms (e.g., macrophytes, plankton, crustaceans, aquatic insects, and early life stages of amphibians), fish, and adult amphibians. For aquatic organisms, potential routes of exposure to surface water COCs include absorption (across respiratory organs, integument or skin, and exoskeleton), adsorption, and ingestion (food and water).

For the most part, the evaluation of ecological risks to aquatic life is based on measurements of concentrations in surface water. Surface water concentrations are compared with surface water quality criteria protective of aquatic life or equivalent threshold concentrations for COCs that have no state or federal water quality criteria. Important considerations in the assessment of risks associated with surface water COCs include the appropriate averaging time for the COC concentrations, temporal and spatial variability and distribution, and the form of the chemical to be measured (e.g., dissolved, total, or ionic).

Like other ecological exposure pathways, there may be reasons to divide the affected property into smaller exposure areas for aquatic receptors, particularly where surface waters may be impacted over a large area. Variations in exposure caused by anthropogenic effects (e.g., releases from off-site sources) and variations in the habitat (hydrology, water chemistry, depth, cover) within the surface water body should largely govern the selection of differing exposure areas for aquatic life.

The TSWQS establish six subcategories of aquatic-life use:

1. minimal
2. limited
3. intermediate
4. high
5. exceptional aquatic life
6. oyster waters

The TSWQS [30 TAC 307.6(b)] specify that water in Texas must not be acutely toxic to aquatic life and must not be chronically toxic to aquatic life if it has designated or existing aquatic life uses of “limited,” “intermediate,” “high,” or “exceptional.”

Each classified segment in the TSWQS is assigned an aquatic life use based on physical, chemical, and biological characteristics of the water body. Unclassified perennial streams, rivers, lakes, bays, estuaries, and other appropriate perennial waters that are not specifically listed in Appendix A or D of the TSWQS are presumed to have a high aquatic life use [30 TAC 307.4(h)(3)]. Additionally, unless specifically listed in Appendix A or D of the TSWQS, unclassified intermittent streams with perennial pools are presumed to have a limited aquatic life use, and intermittent streams are considered to have a minimal aquatic life use except where there is a seasonal aquatic life use [30 TAC

307.4(h)(4)]. Thus, all water bodies must meet acute criteria protective of aquatic life, and all perennial water bodies (including intermittent and ephemeral streams with perennial pools) must meet chronic criteria protective of aquatic life.

In the Houston area, many TRRP sites are located adjacent or close to the Houston Ship Channel. Although Houston Ship Channel Tidal (Segment 1006) and Houston Ship Channel/Buffalo Bayou Tidal (Segment 1007) do not have a designated aquatic life use, the TSWQS (Appendix A) specify that chronic toxic numerical criteria apply. Therefore, it is appropriate to evaluate risks to aquatic life receptors at TRRP sites adjacent to the Houston Ship Channel and its tidal tributaries where site COCs have been released to surface water.

Although less common as COCs in surface water for TRRP sites, specific nutrients (e.g., nitrate nitrogen, total phosphate), salinity, chloride, sulfate, TDS, and pH must be evaluated at an affected property if they are COCs (or degradation products of parent COCs) for the affected property. TCEQ (2007a; TRRP-24) and, to a lesser extent, TCEQ (2010b; TRRP-13) discuss selection of the surface water PCLs and risk-based exposure levels for these types of conventional pollutants.

Surface water is a principal medium to be evaluated in aquatic ecosystems because it directly and indirectly supports wildlife receptors. A primary goal of the surface water investigation and assessment is the protection of wildlife populations and threatened and endangered species. As such, methods and measures employed should reflect the appropriate ecological scale, except for threatened and endangered species, which require individual protection by federal and state law.

Aquatic-based wildlife species can be exposed to COCs in surface water directly (e.g., skin, gills), and from ingestion of water and food. Piscivorous receptors such as the mink, river otter, bald eagle, and kingfisher can be particularly susceptible to risk from bioaccumulative COCs in surface water (e.g., PCBs, dioxins and furans, DDT and its metabolites, selenium, and mercury) as concentrations may biomagnify to levels in fish far greater than ambient concentrations in surface water. Although this discussion focuses on aquatic wildlife receptors, an additional exposure pathway is terrestrial wildlife receptors that may ingest waterborne COCs if impacted surface waters are used as drinking water. While such ingestion is often a complete exposure pathway, it is not likely to be a risk driver (even for bioaccumulative COCs) unless wildlife are likely to regularly contact and consume impacted surface water (e.g., at active impacted groundwater seeps).

Birds and mammals are prominent in risk assessments as aquatic-based wildlife receptors. A qualitative or quantitative evaluation of amphibians and reptiles, depending on available toxicological and life-history information, should also be included in the SLERA if they are expected at the site. A more rigorous evaluation is required where a protected reptile or amphibian species may occur. The TCEQ recognizes that health-effects data for these classes, unlike for birds and mammals, are sparse for many COCs.

The *exposure area* is defined as the surface water area within the affected property over which a measurement receptor may reasonably be assumed to move throughout, and where direct or indirect contact with surface water is likely at all locations. *Indirect exposure* refers to exposure of the wildlife receptor via ingestion of food or prey that contains COCs originating from the affected surface water. Although a wildlife receptor may only use portions of the affected property's surface water (as determined by that receptor's specific habitat and foraging needs), it is usually unnecessary to distinguish different exposure areas because of the dynamic nature of the surface water medium and the mobility of most wildlife receptors and their prey. In rare cases, the exposure area for a wildlife receptor may be modeled as a subset of the affected property's surface water body or bodies.

10.2.5. Groundwater

Because dissolved COC groundwater plumes are dynamic, groundwater concentrations at any given monitoring well are expected to differ from one monitoring event to the next. Current groundwater data should be used for calculation of the EPC and for evaluating compliance with any groundwater PCLs protective of ecological exposure pathways.

The TRRP requires investigation for the presence of groundwater beneath the site of a release. For the ERA, the evaluation of exposure pathways for ecological receptors at the point of groundwater discharge to a surface water body (i.e., at the groundwater–to–surface water–to–sediment interface), is the relevant pathway. The groundwater–to–surface water pathway is also discussed in TRRP-24 and TRRP-15eco, although key concepts are summarized below.

10.2.5.1. Groundwater–to–Surface Water Dilution Factor and Equation

The TRRP rule [30 TAC 350.75(i)(4)] requires the person to establish PCLs for COCs in groundwater that discharge to surface water. The rule also states that this surface water PCL (SWSW) is the lesser of the human health and ecological surface water RBELs. The person may establish a surface water dilution factor (DF) when the concentration of a COC in groundwater at the zone of discharge to surface water exceeds the SWSW for any COC at the time of the affected property assessment (with some limitations). The TRRP rule contains the equation below to establish the groundwater–to–surface water PCL (${}^{SW}GW$):

$${}^{SW}GW = {}^{SW}SW \div DF$$

This equation should be used to adjust the aquatic-life RBEL for dilution as the groundwater mixes with the surface water. TRRP-24 details an approach for evaluating historical groundwater data to determine if a dilution factor can be applied to the surface water PCL.

10.2.5.2. Determining Groundwater Concentrations at the Surface Water Interface

When a groundwater assessment indicates that the groundwater–to–surface water-sediment pathways are complete, the groundwater plume must be evaluated at the groundwater-surface water interface. Since the groundwater–to–surface water POE is defined to be in the groundwater at the interface, an appropriate groundwater-monitoring network should be established along the interface as close as feasible to the surface water body. A conservative representative groundwater concentration may be established based on the highest measured COC concentration from the array of wells at the surface water interface. In lieu of using the highest groundwater concentration measured at the interface for the EPC, a site-specific discharge-weighted groundwater concentration can be determined. See Appendix D of TRRP-15eco for a detailed presentation of the recommended approach.

10.2.5.3. Groundwater-to-Sediment Pathway

The TRRP rule is clear that the monitoring point for the groundwater–to–surface water pathway is within the groundwater rather than the surface water [see 30 TAC 350.51(f)]. The approach is different for the evaluation of potential groundwater impacts to sediment. Here, bulk sediment samples should normally be collected in the area of groundwater discharge. Where groundwater releases are the only site-related cause for potential impacts to sediment, sediment samples should be analyzed for groundwater COCs. If ecological risks are indicated, ${}^{Sed}GW_{eco}$ PCLs should be determined. In some situations, the evaluation of pore water concentrations is preferable to (or should be used in combination with) analyses of bulk sediment for the groundwater-to-sediment exposure pathways. Where sediment pore-water data are used to conservatively reflect groundwater impacts to sediment, the person should provide a rationale for the pore-water sampling locations, and for quantifying sediment pore-water data in the context of the ERA, which could include statistical averaging or a point-to-point comparison, depending on the exposure pathway. TRRP-15eco and TRRP-24 contain additional information on the analysis of the groundwater-to-sediment pathway.

10.3. Exposure Considerations for Hot Spots

The TRRP rule [30 TAC 350.51(l)(5)] states that the presence of hot spots with respect to ecological risk shall be determined on a site-specific basis. The adoption preamble to the 1999 TRRP rule (24 *Texas Register* 7577, September 17, 1999) offers some insight as to the intent of the hot spot provision in the rule: "… to minimize the potential for critical areas of COCs to be 'averaged out' by being combined with sampling data from relatively unimpacted areas."

The evaluation of hot spots was introduced into the Texas ERA process in 2013 with the publication of TRRP-15eco and is now considered standard practice for all ERAs. The process of identifying and evaluating hot spots is described in detail by media and receptor in TRRP-15eco, but a general description is presented below.

A hot spot is not just an area of substantially elevated concentrations relative to surrounding areas; it is also a function of the relative risk to the measurement receptor in question. The standard ERA evaluates the COC concentrations over an area larger than a hot spot to determine potential risk assuming equal distribution of wildlife exposure to COCs across an exposure area. However, this assumption may not be protective if a smaller area with elevated COC contributions either (1) poses a risk of acute toxicity, or (2) is in a location that contributes disproportionately to the receptor's chronic exposure (i.e., a high-quality feeding area). In either case, the area of elevated COC concentration will be considered a hot spot due to the disproportionately elevated risk. The purpose of the hot spot evaluation is to determine the presence or absence of either of these two conditions. An additional concern for managing hot spots is to protect against the excess risk of reducing the viability of local populations.

The identification and early treatment of hot spots (e.g., removal) can be useful in addressing risk management objectives for an affected property. For instance, this may focus the evaluation on those locations that are most important and effective to remediate. The person may choose to address a hot spot up front to minimize future investigation or liability. Hot spots may be removed at any affected property. However, removal is best suited to small sites and small hot spots where the cost is low relative to the cost for conducting a risk assessment. Risk management of hot spots by medium is further discussed in **14.3.**

The person preparing a risk assessment should determine if a hot spot evaluation is needed. The hot spot evaluation should be presented in the problem formulation or the uncertainty analysis in the ERA, depending on site-specific circumstances. A short rationale is required if the person determines a hot spot evaluation is not warranted. The TCEQ will evaluate the adequacy of the hot spot analysis (or the justification for not performing an analysis) and comment as necessary if it needs more detail or clarification. The TCEQ will also evaluate the conclusions of the analysis and the associated risk management recommendation, as appropriate.

Potential risks to protected species are necessarily considered at the level of an individual, rather than the population, because a compromised population is less capable of tolerating the loss of individuals than a healthy population, and it is a violation of the Endangered Species Act to harm or take a protected species or damage critical habitat. Accordingly, the conservatism of the TCEQ review will be greater and the effort put forth in the hot spot analysis may need to be greater where the measurement receptor in question is a protected species or its surrogate. Additionally, any uncertainty associated with the adequacy of the sample density, keeping in mind the ecology of the receptor, may necessitate more data or a field survey of the habitat. The TCEQ may require additional safety factors and conservative assumptions in the ERA calculations where a protected species is potentially affected by a COC hot spot.

10.4. Input Data and Exposure Calculations

As presented in Figure 4.1, input data and exposure calculations for Tier 2 SLERAs are developed in accordance with required element 5 [30 TAC 350.77(c)], which ends with a calculation of the preliminary dose based on conservative assumptions to minimize the potential for overlooking ecological risks. Exposure assumptions should be reasonably conservative in total and may incorporate site-specific data in later stages of the ERA process.

After determining the preliminary dose in required element 5, the NOAEL TRV will be compared to the preliminary dose in required element 6 to determine the conservative HQ. required element 7 allows for a refined exposure assessment (modification of the preliminary dose) and use of both the NOAEL TRV and the LOAEL TRV to determine the HQs. The primary modification to the dose between required elements 6 and 7 is the incorporation of the receptor's home range and seasonality, as appropriate (see **10.4.5**).

This general equation and its modification can be used to estimate oral exposure and adjusted dose, respectively, for wildlife receptors:

$$Dose_{oral} = \frac{[(IR_{food} \times C_{food}) + (IR_{water} \times C_{water}) + (IR_{soil} \times C_{soil}) + (IR_{sed} \times C_{sed})]}{BW}$$

$$Dose_{adjusted} = Dose_{oral} \times EMF$$

where:

$Dose_{oral}$ = estimated dose from ingestion (mg COC/kg body weight/day)

IR_{food} = ingestion rate of food (prey) (kg/day) (see **10.4.3.1**)

C_{food} = COC concentration in food (mg/kg) (see **10.4.1** and **10.4.2**)

IR_{water} = ingestion rate of water (L/day) (see **10.4.3.1**)

C_{water} = COC concentration in water (mg/L)

IR_{soil} = ingestion rate of soil (kg/day) (see **10.4.3.2**)

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

IR_{sed}= ingestion rate of sediment (kg/day) (see **10.4.3.2**)

C_{sed} = COC concentration in sediment (mg/kg)

$Dose_{adjusted}$ = oral dose adjusted by exposure modifying factor (mg COC/kg body weight/day)

EMF= exposure modifying factor (unitless) (applied after the dose is calculated and can be the product of multiple modification factors, see **10.4.5**)

BW= body weight of receptor (kg) (see **10.4.3.3**)

Literature sources used for intake and exposure variables should be clearly indicated and justified in the Tier 2 SLERA. Where literature information is modified for use in a Tier 2 SLERA, the modifications should be indicated in the discussion. For example, a literature-derived food ingestion rate may be adjusted to reflect wet weight or dry weight, as appropriate. Where a variety of choices are available for a receptor (e.g., body weight, dietary composition), the person should indicate how any one reference was selected from those available, particularly where he or she has selected a less-conservative exposure factor. Intake and exposure variables for numerous species are available in the PCL Database and instructions for accessing specific inputs appear throughout the discussion in the rest of this chapter.

10.4.1. COC Uptake in Food Items

A variety of approaches have been developed to estimate COC loads in plant or animal food items consumed by wildlife, including empirical uptake factors, predictive models, and direct measurement of residues in tissue.

These terms describe the transfer of COCs from an external environment to one or more ecological receptors (U.S. EPA, 1997a):

- *Bioaccumulation* is the uptake of COCs by an organism either directly from exposure to a medium or by consumption of food containing the COC. A bioaccumulation factor is the ratio of the concentration of a COC in an organism to the concentration in the ambient environment at steady state, where the organism can take in the COC through both ingestion of food and direct contact. A biota-sediment accumulation factor is a specific type and form of bioaccumulation factor: the ratio of lipid-normalized tissue chemical residue to the chemical concentration in organic carbon–normalized sediment (Rand, 1995).

- *Bioconcentration* is a net accumulation of a COC directly from an exposure medium (usually water) in an organism. It does not include food web transfer. A bioconcentration factor is the ratio of the concentration of a COC in an organism to the concentration in the ambient environment at steady state.

- *Biomagnification* is the result of bioaccumulation and biotransfer by which tissue concentrations of COCs in organisms at one trophic level exceed tissue concentrations in organisms at the next lower trophic level in a food chain.

As presented below and previously discussed in **9.3**, available methods for predicting tissue residues in a Tier 2 SLERA include the use of laboratory- and field-derived BCF and BAFs obtained from the scientific literature, as well as regression models[21] based on literature-derived data, kinetic models, and thermodynamic (equilibrium partitioning) models. Laboratory- or field-derived BCFs and BAFs are available for many compounds and receptors. These values should be closely scrutinized to assess differences between laboratory and field conditions and differences in the biology and ecology of affected properties. The person should consult the original literature whenever possible to determine applicability to conditions at the affected property, giving attention to ensuring that values are based on similar underlying factors (e.g., organic-carbon content, expression as wet versus dry weight, and normalization to a specific lipid content).

For developing BAFs, several methods are available. To determine total dose for persistent and bioaccumulative COCs, exposure from environmental media, food intake, and magnification between successive trophic levels should be

[21] For example, regression models were used in development of the EPA's Eco-SSLs. See Attachment 4-1 of U.S. EPA (2007a).

considered. A BAF simplifies using environmental concentrations (e.g., water or sediment analytical values) to determine a total dose for a wildlife receptor. In general, the estimated dietary concentration is the product of the environmental concentrations and the medium-appropriate BAF. The dietary exposure is then the product of this concentration and the ingestion rate for the receptor.

BAFs are highly site specific, so any BAF based on generalized characteristics will have limited precision. To improve precision, these preferences are usually acceptable:

- Data relevant to Texas and Gulf of Mexico ecosystems and species rather than other generic site data.
- Field data rather than laboratory data.
- Biotic data (e.g., from fish bioconcentration studies) rather than physical- or chemical-based models (e.g., bioconcentration estimated from octanol/water partitioning studies).

The approach selected should reflect the availability of field data, the relevance of tested species, the food chain or trophic structure of the community, and the level of modeling deemed acceptable for a Tier 2 SLERA.

Uptake-factor terminology should distinguish when food exposure is or is not considered in the value (e.g., media-only exposure as BCF and media and food exposure as BAF). For COCs with log K_{ow} values **greater than 5**, biomagnification up to the trophic level of the prey item must be considered in determining total dose to wildlife, unless COC-specific justification is provided (e.g., measurement receptor or feeding guild is capable of metabolizing COC). A default uptake factor of 1 is often assumed when uptake factors are not readily available. This would be inadequate for COCs with log K_{ow} values greater than five, as the potential for biomagnification is present. References used to obtain uptake factors should be documented. Where a reference provides individual values (e.g., mean, median, 90th percentile), the selected value should be clearly identified. Where a formula in a reference is used to derive an uptake factor, that should be clearly indicated. Subsection **10.4.2** describes the sources used to develop uptake factors for the PCL Database. The use of these methods is not required in a Tier 2 SLERA; however, use of alternate values requires sufficient explanation.

10.4.2. COC Uptake in Food Items from the PCL Database

As described previously, COC uptake into food items is a critical part of the exposure analysis. Detailed information can be found in the PCL Database and in the supporting documentation.

> *In the PCL Database on the "PCL Calculator" page, choose the "Chemicals" tab toward the top of the page (not the drop-down menu under Step 2). Find the COC under the "Chemical Name" column at the left side of the screen. Click on the CAS number of the COC of interest and wait for the PDF chemical profile to appear. The PDF contains the bioaccumulation factors for soil to plant, soil to earthworm, soil to arthropod, soil to mammal, sediment to fish, and sediment to benthic invertebrate.*

10.4.3. Ingestion Rates and Body Weights

Ingestion rates for food, water, soil, and sediment, as well as measurement receptor body weights are integral components of the dose equation in **10.4**. Once ingestion rates and a representative (average) body weight have been established for an ecological receptor, they should remain constant throughout the initial and refined assessments. Ingestion rates and COC concentrations in food, including uptake factors, must be expressed on a consistent basis (wet-weight or dry-weight).

10.4.3.1. Food and Water Ingestion Rates

The *Handbook* provides food and water ingestion rates for selected birds, mammals, reptiles, and amphibians (U.S. EPA, 1993a), along with allometric equations for estimating these rates in dry weight, as a function of body weight. The open literature may also be consulted for species-specific food and water ingestion rates.

Because there are several methods for estimation of food ingestion rates, both in wet and dry weight, the method used in the SLERA should be clearly presented. For example, allometric equations as presented in Section 3.1 of the *Handbook* (originally from Nagy 1987), or those from Nagy, 2001, can be used to derive a food ingestion rate for the receptor, even if the receptor is one of the species presented in the *Handbook* (e.g., robin) and a food ingestion rate from other sources is already listed. For example, from Nagy, 1987, the person may use the general equations for all birds (Equation 3-3) and all mammals (Equation 3-7); however, additional equations are available for different types of birds (passerines, nonpasserines, and seabirds), mammals (rodents and herbivores), and iguanid lizards (herbivores and insectivores). Nagy (2001) also presents food ingestion equations by groups for mammals (e.g., all mammals, herbivores, insectivores, carnivores); birds (e.g., all birds, herbivores, insectivores, carnivores); and reptiles (e.g., all reptiles, herbivores, insectivores, carnivores).

Nagy (2001) also includes some unique groupings for each of the three classes such as desert rodents, marine birds, and Phrynosomatidae, which includes horned lizards.

The food ingestion rate (IR_{food}) and COC concentration in food (C_{food}) must be expressed on a consistent basis (wet weight or dry weight). The *Handbook* provides the water content (in percent) of a variety of plant and animal foods. Equations for converting food ingestion rates and food concentrations between dry and wet weight are:

$$IR_{food}\text{ (wet weight)} = IR_{food}\text{ (dry weight)} \div (1 - \text{percent water})$$

$$C_{food}\text{ (wet weight)} = C_{food}\text{ (dry weight)} \times (1 - \text{percent water})$$

The Nagy (1987 and 2001) equations are unit specific (grams or kilograms) for body weight and that conversion from grams to kilograms and normalization to body weight should only occur after the equation has been solved, as shown in Table 10.1. Also, the resultant allometric food ingestion rate is in dry weight. The food ingestion rate can be converted to wet weight as needed to be consistent with the individual food components (i.e., prey and vegetation) to calculate the dose from food. Nagy 2001 presents food intake in both dry matter intake and wet matter intake. The person should document which equations are used, the units and if the resulting food ingestion rate is in wet or dry weight.

As presented in the *Combustion* guide, the moisture content of food is assumed to be 88 percent for plant matter (herbivores), 68 percent for animal matter (carnivores), and 78 percent for an equal portion of plant and animal matter (omnivores). This means that the dry weight food ingestion rate would need to be divided by 0.12, 0.32, or 0.22, respectively, to obtain the corresponding wet weight value. Other moisture content values (as a percent) are presented in Table 4.1 of the *Handbook*. These include: 68 percent for small fish (piscivores), 79 percent for aquatic invertebrates (aquatic invertivores), and 71 percent for terrestrial invertebrates (terrestrial invertivores).

Water ingestion rates for specific species and allometric equations for estimating these rates for groups of receptors are available from the sources discussed above. However, where surface waters or groundwater seeps associated with the affected property are not impacted or when this part of the dose is minute, it is often omitted in the exposure calculation. Of course, where ingestion of contaminated water can represent a significant dose, it will need to be included as part of the overall exposure evaluation. For instance, if a stock pond that is the only source of wildlife drinking water for miles around becomes contaminated by site activities, this exposure would need to be incorporated into the evaluation.

In the PCL Database on the "PCL Calculator" page, click on the "Species" tab toward the top of the page (not the "Species" button under Step 1). Food and water ingestion rates appear near the middle of the screen. Note that all ingestion rates are normalized for body weight. Derivation of ingestion rates is provided on the individual species profile PDF, which can be accessed by clicking on the underlined species name.

10.4.3.2. Soil and Sediment Ingestion Rates

Wildlife may ingest soil or sediment intentionally to obtain nutrients or incidentally during feeding, grazing, preening, cleaning, or burrowing. Information on soil or sediment ingestion rates is limited, and unlike food and water consumption, generalized models do not exist for estimating soil ingestion by wildlife receptors. Several references—such as the EPA's *Handbook* (1993a), Suter et al. (2000), Beyer et al. (1994, 2008) and Beyer and Fries (2003)—estimate soil and sediment ingestion for a variety of wildlife species; however, current literature should be reviewed.

Both the soil-sediment ingestion rate ($IR_{soil/sediment}$) and COC concentration in soil-sediment ($C_{soil/sediment}$) must be expressed on a consistent basis (typically dry weight). Note that the dose received from either food ingestion or soil-sediment ingestion is usually expressed in mg COC/kg body weight/day. Thus, $IR_{soil/sediment}$ and $C_{soil/sediment}$ are expressed on a dry-weight basis, while IR_{food} and C_{food} are expressed either on a wet- or dry-weight basis.

When information on a measurement receptor's soil or sediment ingestion rate is stated as a percentage of dry matter in the gut (as is common), the converted fractional value should be multiplied by the food-ingestion rate to obtain the soil-sediment ingestion rate. Percentages of food items in the diet of the measurement receptor should sum to 100 and should not be normalized to include the soil or sediment ingestion portion.

For consistency, the soil or sediment ingestion (percent) for a receptor should be obtained, extrapolated, or estimated from the Beyer et al. (1994, 2008), Beyer and Fries (2003), or other comparable sources. The premise is that, when comparing diets and feeding strategies of ecological receptors, it is much easier to comprehend the relative percentages of soil or sediment in the diet than the relative rates of soil or sediment ingestion.

For receptors for which no source can be found for soil or sediment ingestion, reasonable surrogates can be used (e.g., the red fox for the coyote). When no source can be found and no surrogate seems appropriate, a reasonable estimation can be proposed. For example, raptors will have a low percentage of soil or sediment ingestion while receptors with diets of soil or benthic invertebrates will have higher ingestion. For example, Beyer et al. (1994) did not evaluate the robin but did evaluate the woodcock. If the woodcock is assumed to eat 100 percent soil invertebrates resulting in 10.4 percent soil ingestion, then a robin eating 50 percent invertebrates could be assumed to ingest 5.2 percent soil. This percentage of soil or sediment value, in dry weight, should then be multiplied by the allometric food ingestion rate (dry weight) to obtain the soil or sediment ingestion rate (dry weight), as shown in the raccoon example in Table 10.1. This rate can then be multiplied by the representative concentration of the COC to obtain the dose from the medium.

When using empirical food ingestion rates, the person should convert the ingestion rate to dry weight by dividing out the moisture content of the prey. This converted food ingestion rate may then be multiplied by the percentage of soil or sediment in diet from Beyer et al. (1994) (or a similar source) to derive a soil or sediment ingestion rate.

In the PCL Database on the "PCL Calculator" page, click on the "Species" tab toward the top of the page (not the "Species" radio button under Step 1). Soil (or sediment) ingestion rates are in the middle of the screen. Ingestion rates are normalized for body weight. Derivation of ingestion rates are in the individual species-profile PDF, which is accessed by clicking on the underlined species name.

10.4.3.3. Body Weights

Body weights are generally reported in the literature as fresh weight, as would be obtained by weighing a live animal in the field. In addition to the *Handbook*, literature sources of wildlife body weights include Davis and Schmidly (1994), Dunning (1984, 1993), Burt and Grossenheider (1980), and Silva and Downing (1995), as well as species-specific peer-reviewed papers.

As shown in Table 10.1, the body weight can also be a statistically derived value. In the raccoon example, the body weight is an arithmetic mean of data derived from two studies. Professional judgment should be applied in the derivation of body weights for use in an ERA. For example, mean body weights of the same species can vary greatly between regions of the U.S. Therefore, preference should be given to stated body weights from studies closest to the affected property and with similar habitat. When available, male and female weights should be averaged to compute the body weight used in the dose equation.

In the PCL Database on the "PCL Calculator" page, click on the "Species" tab toward the top of the page (not the "Species" radio button under Step 1). Body weights for each species are in the middle of the screen. Derivation of body weights appears in the individual species-profile, which can be accessed by clicking on the underlined species name.

10.4.3.4. Example Calculation of Ingestion Rates

Table 10.1 is an example of the derivation of body weight, food ingestion rate, and soil ingestion rate for the raccoon. This example highlights some of the issues discussed in **10.4.3.1** through **10.4.3.3**. This example uses equations from Nagy (1987) to derive the food ingestion rate; however, Nagy (2001) could also be used.

Table 10.1. Raccoon food and soil ingestion rate example calculations.

Step	**Example**
Select a representative BW (in grams or kilograms)	BW = 5411 g or 5.411 kg (arithmetic mean of Illinois and Alabama BW data from the Handbook)
Obtain the Nagy (1987) allometric equation (Equation 3-7) for all mammals from the Handbook	IR_{food} (g/day DW) = $0.235 \times BW^{0.822}$ or IR_{food} (kg/day DW) = $0.0687 \times BW^{0.822}$
Calculate IR_{food}	IR_{food} g/day DW = $0.235 \times (5411^{0.822})$ IR_{food} g/day DW = 0.235×1171.53 IR_{food} g/day DW = 275.31 or IR_{food} kg/day DW = $0.0687 \times (5.411^{0.822})$ IR_{food} kg/day DW = 0.0687×4.01 IR_{food} kg/day DW = 0.275
Convert IR_{food} g/day into kg/day (if the gram-specific equation is used)	IR_{food} kg/day DW = 275.31 ÷ 1000 IR_{food} kg/day DW= 0.275
Normalize IR_{food} to BW of raccoon	IR_{food} kg DW/kg BW-day = 0.275 kg/day DW ÷ 5.411 kg IR_{food} kg DW/kg BW-day = 0.0508
Obtain percent soil ingestion from Beyer et al. (1994) or comparable source	Percent soil ingestion for raccoon = 9.4 percent (or 0.094)
Multiply normalized IR_{food} DW by the percent (fraction) soil ingestion to obtain a soil ingestion rate (IR_{soil})	IR_{soil} kg DW/kg BW-day = 0.0508×0.094 IR_{soil} kg DW/kg BW-day = 0.00478
Determine IR_{food} WW by dividing the BW–normalized IR_{food} DW by the fraction of DW in food	IR_{food} kg WW/kg BW-day = 0.0508 ÷ 0.22 (for omnivore) IR_{food} kg WW/kg BW-day = 0.231

10.4.4. Distinguishing between Inputs Based on Wet and Dry Weight

The TCEQ requires that soil and sediment data be reported on a dry weight basis. Therefore, it is important to ensure that ingestion rates for soil and sediment are also presented on a dry-weight basis. A common mistake is to estimate media ingestion rates as a percentage of a wet weight-based food ingestion rate. Food ingestion rates may be expressed on either a dry-weight or wet-weight basis. Therefore, the person should ensure that this is consistent with any uptake values used. In other words, both the food ingestion rate and the uptake factor should be expressed on a dry-weight or wet-weight basis. Thus, where allometric equations are used to determine food ingestion rates, the person should be aware that these values are usually expressed on a dry weight basis. Body weight is always expressed on a fresh (wet weight) basis. Any dry- or wet- weight conversions should be clearly explained in the text or tables of the risk assessment, including the reference and assumptions for percentage of moisture in the food or prey.

10.4.5. Exposure-Modifying Factors

In this document, exposure modifying factor (EMF, unitless) is a general term that primarily addresses species-specific adjustments to home range and seasonality after the exposure (dose) has been calculated. A default 100 percent EMF (i.e., 1) is used in the initial exposure assessment of required element 6 of the Tier 2 SLERA. If an EMF of 100 percent is assumed, then no further justification is required. The EMF is used to reduce the receptor’s exposure in the refined portion (required element 7) of the Tier 2 SLERA. The person must justify the use of any EMF applied to the receptor’s dose in required element 7.

10.4.5.1. Home Range and Area Use Factors

Home range is defined as the area that a typical individual of a given species travels over as part of its daily excursions from shelter for food, water, and mates. The *foraging range* is a subset of the home range restricted to gathering of food and water. A wildlife receptor’s home range may be larger or smaller than the exposure area. Therefore, the exposure area is not defined by a wildlife receptor’s home range.

Home range can include foraging range, territory size and nesting area; however, the value chosen to represent home range in the ERA should focus on where the receptor would most likely be exposed to contaminated media. A variety of factors may influence home range, including habitat quality, prey abundance, and population density. Area-use factor (AUF) is the ratio of size of the exposure area to the home range of the receptor. For example, suppose the size of an affected property is 16.0 acres and consists of 5 acres of non-ecological habitat (e.g., disturbed ground), 7.5 acres of short grass prairie and 3.5 acres of a stock pond. If a black-crowned night heron, which feeds equally on fish and benthic invertebrates, is one of the measurement receptors, only the 3.5 acres of the pond will be considered suitable habitat for the calculation of its AUF. Since the home range of the black-crowned night heron is 11 acres and the pond is 3.5 acres, then the AUF for the sandpiper is 0.32 or 32 percent. If the home range of another receptor is smaller than the applicable ecological habitat, the AUF will not be adjusted (the default value of 1 would be used). As illustrated by the example above, caution is necessary when estimating a receptor's AUF at affected properties that comprise a fraction of the receptor's home range. Receptors may also forage in unimpacted habitats adjacent to the affected property. When applying an AUF to receptors in these situations, the person should avoid additional exposure adjustments to the food intake that would duplicate the adjustment already factored into the AUF. The *Handbook* has species-specific home-range values, etc., for selected wildlife receptors, and Sample et al. (1997) provides regression models relating home range to body weight for various receptor types. Information on home range is also available in the peer-reviewed literature.

Based on literature values for the foraging or home range of a measurement receptor, AUFs are often indiscriminately applied without consideration of the receptor's ecology. Sometimes these adjustments fail to recognize that ecological receptors will only forage in areas of suitable habitat (e.g., wooded areas for the gray fox; smaller waterways and lakes for the mink) in and adjacent to an affected property. However, almost always, the affected property does not consist entirely of suitable habitat. Therefore, the TCEQ prefers that, for the refined assessment (required element 7), the person only consider the amount of available suitable habitat on the affected property when determining an appropriate AUF. Occasionally, the affected property may be adjacent to highly developed industrial areas or other land or topography that is not suitable for a receptor. In these cases, an ecological receptor may be restricted to the affected property, regardless of the size of its typical home range, because there is no other suitable habitat nearby (i.e., the suitable habitat of the affected property becomes an ecological island). In that case, the default AUF value of 1 would be appropriate.

10.4.5.2. Exposure Frequency and Seasonality

Seasonality, usually expressed as exposure frequency (EF), accounts for migration or other seasonal activity patterns (U.S. ACE, 2010). As discussed in **10.4.5.3**, seasonality for migrating receptors is most applicable to species with special status. Migration is rarely assumed in the dose calculation for measurement receptors because they are intended to represent the entire guild, and some guild members are likely present all year. However, other EF scenarios are possible. For instance, where aquatic-based receptors forage in an impacted intermittent stream, it may be appropriate to adjust the EF to reflect the period when the stream contains water.

Continuing with the black-crowned night heron as a measurement receptor, if the heron is expected to be present only 10 months out of the year because of migration, the EF would be 0.83 or 83 percent. However, the heron represents the entire carnivorous shorebird feeding guild and there could be permanent residents from the same guild at the affected property. Thus, the heron should be evaluated as a permanent resident, unless it can be shown that it is the most exposed member of the guild even with the adjustment for migration. However, if the pond only contains water for nine months and is dry the rest of the year, then an EF of 0.75, or 75 percent, would be appropriate.

The PCL Database presents receptor home ranges and migration information, when applicable. Go to the "Species" tab toward the top of the page and click on the individual underlined species listed under the "Species Name" column. EMFs can be incorporated into the PCL calculation by using the input columns at the far right of the "Analysis" page. There are columns for "AUF" and "EF." Using the black-crowned night heron, the AUF would be input as 32, as all inputs are entered as percentages. Any modifications to the default PCLs should be justified.

10.4.5.3. EMFs for Threatened and Endangered Species

Where a protected species is present and is migratory, it is appropriate to make an exposure adjustment for this receptor only (**not** the guild as a whole as discussed in **10.4.5.2**) provided that any resulting PCL is based solely on the NOAEL TRV. Of course, this would dictate that there are separate calculations made for the protected species and a different receptor representing the entire guild.

*The exposure frequency (entered as a percent) can modify the seasonality in the "EF" column as described in **10.4.5.2.***

10.4.5.4. Other Modifications

Other site-specific circumstances may justify an additional exposure modification. For example, a bioavailability of less than 100 percent can be incorporated into the exposure dose; however, as discussed in **10.4.7**, this application of bioavailability as an EMF is currently limited to metals, primarily arsenic and lead in soil or sediment, and must be based on site-specific studies. Although there may be other EMF scenarios, the person should consult with the TCEQ's ERA staff before adjusting the exposure.

10.4.6. Considerations for Species with Exposure and Toxicity Data Gaps

As previously mentioned, the TCEQ recognizes reptiles, amphibians, livestock, and cave-dwelling receptors as species that have not been traditionally evaluated in ERAs but should be assessed as to their potential presence at the affected property. The TCEQ acknowledges that a risk evaluation of these species is currently limited by the availability of data on exposure and toxicity. The exposure-pathway analysis for these general groups of species is presented in **6.6** and toxicity data as they apply to these species are discussed in **9.2.3**. The rest of **10.4.6** presents the exposure-evaluation procedures for these often-overlooked species.

10.4.6.1. Reptiles and Amphibians

Reptiles and amphibians are often not identified as measurement receptors in the ERA or are not evaluated in any appreciable manner. The TCEQ recognizes that health-effects data for these classes, unlike birds and mammals, are sparse for many COCs.

Reptiles. Because reptiles can have a significantly different type of exposure from birds or mammals, the dose equation can be adjusted for species like snakes that may eat intermittently. One approach assumes that a top predator snake may only eat once a month and therefore exposure frequency expressed as meals per year, along with a likely lifespan of 25 years, could be used to modify the standard mammalian and avian exposure equation. The resulting dose equation would become:

$$\text{Dose} = \frac{IR_{food} \times C_{food} \times EF \times ED}{BW \times AT}$$

where:

Dose = Daily dose (mg/kg- day)

IR_{food} = Ingestion rate of food (kg/meal)

C_{food} = Concentration of food (mg/kg)

EF = Exposure frequency (meals/year)

ED = Exposure duration (years)

BW = Body weight (kg)

AT = Averaging time (days)—period over which exposure is averaged (ED × 365 days)

The approach described above could be used in conjunction with a more traditional dose equation (such as the one provided in the PCL Database that does not account for exposure frequency or duration) to bound the risk. The state of the science for the evaluation of reptiles is developing; for example, Weir et al. (2010) developed a dermal-exposure model for birds and lizards, but there are few relevant and available TRVs. Because of the lack of accepted toxicity data or exposure models, the TCEQ recommends qualitative evaluations of potential risks to reptiles as well as calculations of dose appropriate to the most likely receptors at a site.

A qualitative assessment could include an evaluation of the literature to identify:

- General information concerning reptilian sensitivity to broad classes of chemicals, as appropriate to the affected property.
- Body–tissue-residue and egg studies associated with COC effects.
- Residue studies at COC-impacted and non-impacted sites.

- General population studies at impacted and non-impacted sites with similar COCs.

In any case, the ERA could also discuss the likelihood of exposure to site COCs, given the niche of the reptiles, and the fate and transport characteristics of the COCs in affected media.

As previously mentioned, for reptiles with no toxicity data, a TRV for a bird with a similar diet can be used in combination with reptile life-history information (e.g., body weight, food ingestion rate) to calculate a dose and an HQ. This is the only scenario where across-class extrapolations may occur and will be preferred where a protected species may occur at a site. If this approach is proposed, the TCEQ recommends using a UF of 0.1 for the extrapolation. Exposure factors for the reptiles should be documented and justified and all assumptions will need discussion in the uncertainty analysis.

Amphibians. Immersion and dermal absorption are appropriate pathways for evaluation in place of or in conjunction with oral-dose data, particularly for amphibians. If no amphibian toxicity data (e.g., LC_{50} data, sediment or soil-effect concentrations) for the specific COCs can be found, if surface water concentrations demonstrably meet water quality criteria (or surface water benchmark screening values) and sediment concentrations are protective of benthic invertebrates, then amphibians can be assumed to be protected. Be aware that delayed metamorphosis as an effect should be considered where surface waters or pools are only present for short periods. Threatened and endangered species may require additional effort. Where a protected amphibian species could be exposed to a COC that does not have a state-adopted or federal criterion, the person should further evaluate potential risk to that species through effects data and apply an uncertainty factor of 0.1 if non-amphibian effects data are used and the COC is known to be more toxic to amphibians than the test species (see Weltje et al., 2013). Additionally, for example, the person can be more rigorous in evaluating data (e.g., use maximum concentrations or other more conservative statistics). Many protected amphibians (frogs and salamanders) could occur in many Texas counties, particularly along the Texas-Mexico border and in association with springs and karst-cave features (TPWD, 2016b; Gunnar, 2002).

10.4.6.2. Livestock

Water screening values. An adequate and safe water supply is necessary to the production of healthy livestock. Surface water (and groundwater) used by livestock for watering can be impacted by COCs. Many resources were surveyed to identify drinking-water screening levels for livestock. These screening levels, which are largely for metals and metalloids, are summarized in Table 10.2 and are applicable to surface water and groundwater. Other screening values can be proposed with appropriate justification. Comparing these levels to TRRP groundwater-ingestion PCLs protective of human health (residential), the human health PCLs are generally protective of livestock. However, livestock may be more sensitive than humans in the case of aluminum, copper, cyanide, manganese, selenium, uranium, and vanadium. Alternatively, risks to livestock from COCs in drinking water can also be evaluated in a manner like wildlife receptors using dose and HQ calculations based on appropriate exposure assumptions and toxicity values.

Livestock exposure assumptions. Various ingestion rates for water, soil (or sediment), and food for livestock appear in Table 10.3. Where ranges are provided, the person should conservatively select a value that best represents the expected exposure for a site. Alternative exposure assumptions can be used with adequate documentation. Note that livestock water ingestion rates are highly variable. Water consumption rates will differ depending on the dissolved salts in the water, season, shade availability, feed moisture content, age, reproductive status (e.g., pregnant or lactating females), and body weight. The assumed water ingestion rate should be adjusted to reflect the expected site conditions.

Table 10.2. Recommended concentration limits for substances in drinking water for livestock.

COC	Value (mg/L)	Source	Notes
aluminum	5	c	No data
arsenic	0.2	b	No data
beryllium	0.1	a	No data
boron	5	c	No data
cadmium	0.05	b	No data
calcium	1000	c	Assumes calcium is dominant cation and dietary phosphorus levels are adequate. Tolerable levels may be lower with elevated dietary magnesium and sodium or if calcium is added as a feed supplement.
chromium	0.05	a	No data
cobalt	1	b	No data
copper	0.5–5.0	a	1.0 mg/L (cattle); 5.0 mg/L (avian, swine); 0.5 mg/L (sheep)
cyanide	0.01	e	specific to horses
fluoride	2.0	c	1.0 mg/L if feed contains F-
lead	0.1	b	No data
magnesium	125	e	specific to horses; decreased palatability rather than toxicity
manganese	0.05	e, f	specific to horses and cattle
mercury	0.003	a	No data
molybdenum	0.15	c	No data
nickel	1	b	No data
nitrate (NO_3^-)	400	c	No data
nitrite (NO_2^-)	30	c	No data
nitrate + nitrite N	100	a	as NO_3 + NO_2-N
selenium	0.05	a, e	0.01 mg/L for horses

COC	Value (mg/L)	Source	Notes
silver	0.05	e	specific to horses
sulfate (SO_4^{2-})	500	d	important to consider total dietary contribution of sulfur
uranium	0.2	c	No data
vanadium	0.1	b	No data
zinc	20	c	No data

a. CCME, 2016.

b. NAS, 1974 (see Table 13).

c. ANZECC, 2000.

d. Morgan, 2011.

e. Lewis, 1996.

f. Higgins et al., 2008.

Soil ingestion rates, as a percent of food ingestion, are suggested in Table 10.3 for some livestock animals. Although livestock may intentionally ingest soil to satisfy a mineral nutrient imbalance, soil ingested accidentally by grazing livestock can sometimes form a considerable proportion of the diet. Grazing animals can ingest soil (or sediment) that adheres to vegetation or ingest soil directly from the surface while they feed or lick their snouts and fur. See Abrahams (2005) and Herlin and Andersson (1996) for a general discussion of soil ingestion by livestock. Soil ingestion will generally increase with grazing intensity, root intake, management practices that may increase the amount of soil on vegetation, and decreased forage availability. Additionally, assumed soil ingestion percentages may need to be conservatively adjusted where the pasture vegetative cover is decreased (e.g., more exposure to soil while grazing).

Food intake rates also appear in Table 10.3. Rates will vary with body weight, age, season, water availability, animal condition, activity level, level of milk production, stage of production, forage quality, amount and type of supplement or feed provided, and shelter. Rates presented on a body-weight basis are preferred. Additionally, exposure assumptions may be modified to account for exposure duration and exposure areas relative to specific livestock-management practices. Where dose calculations are performed, they should ideally consider the total COC dose to the animal [i.e., in water, soil, forage, and feed supplements (if known or can be predicted)].

Table 10.3. Livestock exposure assumptions.

Livestock	Water Intake	Water Intake Notes	Body Weight (kg)	Body Weight Notes	Food Intake	Food Intake Notes	Soil Ingestion	Soil Ingestion Notes
Beef Cattle	0.121–0.20 L/kg BW-day	a; finished cattle, 70–90°F	454	f, mature cattle	2.5	f; kg/100 kg BW	10.9 %	j
Beef Cattle	0.156–0.166 L/kg BW-day	a; lactating cows, 70–90°F	--	--	--	--	--	--
Beef Cattle	0.07–0.113 L/kg BW-day	a; mature bulls, 70–90°F	890	a, mature bulls	--	--	--	--
Dairy Cattle	No data	--	680 454	g, Mature Holstein g, Mature Jersey	--	--	4–8 %	k
Dairy Cattle, Lactating	83.3–121.9 L/day	b, based on 1,500 lb cow producing 40–80 lbs milk/day between 60–80°F	--	--	11–26	h; kg/day	--	--
Dairy Cattle, Pregnant Heifers	40.9–54.9 L/day	c, based on 1,200 lb cow between 60–80°F	--	--	--	--	--	--
Horses	9.6	d; (L/100 kg BW, idle 30° C ambient); hay fed	500–600	h	2–3	i; kg DMI/100 kg BW	5.8 %	l
Sheep	--	--	--	--	--	--	7.6 %	m

Livestock	Water Intake	Water Intake Notes	Body Weight (kg)	Body Weight Notes	Food Intake	Food Intake Notes	Soil Ingestion	Soil Ingestion Notes
Mature Ewes, Lactating	0.296	e; L/kg $BW^{0.75}$ day	80–90	e	0.029	s; kg/kg BW-d (80–90 kg, mid-lactation (twins)	--	--
Mature Ewes, Maintenance	0.146	e; L/kg $BW^{0.75}$ day	80–90	e	0.016	e; kg/kg BW-day (80–90 kg)	--	--
Goats, Maintenance	0.146	e; L/kg $BW^{0.75}$ day	60–70	e	0.019	e; kg/kg BW-day (60–70 kg) c	--	--
Goats, Lactating	0.296	e; L/kg $BW^{0.75}$ day	60–70	e	0.032	e; kg/kg BW-day (60–70 kg, mid-lactation kid)	--	--

a. NRC, 2001 (finishing cattle; 70-90ºF, variable weights).

b. Looper and Waldner, 2002.

c. Falk, 2014.

d. NRC, 2007a.

e. NRC, 2007b.

f. Lyons et al., 1999.

g. NRC, 2001.

h. CCME, 1993 (mature weights for small-breed [Jersey] and large-breed [Holstein] cows).

i. Freeman, 2007.

j. Kirby and Stuth, 1980.

k. Fries et al., 1982.

l. Sneva et al., 1983 (value for wild horses; soil intake as a percentage of forage intake per year).

m. Smith et al., 2009 (overall median value; range 0.1–1.8 percent).

10.4.6.3. Cave-Dwelling Receptors

Effects from COC exposure to cave-dwelling receptors cannot be evaluated in any meaningful way. Consequently, troglobites are not usually identified as measurement receptors in the ERA.

The TCEQ recognizes these limitations and suggests that the best way to protect these species from harmful exposure is to identify the circumstances that threaten them. One of the main threats to troglobites is habitat loss due to increasing urbanization and human population growth. Effects of urbanization include habitat loss from filling and collapsing caves, habitat degradation through alteration of drainage patterns, alteration of surface plant and animal communities, edge effects, contamination from pollutants, human visitation, vandalism, and activities associated with mining and quarrying and with predation (U.S. FWS, 2011b).

The extremely limited distributions of many troglobites make them particularly susceptible to extinction. This is especially true in areas undergoing rapid urbanization. The first species to be placed on the U.S. FWS endangered species list was the Texas blind salamander (*Typhlomolge rathbuni*) in 1967. This salamander, restricted to the San Marcos Pool of the Edwards Aquifer, is threatened by pollution as the city of San Marcos, lying directly above the aquifer, continues to grow (Reddell, 1994). It is also threatened by declining water levels in the Edwards Aquifer.

Alteration of drainage patterns. Cave organisms are adapted to live in a narrow range of temperature, humidity, and nutrients that are washed into caves. To sustain these conditions, both natural surface and subsurface flow of water and nutrients should be maintained. Decreases in water flow or infiltration can result in excessive drying and may slow decomposition, while increases can cause flooding that drowns air-breathing species and carries away available nutrients (U.S. FWS, 2011b). Alterations to surface topography, including decreasing or increasing soil depth or adding non-native fill, can change the nutrient flow into the cave and affect the cave community (Howarth, 1983). Impermeable cover, collection of water in devices like storm sewers, increased erosion and sedimentation, and irrigation and sprinkler systems can affect water flow to caves and karst ecosystems (U.S. FWS, 2011b). Altering the quantity or timing of water input to the karst ecosystem, or its organic content, may negatively impact these species.

Alteration of surface plant and animal communities. Karst ecosystems are heavily reliant on surface plant and animal communities to maintain nutrient flows, reduce sedimentation, and resist exotic and invasive species (U.S. FWS, 2011b). As the surface around a cave entrance becomes developed, native plant communities are often replaced with impermeable cover or exotic plants from nurseries. The abundance and diversity of native animals may decline due to decreased food and habitat combined with increased competition and predation from urban, exotic, and pet species. The leaf litter and wood that make up most of the detritus may also be reduced or altered, resulting in a reduction of nutrient and energy flow into the cave. A study at the Lakeline Cave in Travis

County showed a decline in cave crickets as the area around the cave was developed (Zara Environmental, 2008).

Edge effects. These are changes to the floral and faunal communities where different habitats meet. Edge effects associated with soil disturbance and disruption to native communities that accompany urbanization may attract red-imported fire ants (RIFA) or other surface species that prey on or compete with cave species (Reddell, 1993). The invasion of RIFA is aided by “any disturbance that clears a site of heavy vegetation and disrupts the native ant community” (Porter et al., 1988) such as road building, site remediation, and urbanization. Development and edges often allow enough disruption for invasive or exotic species to displace native communities that had previously prevented their spread (Kotanen et al., 1998; Suarez et al., 1998; Meiners and Steward, 1999).

Contamination. Karst landscapes are particularly susceptible to groundwater contamination because water penetrates rapidly through bedrock conduits and little or no filtration occurs (White, 1988). In some areas, the water that moves through the habitat of these species percolates to major aquifers (e.g., the Edwards Aquifer). The Edwards Aquifer is an important source of drinking water for nearly 2 million people (<edwardsaquifer.net/>, accessed October 20, 2016). So, sources of water contamination of the Edwards Aquifer may indicate sources of contamination of karst habitat (U.S. FWS, 2011b). As the ranges of troglobitic species become increasingly urbanized they become more susceptible to contaminants including sewage, oil, fertilizers, pesticides, herbicides, seepage from landfills, pipeline leaks, or leaks in storage structures and retaining ponds. Activities on the surface, such as disposing of toxic chemicals or motor oil, can also contaminate caves (White, 1988). Continued urbanization will increase the likelihood that karst ecosystems are polluted by contamination from leaks and spills.

Human visitation and vandalism. Visitation can impact caves by increasing soil compaction, trash deposition, and vandalism; altering airflow as entrances are expanded and excavated; scaring away trogloxenes (Culver, 1986; Elliott 2000); and may also lead to direct mortality of cave organisms crushed by human disturbance (Crawford and Senger, 1988). Commercialization of caves affects cave communities due to (1) competition with introduced surface species; (2) harmful effects of commercial lighting—for example, increased temperature and decreased humidity near lights; (3) substrate changes around trails; (4) changes in microclimate due to cave ventilation; and (5) increases in the nutrient regime that favor surface species (Culver, 1986; Reddell, 1993; Krejca and Myers, 2005; Mulec and Kosi, 2009).

Quarrying and mining. These operations take place in all the karst-bearing counties in the state. In Bexar County, some of these occur in the northern half, where most of the listed species for the county occur (U.S. FWS, 2011b). While quarrying activities have revealed some caves, which can lead to protecting these locations, they have also destroyed others (Elliott, 2000). As caves and mesocavernous spaces are destroyed at mines and quarries, karst species, possibly including some listed species, will also be lost.

Predation. RIFA are members of a pervasive, non-native ant species originally introduced to the U.S. from South America (Vinson and Sorensen, 1986) in the

1940s (Porter and Savignano, 1990). This ant is an aggressive predator and competitor that has spread across the southern United States. This predator often replaces native species, and evidence shows that overall species richness and abundance drops in infested areas (Vinson and Sorenson, 1986; Porter and Savignano, 1990). Also, several rare and threatened ant species may be disproportionately impacted by RIFA (Porter and Savignano, 1990).

Karst invertebrates in central Texas are especially susceptible to RIFA predation because some of the caves that karst invertebrates inhabit are relatively short and shallow (U.S. FWS, 2011b). The hot, dry weather may also encourage RIFA to move into caves during summer months or seek refuge or prey in caves during colder periods in the winter. RIFA have been found within and near many caves in central Texas and have been observed feeding on dead troglobites, cave crickets, and other species within caves (Elliott, 1992, 1994, 2000; Reddell, 1993; Taylor et al., 2003). A quantitative study of RIFA at six central Texas caves showed that they primarily used the entrance, but during cooler months were occasionally found deep into caves, not necessarily using human entrances as access points (Taylor et al., 2003). Of 36 caves Veni and Reddell visited during status surveys for the nine Bexar County karst invertebrates, RIFA were found inside 26 of them (Reddell, 1993). Karst fauna life stages that are likely most vulnerable to RIFA predation are the immature stages, eggs, and slower-moving adults.

Besides direct predation, RIFA threaten listed invertebrates by reducing the nutrient input that fuels the karst ecosystem (U.S. FWS, 2011b). Cave species rely on nutrients from the surface that are either washed in the entrance or carried in by trogloxenes like cave crickets. A study that assessed foraging behavior of cave crickets at Government Canyon State Natural Area found less competition between cave crickets and RIFA at caves that were managed to reduce RIFA mounds (Lavoie et al., 2007). Because RIFA are voracious, they can out-compete crickets for food resources (Taylor et al., 2003), leading to a reduction in overall productivity in the caves.

TCEQ recommendations. The person should be cognizant of the site's location with respect to adjacent karst environments, especially in Travis, Williamson, Hays, and Bexar Counties. Site activities associated with the known threats to karst ecosystems described above should be avoided if karst habitat is suspected within the vicinity. ERA Program staff will be particularly watchful of site-related impacts to groundwater in karst areas. TCEQ (2007b) presents optional enhanced water quality measures and best management practices for protecting the Edwards Aquifer that will also result in the protection of the habitat of certain endangered and candidate karst-dwelling invertebrates. The BMPs contained in this document have been reviewed by the U.S. FWS, which concurs that these voluntary, enhanced water quality measures will protect endangered and candidate karst-dwelling species from impacts due to water quality degradation.

10.4.7 Bioavailability

Bioavailability is the ratio of a COC that reaches a site of toxic action or biological response in an organism to the total load of that COC in the environment. Uptake and elimination rates of the bioavailable form are important, since the combined effects of these factors determine whether the material is accumulated or eliminated. For example, at the extremes, high uptake and low elimination rates suggest a high bioaccumulation potential, whereas low uptake and high elimination rates suggest low potential.

Bioavailability is the cumulative expression of physical, chemical, and biological processes evident in air, water, soil, and sediment, as well as biological factors present in the bodies, organs, tissues, or cells of exposed organisms that act to change that organism's rate of COC exposure. For example, the form of a COC can affect the degree of stomach absorption—e.g., soluble (e.g., barium sulfate) versus insoluble (e.g., barium carbonate) metal compounds. Zhang et al. (2014) lists factors that can influence bioavailability and toxicity of metals in sediments and waters: (1) solid phases, especially metal bindings, such as acid volatile sulfides, particulate organic carbon, iron and manganese oxyhydroxides; (2) aquatic phase, i.e., overlying and pore water physical-chemical attributes, such as pH, redox potential, hardness, salinity, and ligand complexes and (3) sensitivity and behavior of benthic organisms, e.g., taxon, lifestyle (such as bioturbation and burrowing) and prior exposure. For aquatic receptors, the bioavailable fraction of COCs is closely related to the concentration dissolved in water.

Although bioavailability is a key factor in generating realistic and quantitative exposure estimates for ecological resources, there are few universally accepted ways to quantify bioavailability. Hence, a default of 100 percent is always used in required element 6 of the Tier 2 SLERA (see **11.2**). If a bioavailability of 100 percent is assumed, then no further justification is required. However, if appropriate data are available for site conditions (e.g., a site-specific soil study), then modifications can be incorporated to reflect site-specific bioavailability, preferably in a Tier 3 SSERA[22] (see **10.4.7**). The use of a bioavailability factor less than 100 percent would need to be justified in the ERA. Based on the current state of the science, it is anticipated that these studies will be limited primarily to evaluations of lead and arsenic, as discussed in **10.4.7.1**.

10.4.7.1 Site-Specific Bioaccessibility

U.S. EPA (2007c) defines *oral bioavailability* as "the fraction of an ingested dose that crosses the gastrointestinal epithelium and becomes available for distribution to internal target tissues or organs." Many factors may affect the bioavailability of COCs in soil: chemical and physical factors related to the soil, as well as biological and physiological variables of the exposed receptor. Examples of physical factors include pH, particle size, other COCs, and how long the COC has been in contact with the soil. A related term is *bioaccessibility*, which is defined as the fraction of the chemical extractable from its matrix (e.g., soil, sediment, food) into the gastrointestinal tract that is available for absorption (Koch and Reimer, 2012). U.S. EPA (2007c) defines *bioaccessibility* as "a measure of the physiological solubility of the metal at the portal of entry into the body". For the default of 100 percent bioavailability used in required

[22] A site-specific soil or sediment study to determine percent bioavailability would normally be considered a Tier 3 evaluation. However, if time and resources allow, this effort may be conducted in required element 7 of the SLERA.

element 6 of the Tier 2 SLERA to be reduced in required element 7, site-specific bioavailability or bioaccessibility factors can be developed. This publication focuses on bioaccessibility, although the two terms are often used interchangeably modifying exposure assessments. Literature-based adjustments of bioaccessibility will not be accepted by the TCEQ because of the significant number of variables associated with the medium (e.g., soil matrix) and the receptors. A bioaccessibility factor for a COC is not a one-size-fits-all value for different soil types and different species (i.e., a wide variety of gastrointestinal processes).

Site-specific data may be developed *in vivo* or *in vitro*. *In vivo* evaluation involves administration of COC-contaminated soil or food via gavage to live laboratory animals such as rats, mice, rabbits, swine, and monkeys. Because *in vivo* measurements are expensive and time consuming, *in vitro* bioaccessibility techniques are used much more frequently to support ERAs. *In vitro* studies can be done more rapidly than *in vivo* studies, at a lower expense, and with more soils or sediments. One major uncertainty with *in vitro* studies is how well they correlate with *in vivo* studies.

In vitro methods are laboratory bench-top methods that consist of mixing aliquots of the selected ingested matrix (soil, sediment, or food) with a solution of salts and enzymes whose composition and pH are intended to simulate the conditions within the gastrointestinal tract of a specific species. This mixture is maintained at a specific temperature for a specified time (based on the digestive processes for the species of interest), after which a solute sample is extracted for analysis. Total analyses of the soil or food matrix are carried out concurrently. The concentrations are compared, and the bioaccessible percentage determined.

Most research in bioaccessibility has focused on supporting assessments of human health exposure. These human health-based models are suitable for mammalian receptors with a simple monogastric digestive physiology but are sources of considerable uncertainty in ecological risk assessments due to the diversity of the digestive physiology among mammalian taxa (Sample et al., 2014). For example, Walraven et al. (2015) found that oral bioaccessibility of lead depends on (1) the chemical composition of the lead source and its solubility, (2) the specific reactive surface of lead in soils, and (3) the type of soil.

Most bioaccessibility research on metals has focused largely on lead and arsenic, although there are published papers available on other metals and some organics. The literature shows that metals bioaccessibility vary by metal, the chemical form of the metal, source of the metal, soil parameters (e.g., pH, organic matter, clay, reactive iron, and calcium carbonate composition), aging processes, and the extraction method used. For example, Bradham et al. (2011) developed a mouse *in vivo* model to assess bioavailability of soil-bound arsenic and developed an *in vitro* bioaccessibility assay using simulated gastric fluid. The researchers found that the bioavailability and bioaccessibility estimates using the two methods were highly correlated and the soil physiochemical properties (sorption of arsenic to iron and aluminum oxides) were also correlated with both the bioavailability and bioaccessibility estimates.

Cadmium, nickel, chromium and mercury are the most frequently studied metals besides lead and arsenic; however, there is no consensus on study design or coordination between *in vivo* and *in vitro* studies. These metals illustrate a wide range of characteristics that affect *in vivo* and *in vitro* study design.

- *In vivo* study design must consider characteristics that affect bioavailability. For example, cadmium in soil is poorly absorbed and has a long half-life. Both nickel and chromium are primarily excreted in the urine, making urinary-excretion measurements a viable approach; however, the naturally occurring soil contribution to urine concentration will be difficult to detect, because background exposures from diet typically exceed the contribution from soil to total exposures.

- *In vitro* approaches to estimating bioaccessibility may also need to be varied based on both metal characteristic and interactions with soil. The use of gastric phases has focused on the human model and not wildlife.

- Metals such as mercury and chromium have varied toxicity for different forms (e.g., oxidation states). This presents challenges to ensure that forms with comparable toxicity are being assessed (ENVIRON, 2011).

- Toxicokinetic factors—including degree of absorption of the soluble metal form, disposition, and long or short half-life—are most important for *in vivo* method design, as illustrated by methods used for lead that typically include measurements of concentrations in blood, liver, and kidney after repeated-dose studies (U.S. EPA, 2007c), compared with studies of arsenic based on urine excretions or blood concentrations after either a single dose or more than one (Bradham et al., 2011; Casteel et al., 2003; Freeman et al., 1995; Roberts et al., 2007).

Information on the processes that influence bioaccessibility and bioavailability to wildlife is in its early stages of development, so medium-specific default values are not available for the calculation of exposure dose (Sample et al., 2014). However, the TCEQ will consider site-specific bioaccessibility studies for lead or arsenic as part of a Tier 3 SSERA. In that case, the person is urged to provide a methodology in the Tier 3 SSERA work plan.

The TCEQ will accept site-specific *in vitro* bioaccessibility assays for lead using U.S. EPA Method 1340 (U.S. EPA, 2013b). For arsenic bioaccessibility, the TCEQ will accept procedures developed by U.S. EPA Region 8 with the University of Colorado (Drexler, 2012; Brattin et al., 2012). Both test procedures are similar. Briefly, COCs are extracted from soil or sediment (or food), filtered, and analyzed to quantify the fraction of lead or arsenic in the sample that had dissolved. The TCEQ requires that the sampling design be included in the Tier 3 SSERA work plan and all laboratory reports be included in the Tier 3 SSERA. In the future, the TCEQ will accept studies on other COCs as methods are developed and accepted by the U.S. EPA.

10.4.7.2 AVS and SEM

The acid volatile sulfide and simultaneously extracted metal (AVS and SEM) method is widely used to estimate if metals in sediments are toxic. AVS refers to the fraction of sulfide in sediments extracted by cold hydrochloric acid and heavy metals released during this treatment are defined as SEM. AVS exists in natural sediments primarily as iron and manganese sulfide complexes. The AVS and SEM approach has become widely applied to predict the bioavailability of divalent metals in sediments, normally including cadmium, copper, lead, nickel, and zinc. Other ions, such as silver (+1), cobalt (+2), mercury (+2), arsenic (+3), and chromium (+2) can also form metal or metalloid sulfides less soluble than iron and manganese monosulfides. Consequently, the bioavailability of these elements in anoxic-sulfide sediments is likely also controlled by sulfide (Fu et al., 2014).

Bioavailability of cadmium, copper, lead, nickel, and zinc in anoxic sediments can be predicted by measuring the 1:1 relationship (in µmoles) between AVS and SEM. Total SEM (ΣSEM) is the sum of cadmium, copper, lead, nickel, and zinc. The TCEQ will consider the inclusion of other metals such as silver and mercury in ΣSEM case by case.

The difference, termed *ΣSEM-AVS*, is useful for predicting bioavailability and toxicity (or lack thereof) of metals to benthic organisms in sediments (Di Toro, 2008). The ΣSEM-AVS paradigm has also been shown to be accurate in predicting the *absence* of mortality in sediment toxicity tests (Di Toro et al., 1990, Hansen et al., 1996, U.S. EPA, 2005a).

The ΣSEM-AVS model is predicated on the same premise as the EqP model, i.e., the toxicity of metals in the sediment is directly related to its equilibrium between activity in sediment and the pore water. For cationic metals, however, solubility is theoretically governed by the strong complexation with sediment sulfides. By comparing the molar quantity of ΣSEM and AVS in a sediment sample, a measure of the bioavailable metal fraction can be estimated (Di Toro et al., 1990), where:

- ΣSEM-AVS < 1 indicates the ΣSEM is bound to sulfide (sulfide is in excess) and are therefore not bioavailable.
- ΣSEM-AVS > 1 indicates the ΣSEM exceeds concentrations of acid-soluble sulfide and therefore may be bioavailable.

Incorporation of organic carbon. Fraction of sediment organic carbon (f_{oc}) plays a major role in the binding of excess divalent metals (Mahoney et al., 1996; U.S. EPA, 2005a). An excess of AVS will ensure that the bioavailability of metals (and the probability for toxicity) is low; an excess of SEM may indicate the potential for toxicity, unless the sediment fraction of organic carbon is enough to act as another binding phase for metals that are not bound by AVS. U.S. EPA (2005a) states that when the ΣSEM-AVS is normalized by dividing by the f_{oc}, toxicity is likely when the (ΣSEM-AVS)/f_{oc} is > 3000 µmol/g_{oc}, uncertain when the concentration is 130–3000 µmol/g_{oc}, and not likely when the concentration is < 130 µmol/g_{oc}.

Sampling and analysis. Proper sample storage and analytical techniques are required when analyzing anoxic sediment for AVS, because a change in redox state may alter sediment parameters. Lasora and Casas (1996) state that sulfide levels are best maintained when samples are handled under a nitrogen atmosphere, stored at 4ºC or frozen at –20ºC and analyzed within two weeks of collection. De Lange et al. (2008) reported that storage conditions and sediment treatment affected AVS but not SEM levels.

The best way to sample the sediment is with an Ekman grab sampler. Immediately store the sediment in a jar without head space and freeze the sample as soon as possible. Hammerschmidt and Burton (2010) reported that to minimize AVS oxidation during transportation and storage, samples should be sent by overnight delivery in a frozen condition and extractions completed within 18 days of receipt. Analysis should follow Allen et al. (1991). This method uses cold 1 normal (N) HCL to extract AVS and SEM. The standard U.S. EPA Method 6020 using inductively coupled plasma mass spectrometry should be used for the SEM determination.

The TCEQ requests that the laboratory provides the method for determination of sulfide quantification (e.g., gravimetric, colorimetric, or electrochemical) and that the same method be used for all the samples from a site to minimize variability. The TCEQ also requires an increased number of duplicate samples and clear documentation on sample collection and handling because of the potential variability in the results. As reported by Hammerschmidt and Burton (2010), there can be significant issues with reproducibility and accuracy of AVS and SEM measurements.

Limitations and uncertainties. The circumstances giving rise to the formation and the distribution of AVS in aquatic sediments are complex (Zhang et al., 2014). The limitations and uncertainties of the ΣSEM-AVS and the ΣSEM–AVS/f_{oc} analyses should be clearly understood and those relevant to the SLERA should be discussed in the Uncertainty Analysis section. These limitations and uncertainties include, but are not limited to:

- *Dynamic nature of sediment and aquatic systems.* Use of ΣSEM–AVS in risk assessment assumes that reducing conditions will be constant; however, AVS varies both spatially and seasonally. AVS concentrations generally increase with sediment depth (Zhang et al., 2014). Yang et al. (2014) found that both AVS and SEM were slightly higher in spring than in winter, corresponding to relatively small temperature variations. ΣSEM–AVS considerations are most applicable in sediment environments characterized by high levels of sulfate (e.g., estuarine or marine environs) and high organic matter where bacterial activity can be expected to minimize oxygen penetration into the sediments, typically generating stable anoxic, reducing conditions (e.g., palustrine wetlands with seasonal die-off). The model does not account for potential dissociation during oxidation of the metal sulfide complexes (and thus increased bioavailability) that may occur with resuspension or the subsequent potential reformation of metal sulfides.

- *Benthic-invertebrate ecology.* The relationship between AVS and metal accumulation in invertebrates is largely determined by feeding behavior. Benthic invertebrates that reside in surface sediments can build an oxidation microenvironment and thus maintain a much lower level of AVS than that of bulk sediment (Zhang et al., 2014). In general, benthic invertebrates tend to concentrate in the oxidized sediments where occurrence of sulfides is not favored. Additionally, the SEM-AVS ratio method does not consider the ingestion of sediment by receptors. For deposit or detritus feeders—a large portion of benthic invertebrates—direct exposure to the sediment particles may be a major route of exposure. For example, the amphipod *Melita plumulosa* ingests large quantities of sediment during foraging for food and dietary exposure may elicit toxicity (Simpson et al., 2012). The bivalve *Tellina deltoidalis* is a deposit feeder and, although it buries in the surficial sediment layer, it has been observed to collect particles, and presumably organic matter, from the surface, by extruding a siphon that actively stirs the nearby sediment. This feeding behavior promotes oxidation of the substrate thus decreasing AVS concentrations (Campana et al, 2013). However, under reducing conditions, direct uptake of metal does occur in some infaunal species after ingestion of the various metal forms found in sediments, including metal sulfides (Lee et al., 2000). Thus, the TCEQ suggests that an assessment of the benthic invertebrate community be considered when conducting AVS and SEM measurements as an additional line of evidence.

- *Specific conditions and limited number of metals.* The methodology is only applicable in anaerobic sediments and for a limited number of metals (cadmium, copper, lead, nickel, and zinc). As previously mentioned, the TCEQ will not consider other metals, except for silver and mercury, and even then, only case by case.

- *Limited to benthic organisms.* An excess of AVS compared to SEM on a molar basis predicts that metals will be bound to the sediments and will not occur in interstitial water and thus will not be bioavailable to benthic organisms through associated exposure routes (dermal, gills, water ingestion). This theory does not address sediment and food ingestion, and subsequent food-chain transfer. In a study of metal bioaccumulation in estuarine food webs, Chen et al., (2016) found that all SEM-AVS values in sediment samples were negative, indicating that metals should all be bound to sediments and not available in porewater; yet, bioaccumulation still occurred, suggesting the importance of dietary sources. The TCEQ will not accept AVS and SEM results to evaluate risks to upper trophic level receptors.

ΣSEM-AVS can be used in an ERA to provide information for site-specific conditions. For ΣSEM-AVS to be considered as an additional line of evidence in a SLERA, the TCEQ requires that (1) only cadmium, copper, lead, nickel, and zinc be considered as SEM, (2) site-specific f_{oc} is determined and is used to normalize the ΣSEM-AVS so that (ΣSEM-AVS)/f_{oc} is presented, (3) the precautions taken to minimize oxidation of the samples are documented, (4) sufficient numbers of samples and duplicate samples are collected and analyzed to demonstrate reproducibility, (5) the laboratory procedures and quality assurance are documented, and (6) the conclusions of the ERA, and potential risk to benthic invertebrates, are not solely based on the ΣSEM-AVS methodology.

10.5. Special COC Classes

Several classes of COCs commonly detected during affected property assessments require unique analytical techniques, interpretation, analysis, or receptor identification. While not comprehensive, this discussion highlights some of these issues and discusses their incorporation into the Tier 2 requirements.

10.5.1. Metals

The chemical form (or *species*) of a metal can be important in determining its toxicity. For example, chromium III is much less toxic than chromium VI. Additionally, certain metals can exist in organic forms (e.g., methylmercury and alkyllead), which are generally more toxic than their elemental forms. Determination of a metal's species, however, is often prohibitively expensive, but in some situations a chemical-speciation model may be appropriate, especially if a species of metal (e.g., chromium VI) was used at the site or is known to be associated with a release.

The quantity of metals in an abiotic medium is not always indicative of its toxicity. The fraction of the total metal concentration that is bioavailable is usually a better indicator of toxicity. Factors that can influence the bioavailability of a metal include organic carbon, cation-exchange capacity, pH, sulfides, and water hardness. For aquatic organisms, the dissolved or soluble fraction of a metal may be a better measure of potential toxicity in both water and pore-water exposures. However, accurate determination of the dissolved fraction can be difficult and expensive and underestimates exposure to receptors with certain feeding behaviors (Lee et al., 2000). Nevertheless, for surface water and groundwater assessments, it is important to determine if the water samples should be analyzed for dissolved or total metals for each COC depending on existing aquatic life criteria and the exposure pathway in question (see **2.5** and **2.6**).

If PCL derivation (and remediation) is unwarranted based on the form of the metal used to derive the TRV (e.g., a soluble metal salt), then the preferred justification is property-specific data documenting the form of the metal present. Alternatively, the person may submit a detailed justification that addresses the known site chemistry and fate processes that influence the chemical form of the COC. In general, metals are assumed to be in the bioavailable form (or bioaccumulative form) unless sufficient analytical data are available to identify the metal species that are present.

10.5.2. Dioxins, Furans, and Polychlorinated Biphenyls

"Dioxin and dioxin-like substances" commonly refers to polychlorinated dibenzodioxins (PCDDs), polychlorinated dibenzofurans (PCDFs) and polychlorinated biphenyls (PCBs). They have two- or three-ring structures that can be chlorinated to varying degrees. PCBs can have up to 10 chlorine atoms substituting for hydrogen atoms, and PCDDs and PCDFs can have up to eight.

These compounds often have similar toxicity profiles and common mechanisms of action and are generally considered together. PCDDS and PCDFs are widely present in the environment as by-products of combustion and various industrial processes (WHO, 2010). They are also generated during the incineration of chlorinated compounds and are associated with the bleach-kraft pulp-mill process (Eisler, 1986a). PCDFs were major contaminants of PCBs, but neither PCDDs nor PCDFs have ever been manufactured deliberately. PCBs are not natural substances but were globally manufactured and used in the past. Although PCB manufacture is prohibited under the Stockholm Convention on Persistent Organic Pollutants, their release into the environment still occurs from the disposal of electrical equipment and waste (WHO, 2010).

There are 75 possible PCDD congeners, 135 possible PCDF congeners, and 209 possible PCB congeners. Their toxicity depends on the number and location of chlorine atoms. Additionally, these COCs have large K_{ow} values that dictate their mobility and partitioning in environmental and biological media (Eisler, 1986a).

"Dioxin-like effects" refers to effects that are like those caused by 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD). These chemicals exert effects through binding to the aryl hydrocarbon receptor (AhR). As reflected in Table 10.4, 7 dioxins, 10 furans, and 12 PCBs exert dioxin-like effects (U.S. EPA, 2008c).

Hazard characterization and risk assessment focus on a subset of PCDDs, PCDFs, and PCBs known as "dioxin-like." PCDDs and PCDFs and dioxin-like PCBs have a wide range of relative potencies and are usually found in complex mixtures in the environment. To simplify this process, internationally recognized toxic-equivalency factors (TEFs) are used (Van den Berg et al., 1998, 2006). The TEFs for mammals, birds and fish appear in Table 10.4. A TEF is an order-of-magnitude estimate of the toxicity of an individual congener relative to that of 2,3,7,8-TCDD. The concentration of each congener is multiplied by its TEF, and the sum of these products is defined as the toxic equivalency (TEQ) and is an estimate of the total 2,3,7,8-TCDD–like activity of the mixture.[23] As shown in the following equation, the 2,3,7,8-TCDD TEQ is the primary expression of exposure to an organism in an ERA involving complex mixtures of PCDDs, PCDFs, and PCBs.

[23] While the TEFs defined in Table 10.4 are appropriate for evaluating potential risks to wildlife exposed to dioxins, furans, and dioxin-like PCBs, a different set of TEFs is used to determine compliance with the surface water criteria protective of human health [see 30 TAC 307.6 (d)] when evaluating surface water data at TRRP site. Additionally, specific approaches to determine human health soil PCLs for these compounds are detailed in subsections 350.76(d, e) of the TRRP rule.

$$TEQ = \sum_{n=1}^{k} (C_n \times TEF_n)$$

where:

C_n = concentration of dioxin-like chemical n

TEF_n = toxic equivalency factor for dioxin-like chemical n (unitless)

k = number of toxic dioxin-like chemicals in mixture

The TEF methodology is appropriate and applicable in ERAs involving both aquatic and terrestrial systems. Once a TEQ has been calculated, it is treated as though it is the concentration of 2,3,7,8-TCDD in a medium (sediment, soil, and tissue). Then the 2,3,7,8-TCDD dose to a wildlife receptor can be compared with an appropriate TRV for 2,3,7,8-TCDD to evaluate relevant potential for population-level risks to the receptor or its guild. The EPA has developed several calculator tools as described at <www.epa.gov/superfund/risk-assessment-dioxin-superfund-sites#tefsteqs>. These spreadsheets calculate TEQs from congener results and take into consideration non-detect and rejected data.

The rationale for the use of TEQs or TEFs is based on a common mechanism of action described for planar halogenated hydrocarbons (PHHs) including dioxins, furans and some PCBs. The intracellular target of PHHs is the AhR, which—while bound to the ligand—mediates the transactivation and inhibition of a variety of target genes, with a wide array of deleterious effects. The structure of a ligand is critical to its affinity for the AhR, and this affinity correlates well with the resulting toxicity and biochemical responses (Safe, 1997). Due to its high potency, the AhR-mediated toxicity of 2,3,7,8-TCDD is the standard to which all other PHH potencies are normalized. The relative AhR-mediated toxicity of PHHs has been determined using a combination of *in vitro* and *in vivo* studies, and TEFs are available for mammalian, avian, and fish species (Van den Berg et al., 1998, 2006).

PCBs are highly lipophilic, bioaccumulative, and persistent in the environment. This class of COCs is generally strongly sorbed to particulates and ultimately resides in sediments or soils. PCBs readily cross most biological membranes and partition into fatty tissues. Concentrations of these COCs are expected to be higher in upper-trophic-level receptors because of food-chain transfer. The toxicity of PCBs depends on the congener as confirmed by the number and position of chlorine substitutions. The mechanism of toxicity of non- and mono-ortho-substituted PCB congeners (planar) is initiated through activation of the AhR in the same manner as TCDD. AhR-mediated toxicity results in a broad spectrum of maladies and has been studied extensively in many species (e.g. Safe, 1994; Poland and Knutson, 1982) but is still an active area of research.

Nonplanar PCBs are neurotoxic and carcinogenic, and elicit a wide range of responses, including behavioral, endocrinal changes (Van den Berg et al., 1998, 2006; Safe, 1994). These responses, which may resemble responses to dioxin in some cases, are, however, not mediated by the AhR and therefore cannot be normalized to the TCDD mechanism of toxicity using TEFs. Planar PCBs generally constitute a small percentage of the total PCBs in technical PCB mixtures but often pose the greatest risk of toxicity in the environment (Tillitt, 1999; Giesy and Kannan, 1998). Dioxin-like activity is generally the more potent in eliciting toxic responses.

PCBs were produced in the United States as specific mixtures of congeners known as Aroclors. PCBs can be analyzed and quantified as Aroclor mixtures or as individual congeners. A common question is whether PCB analyses should be based on Aroclors or individual PCB congeners. The pros and cons of both methods are thoughtfully presented in NAVFAC (2001). Briefly, PCB congener data offer many advantages over Aroclor data for environmental risk analysis including lower detection limits, better accuracy in quantitating individual congeners, and the ability to measure weathered, degraded, and metabolized PCB mixtures. Importantly, the toxicity of PCBs is congener-specific, and, therefore, assessments on an Aroclor basis may not accurately measure toxicity. Generally, the literature indicates that greater toxicity is associated with weathered PCB mixtures and the proportion of TEQs relative to total PCB mass increases with trophic level (Zimmermann et al., 1997; Giesy and Kannan, 1998). Aroclor mixtures will seldom maintain their original composition once they are released into the environment, so the use of reference doses derived from technical mixtures may be under protective of ecological receptors (Giesy and Kannan, 1998) and that the approach based on threshold-effect concentrations (TECs) or TEQ-TEF may more accurately indicate risk (Tillitt, 1999).

Table 10.4. 2,3,7,8-TCDD toxic equivalency factors

Congener	Congener Type	Mammal TEF	Bird TEF	Fish TEF
2,3,7,8-Tetrachloro-dibenzo-*p*-dioxin (TCDD)	Dioxin	1	1	1
1,2,3,7,8-Pentachloro-dibenzo –*p*-dioxin (PeCDD)	Dioxin	1	1	1
1,2,3,4,7,8-Hexachloro-dibenzo-*p*-dioxin (HxCDD)	Dioxin	0.1	0.05	0.5
1,2,3,6,7,8-Hexachloro-dibenzo-*p*-dioxin (HxCDD)	Dioxin	0.1	0.01	0.01
1,2,3,7,8,9-Hexachloro-dibenzo-*p*-dioxin (HxCDD)	Dioxin	0.1	0.1	0.01
1,2,3,7,8,9-Heptachloro-dibenzo-*p*-dioxin (HxCDD)	Dioxin	0.01	< 0.001	0.001
Octachloro-dibenzo-*p*-dioxin (OCDD)	Dioxin	0.0003	0.0001	< 0.0001
2,3,7,8-Tetrachloro-dibenzofuran (TCDF)	Furan	0.1	1	0.05
1,2,3,7,8-Pentachloro-dibenzofuran (PeCDF)	Furan	0.03	0.1	0.05
2,3,4,7,8-Pentachloro-dibenzofuran (PeCDF)	Furan	0.3	1	0.5
1,2,3,4,7,8-Hexachloro-dibenzofuran (HxCDF)	Furan	0.1	0.1	0.1
1,2,3,6,7,8-Hexachloro-dibenzofuran (HxCDF)	Furan	0.1	0.1	0.1
1,2,3,7,8,9-Hexachloro-dibenzofuran (HxCDF)	Furan	0.1	0.1	0.1
2,3,4,6,7,8-Hexachloro-dibenzofuran (HxCDF)	Furan	0.1	0.1	0.1
1,2,3,4,6,7,8-Heptachloro-dibenzofuran (HpCDF)	Furan	0.01	0.01	0.01
1,2,3,4,7,8,9-Heptachloro-dibenzofuran (HpCDF)	Furan	0.01	0.01	0.01
Octachloro-dibenzofuran (OCDF)	Furan	0.0003	0.0001	< 0.0001
3,3',4,4'-Tetrachloro-biphenyl (77)	PCB	0.0001	0.05	0.0001
3,4,4',5-Tetrachloro-biphenyl (81)	PCB	0.0003	0.1	0.0005
3,3',4,4',5-Pentachloro-biphenyl (126)	PCB	0.1	0.1	0.005
3,3',4,4',5,5'-Hexachloro-biphenyl (169)	PCB	0.03	0.001	0.00005
2,3,3',4,4'-Pentachloro-biphenyl (105)	PCB	0.00003	0.0001	< 0.000005

Congener	Congener Type	Mammal TEF	Bird TEF	Fish TEF
2,3,4,4',5-Pentachloro-biphenyl (114)	PCB	0.00003	0.0001	< 0.000005
2,3',4,4',5-Pentachloro-biphenyl (118)	PCB	0.00003	0.00001	< 0.000005
2',3,4,4',5-Pentachloro-biphenyl (123)	PCB	0.00003	0.00001	< 0.000005
2,3,3',4,4',5-Hexachloro-biphenyl (156)	PCB	0.00003	0.0001	< 0.000005
2,3,3',4,4',5'-Hexachloro-biphenyl (157)	PCB	0.00003	0.0001	< 0.000005
2,3',4,4',5,5'-Hexachloro-biphenyl (167)	PCB	0.00003	0.00001	< 0.000005
2,3,3',4,4',5,5'-Heptachloro-biphenyl (189)	PCB	0.00003	0.00001	< 0.000005

The primary concerns of using congener-specific analyses include cost and the general lack of toxicity data for most congeners. Aroclor analysis has been used for decades; therefore, many toxicity studies, effect concentrations, benchmark screening values, and bioaccumulation factors are based on specific Aroclors or total PCBs based on Aroclors. Congener-specific methods generally require the use of TEFs to determine potential adverse effects to ecological receptors. Additionally, there is not sufficient information available to determine conclusively that the risk associated with dioxin-like PCB toxicity is greater than that associated with the non-dioxin-like PCBs. Significant uncertainty could exist if the TEC or TEQ-TEF approach was used as a surrogate for total PCBs exposure overall.

Additionally, congener-specific data are preferred over Aroclor analyses to allow for accurate assessment of risk associated with dioxin-like, non-dioxin-like PCBs and Aroclors. Therefore, the TCEQ prefers both dioxin-like PCBs (using TEC or TEQ-TEF) and non-dioxin-like PCBs be evaluated separately in a Tier 2 SLERA. The PCL Database lists Aroclor 1248, Aroclor 1254, Aroclor 1260, and total PCBs as COCs.

The TCEQ recognizes that Aroclor analyses may be appropriate when:

- The historical information suggests there are no sources of PCBs.
- The project is in the initial stages of investigation to determine presence or absence of PCBs or a preliminary estimation of risk.
- Information on effects for the receptor or receptor group is only available for Aroclors or total PCBs based on Aroclors.
- Aroclor analyses are used in combination with total congener analyses (as a percentage) of the total number of samples.

Extensive information is available on the effects of PCDDs, PCDFs, and PCBs on fish and wildlife, and the TCEQ encourages consulting the literature to determine receptors and effects most appropriate for the affected property. Literature sources for reptiles and amphibians, such as Sparling et al. (2000) and Pauli et al. (2000), should be reviewed for information regarding the effects from exposure to these COCs. Toxic effects on fish-eating birds include developmental abnormalities, malformations, embryo mortality, and edema (White and Hoffman, 1995; Henshel et al., 1997; Grasman et al., 1998). Effects in mammals include cognitive disabilities, wasting syndrome, impaired immune response, decreased reproduction, reduced offspring survival, and mortality (Seo et al., 1999; Leonards et al., 1995; De Swart et al., 1995). Effects on fish include cranial deformities, yolk-sac edema, vascular hemorrhage, fin necrosis, hyperpigmentation, weight loss, and death (Black et al., 1998; Johnson et al., 1998; and Cantrell et al., 1996). Mink and some predatory fish and piscivorous birds are very sensitive to the toxicity of these COCs (Eisler, 1986b).

10.5.3. Polycyclic Aromatic Hydrocarbons

PAHs are multiple-ring structures of carbon and hydrogen, both natural and anthropogenic, and ubiquitous in the environment. There are hundreds of individual PAHs, but the compounds with molecular weights ranging from 128.17 (naphthalene, $C_{10}H_8$) to 300.36 (coronene, $C_{24}H_{12}$) are of particular environmental concern. The higher the molecular weight of the PAH, the more lipophilic, less volatile, and less soluble the PAH will be. Because of the differences in environmental fate, PAHs are divided into two categories: low molecular weight PAHs (LPAHs) and high molecular weight PAHs (HPAHs) (see Table 10.5).

The bioavailability of PAHs is influenced by organic carbon quality and quantity, aging and weathering, microbial action, methylation, adsorption and desorption and interaction with ultraviolet light (U.S. EPA, 2007d). Despite the lipophilic nature of PAHs, biomagnification is considered unlikely because they tend to be rapidly transformed or eliminated in many fish and mammalian species (Nezda et al., 1997; Yuan et al., 1999); however, accumulation and acute exposure can occur for lower trophic level species (e.g., invertebrates and those receptors that consume the invertebrates).

Generally, PAH toxicity involves the disruption of the normal function of enzyme systems or DNA damage by reactive metabolic intermediates. LPAHs (generally, with 2–3 benzene rings) are acutely toxic to many invertebrates, possibly because of their higher solubility, but are generally considered noncarcinogenic. HPAHs (generally, with 4–7 benzene rings) are less acutely toxic, but many are mutagenic, teratogenic, or carcinogenic. Additionally, HPAHs are more recalcitrant in abiotic and some biotic media. Genetic injury resulting in tumor formation depends on the metabolic transformation of the parent compound to one or more reactive intermediates.

Individual species have widely differing abilities to metabolize PAHs to carcinogenic intermediate compounds (Yuan et al., 1999; Livingston, 1998; Eisler, 1987). Invertebrate species are susceptible to acute toxicity (Thompson et al., 1999) and metabolic disruption (Saint-Denis et al., 1999) upon exposure to PAHs; however, stimulatory effects on invertebrate communities have been reported (Erstfeld and Snow-Ashbrook, 1999).

The effects of PAHs on mammals—particularly carcinogenesis—have been extensively studied (Eisler, 1987). Many fish species (especially bottom-dwelling fishes) are also very sensitive to the carcinogenic effects of PAHs (Anulacion et al., 1998; Baumann, 1998). Additionally, decreased circulating hormones, disruption of vitellogenesis and oocyte maturation, decreased reproductive success, and altered immune function have been reported in fish exposed to PAHs (Nicolas, 1998; Karrow, 1999). Cousin and Cachot (2014) reviewed numerous articles on the effects of PAHs in fish at several developmental stages and via many different routes of exposure and concluded that whatever the route and the mixture used, exposure to environmental concentrations of PAHs reduces fish fitness and participation in recruitment (i.e., their ability to contribute to the next generation). Very little information exists on their effects on avian species (Eisler, 1987; U.S. EPA, 2007d). Literature sources for reptiles and amphibians, such as Sparling et al. (2000) and Pauli et al. (2000), should be reviewed for information regarding effects of PAH exposure.

Depending on the media, screening-level benchmarks and PCLs are available for individual PAH compounds, LPAHs, HPAHs, and total PAHs (TPAHs). **However, the benchmarks and PCLs provided for TPAHs are the most relevant for evaluating risk in an ERA.** This is because PAHs almost always occur in the environment as mixtures and, as such, the piecemeal elimination of components that constitute the mixture should be avoided. Therefore, the screening-out of individual PAHs, LPAHs, and HPAHs based on benchmarks and PCLs is not appropriate where same-media values for TPAHs exist. This becomes clearer when remediation is considered. For example, in most cases where PAH-contaminated sediments or soils are being removed as a response action, it is not possible to distinguish between individual PAHs such as, benzo[a]pyrene and naphthalene, or HPAHs from LPAHs, as these are all part of the TPAH footprint.

Values for individual, LPAHs, and HPAHs should be used where there are no benchmarks or PCLs available for TPAHs. Otherwise, these values should be used as guidelines to aid in the determination of disproportionate concentrations of more toxic individual PAHs within the mixture that may be masked by the total. However, for sediment, any exceedances of individual, LPAHs, or HPAHs second-effects levels (see the sediment table in RG-263b) may indicate adverse effects and therefore should be further discussed (e.g., in the uncertainty analysis). Also, for surface water, the only available benchmarks are for individual PAHs, with phenanthrene having the only State-adopted criteria.

In the PCL Database, sediment and soil wildlife PCLs for TPAHs, LPAHs, and HPAHs are available, but preference should be given to the TPAH values. Sediment-based wildlife PCLs for TPAHs should be compared to the corresponding benthic PCLs for determination of the final ecological PCL for TPAHs in sediment (see discussion of comparative and final ecological PCLs in **13.4**). Where highly-weathered PAHs are site COCs and the LPAHs have degraded or volatized (as demonstrated by laboratory analysis), the person may use the HPAHs PCLs from the PCL Database for evaluating risk to wildlife.

Tables 10.5. Classification of PAHs for a Tier 2 SLERA

PAH	PAH Class	Molecular Weight (g/mole)	Number of Rings	Solubility (mg/L)	Log K_{ow}
Naphthalene	Low	128	2	31.4	3.17
1-Methyl naphthalene	Low	142	2	28.0	3.72
2-Methyl naphthalene	Low	142	2	25.4	3.72
Acenaphthylene	Low	152	3	3.93	3.94
Acenaphthene	Low	154	3	4.24	4.15
Fluorene	Low	166	3	1.98	4.02
Anthracene	Low	178	3	0.0434	4.34
Phenanthrene	Low	178	3	0.994	4.34
Fluoranthene	High	202	3	0.26	4.93
Pyrene	High	202	4	0.135	4.93
Benzo[*a*]anthracene	High	228	4	0.01	5.52
Chrysene	High	228	4	0.002	5.52
Benzo[*a*]pyrene	High	252	5	0.00162	6.11
Benzo[*b*]fluoranthene	High	252	5	0.0015	6.11
Benzo[*k*]fluoranthene	High	252	5	0.00055	6.11
Indeno[1,2,3-*c*,*d*]pyrene	High	276	6	0.00375	6.70
Benzo[*g*,*h*,*i*]perylene	High	276	6	0.00026	6.70
Dibenzo[*a*,*h*]anthracene	High	278	5	0.0005	6.70

Molecular weight, solubility and log K_{ow} taken from the TRRP chemical physical properties table (April 2018).

10.5.4. Total Petroleum Hydrocarbons

Total petroleum hydrocarbons (TPHs) comprise several hundred COCs and can be broadly categorized as *aliphatic hydrocarbons* (straight-chained alkanes, alkenes, alkynes, and cyclic compounds) and *aromatic hydrocarbons* (BTEX and PAHs) (ATSDR, 1995, 1999). Additionally, petroleum products may contain metals; nitrogen-, sulfur-, and oxygen-containing organics and additives such as oxidants, scavengers, and organolead compounds (Suter, 1997).

The origin of a TPH cannot always be determined due to differences in crude oil, refinery processes, and chemical, physical, and biological weathering. Due to the complexity and variety of these hydrocarbon mixtures, TPH analysis generally offers little necessary information for an ERA because it does not elucidate the properties that determine potential fate and toxicity of the material. Therefore, characterization of the individual components of TPH is necessary to facilitate a defensible assessment of ecological risk (Suter, 1997).

It is anticipated that advancements in the assessment of the ecological risk posed by TPHs will continue and readers are encouraged to search for the latest information. Until toxicological information becomes available specific to TPH fractions, the TCEQ will continue to recommend constituent-specific analysis with a focus on PAHs.

10.5.5. Munitions and Explosives

Because of military training and weapons testing activities, munitions are present at numerous current and former U.S. DOD sites. Munitions constituents are defined in 10 U.S.C. 2710(e)(4) as "Any materials originating from unexploded ordnance, discarded military munitions, or other military munitions, including explosive and nonexplosive material, and emission, degradation, or breakdown elements of such ordnance or munitions" (U.S. ACE, 2013).

Many active and former military installations have ranges and training areas that include aquatic environments, such as ponds, lakes, rivers, estuaries, and coastal areas. Areas where explosives impact soil or sediment quality are sometimes extensive; some artillery ranges are several square miles in area (U.S. ACE, 2013). Manufacturing of explosives and their loading, assembling, packing into munitions for use in testing, training, and combat has impacted terrestrial and aquatic systems (Montreil-Rivera et al., 2009).

Explosives are classified as primary or secondary based on their susceptibility to initiation. Primary explosives, which include lead azide, are highly susceptible to initiation. Primary explosives are referred to as initiating explosives because they can be used to ignite secondary explosives. Secondary explosives, which include 2,4,6-trinitrotoluene (TNT), cyclonite or Royal Demolition Explosive (RDX) and High Melting Explosive (HMX) are more prevalent than primary explosives in environmental media. Secondary explosives can also be classified according to their chemical structure as nitroaromatics (e.g., TNT) and nitramines (e.g., RDX) (U.S. EPA, 1993b). Benchmarks for common munitions are presented in the Benchmark Tables for surface water, sediment and soil. Where these types of COCs exceed screening benchmarks, they should be carried through the SLERA for the evaluation of risks to wildlife receptors. Wildlife TRVs are generally available from the open literature.

10.5.6. Radionuclides

Radioactive materials are regulated primarily under 30 TAC Chapter 336 (Radioactive Substances Rules). Pursuant to these rules, contamination limits are specified for media and vegetation, and are based on the protection of human health. For the protection of fish and wildlife, the person should consider the following sources. The U.S. Department of Energy (U.S. DOE) has developed RESRAD-BIOTA, which is a user-friendly tool that implements U.S. DOE's methodology described in U.S. DOE Technical Standard DOE-STD-1153-2002, A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota. The RESRAD-BIOTA program can be downloaded at <resrad.evs.anl.gov/codes/resrad-biota/>. The associated Technical Standard DOE-STD-1153-2002 can be found at <www.standards.doe.gov/standards-documents/1100/1153-AStd-2002>.

RESRAD-BIOTA analyzes radiation exposures to biota in terrestrial or aquatic systems. A range of organisms were evaluated to develop default exposure parameter values. These reference organisms are categorized into terrestrial animals and terrestrial plants for a terrestrial system and aquatic animals and riparian animals for an aquatic system. There are 45 radionuclides in the RESRAD-BIOTA database and three levels of assessment.

At Level 1, conservative screening values known as biota concentration guides (BCGs) are radionuclide concentrations in environmental media that do not exceed recommended dose rate guidelines. These generic screening values developed by U.S. DOE are for the general categories of terrestrial animals, terrestrial plants, aquatic animals, and riparian animals.

The first step in the graded approach is the comparison of the radiological concentrations in soil, water, and sediment to the U.S DOE conservative screening values, or BCGs. The TCEQ has incorporated the BCGs as benchmarks. For each medium, for radionuclide a, b, … *n*, with concentrations C_a, C_b …C_n, and for corresponding screening BCG values BCG_a, BCG_b …BCG_n, this relationship for aquatic and terrestrial systems is:

$$\left[\frac{C_a}{BCG_a} + \frac{C_b}{BCG_b} + \ldots + \frac{C_n}{BCG_n}\right] \text{water} + \left[\frac{C_a}{BCG_a} + \frac{C_b}{BCG_b} + \ldots + \frac{C_n}{BCG_n}\right] \text{sediment} < 1.0$$

$$\left[\frac{C_a}{BCG_a} + \frac{C_b}{BCG_b} + \ldots + \frac{C_n}{BCG_n}\right] \text{water} + \left[\frac{C_a}{BCG_a} + \frac{C_b}{BCG_b} + \ldots + \frac{C_n}{BCG_n}\right] \text{soil} < 1.0$$

If the sum of the fractions (the summed ratios between the radionuclide concentrations in environmental media and the radionuclide-specific BCGs) is less than 1.0, the dose to an aquatic or terrestrial receptor is below the biota dose limit. Note that BCGs for water are included in both the aquatic and terrestrial evaluations. This accounts for the total exposure that an organism would encounter. If the sum of the ratios for all media is greater than 1.0, then the person should consider a background evaluation and use of the U.S. DOE's publicly available program RESRAD-BIOTA. Level 2 in RESRAD-BIOTA allows for site specific adjustments. Level 3 is a kinetic-allometric modeling tool. The Level 3 assessment allows development of a specific receptor using the New Organism Wizard. If it is determined that a certain species is of concern at a site, then this species can be evaluated using site specific input parameters.

In addition to RESRAD-BIOTA, the International Atomic Energy Agency has developed the Environmental Modelling for Radiation Safety process as (2010, 2012, 2014a and 2014b). The U.S. DOE, the IAEA, and the International Commission on Radiation Protection (ICRP, 2008, 2009) all use a simple concentration ratio to predict radionuclide activity concentrations in wildlife from those in soil or water. The concentration-ratio approach has many limitations in that it does not account for site-specific factors (e.g., water or soil chemistry). Beresford et al. (2013) developed a mixed-model regression approach that used an existing data set for 53 freshwater fish species from 67 sites for Cesium-137. Other models and methods are available from the open literature as described by Stark et al. (2015) for wetlands habitats, Wood et al. (2009) for coastal sand-dune habitat, Yankovich et al. (2010) for a freshwater system and Johansen et al. (2012) for terrestrial wildlife. Because this technical area is continuing to develop, the person could consider methods of analysis that allow for site-specific information to be incorporated; however, the TCEQ recommends use of the U.S. DOE's BCG values as screening values.

10.5.7. Emerging Contaminants

There is no consistent definition of the term "emerging contaminants" (ECs) in the industry, primarily because the definition is based on one's perspective. ECs are chemicals that do not appear on the usual TRRP COC list but may have an impact on the environment. They are often not included in environmental legislation and their environmental fate is not always understood. Furthermore, analytical methods are often not sensitive enough to detect low concentrations in environmental media (Zenker et al., 2014). As an example, EPA Method 8270 for semivolatile organic compounds would not report concentrations of caffeine. However, there is increasing concern about the presence and potential impacts of ECs on the environment.

Thus far, much of the identification of ECs is from their presence in treated and discharged wastewaters. A study conducted by the United States Geological Survey between 1999 and 2000 sampled 139 streams across 30 states, with the compounds detected representing a wide range of residential, industrial, and agricultural origins. The most frequently detected compounds were coprostanol (fecal steroid), cholesterol (plant and animal steroid), *n,n*-diethyltoluamide (insect repellant), caffeine (stimulant), triclosan (antimicrobial disinfectant), tri(2-chlorethyl)phosphate (fire retardant), and 4-nonylphenol (nonionic detergent metabolite) (Kolpin et al., 2002). In 2008, Focazio et al. published findings from another national reconnaissance for ECs in untreated drinking water sources. The five most frequently detected chemicals included cholesterol (59 percent, natural sterol), metolachlor (53 percent, herbicide), cotinine (51 percent nicotine metabolite), β-sitosterol (37 percent, natural plant sterol), and 1,7-dimethylxanthine (27 percent, caffeine metabolite).

As discussed below, ECs include pharmaceuticals, personal care products, brominated flame retardants, perfluorinated compounds, and nanomaterials. Another EC of interest is 1,4-dioxane, which is a synthetic industrial chemical used as a stabilizer for chlorinated solvents and is completely miscible in water. It is being analyzed for at chlorinated solvent groundwater sites more commonly. It is relatively resistant to biodegradation in water and soil and does not bioconcentrate (U.S. EPA, 2014a); however, 1,4-dioxane is currently considered more of a human health than an ecological concern. As knowledge of the environmental effects of these compounds increases, the TCEQ will consider evaluating ECs as site COCs on a case by case basis where they appear to be site-related.

10.5.7.1. Pharmaceuticals

Pharmaceuticals are of concern for possible impact on aquatic ecosystems due to their universal use and their capacity to be incompletely reduced in wastewater treatment (Zenker et al., 2014). Pharmaceuticals as ECs include a wide variety of compounds such as antidepressants, analgesics, antivirals, antibiotics, beta-blockers, and antiepileptic and anti-inflammatory drugs but also include antiparasitic compounds and hormonal preparations (Cardoso et al., 2014; Frederic and Yves, 2014). Data are currently insufficient to determine if long-term exposure to these constituents poses a significant risk to wildlife populations (Taylor and Senac, 2014).

10.5.7.2. Personal-Care Products

Personal-care products are a diverse group of compounds in soaps, lotions, toothpaste, fragrances, and sunscreens, for example. The primary classes of personal care products and associated compounds include disinfectants (e.g., triclosan), fragrances (e.g., musks), insect repellants (e.g., DEET), preservatives (e.g., parabens), and ultraviolet filters (e.g., methylbenzylidene camphor). Unlike pharmaceuticals, which are generally intended for internal use, personal-care products are intended for external use on the body and thus are not subject to metabolic alterations. Data developed thus far indicate that most personal-care products are relatively nontoxic to aquatic organisms at expected environmental concentrations. However, the primary concern involving these substances is their potential to cause estrogenic effects at low concentrations (Brausch and Rand, 2011).

10.5.7.3. Polybrominated Compounds

Polybrominated diphenyl ethers (PBDEs) and polybrominated biphenyls (PBBs) are classes of brominated hydrocarbons, also referred to as brominated flame retardants chemicals. PBBs were formerly used as additive flame retardants in synthetic fibers and molded plastics. PBBs were banned in the United States in 1976. PBDEs are used as flame retardants in a wide variety of products including plastics, furniture, upholstery, electoral equipment, electronic devices, textiles, and other household products. Both PBDE and PBB are structurally like PCBs in that they are hydrophobic and bioaccumulative. PBDEs may enter the environment through emissions from manufacturing processes, volatilization from various products that contain PBDEs, recycling wastes, and leachate from waste-disposal sites (U.S. EPA, 2014b). Concerns about the massive use of these substances have increased due to their possible toxicity, endocrine-disrupting properties, and occurrence in almost all environmental media. The ecotoxicity mechanisms and environmental fate of these compounds are poorly understood (Ezechiáš et al., 2014).

10.5.7.4. Perfluorinated Compounds

Perfluorinated compounds (PFCs), also known as per- or polyfluoroalkyl substances (PFASs), are a class of synthetic compounds containing thousands of chemicals formed from carbon chains with fluorine attached to these chains. PFOA (perfluorooctanic acid) and PFOS (perfluorooctane sulfonate) are two of the best-known. The chemical structure of PFCs gives them unique properties, such as thermal stability and the ability to repel water and oil, making them useful in a wide variety of consumer and industrial products, including non-stick cookware, food packaging, waterproof clothing, fabric stain protectors, lubricants, paint, and firefighting foams. Large volumes of PFCs have been produced since the 1950s. Their high production volume led to widespread distribution in the environment, particularly in water where they are readily transported (U.S. EPA, 2013a).

Several PFCs (i.e., long-chained such as PFOA and PFOS) are considered persistent, bioaccumulative, and toxic (U.S. EPA, 2018). Environmental fate characteristics vary between specific PFC compounds (Giesy et al., 2010 and Valsecchi et al., 2017). Note: the scientific understanding of PFCs is evolving and the person should consult the open literature for the latest information when evaluating these compounds in an ERA.

10.5.7.5. Nanomaterials

Nanoparticles are widely used in commercial products and in industry owing to their small size and interesting properties. Nanoparticle research is currently an area of intense scientific study, due to a wide variety of potential applications in biomedical, optical, and electronic fields. For example, silver nanoparticles are used in food packaging, clothing, disinfectants, and household appliances. Titanium dioxide and zinc oxide nanoparticles are used in sunscreens, other cosmetics, and food products. Kwak and An (2015) collated information from studies on the toxicity of metal- and carbon-based nanomaterials to earthworms in the soil matrix. These researchers found that survival and growth of adult earthworms are negligibly affected by nanomaterials in soil. According to Kwak and An (2015), many studies have reported that nanomaterials may reduce the reproduction of earthworms. In another example, Oberholster et al. (2011) determined threshold concentrations of nanoparticles in spiked sediment had the potential to negatively affect survival and behavior of benthic organisms. These researchers tested α-alumina, γ-alumina, several forms of silica, antimony pentoxide, and superfine amorphous ferric oxide on the sediment-dwelling invertebrate *Chironomus tentans* (Oberholster et al., 2011).

10.6. Documentation of Inputs and Assumptions

In general, an ERA should track the required elements of the agency's ERA process and focus on describing the project-specific approach for addressing those elements. For ease of review and transparency, all dose, HQ, and PCL calculations should be included. These may be presented in tabular form or spreadsheets in a stepwise manner, with each component of the equation listed with its corresponding result. Paramount to this is a clear indication of the values used for the exposure point concentrations (e.g., 95 percent UCL, maximum) for each exposure medium along with all exposure parameters. A reference and rationale (where appropriate) should be given for all receptor exposure assumptions (e.g., home range, preferred habitat, seasonality). Where multiple values or equations for an input are provided in a reference such as those listed in the *Handbook* or the *Combustion* guide, a rationale for the selected input should be made (e.g., selection based on average of adult body weights or selection based on the most appropriate habitat type for the affected property in question).

Changes to the inputs with the second round of HQ calculations (i.e., required element 7) should be explained. As the risk assessment progresses through the refined HQ calculations, tables should reflect what COC and receptor combinations are dropped or retained. Without all this information, it is impossible for the TCEQ to adequately verify the HQs and PCLs determined in the ERA.

11.0 Hazard Quotient Analyses (Required Elements 6 and 7)

This text addresses required elements 6 and 7 from the TRRP rule, which are specific to calculations of the HQ. As discussed below, the methodologies utilized in required elements 6 and 7 result in a conservative assessment (**11.2**) and a less-conservative assessment (**11.3**), respectively.

11.1. Process for Calculating HQs Using NOAEL and LOAEL TRVs

ERAs often exhibit some procedural inconsistencies in the risk calculations regarding these two required elements. HQs using the NOAEL TRV (6) **and both** the NOAEL and LOAEL TRVs (7) are to be calculated according to 30 TAC 350.77(c)(6, 7) of the TRRP rule. Where only one of these TRVs is available from the literature, uncertainty factors presented in this guidance can be used for extrapolation to the other (see **9.2.1.1**).

The initial HQ calculation (required element 6) should use the NOAEL and conservative exposure assumptions. In the refined HQ calculation (required element 7), the exposure assumptions (e.g., home range) can be adjusted and both NOAEL and LOAEL TRVs are used in the estimate. If the results of these calculations indicate that NOAEL HQ > 1 > LOAEL HQ, then the development of PCLs may not be warranted, as the PCL would normally lie between a lower-bound NOAEL-based value and an upper-bound LOAEL-based value. However, even in this event, the justification for not developing a PCL should be based on the strengths and weaknesses of the data and should be discussed in the uncertainty analysis (**12.0**). This guidance outlines the methods for determining a final ecological PCL for a COC that lies between the NOAEL and the LOAEL-based values and should be reviewed before PCL calculation (see **13.4**). If a receptor is a threatened or endangered species (or is a surrogate for a protected species), then the PCL **must** be based on a NOAEL TRV only.

11.2. Conservative Assumptions and HQs

HQs (and HIs, as appropriate) must be calculated for each COC-receptor pair, as identified in required element 6. An HQ reflects the ratio of the predicted exposure to an acceptable exposure, for a specific COC and a specific representative measurement receptor. An HQ (unitless) is calculated as:

$$HQ = Exposure \div TRV$$

where:

Exposure = measured or estimated exposure point concentration (e.g., mg/L, mg/kg) or dose (e.g., mg/kg body weight/day);

TRV = toxicity reference value (e.g., based on a NOAEL or LOAEL) in units matching the exposure-point concentration or dose.

In the risk estimate generated in required element 6, an HQ is based on conservative exposure assumptions (e.g., 100 percent bioavailability and home range) and a representative NOAEL-based TRV (TRV_{NOAEL}). COCs with an HQ (and any associated HI) ≤ 1 are dropped from further evaluation for that measurement receptor for that medium. If all COCs associated with a receptor are eliminated from the Tier 2 SLERA, no further evaluation of that receptor is required. HQ approaches are useful tools in screening-level risk assessments, as a relatively simple and transparent means of deciding which COCs might be carried forward in more detailed evaluations. Often, HQs can be calculated with minimal effort using existing site-characterization data and literature values on toxic effects.

Generally, the risk assessment is purposefully conservative in its measure of exposure and selection of TRVs. However, the TCEQ supports derivation of HQs in required element 6 using the most appropriate TRV_{NOAEL} that can be technically defended. Although this may result in the selection of a TRV that is not necessarily the most conservative, it is important that the selected TRV be the most applicable to the evaluated receptor since the TRV is to remain constant throughout the risk assessment. Selection of the TRV should reflect the concepts discussed in **9.2** regarding TRV selection from multiple studies of differing design, quality, and scope. Often this involves assessments of tests done with surrogate species or differing forms of the COC, tests with differing durations or exposure-dose-response regimes, or tests with different or conflicting results. These considerations should be fully documented in derivation and selection of the TRV values in required element 5 of the Tier 2 SLERA. Once the TRV_{NOAEL} is selected, it should be carried through the Tier 2 SLERA.

Several limitations and cautions apply to using HQs in risk characterization (Allard et al., 2010 and Tannenbaum, 2010). The quotient usually is calculated from point estimates of exposure and toxicity. Thus, no quantitative assessment of risk should be inferred. For example, an HQ based on a point estimate of toxicity such as a TRV_{NOAEL} may indicate the presence of an adverse ecological effect but not its magnitude, because the slope of the dose-response curve is not used. Note that the hazards posed by exposure to a COC do not increase linearly as the HQ increases linearly. Also, an HQ should not be viewed as a statistical value. For example, an HQ of 0.01 indicates that an exposure is 100 times less than the associated TRV, not a 1-in-100 probability of an adverse ecological effect occurring (U.S. ACE, 2010).

Because HQs focus only on individual COCs, they do not represent the potential for toxic effects on ecological receptors from a combination of COCs. Subsection **10.5** gives example classes of COCs where such concerns may arise.

For additive effects, risk assessors sum individual HQs to form a cumulative expression of risk in an HI. The HI is the sum of two or more HQs for different COCs:

$$HI = \sum HQ_i$$

where:

i = COCs with a common toxic mechanism

HIs are calculated as a measure of the potential for impacts of multiple COCs, if the effects are additive. Therefore, this computation is limited to COCs with the same toxic mechanism (i.e., the same mode and site of action). For example, it is not appropriate to add the HQs of two COCs that are both reproductive toxicants if one affects fertilization potential and the other reduces egg production; the risks are independent, and therefore not additive. Note that ecological benchmarks may have been derived from situations where multiple COCs were present (e.g., sediment “effects range—low” values) and from additional conservative generalizations. Consequently, comparisons against ecological benchmarks do not routinely sum different COC concentrations unless the benchmark was explicitly derived for a class of COCs (e.g., TPAHs).

When sufficient information is available, an HI should be calculated for any suite or class of COCs with the same toxicological mechanisms. However, this information is not often available outside of a few well-studied groups of COCs, such as those discussed in **10.5**. Thus, HIs commonly are considered only for a few groups of COCs known to act through a common toxic mechanism for common test species. For example, HIs are often appropriate for PCBs, chlorinated benzenes, dioxins and furans, and TPAHs. Alternatively, if toxic equivalency factors are used to combine exposures within a class of COCs (e.g., dioxins, furans and dioxin-like PCBs), then the HQ of the surrogate already represents the cumulative exposure for the whole class adjusted for constituent-specific toxicity potency, and calculation of an HI is not appropriate. Where evidence exists that these groups of COCs do not act through a common toxic mechanism, the assumption of additivity may be dropped if the technical justification can be defended and the rationale discussed in the uncertainty analysis (see **12.0**).

HQs and HIs should be calculated using reasonable and representative exposure estimates for the appropriate media and habitats considered for measurement receptors. In most cases, the representative value will be an estimate of the mean exposure concentration. TRRP-15eco should be consulted for a discussion of the exposure point concentrations. For any COCs with the same toxic mechanism, the corresponding HI must also equal 1 for their elimination from further consideration in the Tier 2 SLERA.

11.3. Less-Conservative Assumptions and HQs

Required element 7 of the Tier 2 SLERA [30 TAC 350.77(c)(7)] allows for calculation of HQs using less-conservative exposure assumptions and TRVs based on data from both NOAEL and LOAEL effects. Applicable exposure variables for the refined risk estimate generally consist of EMFs for home range, seasonality, or bioavailability. These variables should be less conservative in their totality, and the person must justify the use of such data based on site-specific information or some other clear rationale. Additional discussion appears in **10.4.5**.

If comparison of a less-conservative exposure estimate with a NOAEL-based TRV results in an HQ (and any HI) ≤ 1, the COC may be dropped from further evaluation in the Tier 2 SLERA. The person may propose to drop COCs with HQs or HIs derived from technically defensible, LOAEL-based TRVs that are < 1 provided that supporting information is included in the uncertainty discussion (**12.0**). HQs > 1 based on less-conservative exposure assumptions and LOAEL-based TRVs form a reasonable basis for beginning remedial planning, as ecological impacts may be expected. In lieu of initiating response actions based on Tier 2 evaluations, COCs with HQs > 1 (and any associated HI) may be carried forward to Tier 3 for more site-specific risk evaluations. Consultations with the TCEQ are recommended at this stage to determine a course of action based on the identified risk drivers.

Adjustments of exposure estimates in required element 7 based on differing assumptions or new data are considered a less-conservative approach [30 TAC 350.77(c)(7)]. These adjustments are appropriate and allowed in the TRRP with sufficient justification; however, the level of acceptable risk does not change. That is, the decision point is still whether the HQ exceeds 1 for the specified measurement receptor. The less-conservative, yet reasonable, exposure adjustments focus the screening-level effort and avoid more complicated and costly evaluations often associated with the higher-tier risk assessments. As stated in the TRRP rule, less-conservative HQs present a more representative or certain estimate of exposure or hazard based on site-specific considerations.

12.0 Uncertainty Analysis (Required Element 8)

After calculating the HQs in required element 7 and analyzing the results of the risk assessment, the person will need to evaluate the uncertainty associated with the ERA in required element 8. The results of an ERA are influenced to some degree by variability and uncertainty that should be considered when interpreting these results. Major sources of uncertainty include natural variability and incomplete knowledge of the site-specific biological processes and fate and transport mechanisms. The person should carefully consider the impact of the uncertainty on the risk conclusions and recommendations.

12.1. Using the Uncertainty Analysis Properly

The nature of the uncertainties should be clearly summarized in the SLERA (or SSERA, as appropriate). Uncertainty analysis can be used to justify the need for calculating or not calculating a PCL for a given COC (required element 9), considering indications of potential ecological risk in context with the likelihood of that risk. Factors to be evaluated include the location and extent of the COCs, the degree to which the TRV is exceeded, and the expected half-life of the COCs in the environment. If, after completing the HQ exercises in **11.0** for a COC, the NOAEL HQ or HI > 1 but the LOAEL HQ or HI < 1, the person may state in the uncertainty analysis that no PCL is necessary for that COC. This is justified because, ideally, any potential remediation of a medium would be to a PCL that is bounded by those two effect levels. However, justification is required when the LOAEL HQ or HI approaches unity and there are indications that risk may have been underestimated in other areas, or there was uncertainty in the assessment in general (e.g., elevated detection levels, limited number of samples, inadequate determination of nature and extent). PCL calculations for a given COC can be justified qualitatively or quantitatively, based on strengths and weaknesses in the data. In most cases, the uncertainty analysis will be qualitative.

More traditional sources of uncertainty include:

1. uncertainty in the conceptual site model (CSM)
2. uncertainty in the parameters used to evaluate risk
3. uncertainty in the models used to interpret risk
4. stochasticity (natural variability)

Uncertainty in the development of the CSM may be one of the most important sources of uncertainty in the entire ERA process. If relationships between sources and receptors are missing or incorrectly identified in the CSM, then risks could be underestimated or overestimated. Proper CSM development can help reduce this source of uncertainty. Sources of uncertainty can also often be reduced by collecting additional data, when practicable.

Usually, the primary function of the uncertainty analysis is to describe the potential for underestimation or overestimation of risks. This element of the discussion should not be omitted for COCs and exposure pathways retained at required element 6 and beyond. The uncertainty analysis serves other purposes in the context of the TRRP ERA requirements. As previously stated, the TRRP rule specifies that the uncertainty analysis, when used properly, can be used to justify the need for calculating **or not** calculating a PCL for a given COC-and-receptor pair. In any event, the uncertainty analysis should not be used to dismiss the need for a PCL where conservative assumptions are not adjusted throughout the risk assessment, such that the HQ calculations are inflated. Rather than carry this over-conservatism into the discussion of uncertainty analysis, the TCEQ prefers that the exposure assumptions available for adjustment be modified in required element 7 such that the results are best estimates of either acceptable risk or risk that requires a risk management decision. HQs or HIs greater than 1 based on less-conservative exposure assumptions and LOAEL-based TRVs are a reasonable basis for developing PCLs for consideration in remediation planning, as ecological impacts may be expected.

12.2. Uncertainties in Hot-Spot Analysis

As discussed in TRRP-15eco, the purpose of a hot spot evaluation is to identify potential risks to receptors that would not be identified and mitigated through the standard risk evaluation, which is based on a statistical representation of the data (i.e., using a 95 percent UCL as the EPC). The standard ERA evaluates potential risks associated with COC concentrations over an exposure area larger than a hot spot based on the assumption that exposure will be equally distributed across an area. An evaluation of hot spots or their nonexistence, is now recommended in each SLERA. A hot spot evaluation should be discussed in the uncertainty analysis. TRRP-15eco should be consulted for determining when it is appropriate to evaluate hot spots relating to benthic invertebrates, fish, aquatic life, and wildlife.

Because the determination of hot spots is based on site-specific conditions, there can be numerous areas of uncertainty. For example, soil contamination can be highly variable. Discrete soil samples from locations near one another can be expected to give considerably different results when tested for the same compound. Subsamples from a container of soil brought to the laboratory can yield significantly different results. Even split samples of the same few grams of soil can give very different analytical results (Hadley and Mueller, 2012). The person should consider site history, the CSM, fate and transport mechanisms of COCs, ecological resources at the site, potential receptors at the site, and potential routes of exposure.

12.3. Limitations and Uncertainties Typical of a SLERA

The selection of measurement receptors is partially based on the premise that if key components of the ecosystem are protected, protection will be conferred to populations and, by extension, communities and the ecosystem. However, the representative species may not be the most sensitive to some COCs, but may have been chosen as a function of their ecological significance and the availability of information on natural history and toxicology.

The toxicity of COCs varies with the measurement receptors and with the availability and form of a given COC. Availability and chemical form are affected by factors such as pH, temperature, alkalinity, seasonal variation, microbial activity, organic carbon content, and complexation with other COCs. In the SLERA, bioavailability of COCs is assumed to be like that observed in the toxicity studies reported in the literature. Thus, toxicity may be overestimated or underestimated, depending in part on the extent to which site-specific COC bioavailability differs from those in studies reported in the literature.

Attempts to quantify and correct for uncertainty resulting from the use of surrogate species are common, but controversial. Calabrese and Baldwin (1993) discuss the use of UFs to adjust for extrapolations among taxa, between laboratory and field responses, and between acute and chronic responses. Wentsel et al. (1996) also describes the use of UFs. These multipliers are expected to adjust for differences in responses among taxa resulting from differences in physiology and metabolism. When extrapolating from laboratory to field settings, important considerations are differences in physical environment, organism behavior, and interactions with other ecological components. Extrapolation between responses may be necessary, particularly when data on relevant endpoints are not available (most commonly when extrapolating from LOAEL to NOAEL TRVs). The net effect of UFs on the accuracy of the SLERA depends on the accuracy of assumptions.

The SLERA typically uses some default parameter values in place of site-specific measured data, incorporating assumptions because of data gaps. The absence of site-specific information and the need for these assumptions may result in uncertainty in calculating HQs. After identifying the major uncertainties associated with the SLERA results, their significance to the computed HQs should be evaluated. Uncertainties that generally should be evaluated in a SLERA include:

- Uncertainties in the site assessment.
- Use of non-detected results.
- Site-specific representativeness of food webs.
- Exposure potential of the measurement receptors.
- Representativeness of exposure assumptions for measurement receptors.
- Extrapolation of effects studies to measurement receptors.
- Effect of COC physicochemical properties on fate and bioavailability.
- Effect of site-specific environmental conditions affecting the fate, transport, and bioavailability of the COCs.
- An assumption that a measurement receptor does not metabolize or eliminate a COC.

- Potential risk to measurement receptors or communities from COCs with no TRVs or effects data.
- Use of samples collected from non-ecological habitat.

13.0 Development of Ecological PCLs (Required Element 9)

Ecological PCLs must be calculated for each COC that has not been eliminated from consideration under required elements 1, 6, 7, or 8 of the Tier 2 SLERA [see 30 TAC 350.77(c)]. The ecological PCL must be protective of wide-ranging ecological receptors that may live on or frequent the affected property in search of food and, where appropriate, benthic invertebrate communities within the waters in the state. The ecological PCL is not directly intended to be protective of on-site receptors with limited mobility or range. Text in **13.1** discusses receptors with limited mobility or range.

Since exposures for community-level receptors such as fish and benthic macroinvertebrates is generally expressed in terms of media concentrations, any PCLs related to such receptors are based on a simple comparison of representative media concentrations to applicable TRVs. For wildlife receptors where exposure may be due to ingestion of impacted food or media, **13.2** describes techniques for deriving media-specific PCLs. For comparison purposes, the derivation of PCLs from the PCL Database is discussed in **13.3**. The ERA should be conducted in a manner that results in the protection of ecological receptors subject to management by other federal and state agencies and consistent with those agencies' statutory authority. For each COC not eliminated from consideration under required elements 1, 6, 7, or 8, a medium-specific PCL bounded by the NOAEL and LOAEL is calculated for each relevant measurement receptor. Guidelines for deriving ecological PCLs bounded by the NOAEL–LOAEL range appear in **13.4**.

13.1. Ecological PCL Definition and Small-Ranging Receptors

Regarding the definition of an ecological protective concentration level, the TRRP rule [30 TAC 350.4(a)(27)] states:

> These concentration levels are primarily intended to be protective for more mobile or wide-ranging ecological receptors and, where appropriate, benthic invertebrate communities within the waters in the state. These concentration levels are not intended to be directly protective of receptors with limited mobility or range (e.g., plants, soil invertebrates, and small rodents), particularly those residing within active areas of a facility, unless these receptors are threatened or endangered species or unless impacts to these receptors result in disruption of the ecosystem or other unacceptable consequences for the more mobile or wide-ranging receptors (e.g., impacts to an off-site grassland habitat eliminate rodents which causes a desirable owl population to leave the area).

Although it is clear what is meant by "plants" and "soil invertebrates," generally there has been much confusion over the implementation of the remainder of this definition regarding wildlife receptors. Over the years, ERAs submitted to the TCEQ have presented a variety of approaches and interpretations as to which wildlife receptors do not require PCL development. In general, *small-ranging receptors* are those with a home range less than or equal to 1 hectare (approximately 2.5 acres), and *wide-ranging receptors* are those with a home range greater than 1 hectare.

The preceding definition of "ecological PCL" is based on numerous TRRP stakeholder meetings held before the initial rulemaking. At these meetings, many stakeholders made it clear they wanted to avoid situations where ecologically- based cleanups at affected properties would be for the protection of earthworms, rats, or other "nuisance species." However, *nuisance species* means different things to different people. For instance, some property owners may consider grackles, swifts, swallows, snakes, toads, raccoons, rabbits, deer, foxes, skunks, and coyotes as nuisances. For the purposes of the TRRP and this guide, *nuisance species* is narrowly defined as those rodent species commonly controlled for protection of human health and property.

The TCEQ asserts that the ecological PCL exclusion is only applicable to some rodents and should not be applied to any receptor that is a state-listed or federally listed protected species, or any measurement receptor that was evaluated as a surrogate for such species. This rodent exclusion, as a matter of policy, is based on the stakeholder feedback during the TRRP rule development. When conducting an ERA under the TRRP process, birds, reptiles, amphibians, and non-rodent mammals (including shrews) **do not** qualify for exclusion from ecological PCL development. All rats, mice, voles, squirrels, gophers, chipmunks, and nutria are rodents that may qualify for PCL exclusion.

When a qualifying rodent (e.g., deer mouse, cotton rat) is selected as the measurement receptor for the herbivorous or omnivorous mammal guild in an ERA, it represents the entire feeding guild, including receptors that do not qualify for PCL exclusion. If it can be shown that the qualifying rodent is a good guild representative and is the only member of the guild at the affected property that could be at risk, then PCL development will not be necessary for that guild, unless the rodent is a keystone species, as described below. However, under no circumstances can the rodent be eliminated as prey from the calculations of food chain exposure.

Keystone species are crucial in maintaining the organization and diversity of their ecological communities and they are exceptional, relative to the rest of the community in their importance (Paine, 1969). Removal of a keystone species may lead indirectly to the loss of other species in the community, as exemplified by the rodent-and-owl scenario in the ecological PCL definition.

Keystone species can exert effects through consumption, competition, mutualism, dispersal, pollination, disease, and by modifying habitats (ecosystem engineers) (see Power et al., 1996 and Jones et al., 1994). Rodents that are considered keystone species in Texas include, among others, the black-tailed prairie dog (see Hoogland, 2006) and the American beaver. Site-specific circumstances may dictate that other rodents that would normally qualify for PCL exclusion be considered keystone species, particularly when they are a primary food source that is significantly diminished.

Before proposing to eliminate a rodent from PCL development, the person should evaluate the factors below to determine if the selected rodent measurement receptor could be considered a keystone species or if the guild it represents would still be protected. As appropriate, some combination of the topics presented in the bullets below should be discussed in the uncertainty analysis of the SLERA.

- Distribution of the impacted area: What is the distribution of the contamination at the affected property relative to the home range of the rodent? Is there sufficient adjacent, unimpacted habitat that a rodent population could use relative to the home range of the rodent, or are they restricted to a narrow corridor between impacted areas?

- Demonstration of protectiveness of the guild: Is there a more representative guild member that could be evaluated (e.g., has smaller body weight, is more sensitive) that would better ensure that the guild is protected?

- Exceedance of the LOAEL HQ: Is the perceived risk specific to the rodent measurement receptor? Are other members of the guild (e.g., rabbits, deer, and livestock) not at risk because of their larger home range?

- Site information: Is there indication that the rodent population is likely to be diverse and thriving at the affected property? Are there predators that rely primarily on rodents from the affected property as a food source?

- Sample location: Does the exposure concentration reflect samples collected primarily from ecological habitat?

13.2. PCL Calculation Method

With multimedia exposure (e.g., sediment, soil and water), there is no single set of valid PCLs since their derivation requires solving one equation (the general dose equation discussed in **10.4**) for multiple unknowns (media-specific PCLs). The following text describes a method for deriving multimedia PCLs, although other methods may be used.

The most common method for deriving media-specific PCLs for wildlife measurement receptors starts with the wildlife dose equation presented in **10.4**, with the equation rearranged to solve for a single medium-specific PCL (e.g., soil) while COC concentrations in other media are held constant. This may be accomplished by using a background value, a surface water criterion, an ecological benchmark, or the lowest value in the data set. For example, if a receptor has exposure to all three media and the soil PCL is to be determined, the concentrations in water and sediment would be held constant and the person would solve for the soil concentration as the PCL.

$$\mathrm{Dose_{oral}} = \frac{[(\mathrm{IR_{food}} \times \mathrm{C_{food}}) + (\mathrm{IR_{water}} \times \mathrm{C_{water}}) + (\mathrm{IR_{soil}} \times \mathrm{C_{soil}}) + (\mathrm{IR_{sed}} \times \mathrm{C_{sed}})]}{\mathrm{BW}}$$

As described in **11.2**, an HQ compares exposure (e.g., dose) to a TRV:

$$HQ = \frac{\text{Exposure}}{\text{TRV}} = \frac{\text{Dose}}{\text{TRV}}$$

The derivation of a PCL requires the HQ to equal 1.0. If the COC concentration in the primary food or prey of the wildlife receptor has not been measured (as would typically be the case for Tier 2 SLERAs), then the COC concentration in food (C_{food}) can be predicted from the applicable media concentration and an uptake factor as discussed in **10.4.1**. For the purposes of this example, C_{food} is represented solely in terms of C_{soil} (i.e., PCL_{soil}) and an applicable uptake factor. The resulting equation is solved for PCL_{soil}:

$$PCL_{soil} = \frac{TRV \times BW - [(IR_{water} \times C_{water}) + (IR_{sed} \times C_{sed})]}{[(IR_{food} \times \text{Uptake Factor}) + (IR_{soil})]}$$

The PCL_{soil} term will reflect the NOAEL PCL if the NOAEL was used as the TRV, or the LOAEL PCL if the LOAEL was used as the TRV; however, the TRRP rule requires that both PCLs must be calculated.

The PCL calculations may incorporate any EMF used in the Tier 2 or Tier 3 HQ calculations such as AUF or EF (**10.4.5**). PCL values can be adjusted as shown below:

$$PCL_{soil\text{-}adjusted} = PCL_{soil} \times \frac{1}{EMF}$$

13.3. Ecological PCL Database

In the PCL Database, receptor-specific and COC-specific data are compiled into an equation that accounts for the toxicity of the COC and exposure of the receptor. However, a slightly different format is used to calculate PCLs. Ingestion rates have been normalized for body weight and the default PCLs do not reflect an EMF adjustment; however, EMFs can be manually inserted into the calculation of the refined PCL using the methodology described in **13.2**.

For each COC-receptor pair, the PCL is calculated as:

$$PCL_{soil/sediment} = \frac{TRV}{[(BAF \times FIR) + SSIR]}$$

Where:

$PCL_{soil/sediment}$ = the protective concentration level for soil or sediment (mg/kg dry weight)

TRV = the toxicity reference value of the chemical (mg/kg-day)

BAF = bioaccumulation factor

FIR = food ingestion rate (kg/kg BW-day)

SSIR = soil or sediment ingestion rate (kg/kg BW-day)

A step-by-step development of PCLs is as follows:

Go to the PCL Database at <pcl.wtamu.edu/pcl/login.jsp>

For the initial use of the PCL Database, please use the "Register Now!" button and then login as "Guest" until your registered login is approved. (You will receive an e-mail.)

After logging in, the "PCL Calculator" page appears. Select the appropriate habitat (see ***6.2****) for your site (only one per calculation) from the drop-down list after selecting the "Habitat" radio button, or select the "Species" radio button, and hold down the control key to select multiple species.*

Choose the chemical.

Click on the "Next" box.

Allow the PCL analysis to run.

Scroll down the "Analysis" page and look for the number outlined by a box under the "Conservative PCL" column. This value represents the lowest PCL and does not reflect any site-specific adjustments. Use this PCL as the ecological assessment level if it is lower than the soil or sediment benchmark; see ***2.1*** *for additional discussion on assessment levels. Note that, if multiple habitats are present at the site, you will need to identify the lowest conservative wildlife PCL among those applicable habitats for the COC to determine the assessment level. If the EPC is at, or lower than, this value, then no wildlife (including protected species) are at risk and the COC can be eliminated from further evaluation.*

Find the "Average TRV PCL" column and look for the lowest value. This PCL is based on the average of the NOAEL and LOAEL TRVs without any exposure adjustments. If the site EPC is at or lower than this PCL, that COC can be eliminated, if protected species have been addressed via application of EMFs. To include site-specific information for a species home range or seasonality, enter these values in the white boxes to the right of the "Average TRV PCL" column. The modified PCL will appear under the column "Refined PCL." If the site EPC is at or lower than this PCL, that COC can be eliminated, provided protected species have been addressed.

As noted, if more than one habitat is present at the site, you will need to identify the lowest average wildlife PCL among those applicable habitats for the COC.

Click on the "Export" box to export to Excel or to print (registered users only).

Click on the "Close Analysis" box to return to the PCL calculator page to evaluate another COC.

13.4. Selection of Comparative and Final Ecological PCLs

Under 30 TAC 350.77(c)(9), the person must develop medium-specific PCLs bounded by the NOAEL and the LOAEL before choosing a final PCL. These *comparative PCLs* are developed for each remaining COC associated with each relevant measurement receptor for a medium and, where appropriate, for the medium itself in the case of benthic invertebrates and aquatic life. The final ecological PCL[24] for a COC in a medium should be the lowest of the comparative PCLs and, except as discussed below, should lie between the NOAEL and the LOAEL for the most susceptible measurement receptor or community. Because the ERA process allows for realistic exposure assumptions (and incorporation of any site-specific data) before PCLs are developed, the TCEQ can be reasonably confident that any COC with a LOAEL HQ ≥ 1 (resulting from the exercise in required element 7) may pose unacceptable ecological risk. This also means that remediation to a LOAEL-based PCL (derived from realistic exposure assumptions) may result in a continuing unacceptable ecological risk scenario. As discussed below (see item 5), the rationale for selecting a comparative PCL that is skewed toward the NOAEL-based or LOAEL-based PCL should be made in the uncertainty analysis.

To allow flexibility, the TRRP rule is intentionally silent on how to select a comparative ecological PCL that is bounded by the NOAEL and LOAEL. However, the TCEQ has developed guidelines to assist in this determination that may be adapted to site-specific circumstances. Generally, when establishing comparative ecological PCLs for the relevant measurement receptors and COC pairs, the guidelines below should be followed. Guidelines 1 and 2 apply to wildlife, guidelines 3 through 6 apply to benthic invertebrates in sediment, guideline 7 applies to aquatic-life exposure to surface water (and groundwater), and guideline 8 applies to the development of the final ecological PCL.

1. For wildlife, the average between the NOAEL- and LOAEL-based PCLs for a COC in a specific medium can be used as the comparative PCL, provided that the NOAEL and LOAEL do not differ by more than a factor of 10. For example, if the NOAEL-based PCL for a COC for a measurement receptor was determined to be 12 mg/kg and the LOAEL-based PCL was 60 mg/kg, then the average (i.e., comparative PCL) would be (12 + 60) mg/kg ÷ 2 = 36 mg/kg. Selection of this average value does not require any further justification other than a statement identifying the comparative PCL as the average of the NOAEL- and LOAEL-based PCLs.
2. One exception to having the comparative PCL lie between the NOAEL- and LOAEL-based PCLs occurs whenever a protected species is potentially at risk. In this case, the NOAEL-based PCL (corresponding to the Conservative PCL in the PCL Database) should be chosen as the comparative PCL because the TCEQ will not approve a risk

[24]The final ecological PCL should not be confused with the critical PCL. The critical PCL is the lower of the human-health PCL and the final ecological PCL for a COC within a specific medium. See additional discussion of the critical PCL in **14.1** and in TRRP-25 (Critical PCLs, TCEQ, 2009).

management decision to leave COCs in place at concentrations likely to cause adverse effect on individuals of a protected species (i.e., greater than the NOAEL-based PCL).

3. A sediment PCL may be necessary to preclude risks to a benthic invertebrate community potentially harmed by a release. As stated before, the sediment benchmark concentrations (in the Benchmark Tables) should not be equated with cleanup goals because of their conservatism. Rather, a midpoint PCL may be used as described here. First, for the sediment benchmarks there is usually a corresponding second effect level value from the same source (see the sediment benchmark table). For example, some of the primary benchmarks used for freshwater sediment are the threshold effect concentrations from MacDonald et al. (2000). The second effect levels from this same source are called *probable effect concentrations* (PECs). The TEC is intended to estimate the concentration for a given COC below which adverse biological affects rarely occur. The PEC is intended to represent the concentration for a given COC above which adverse biological affects frequently occur. For the development of benthic midpoint PCLs, the TCEQ recommends using the sediment benchmarks as NOAELs, and the second effect levels as LOAELs. The average of the two values becomes the midpoint PCL. For instance, the TEC for copper in freshwater is 31.6 mg/kg; the PEC is 149 mg/kg. The average to be proposed as the comparative PCL would be $(31.6 + 149) \div 2 = 90.3$ mg/kg. The midpoint value becomes the default PCL, with one notable exception (see 4).
4. Usually the person may propose the midpoint as the comparative PCL without further justification. However, the midpoint will not be considered the default when the weight-of-evidence suggests the value is not protective of the benthic community. Continuing with the copper example in item 3, if a more relevant (e.g., local) study from the literature showed that sediment copper concentrations above 80 mg/kg caused unacceptable impacts to the benthic community, then the midpoint value of 90.3 mg/kg would not be suitable. Here, the TCEQ would require a PCL lower than the default to be protective of the benthic community.
5. Continuing with the sediment PCL protective of benthos, the person may propose a value different from the default midpoint concentration. Here the TCEQ suggests that the person evaluate the individual studies that comprise the effects-level databases, find the most applicable study, and recommend a PCL that is more suited to the actual circumstances (e.g., affected species, sediment composition) of the affected property. Of course, such a recommendation will need adequate justification and documentation.
6. Finally, the person may also choose an EqP method for developing a benthic PCL. Based on the inherent uncertainties, it is recommended as an alternative method only where data gaps necessitate this method. The person proposing an EqP-based sediment PCL should clearly discuss the uncertainties, including the gaps in available

benthic invertebrate toxicity information or sediment screening values that preclude derivation of a default midpoint PCL.

7. The derivation of PCLs for aquatic life (water column receptors) does not parallel the derivation of PCLs for other media (e.g., benthic receptors in sediment) where a range is determined. Surface water PCLs are point values representing the TSWQS, values derived in accordance with the TSWQS, or federal criteria. Here, it is not appropriate to use the midpoint between the acute and chronic values.
8. The final media-specific ecological PCL for a COC should be the lowest concentration among the comparative PCLs (corresponding to the Refined PCL in the PCL Database) determined for each relevant measurement receptor and the benthic community, where appropriate. Accordingly, the measurement receptor or benthic community requiring the lowest comparative PCL is considered the most susceptible for that medium. Cost and remediation technology should never be factored into the determination of the final ecological PCL. These are risk-management considerations.

As stated previously, application of these guidelines listed above may vary between affected properties. This list is **not** comprehensive. The person should rely on site-specific circumstances and the availability of toxicological data for selection of comparative and final ecological PCLs.

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14.0 Ecological Risk Management (Required Element 10)

The Tier 2 SLERA concludes with required element 10, a recommendation on how to manage ecological risk at the affected property. If all COCs and pathways have been eliminated by this point, there is no unacceptable ecological risk at the affected property. However, if ecological PCLs were calculated in Tier 2, the person must do one or more of the following:

- Proceed to additional risk assessment under Tier 3 to develop site-specific ecological PCLs or to determine that there is no apparent unacceptable ecological risk at the affected property.
- Compare the PCL values generated in Tier 2 to relevant levels protective of human health (e.g., values generated from a baseline risk assessment, or TRRP human health PCLs generated at any tier) to determine the critical PCL and remediate to those levels.
- Evaluate and state whether the human health remedy would eliminate all ecological exposure pathways.
- Request permission to conduct an ecological services analysis (**14.3**).

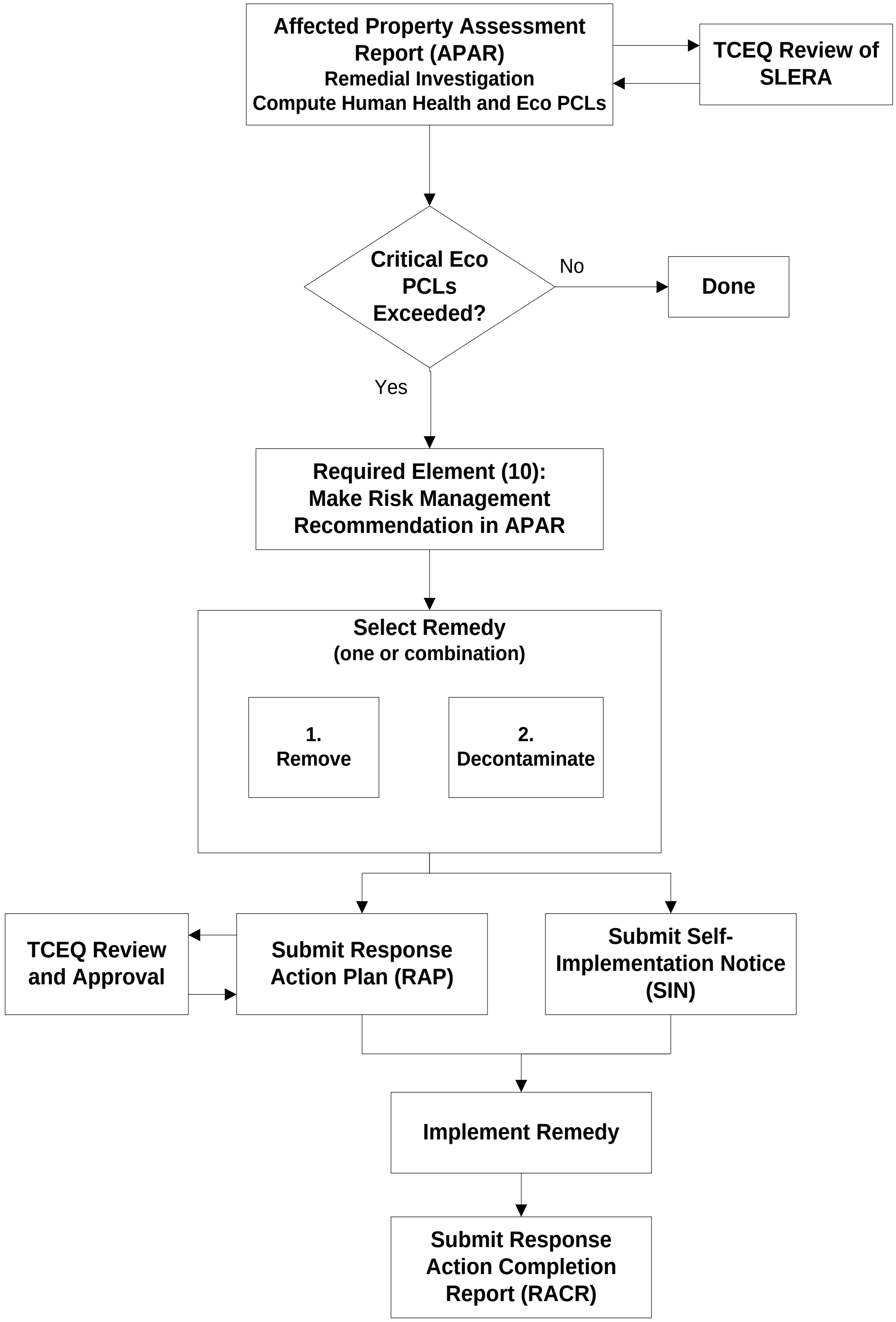
Other management strategies may be possible, but the ecological risk management recommendation must describe an action that will address any exceedances of ecological PCLs.

14.1. Risk Management under the TRRP Rule

As a component of required element 10 of the Tier 2 SLERA, the person must make risk management recommendations for the affected property if the affected medium contains COCs at concentrations greater than applicable PCLs. Decisions about ecological risk management can be made using either the RRR or the TRRP (but not both). The RRR standards or the TRRP Remedy Standards (A or B) must include protection of ecological receptors as a remedy goal. This document focuses on implementation of the TRRP, although if the RRR is the applicable rule then this guidance still applies. The person should contact the TCEQ for more information on coordination of the ERA risk management process with the RRR.

Under TRRP, risk management recommendations are confined to the response options available under Remedy Standard A (Figure 14.1) or B (Figure 14.2). The remedy must address both human health and ecological exposure. For this purpose, human health-based and ecological PCLs are compared to identify critical PCLs (i.e., the lowest concentration levels) for each COC and affected medium, and the remedy is directed toward addressing critical protective concentration level exceedance (PCLE) zones. The remedy is complete when either Standard A or Standard B response objectives have been achieved, the TCEQ has approved all requisite reports, and any necessary post-response care has been performed and financial assurance is maintained (see 30 TAC 350.34 and 30 TAC 350.91–96).

It is the responsibility of the person to select the appropriate remedy, and, if selecting Remedy Standard B, to submit a response action plan for review and approval by the TCEQ.



*Optional unless required by another agency rule, permit, or order.

Figure 14.1. Remedy Standard A for ecological exposures.

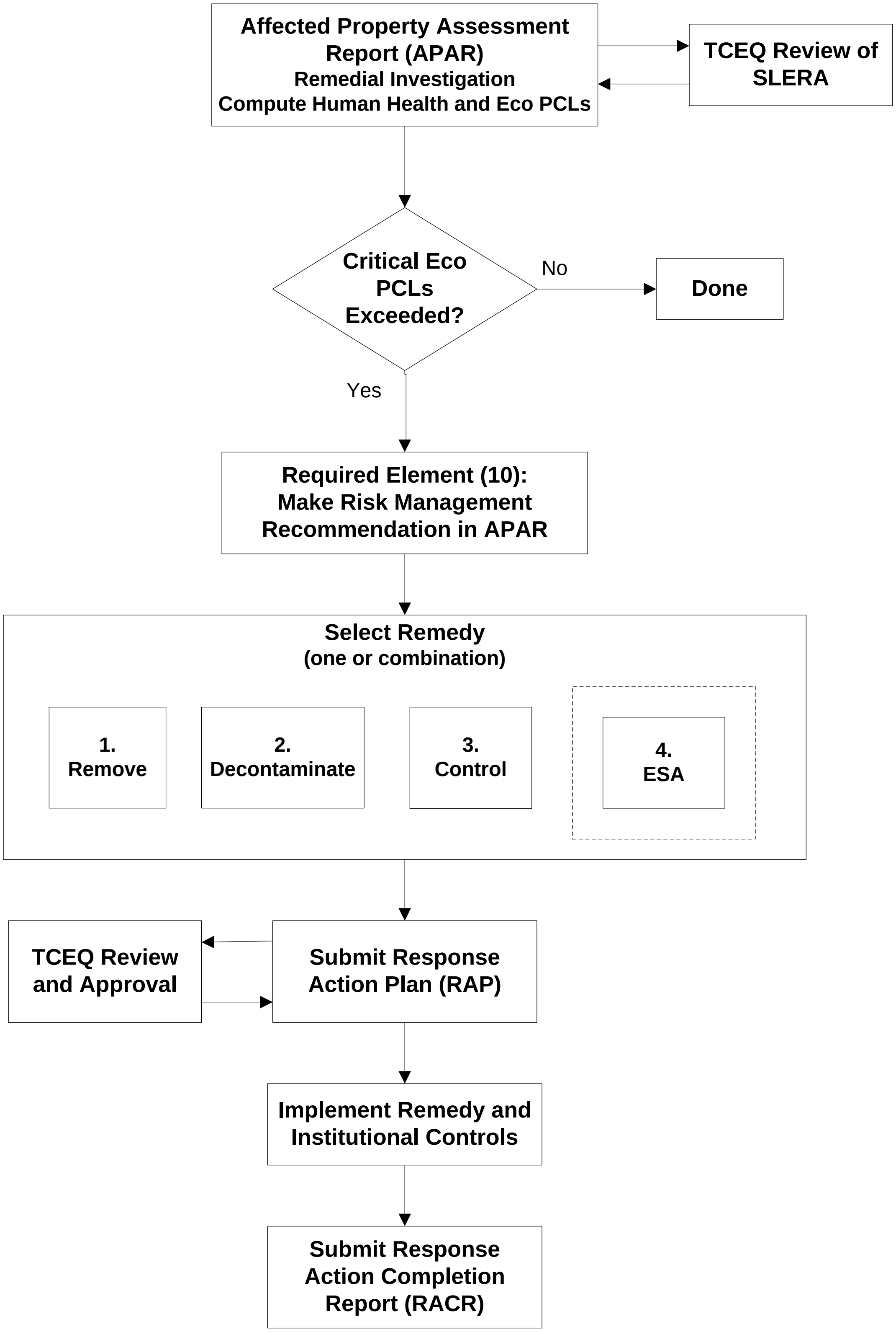


Figure 14.2. Remedy Standard B for ecological exposures.

For each COC where the ecological PCLs are determined to be the critical PCL and the corresponding media concentration of that COC exceeds the critical PCL, the person must consider the need for further assessment (i.e., a Tier 3 SSERA) or select one or a combination of the available remedies available under the TRRP rule.

The person may propose a remedy for managing the PCLE zone. The specific remedy options may be summarized as (see the full presentation in 30 TAC 350, Subchapter B):

- Removal (Remedy Standards A or B) of media with concentrations of COCs that exceed the critical PCLs. Examples include excavation or dredging and subsequent placement or disposal in a manner protective of ecological risks. Waste or affected environmental media must be removed and taken to another location.

- Decontamination (Remedy Standards A or B)—meaning a permanent and irreversible treatment that eliminates concentrations of COCs that exceed their respective critical PCLs. Examples include biological degradation (natural and enhanced), chemical fixation or sequestration, detoxification, or natural attenuation. (Other remedies are possible.)

- Control (Remedy Standard B only) considers physical or institutional controls that prevent the exposure of ecological receptors to concentrations of COCs that exceed their respective ecological PCLs. Examples include capping an affected area with an impermeable or semi-impermeable cap or layer (e.g., concrete, clay, geotextiles), physical containment of potential sources of COCs, or institutional controls that establish requirements for maintenance and housekeeping at the property (e.g., remedy-related requirements, habitat maintenance in commercial or industrial areas to ensure appropriate ecological protection is achieved).

- Ecological services analysis (Remedy Standard B only) considers the potential impacts of the remediation as well as risks associated with exposure to COCs that exceed their respective PCLs. Involves an analysis of the ecological service flows associated with options that include, but are not limited to, natural attenuation and partial or full-scale remedial actions. In certain cases, ecological restoration or a combination of actions may be used to compensate for potential losses of ecological services associated with a selected remedy. This option must be conducted whenever concentrations of COCs that exceed ecological PCLs are proposed to be left in place.

For practical purposes, there is little technical difference in removal or decontamination under Standards A and B. The key technical differences deal with assumptions based on calculations of human health risk (e.g., “assume direct contact, no lateral transport”). In application, though, there are many differences between Standards A and B. First, Remedy Standard A is self-implementing. The person need only submit a self-implementation notice to the

TCEQ before initiating remediation. Second, post-closure care, institutional controls, and possibly financial assurance[25] are required for a Standard B response action.

Neither of those requirements affects Remedy A. However, when submitting a SIN for undertaking a remedy that addresses ecological PCLs, it is recommended that the person consult with the TCEQ and gain approval of the Tier 2 or 3 ERA before initiating the remedy. Otherwise, there is the risk of being required to perform additional response work or assessment if the TCEQ disagrees with or disapproves the assumptions or calculations made in the Tier 2 or 3 ERA.

In recommending a specific remedy (per required element 10), the person should consider existing background levels (if not previously considered); current and likely future land uses; current and likely future resource uses in the area; the local, regional, and national ecological significance of the affected property; and the potential impacts of available response actions, including the impacts associated with leaving COCs in place.

When human health PCLs are exceeded, and if human health risks demonstrably are minimal and a human health–based response action would have a "significant and highly disproportionate effect" on ecological receptors, the person may propose that the response not be performed [see 30 TAC 350.33(a)(3)]. In addition to those options, the TCEQ may require some form of post-response sampling or monitoring.

14.2. Ecological Services Analysis

As stated in the TRRP rule, after the ecological risk has been quantified, PCLs established, and the ecological PCL determined to be the critical PCL (i.e., the risk driver) or the only PCL, the person may act to remove, decontaminate, or control contaminated media and COCs. However, to afford additional flexibility where concentrations of COCs do not exceed human health-based levels[26] (either before or after a response action) but do exceed ecological PCLs, the TCEQ allows an ESA to be conducted, as described below and at 30 TAC 350.33(a)(3)(B).[27] The performance of the ESA and any required compensatory ecological restoration must be done in cooperation with and approval from the Natural Resource Trustees for Texas, including the TCEQ, the TPWD, the Texas General Land Office, the U.S. Department of Commerce represented by the National Oceanic and Atmospheric Administration (NOAA), and the U.S. Department of the Interior (U.S. DOI represented by the U.S. FWS), hereinafter collectively referred to as the "Trustees." Additional information on the various Trustee programs, including their legal authority, can be reviewed at <www.tceq.texas.gov/goto/nrtp>. The case study includes an example ESA (see **1.3.8**).

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[25] Financial assurance is only required if a physical control is used.

[26] Except as allowable under 30 TAC 350.33(a)(3).

[27] According to 30 TAC 350.33(a)(3)(B), an ESA must be conducted whenever concentrations of COCs that exceed ecological PCLs are proposed to be left in place with the potential for continuing exposure.

The ESA considers the present and predicted ecological services of the affected property, as well as beneficial and detrimental effects on services associated with potential responses to address residual ecological risks. Furthermore, where appropriate and based upon the results of the ESA, a plan for compensatory ecological restoration may also be combined with some type of response action (e.g., hot-spot removal, monitored natural attenuation) for the affected property. Compensatory ecological restoration provides or restores alternative services when a response at the affected property is likely to cause additional, unwarranted risks to ecological receptors.

The key tenets of the ESA are:

1. impaired habitats can provide valuable ecological services (e.g., food source, breeding, rearing, shelter),
2. the “environment” is an ecosystem that extends beyond the perimeter of an affected property, and
3. reduction in habitat services in one location can be addressed by increased services elsewhere in the same ecosystem.

The advantage of this option is a net environmental gain (in the form of restoration and conservation of unaffected habitat) with potentially lower associated costs than full-scale remediation. According to the TRRP rule [350.33(a)(3)(B)], the ecological services produced by the restoration must exceed the future ecological service decreases potentially associated with the continued exposure to COCs or any selected response action at the affected property (or both)—i.e., the person is required to compensate beyond actual impacts. Figure 14.3 gives an overview of the ESA as outlined herein, consistent with Remedy Standard B.

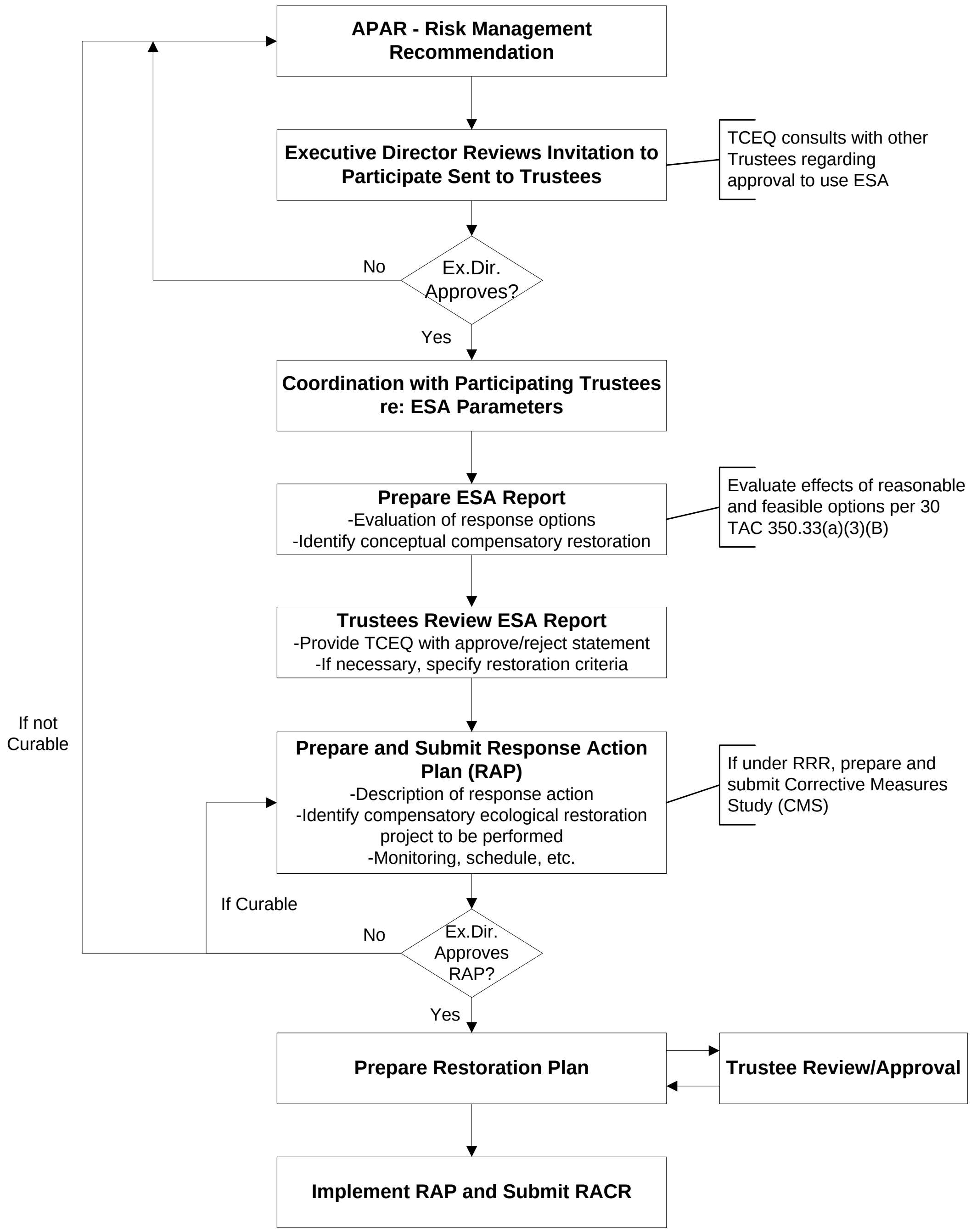

Figure 14.3. Ecological services analysis: response action.

14.2.1. Coordination with the TCEQ and the Natural Resource Trustees

If the ESA process is pursued, the person is required to consult with and obtain approval from the TCEQ or other Trustees at two points in the process. First, when the person requests to perform an ESA, he or she must obtain the approval of the TCEQ after it consults with the Trustees. Second, if compensatory ecological restoration is proposed by the person or required by the Trustees as part of the remedy under the ESA option, the person must obtain approval to proceed with an ESA from both the TCEQ and the Trustees [see 30 TAC 350.33(a)(3)(B)]. At any time, the person may elect to withdraw from the ESA process and revert to another appropriate TRRP response action.

14.2.1.1. TCEQ Approval to Pursue an ESA

Regarding the first requirement [30 TAC 350.77(c)(10) and (f)(2)], the preferred method for requesting approval to pursue an ESA is by making the request to the TCEQ as part of the risk-management recommendation under required element 10. Alternatively, the person may submit a separate written request as part of the APAR. In either case, the request should minimally include: (1) the location within the affected property where COCs are proposed to be left in place above ecological PCLs, (2) a description of the habitats within the area, (3) a list of receptors at risk (as defined in the ERA), and (4) a list of COCs within the area. To the extent possible, any other relevant information that may assist the TCEQ in evaluating the request should be included or referenced from the ERA.

The TRRP rule requires that the TCEQ consult with the Trustees before approval of a request to conduct an ESA [30 TAC 350.33(a)(3)(B) and 30 TAC 350.77(f)(2)]. After receiving the request, the TCEQ will give timely notification to pre-designated contacts within each Trustee agency. Each agency must then respond within a designated time and state whether it needs to be consulted on the request. If all the other Trustees opt not to participate or fail to give a timely response, the person has the choice of continuing with the ESA through collaboration with the TCEQ Trustee Program or discontinuing the approach and reverting to a standard TRRP response.

Upon receipt of a Trustee’s intent to be consulted, the TCEQ will coordinate with the applicable Trustee agencies to determine whether the ESA request is appropriate. Any Trustee that recommends against TCEQ approval of the request must provide a reasoned explanation. The TCEQ ecological risk assessor will evaluate the Trustee comments on the request to perform an ESA and the TCEQ project manager will incorporate them into the TCEQ’s response to the person, as appropriate. If the TCEQ disapproves the request to perform an ESA, the person must recommend an alternate response action.

14.2.1.2. ESA Development and Reporting Requirements

A memorandum of understanding (MOU) between the TCEQ and the Trustees has been developed to ensure the timely and efficient coordination of the consultation with the Trustees regarding a request to perform an ESA. This MOU has been adopted as 30 TAC 7.124, available online at:

<https://texreg.sos.state.tx.us/public/readtac$ext.TacPage?sl=R&app=9&p_dir=&p_rloc=&p_tloc=&p_ploc=&pg=1&p_tac=&ti=30&pt=1&ch=7&rl=124.>. The MOU describes procedures for the distribution of relevant documents and coordination of meetings, sets deadlines for the submission of Trustee comments, and outlines a process for the resolution of conflicting comments. The MOU also institutes a mechanism for the reentry of any Trustee agency into the process even if it elects not to participate at the outset or exits early.

Upon TCEQ approval of the request to perform an ESA, the person must work directly with the Trustees as the ESA is prepared. The Trustees have the responsibility of informing the TCEQ remedial-project manager of all ESA activities, copying the manager on all comments, and inviting the manager to all meetings with the person concerning the ESA. To enhance coordination and ensure efficient development of the ESA, the person should initiate a dialogue with the Trustees as soon as the ESA option is considered and maintain open communications with them throughout the process.

The MOU also addresses timely and efficient coordination between the Trustees and the person in the development and implementation of the ESA, prescribes procedures for review and approval of the ESA, includes designation of a lead Trustee representative as liaison with the person, and specifies a mechanism for ensuring a unified Trustee position on all written comments and statements to the person. If the Trustees cannot reach agreement with the person or he or she fails to perform the ESA as proposed, the Trustees will refer the affected property to the TCEQ for further decisions on an appropriate remedial action.

The culmination of the ESA is the preparation of a report recommending a final remedy for the affected property (e.g., removal, decontamination, control, natural recovery, compensatory ecological restoration, or some combination of these responses). If the ESA demonstrates that compensatory ecological restoration is required, or the person proposes it, the person must submit a restoration project that produces ecological services greater than the ecological service decreases potentially associated with the continued exposure to COCs or any selected response action at the affected property [see 30 TAC 350.33(a)(3)(B)]. The Trustees will review the ESA report and will forward a written statement to the person and the TCEQ project manager approving or rejecting the conclusions in the report. If approved, the Trustees will specify necessary restoration criteria for the project, as applicable. The person will then prepare a response action plan giving details of any actions selected, including a conceptual plan of any necessary restoration work and other RAP requirements as specified by 30 TAC 350.94.

14.2.1.3. Compensatory Ecological Restoration as a Response

When any necessary compensatory ecological restoration is completed consistent with Trustee-approved criteria, the Trustees will also send a written statement to both the person and the TCEQ project manager confirming satisfactory completion of the compensatory ecological restoration (under the performance criteria established in the RAP). If the compensatory ecological restoration cannot be completed to the Trustees' satisfaction, they will refer the affected property to the TCEQ for further decisions on remedial actions.

Frequent coordination with Trustees on the items discussed here will facilitate timely and successful completion of the ESA.

Unless otherwise negotiated with the Trustees, the ESA framework addresses only potential prospective losses of ecological service (e.g., commencing on the date the affected property assessment data were collected). The TCEQ will not construe a decision to undertake an ESA as an admission that natural resource injuries have occurred or are associated with the affected property.

14.2.2. Evaluation of Response Actions under the ESA

To evaluate which response actions will be selected, the person should compare realistically feasible active and passive alternatives available under Remedy Standard B. Realistic estimates of the positive and negative effects of implementing an option must be demonstrated as part of the evaluation. Any responses should allow environmental conditions to return to full function in a reasonable period. Refer to Application of Remedy Standards A and B (TCEQ publication RG-366/TRRP-28) for a discussion of reasonable time periods.

Responses to consider should include removal (e.g., excavation), isolation (e.g., capping), and recovery via natural attenuation (e.g., burial by sedimentation or COC degradation). A decision to select natural attenuation should consider the magnitude and spatial scale.

As set forth in 30 TAC 350.33(a)(3)(B), combinations of active and passive remedies, with or without compensatory restoration, may be appropriate. For example, highly contaminated sediments could be removed, while the remainder of the affected property may best be addressed through a combination of natural attenuation and compensatory ecological restoration.

> The ecological services analysis must, at a minimum, include an evaluation of the effects of reasonable and feasible remediation alternatives, including complete removal/decontamination to PCLs and a control measure to prevent ecological exposure to COCs in excess of ecological PCLs, with respect to present and predicted losses of ecological services; and clear justification for leaving COCs in place above ecological PCLs. [30 TAC 350.33(a)(3)(B)]

Equivalency-analysis tools should be used to compare the negative and positive effects of implementation of these various remedial options. One of the most readily accessible tools is the habitat-equivalency analysis. HEA is an economic model originally developed by NOAA and the U.S. DOI for use in scaling restoration projects to compensate for potential ecological injuries in actions related to natural resource damage. HEA can be used to determine the net present value of ecological services provided by 1 acre of habitat over a specified time measured in discounted service-acre-years. The HEA sees frequent use, including at several sites in Texas by the TCEQ, TPWD, and the Texas General Land Office.

The implicit assumption behind HEA is that the public is willing to accept a trade-off between lost ecological services and restoration project services. Out-of-kind services can often be normalized for comparison. HEA is applicable when productivity services are considered comparable. A thorough explanation of HEA and associated inputs appears at <yosemite.epa.gov/Sab/Sabproduct.nsf/WebFiles/HEA/$File/HEA-03-09-09.pdf>. Other equivalency-analysis tools (e.g., resource-to-resource, service-to-service or valuation scaling approach) and methods for scaling compensatory ecological restoration may be considered by the person but will need review and approval by the Trustees prior to the analysis.

Inputs to the HEA and other equivalency models typically include:

1. Date the assessment begins.
2. Area of the impact (as defined by exceedances of ecological PCLs).
3. Severity of the impact (based upon the results of the Tier 2 or 3 ERA).
4. Duration of the impact and recovery time from it (in years).
5. A discounting factor.

For the ESA, information generated as part of the ERA is critical (e.g., delineation of the extent of the affected area as defined by exceedances of the ecological PCLs in the various environmental media including surficial sediment, surface water, or shallow soil). The quantification of risk in terms of potential ecological-service losses is based on site-specific factors and the resources involved. The process may differ for each site, each case, and each negotiated resolution. While ecological risk and ecological-service losses may not be equivalent, for expediency and cost-effectiveness, the ESA process intends that risk estimation and remedial effects be used to determine potential losses; therefore, areas of significant ecological risk may be useful in initial consideration of the ESA process. As appropriate and after consultation with the Trustees, other factors (e.g., biological effects) may also be used to determine potential ecological-services losses and their associated potential responses.

Generally, mortality, reproductive effects (e.g., fecundity reductions, sterility), and growth effects that are used in the ERA as effects endpoints should be interpreted as resulting in greater ecological-service losses. Behavioral effects, such as avoidance of contaminated media, are also interpreted as service losses, but to a lesser degree than those involving mortality and reproductive and growth effects.

The *ecological debit* is the amount of restoration necessary to offset the ecological-service losses resulting from continued exceedances of the ecological PCLs in affected media and impacts from any response actions. One key feature of the ESA is a comparison among the ecological debits for each of the alternatives. Generally, the alternative selected should balance the severity of remaining ecological risk, the length of time necessary for the affected property to recover to pre-release conditions, appropriate compensation for the public, and cost. At any time during the ESA, a person may elect to withdraw from the process and initiate another response under Remedy Standards A or B.

In developing the inputs for the debit calculation, geographic information systems (GIS) can help identify, describe, and measure the spatial extent of potential ecological-service losses, using information from the ERA. In many cases, literature cited in the ERA will be useful in developing conservative service-loss values, from the perspective of public trust. Estimates of recovery time may come from the literature or site-specific information. Some properties may have developed site-specific interpretive risk data on sediment-quality triads or community structure, etc. Regardless, all estimates should be completely justified, thoroughly documented, and reasonably conservative.

14.2.3. Restoration Planning

Three steps in the ESA process involve restoration planning. The first step is the preparation of the ESA report. If the evaluation of the ESA remedy shows that compensatory ecological restoration is appropriate as a response, the person must first indicate in the ESA report that such restoration will be carried out to compensate for losses of ecological service associated with the continued exceedances of COCs in environmental media. The discussion of restoration at this stage will be conceptual and will identify the habitat types to be addressed as part of the restoration planning (e.g., intertidal marsh, upland forest, shrub and scrub) and the ecological credits necessary to offset the debit associated with the PCL exceedances. In the second step, if Trustees approve the compensatory ecological restoration proposed in the ESA report, the person will then incorporate more details on the project in the RAP, in coordination with the Trustees. The compensatory ecological restoration is the response. It may be coupled with other response actions as the ESA remedy evaluation indicates.

The RAP should include this general information regarding the restoration project:

1. Ecological credits required based upon remedy evaluation.
2. Discussion of the candidate restoration project (e.g., location, habitat types, proposed restoration actions, acreage, duration, maintenance).
3. Ecological credits generated by the project.

The TCEQ strongly recommends that the person coordinate with the Trustees before submitting this information in the RAP to avoid delay in its approval. In addition, the agency suggests that, if the ESA is part of the RAP, the ESA portion should be submitted to the Trustees concurrently with the RAP to the TCEQ project manager. In the last step after the RAP is approved, the person will then develop a detailed restoration plan (detailing the tasks to be conducted and the performance criteria, like construction details for other responses) for review and approval by the Trustees. Once approved, the person will then implement the restoration project pursuant to the plan.

To facilitate the identification and selection of a restoration project that compensates for the future ecological risks and lost ecological services, the person should screen candidate restoration projects against certain criteria. Factors to be considered—to ensure maximum long- and short-term benefits to the ecosystem—include (but are not limited to) proximity of the restoration site to the affected property, hydrology, current uses of the site, and topography. Examples of specific criteria for the selection include:

- The preferred option is for the restoration site to be within the same watershed or ecosystem as the affected property.

- The site should benefit from the enhancement, acquisition, or preservation of the same or similar types of habitat (e.g., vegetation and soil types) as at the affected property.
- The project should be designed to produce the same type of ecological services as part of the remedy evaluation for the affected property.
- The project must allow for reasonable scaling relative to the potential injury at the site and addresses comparable natural resources where PCL exceedances occur.
- The project and site must have the capacity for long-term success.

These criteria were developed specifically to identify a project site with the potential for habitat restoration, to ensure the project has more than enough acreage to compensate for the future potential ecological risks and associated lost services being offset, and to allow for a timely and cost-effective project. The ecological credits to be generated by candidate restoration projects should be determined using the same equivalency process used with the remedy evaluation to ensure that the scaling remains consistent.

To most effectively move through the ESA process, coordination between all parties (i.e., the person, TCEQ project manager, TCEQ ecological risk assessor, and Trustees) is critical.

14.3. Risk Management for Hot Spots

Risk managers should consider all the available information on the affected property when evaluating risk management alternatives. Because the size of a hot spot is most likely restricted (i.e., it does not usually comprise the entire affected property), hot spots can be considered as a separate component for remediation.

If hot spots are identified within the exposure area, the person will need to recommend appropriate risk management practices. Where hot spots are identified and will be separately addressed with a remedy (e.g., removal, capping, ESA), these data points should be removed from the 95 percent UCL calculation and the resulting 95 percent UCL should be used as the exposure point concentration.

The person can suggest to the TCEQ what response actions are appropriate and can provide a rationale. They person may also recommend additional sampling or more property-specific analyses to ultimately refine the potential risk-management alternatives. For more details on risk management of hot spots see 2.4.4.4, 3.2.3.2, and 3.3.4.4 in TRRP-15eco. Key points are:

- Determining what constitutes a hot spot and the appropriate response action are ultimately risk-management decisions specific to the property.

- The response action for a hot spot may be different from the response for the rest of the affected property.
- The hot-spot evaluation may be iterative. Initially, the evaluation may dictate the need for more sampling (to determine if data were in error, to establish the area of a hot spot, to establish a more appropriate sampling density, or to address a specific exposure pathway).
- The person may pursue a limited removal without any corresponding evaluation of risk associated with a hot spot, followed by a standard risk evaluation of the remaining impacted soil and relevant exposure pathways.
- If soil removal is implemented and cleanup is completed to the TCEQ's satisfaction, the associated area will be removed from further ecological evaluation.
- If sediment is removed and cleanup is completed to the TCEQ's satisfaction, the associated hot spot will be removed from further consideration of wildlife risk if there is no potential for recontamination from the affected property.
- Soil hot spot removal may be undertaken at any affected property. However, it is best suited to small sites or small hot spots where the cost of removal action is low relative to the cost of a risk assessment.
- The response action must consider the source of the contamination and the conceptual site model, as sources may include stormwater runoff or contaminated groundwater releases to the affected property. Coordinate the response action for the sediment hot spot with the overall project objectives to prevent recontamination.
- The person may pursue limited sediment removal without any corresponding evaluation of risk from a hot spot followed by a standard risk evaluation of the remaining impacted sediment and relevant exposure pathways. Before removal, an understanding of the CSM and sediment dynamics at the affected property is crucial to ensure that the remediated hot spot will not become re-contaminated by new releases of COCs.

14.4. Implementing the Response

Under Remedy Standard A, implementation begins 10 days after submission of the SIN to the TCEQ. Under Standard B, implementation begins upon approval of the RAP by the TCEQ.

As described in TRRP-25 (Critical PCLs, TCEQ, 2009), for each COC and each affected medium (surface soil, subsurface soil, groundwater, sediment and surface water), the critical PCL is the lowest PCL value among all applicable pathways for human health and ecological exposure. Note that, when ecological exposure pathways are applicable, the timing for development of human health-based PCLs and ecological PCLs may not be the same.

The soil intervals for human and ecological exposures vary, and consideration of the application of ecological PCLs should include an understanding of these depths. For instance, most ecological assessments will focus on surface soil (0–0.5 feet below ground surface) but may also contain an ecological subsurface component (0.5–5 feet below ground surface). Human health exposure pathways may not correspond to the ecological soil intervals. The person should consider these differences when designing a remediation strategy for a site. See TRRP-29 (Soil and Groundwater Response Objectives, TCEQ, 2013b) for further discussion.

14.5. Confirmation Sampling and Monitoring

Final details on the need for confirmation sampling and monitoring after remedy implementation should be discussed with the TCEQ project manager and the TCEQ risk assessor, and agreement reached before sampling and monitoring begin. Under Remedy Standard B, the sampling-and-monitoring plan should be included within the RAP for approval [30 TAC 350.94(d, f)]. To ensure consistency with previous work for the affected property, all sampling and analytical methods should comply with 30 TAC 350.31 and 350.54.

Confirmation sampling of soil, sediment, groundwater, and surface water may be needed to determine whether concentration-based remedial goals have been achieved. The confirmation sampling is likely to be less robust than that needed to delineate the PCLE zone.

In designing an approach to confirmation sampling, it is important to consider an appropriate level of statistical significance that would support any conclusions drawn from the sampling. For example, confirmatory samples may be necessary only at 10 percent of the locations sampled previously, provided that satisfies the statistical criteria used. In some situations, particularly where a natural-attenuation remedy is implemented, confirmatory sampling will not be needed immediately after the remedy is implemented, since there may be no immediate measurable change. Where more active remediation is implemented, confirmatory sampling may need to begin once a discrete area is cleaned, but before the remedial equipment is removed from the affected property. This may prove cost effective at large properties where equipment is being moved from one area to another over time. In other situations, it may not be feasible, due to the lag time between taking of samples and obtaining the results.

In addition, it is unlikely that biological tissues will need to be sampled immediately after remedial actions are completed. More importantly, where no tissue-specific remedial goal is involved, tissue sampling may be unnecessary. Whether natural attenuation or more active remediation is undertaken, a decline in tissue-specific concentrations may require months or longer to appear and may be highly variable when sampling animals whose home range is greater than the remedial area. This type of sampling approach requires great caution.

Often, confirmation sampling and monitoring are linked, but they are not the same. Confirmation sampling may be implemented to determine whether a concentration-based remedial goal has been met, whereas post-remedial monitoring generally involves an examination of the chemical and biological characteristics of the affected property over some time to confirm that continuing conditions conform to the requirements of an approved RAP. Monitoring may involve COC analysis of remediated media such as soil, sediment, groundwater, and surface water.

In addition, some monitoring plans may include biological assessments to determine whether the remedy has achieved improvements or remedial targets in biological parameters. Some biological assessments may involve simple surveys of the number and types of flora and fauna on the affected property once remediation is completed (greater abundance and diversity for example), or—in more complex situations—may involve tissue-specific monitoring, as noted previously.

An important consideration, however, is that the results of biological assessments may vary widely depending on the type of parameter monitored. The number and type of organisms on an affected property will vary with season and local conditions (food, competition, weather, etc.). Given that animals can forage over large areas, tissue-specific monitoring programs may obtain highly variable and perhaps uninterpretable results if the life history and home range of an animal are not well known. Plants and other sessile organisms will likely serve as more reliable surrogates for tissue-specific concentration monitoring.

The duration of a monitoring program depends on its remedial goals. Some goals can be reached quickly and the need for extensive, long-term monitoring may be limited. Where natural attenuation is applied, monitoring may be needed over several years, at some frequency, to determine if the remedial goals have been achieved within the predicted time frame. In general, monitoring programs should be consistent with the predicted recovery periods to ensure that goals have been met.

14.6. Reporting and Documentation

There are numerous reporting requirements under the TRRP rule and, in many cases, under the controlling program (e.g., Superfund) for the affected property. Details of the various TRRP reports are described in 30 TAC 350, Subchapter E. The person must present the PCLs and associated backup information in the APAR. The APAR presents the results of the investigation of the affected property, as well as the human-health and ecological PCL calculations (if the ERA is included within the APAR). In the APAR, the person is required to make a risk-management recommendation per required element 10 for Tier 2 and Tier 3 ERAs. While the APAR may not specifically identify all aspects of the proposed remedy, it should include enough detail so that the TCEQ can make an informed evaluation of the risk assessment and associated calculations.

For Remedy Standard A, the person must submit a self-implementation notice, available at: <www.tceq.texas.gov/assets/public/remediation/trrp/forms/10323sin.pdf> A person submitting a SIN for undertaking a remedy that addresses ecological PCLs is strongly counseled to consult with the TCEQ and gain its approval of the Tier 2 SLERA before initiating the remedy. Otherwise, the person runs the risk of being required to perform additional response work or assessment activities if the TCEQ disagrees with, or disapproves of, the assumptions or calculations made in the Tier 2 ERA.

A person who elects a remedy under Standard B must submit a RAP to the TCEQ for review and approval. A person pursuing an ESA must prepare and submit an ESA report to the Trustees for review and approval. Once approved and where restoration is required, the person must include in—or submit concurrently with the RAP—a compensatory ecological restoration plan. The response may not commence until the person receives TCEQ approval or approval with modification. This does not preclude interim measures. Additionally, the Trustees must approve any restoration under the ESA option.

Upon approval of the RAP, the person must report on response action effectiveness reports every three years until the response action is complete and, once complete, must then submit a response action completion report. After the response action is completed, if post response action care continues, reports will be required.

14.7. Institutional Controls

The TRRP requires placement of institutional controls such as deed notices or restrictive covenants on affected properties in some circumstances. See TRRP-16 (Institutional Controls under TRRP, TCEQ 2010d) for a discussion of the use of institutional controls to address other exposure pathways that are reasonably anticipated to be completed, including agricultural exposure pathways. The fundamental purposes of an institutional control are to:

- give permanent notice to subsequent owners that residual COCs are present at the affected property above PCLs; and
- impose conditions on the future use of the affected property to ensure protective use.

An example of an institutional control applicable to ecological receptors may be a restriction on sediment removal after a cap has been placed over contaminated sediment.

15.0 Tier 3: Site-Specific Ecological Risk Assessment

In accordance with the TRRP rule [30 TAC 350.77(d)], the purpose of the optional SSERA is to incorporate additional information obtained through site-specific studies designed to support an empirical evaluation of ecological risk at the affected property. An SSERA can be conducted when the person believes that any of the Tier 2 PCLs are inappropriate or do not reflect existing conditions at the affected property, or when otherwise elected. Where the ecological evaluation for an affected property begins with a SSERA, the person must incorporate applicable components of a Tier 2 SLERA, including required elements 2, 3, 4, 8, 10, and any other requirements as determined appropriate by the TCEQ. Completion of a Tier 1 Exclusion Criteria Checklist may also be useful to focus the SSERA about the applicable exposure pathways. If a Tier 2 SLERA has been completed, some aspects of the SLERA may need to be refined to reflect the focused specificity of the Tier 3 SSERA. Commonly, the SSERA will focus on a single COC and receptor pair.

The result of the SSERA will be the development of site-specific Tier 3 PCLs accompanied by an ecological risk management recommendation, a determination that there is no ecological risk, or a conclusion that ecological risk is not apparent based on site-specific information. The Tier 3 SSERA can include, but is not limited to:

- Development of site-specific bioaccumulation factors or measurement of actual COC concentrations in tissue through the collection and analysis of tissue samples (e.g., fish, benthic or terrestrial invertebrates, plants).

- Performance of media-specific (i.e., soil, sediment, water) laboratory toxicity tests using an appropriate test species.

- Comparison of site data (e.g., macroinvertebrate-diversity surveys) to like data from a reference area.

- Other studies designed to obtain a preponderance of evidence for conclusions about ecological risk.

Examples of studies that may qualify as Tier 3 assessments are discussed in this chapter.

As indicated, the Tier 3 SSERA is optional and is only conducted at the discretion of the person. Occasionally, the TCEQ may suggest that a Tier 3 SSERA be performed where the agency believes there is too much uncertainty associated with the conclusions of a Tier 2 SLERA. Because Tier 3 involves the collection of site-specific information, it can be costly and time-consuming; therefore, the person is strongly encouraged to communicate with the TCEQ ERA staff regarding the study objectives, conceptual model, study methodology, decision criteria, and additional sampling and site investigations before proceeding. Submission of a work plan is strongly encouraged, but not required.

15.1. Types of Studies

This is a general listing of the types of studies commonly used in Tier 3 SSERAs. Other approaches are possible, and the state of the science continues to evolve. The person is encouraged to seek out additional scientific literature.

15.1.1. COC-Residue Studies

Tissue residue analysis and bioaccumulation studies can be used to evaluate COC transfer through the food chain, to measure the bioavailability of COCs and their concentrations in foods consumed by receptors of concern, to generate site-specific estimates of exposure to higher trophic level organisms, and to relate the tissue concentrations to concentrations in environmental media (U.S. EPA, 1997a). Sometimes residue studies are necessary to generate site-specific uptake factors that are used to back-calculate a source medium PCL. Residue studies are appropriate if COCs in question are expected to bioaccumulate.

15.1.2. Toxicity Tests

Toxicity tests, when combined with COC analyses, can be an important tool in a Tier 3 SSERA to determine if COCs present in an exposure medium are toxic (and bioavailable) to test organisms. To measure toxicity, a specific biological endpoint (e.g., mortality, reductions in growth or reproduction, relevant changes in behavior) is used to assess the response of the test organisms to COCs in the affected media. Toxicity tests can be used to:

- Determine the relative bioavailability of a COC.
- Evaluate the aggregate toxic effects of all COCs in a medium and the toxicity of substances whose toxicity is not well characterized or known.
- Characterize the nature of the toxic effect (lethal or sublethal).
- Develop PCLs and facilitate remediation decisions.

Toxicity tests may be more sensitive to low levels of contamination than other monitoring methods (e.g., COC analyses of media). Most standard toxicity tests are performed at laboratories on media transported from the affected property. This allows for constant conditions, standardized test protocols, and readily available equipment. When toxicity tests are proposed for a Tier 3 SSERA, key upfront decisions include test laboratory, test organisms and endpoints, test duration, decision criteria and applicable statistical tests, and designation of controls. Also consider if the toxicity test is intended to be used to simply determine if COC concentration in affected media are toxic, to develop PCLs, or both. Test design should be planned accordingly.

15.1.3. Field Studies

Ecological field studies take place at the actual affected property, focusing on its habitats and biota. Field studies generally focus on populations and communities and the associated habitats rather than individual organisms.

Results are usually analyzed by comparing the affected property biological metrics (e.g., biomass, abundance, diversity, richness of species and communities, age structure, and trophic structure) to corresponding measurements from a reference or control area.

Population metrics include measurements of density patterns, abundance, biomass, rates of recruitment, size and age distribution, spatial distribution, growth, and survival. Community metrics include measurements of species composition, richness, diversity, dominance, abundance, community structure, trophic dynamics, seasonal patterns, and age classes. Behavioral and physiological measurements such as respiration, photosynthesis, reproduction, burrowing, predation, and courtship may also be evaluated. These measurements are typically compared to those of a reference area or are evaluated for changes along a concentration gradient.

15.1.4. Reference Area

A *reference site* or *area* is defined as an area that is outside the COC influence of the affected property, but possesses similar characteristics, such as habitat and substrate type, allowing for comparison of areas with and without impacts. This definition is applicable to a reference area that is used for a community or population study, or for toxicity tests.

Mortality, vegetation stress, tissue data (histopathologic and COC concentrations), habitat degradation, presence or absence of key species, population assessment of key species, community indices, and ecosystem function determined at the affected property can be compared with the reference site.

Reference areas give valuable information about naturally occurring compounds or ubiquitous COCs. The area selected must have similar habitats to those of the affected property and should lie outside the area of influence of the affected property, preferably in an area of minimal impact or disturbance (U.S. EPA, 1997a: Appendix B). Sampling and surveying of reference areas should be equivalent to that employed at the affected property to ensure a valid comparison.

15.1.5. Other Studies, Weight-of-Evidence

For Tier 3 SSERAs that entail more than one type of study (or line of evidence), a weight-of-evidence approach can be used to integrate multiple types of data to support a conclusion. Generally, confidence in the risk assessment conclusions will be increased using several lines-of-evidence. Balancing and interpreting the different types of data can be major tasks requiring professional judgment, as not all data are of equal importance or certainty. The weight-of-evidence decision criteria should be established before initiating the SSERA. This ensures that data interpretation is objective, not biased to support a preconceived answer. Additional considerations at this stage include the degree to which data-quality objectives were met, and whether confounding factors became evident in the analysis phase.

Menzie et al. (1996) describes a weight-of-evidence methodology for reconciling or balancing multiple lines of evidence pertaining to an assessment endpoint. Other references include Exponent Inc. (2010), McDonald et al. (2007) and Suter and Cormier (2011).

15.2. Tier 3 Reporting and Conclusions

Upon conclusion of a Tier 3 study, the person must submit the SSERA, including recommendations for managing ecological risk, to the TCEQ as part of the APAR. The person should ensure that the SSERA supports the recommended decision for risk management. This determination may require technical advice from the TCEQ ERA staff. At this point, possible recommendations for risk management are the information:

- Is adequate to conclude that there is negligible ecological risk and, therefore, there is no need for remediation.
- Indicates there may be risk and, therefore, a specific response action or control mechanism (possibly to protect human health also) should be implemented to manage ecological risk.
- Indicates there may be risk and, therefore, Tier 3 PCLs will be evaluated in combination with applicable human health PCLs.
- Indicates that an ESA may be appropriate.

These recommendations should be specific to the Tier 3 SSERA and the COC and receptor pairs evaluated. In some cases, they may be combined with risk management decisions established for other COC and receptor pairs from the Tier 2 SLERA.

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17.0 Definitions

[R] = Definition as it appears in the Texas Risk Reduction Program rule (30 TAC 350). Current at time of publication.

95 percent upper confidence limit (of a mean)—A value that, when calculated repeatedly for randomly drawn subsets of site data, equals or exceeds the true mean 95 percent of the time.

affected property[R]—The entire area (i.e., on-site and off-site; including all environmental media) which contains releases of chemicals of concern at concentrations equal to or greater than the assessment level applicable for residential land use and groundwater classification. The terms "site" and "affected property" are used interchangeably in this document but both terms are used to denote the entire area of contamination [see 30 TAC 350.4(a)(1)].

area-use factor—The ratio of an organism's home range, breeding range, or feeding and foraging range to the area of contamination of the site under investigation.

assessment endpoint—An explicit expression of an environmental value to be protected.

assessment level[R]—A critical protective concentration level for a chemical of concern used for affected property assessments where the human health protective concentration level is established under a Tier 1 evaluation as described in §350.75(b) of [Title 30, TAC] (relating to Tiered Human Health Protective Concentration Level Evaluation), except for the protective concentration level for the soil-to-groundwater exposure pathway which may be established under Tier 1, 2, or 3 as described in §350.75(i)(7) of [Title 30], and ecological protective concentration levels which are developed, when necessary, under Tier 2 and/or 3 in accordance with §350.77(c) and/or (d), respectively, of [Title 30] (relating to Ecological Risk Assessment and Development of Ecological Protective Concentration Levels).

background[R]—A population of concentrations characterized from samples in an environmental medium containing a chemical of concern that is naturally occurring (i.e., the concentration is not due to a release of chemicals of concern from human activities) or anthropogenic (i.e., the presence of a chemical of concern in the environment which is due to human activities, but is not the result of site-specific use or release of waste or products, or industrial activity). Examples of anthropogenic sources include non-site specific sources such as lead from automobile emissions, arsenic from use of defoliants, and polynuclear aromatic hydrocarbons resulting from combustion of hydrocarbons. There are some commonalities regardless of the activity; specifically, the chemicals of concern have resulted from the use of a product in its intended manner and may be present at generally low levels over large areas (tens of square miles up to hundreds of square miles). Background is required for use in a statistical model appropriate for testing the hypothesis that the background area characterized by these kinds of models has the same concentrations of the chemical of concern as the affected property. The background area characterized is as "close" as possible to the affected property, in either space or time, as required.

bedrock[R]—The solid rock (i.e., consolidated, coherent, and relatively hard naturally formed material than cannot normally be excavated by manual methods alone) that underlies gravel, soil or other surficial material.

benthic community—The community of organisms dwelling at the bottom of a pond, river, lake, or ocean.

bioaccumulation—General term describing uptake of chemicals by an organism, either directly from exposure to a medium or by consumption of food containing the chemical.

bioccumulation factor—The ratio of the concentration of a chemical of concern in an organism to the concentration in the ambient environment at steady state.

bioaccumulative chemical of concern[R]—A chemical of concern which has the tendency to accumulate in the tissues of an organism as a result of food consumption or dietary exposure and/or direct exposure (e.g., gills and epithelial tissue) to an environmental medium.

bioavailability—The degree to which a material in environmental media can be assimilated by an organism.

bioconcentration—Net accumulation of a chemical directly from an exposure medium (e.g., water) into an organism; does not include food web transfer.

biomagnification—The result of bioaccumulation and biotransfer by which tissue concentrations of chemicals in organisms at one trophic level exceed tissue concentrations in organisms at the next-lower trophic level in a food chain.

chemical of concern[R]—Any chemical that has the potential to adversely affect ecological or human receptors due to its concentration, distribution, and mode of toxicity. Depending on the program area, chemicals of concern may include the following: solid waste, industrial solid waste, municipal solid waste, and hazardous waste as defined in the Texas Health and Safety Code, §361.003, as amended; hazardous constituents as listed in 40 Code of Federal Regulations Part 261, Appendix VIII, as amended; constituents on the groundwater monitoring list in 40 Code of Federal Regulations Part 264, Appendix IX, as amended; constituents as listed in 40 Code of Federal Regulations Part 258 Appendices I and II, as amended; pollutant as defined in Texas Water Code, §26.001, as amended; hazardous substance as defined in the Texas Health and Safety Code, §361.003, as amended, and Texas Water Code, §26.263, as amended; other substances as defined in Texas Water Code §26.039(a), as amended; and daughter products of the aforementioned constituents.

closure[R]—The act of permanently taking a waste management unit or facility out of service.

community[R]—An assemblage of plant and animal populations occupying the same habitat in which the various species interact via spatial and trophic relationships (e.g., a desert community or a pond community).

compensatory ecological restoration[R]—The creation of ecological services by or through restoration or the setting aside of, preferably, a comparable type of habitat as that which is impacted to offset residual ecological risk at an affected property. A net environmental benefits analysis or similar evaluation of ecological services may be used in the determination of the appropriate level of compensation.

complete exposure pathway[R]—An exposure pathway where a human or ecological receptor is exposed to a chemical of concern via an exposure route (e.g., incidental soil ingestion, inhalation of volatiles and particulates, consumption of prey, etc.).

conceptual model—A series of working hypotheses of how a stressor might affect ecological components. Describes an ecosystem or ecosystem components potentially at risk and the relationships between measurement and assessment endpoints and exposure scenarios.

control[R]—To apply physical or institutional controls to prevent exposure to chemicals of concern. Control measures must be combined with appropriate maintenance, monitoring, and any necessary further response action to be protective of human health and the environment.

critical protective concentration level[R]—The lowest protective concentration level for a chemical of concern within a source medium determined from all of the applicable human health exposure pathways as described in 350.71 of [Title 30, TAC] (relating to General Requirements), and when necessary, protective concentration levels for applicable ecological exposure pathways as required in 350.77 of [Title 30] (relating to Ecological Risk Assessment and Development of Ecological Protective Concentration Levels).

decontaminate[R]—Application or occurrence of a permanent and irreversible treatment process to a waste or environmental medium so that the threat of release of chemicals of concern at concentrations above the critical protective concentration levels is eliminated.

de minimus—The description of an area of affected property comprised of one acre or less where the ecological risk is insignificant because of the small extent of contamination, the absence of protected species, the availability of similar unimpacted habitat nearby, and the lack of adjacent sensitive environmental areas.

disturbed ground (also **disturbed area** or **setting**)—A location where any ecological habitat that may have once existed has been altered, changed, or reduced to a degree that it is no longer conducive to use by ecological receptors (e.g., pavement, process areas, buildings). These locations are predominantly urban or commercial/industrial and are often characterized by human presence and activities.

dose—A measure of exposure. Examples include the amount of a chemical ingested, the amount of a chemical absorbed, and the product of ambient exposure concentration and the duration of exposure.

dose-response curve—The relationship between a change in effect on an organism caused by differing levels of exposure (or doses) to a stressor (usually a chemical) after a certain exposure time.

ecological benchmark[R]—A state standard, federal guideline, or other exposure level for a chemical of concern in water, sediment, or soil that represents a protective threshold from adverse ecological effects. An ecological benchmark may also be a toxicity reference value that is established by the person based on scientific studies in the literature.

ecological hazard index[R]—The sum of individual ecological hazard quotients of COCs within a class of compounds that exert ecological effects which have the same toxicological mechanism or endpoint (e.g., PAHs, PCBs).

ecological hazard quotient[R]—The ratio of an exposure level to a chemical of concern to a toxicity value selected for the risk assessment for that chemical of concern (e.g., a "no observed adverse effects level").

ecological protective concentration level[R]—The concentration of a chemical of concern at the point of exposure within an exposure medium (e.g., soil, sediment, groundwater, or surface water) which is determined in accordance with §350.77(c) or (d) of [Title 30, TAC] (relating to Ecological Risk Assessment and Development of Ecological Protective Concentration Levels) to be protective for ecological receptors. These concentration levels are primarily intended to be protective for more mobile or wide-ranging ecological receptors and, where appropriate, benthic invertebrate communities within the waters in the state. These concentration levels are not intended to be directly protective of receptors with limited mobility or range (e.g., plants, soil invertebrates, and small rodents), particularly those residing within active areas of a facility, unless these receptors are threatened/endangered species or unless impacts to these receptors result in disruption of the ecosystem or other unacceptable consequences for the more mobile or wide-ranging receptors (e.g., impacts to an off-site grassland habitat eliminate rodents which causes a desirable owl population to leave the area).

ecological risk assessment[R]—The process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors; however, as used in this context, only chemical stressors (i.e., COCs) are evaluated.

ecological services[R]—The physical, chemical, or biological functions of natural resources that one natural resource provides for another or to the public. Examples include provision of food, protection from predation, and nesting habitat, among others.

ecological services analysis[R]—A measurement of the potential change in ecological services based on considerations which may include but are not limited to: the percent change in ecological services at the affected property that are attributable to COCs and/or potential response actions; the spatial extent of the affected property; and the recovery period.

ecosystem—The biotic community and abiotic environment at a specified location and time.

ecotoxicity—Toxic effects on nonhuman organisms, populations, or communities.

environmental medium[R]—A material found in the natural environment such as soil (including non-waste fill materials), groundwater, air, surface water, and sediments, or a mixture of such materials with liquids, sludges, gases, or solids, including hazardous waste which is inseparable by simple mechanical removal processes, and is made up primarily of natural environmental material.

exclusion criteria[R]—Those conditions at an affected property which preclude the need to establish a protective concentration level for an ecological exposure pathway because the exposure pathway between the chemical of concern and the ecological receptors is not complete or is insignificant.

exposure—Co-occurrence of or contact between a stressor and an ecological component. The contact reaction between a chemical and a biological system or organism.

exposure area[R]—The smallest property surface area within which it is believed that exposure to chemicals of concern in soil or air by a receptor would be limited under reasonably anticipated current or future use scenarios.

exposure assessment—The determination or estimation (qualitative or quantitative) of the magnitude, frequency, duration, and route of exposure.

exposure medium[R]—The environmental medium or biologic tissue in which or by which exposure to chemicals of concern by ecological or human receptors occurs.

exposure pathway[R]—The course that a chemical of concern takes from a source area to ecological or human receptors and includes a source area, a point of exposure, and an exposure route (e.g., ingestion), as well as a transport mechanism if the point of exposure is different from the source area.

feeding guilds[R]—Groups of ecological receptors used to represent the variety of species that may be exposed to chemicals of concern at the affected property. The feeding guilds are generally based on function within an ecosystem, potential for exposure, and physiological and taxonomic similarity. Examples include carnivorous mammals, carnivorous birds, and piscivorous birds.

food-chain transfer—A process by which substances in the tissues of lower-trophic-level organisms are transferred to the higher-trophic-level organisms that feed on them.

fossorial—Describes an animal that is adapted for burrowing.

groundwater-bearing unit[R]—A saturated geologic formation, group of formations, or part of a formation which has a hydraulic conductivity equal to or greater than 1×10^{-5} centimeters/second.

habitat—Place where a plant or animal lives, often characterized by a dominant plant form and physical characteristics.

home range—The area to which an animal confines its activities including foraging and nesting.

Implementation Procedures[R]—The most current version of *Procedures to Implement the Texas Surface Water Quality Standards,* as amended.

institutional control[R]—A legal instrument placed in the property records in the form of a deed notice, Voluntary Cleanup Program Certificate of Completion (VCP Certificate of Completion), or restrictive covenant which indicates the limitations on or the conditions governing use of the property which ensures protection of human health and the environment or equivalent zoning and governmental ordinances.

landscaped area[R]—An area of ornamental, introduced, commercially installed, or manicured vegetation which is routinely maintained.

LC_{50}—The concentration of a toxicant that is lethal (fatal) to 50 percent of the organisms tested in a specified time period.

LD_{50}—The dose of a toxicant that is lethal (fatal) to 50 percent of the organisms tested in a specified time period.

lowest observed adverse-effect level (LOAEL)—The lowest level of a stressor evaluated in a toxicity test or biological field survey that has a statistically significant adverse effect on the exposed organisms compared with unexposed organisms in a control or reference site.

measurement endpoint—A measurable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint. Measurement endpoints often are expressed as the statistical or arithmetic summations of the observations that make up the measurement. Can include measures of effect and measures of exposure.

method detection limit[R]—The minimum concentration of a substance that can be measured and reported with 99 % confidence that the analyte concentration is greater than zero and is determined for each COC from the analysis of a sample of a given matrix type containing the COC.

method quantitation limit[R]—The lowest non-zero concentration standard in the laboratory's initial calibration curve and is based on the final volume of extract (or sample) used by the laboratory.

monitored natural attenuation[R]—The use of natural attenuation within the context of a carefully controlled and monitored response action to achieve protective concentration levels at the point of exposure.

natural attenuation[R]—The reduction in mass or concentration of a chemical of concern over time or distance from the source of a chemical of concern due to naturally occurring physical, chemical, and biological processes, such as: biodegradation, dispersion, dilution, adsorption, and volatilization.

Natural Resource Trustees[R]—The federal agencies as designated by the President and the state agencies as designated by the Governor pursuant to the National Contingency Plan, Oil Pollution Act, and CERCLA §107(f)(2)(A) and (B) to act on behalf of the public as trustees of natural resources (e.g., water, air, land, wildlife). The Trustees include TCEQ, Texas Parks and Wildlife Department, Texas General Land Office, National Oceanic and Atmospheric Administration, and the Department of the Interior.

"no observed adverse effect" level (NOAEL)—The highest level of a stressor evaluated in a toxicity test or biological field survey that causes no statistically significant difference in effect compared with the controls or a reference site.

off-site property (off-site)[R]—All environmental media which is outside of the legal boundaries of the on-site property.

on-site property (on-site)[R]—All environmental media within the legal boundaries of a property owned or leased by a person who has filed a self-implementation notice or a response action plan for that property or who has become subject to such action through one of the agency's program areas for that property.

person[R]—An individual, corporation, organization, government or governmental subdivision or agency, business trust, partnership, association, or any other legal entity.

[Specifically, "person," throughout this document, is meant to denote the regulated entity or environmental consultant that is performing the ERA or ecological services analysis.]

physical control[R]—A structure or hydraulic containment action which prevents exposure to and/or migration of chemicals of concern when combined with appropriate post-response action care to protect human health and the environment. Examples of physical controls are caps, slurry walls, sheet piling, hydraulic containment wells, and interceptor trenches, but typically not fences.

point of exposure[R]—The location within an environmental medium where a receptor will be assumed to have a reasonable potential to come into contact with chemicals of concern. The point of exposure may be a discrete point, plane, or an area within or beyond some location.

population—An aggregate of individuals of a species within a specified location in space and time.

practical quantitation limit—The lowest concentration of an analyte that can be reliably quantified within specified limits of precision and accuracy during routine laboratory operating conditions. The PQL minimizes to the extent possible the effects of instrument and operator variability and the influences of the sample matrix and other COCs or substances upon the quantitation of the analyte. "Specified limits of precision and accuracy" are the criteria that have been included in applicable regulations or that are listed in the quality control sections of the analytical method. The PQL may be directly obtained or derived from the following sources with preference given to the most recent scientifically valid method: federal regulations, EPA guidance documents, calculation from interlaboratory studies, and experimentally determined analytical methods not available from other existing sources.

prescribed points of exposure[R]—The prescribed on-site and off-site locations within an environmental medium where an individual human or population will be assumed to come into contact with chemicals of concern from an affected property.

protective concentration level[R]—The concentration of a chemical of concern which can remain within the source medium and not result in levels which exceed the applicable human health risk–based exposure limit or ecological protective concentration level at the point of exposure for that exposure pathway.

protective concentration level exceedance zone[R]—The lateral and vertical extent of all wastes and environmental media which contain chemicals of concern at concentrations greater than the critical protective concentration level determined for that medium, as well as, hazardous waste. A protective concentration level exceedance zone can be thought of as the volume of waste and environmental media which must be removed, decontaminated, and/or controlled in some fashion to adequately protect human health and the environment.

reasonably anticipated to be completed exposure pathway[R]—A situation with a credible chance of occurrence in which an ecological or human receptor may become exposed to a chemical of concern (i.e., complete exposure pathway) without consideration of circumstances which are extreme or improbable based on property characteristics.

reference site—A relatively uncontaminated site used for comparison to contaminated sites in environmental monitoring studies, often incorrectly referred to as a “control.”

remediation[R]—The act of eliminating or reducing the concentration of chemicals of concern in environmental media.

remove[R]—To take waste or environmental media away from the affected property to another location for storage, processing or disposal in accordance with all applicable requirements. Removal is an irreversible process that results in permanent risk reduction at an affected property.

residential land use[R]—Property used for dwellings such as single family houses and multi-family apartments, children’s homes, nursing homes, and residential portions of government-owned lands (local, state or federal). Because of the similarity of exposure potential and the sensitive nature of the potentially exposed population, day care facilities, educational facilities, hospitals, and parks (local, state or federal) shall also be considered residential.

response action[R]—Any activity taken to comply with these regulations to remove, decontaminate and/or control (i.e., physical controls and institutional controls) chemicals of concern in excess of critical PCLs in environmental media, including actions taken in response to releases to environmental media from a waste management unit before, during, or after closure.

risk—The expected frequency or probability of undesirable effects resulting from exposure to known or expected stressors.

risk-based exposure limit[R]—The concentration of a chemical of concern at the point of exposure within an exposure medium (e.g., soil, sediment, vegetables, groundwater, surface water, or air) which is protective for human health. Risk-based exposure limits are the fundamental risk-based values which are initially determined and used in the development of protective concentration levels. Risk-based exposure limits do not account for cumulative effects from exposure to multiple chemicals of concern, combined exposure pathways, and cross-media or lateral transport of chemicals of concern within environmental media.

sample detection limit—The method detection limit, adjusted to reflect sample-specific actions, such as dilution or use of smaller aliquot sizes than prescribed in the analytical method, and to take into account sample characteristics, sample preparation, and analytical adjustments. The term, as used in [30 TAC 350], is analogous to the sample-specific detection limit.

sediment[R]—Non-suspended particulate material lying below surface waters such as bays, the ocean, rivers, streams, lakes, ponds, or other similar surface water body (including intermittent streams). Dredged sediments which have been removed from below surface water bodies and placed on land shall be considered soils.

selected ecological receptors[R]—Species that are to be carried through the ecological risk assessment as representatives of the different feeding guilds and communities that are being evaluated. These species may not actually occur at the affected property, but may be used to represent those within the feeding guild or community that may feed on the affected property.

sensitive environmental areas[R]—Areas that provide unique and often protected habitat for wildlife species. These areas are typically used during critical life stages such as breeding, hatching, rearing of young, and overwintering. Examples include critical habitat for threatened and endangered species, wilderness areas, parks, and wildlife refuges.

site—The terms “site” and “affected property” are used interchangeably in this document but both terms are used to denote the entire area of contamination [see 30 TAC 350.4(a)(1)].

soil protective concentration level exceedance zone[R]—A protective concentration level exceedance zone within the surface soil or subsurface soil which may extend down to a groundwater-bearing unit(s). These protective concentration level exceedance zones may also be present below or between groundwater-bearing units.

stressor[R]—Any physical, chemical, or biological entity that can induce an adverse response; however, as used in this context, only chemical entities apply.

subsurface soil[R]—For human health exposure pathways, the portion of the soil zone between the base of surface soil and the top of the groundwater-bearing unit(s). For ecological exposure pathways, the portion of the soil zone between 0.5 feet and 5 feet in depth.

surface cover[R]—A layer of artificially placed utility material (e.g., shell, gravel).

surface soil[R]—For human health exposure pathways, the soil zone extending from ground surface to 15 feet in depth for residential land use and from ground surface to 5 feet in depth for commercial/industrial land use; or to the top of the uppermost groundwater-bearing unit or bedrock, whichever is less in depth. For ecological exposure pathways, the soil zone extending from ground surface to 0.5 feet in depth.

surface water[R]—Any water meeting the definition of surface water in the state as defined in §307.3 of [Title 30, TAC] (relating to Definitions and Abbreviations), as amended.

toxicity reference value[R]—An exposure level from a valid scientific study that represents a conservative threshold for adverse ecological effects.

trophic level—A functional classification of taxa within a community based on feeding relationships (e.g., aquatic and terrestrial plants make up the first trophic level, and herbivores make up the second).

Appendix A: Food Webs of the Seven Major Habitats in Texas

As discussed in **6.2.1**, although the food webs in this appendix look complex, users need not evaluate risk to all feeding guilds within a web if there is a logical justification. Each of the food webs representing the seven major habitats in Texas are discussed in detail followed by a graphic representation.

Figure A.1 shows an upland forest food web, where soil represent the basis of the food web and contain nutrients, detritus, bacteria, microfauna, and microflora. There are four trophic levels that are supported by soil.

The first trophic level is represented by terrestrial plants including pines, hardwoods, brush trees, grasses, and forbs.

Trophic level two includes herbivorous mammals, terrestrial invertebrates, and herbivorous birds that would all ingest terrestrial plants in trophic level one. Herbivorous mammals include such species as the eastern cottontail and white-tailed deer. Terrestrial invertebrates include insects and arthropods. Herbivorous birds include species like the mourning dove and house finch.

Trophic level three is represented by omnivorous mammals, omnivorous amphibians and reptiles, and omnivorous birds. Omnivorous species ingest a variety of plants and tissues of other species. Omnivorous mammals include the southern short-tailed shrew and raccoon. The omnivorous amphibians and reptiles include the marbled salamander, eastern box turtle, and rough earth snake. Omnivorous birds include the American robin and northern cardinal.

Trophic level four is represented by carnivorous mammals, birds, and reptiles. Carnivorous species ingest other species found in trophic levels two and three. Carnivorous mammals include the red fox and bobcat. Carnivorous birds include the red-tailed hawk and great horned owl. Carnivorous reptiles include the Texas rat snake and southern copperhead.

Some species in the figure (or closely related species) are discussed in EPA's 1993 Wildlife Exposure Factors Handbook and are noted with an asterisk in the figure. These species include the southern short-tailed shrew, raccoon, eastern box turtle, American robin, red fox, and red-tailed hawk.

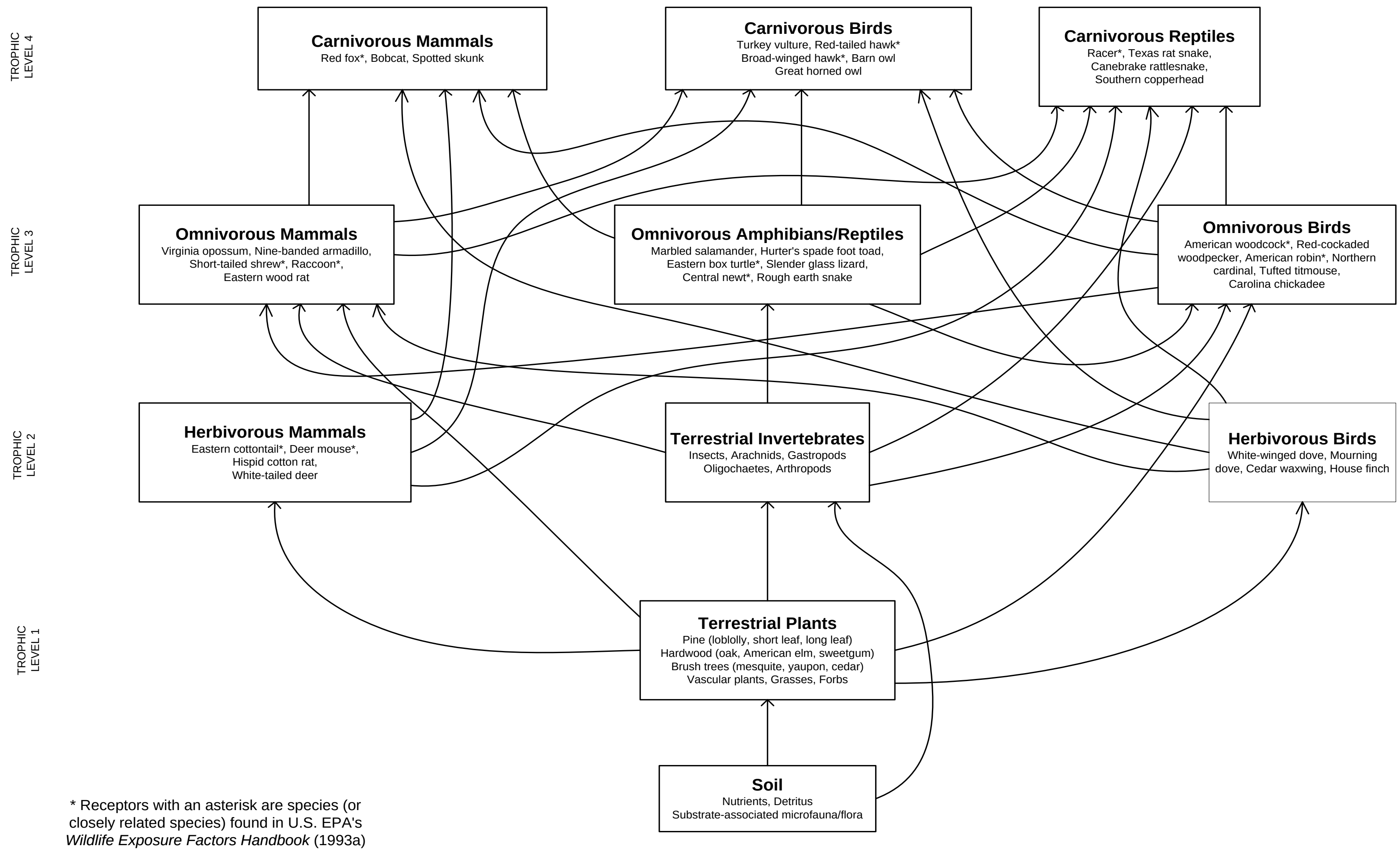


Figure A.1. Upland-forest food web.

Figure A.2 shows a tallgrass prairie food web, where soil is the basis of the food web and contains nutrients, detritus, microfauna, and microflora. There are four trophic levels that are supported by soil.

The first is represented by terrestrial plants including big bluestem and other grasses and forbs.

Trophic level two includes herbivorous mammals, terrestrial invertebrates, and herbivorous birds that would all ingest terrestrial plants in trophic level one. Herbivorous mammals include such species as the plains harvest mouse and hispid cotton rat. Terrestrial invertebrates include oligochaetes and arthropods. Herbivorous birds include species like the mourning dove and sparrow.

Trophic level three is represented by omnivorous mammals, amphibians and reptiles, and birds. Omnivorous species ingest a variety of plants and tissues of other species. Omnivorous mammals include the least shrew and deer mouse. The omnivorous amphibians and reptiles include the Texas toad and plains hognose snake. Omnivorous birds include the western meadowlark and bobwhite quail.

Trophic level four is represented by carnivorous mammals, birds, and reptiles. Carnivorous species prey upon other species found in trophic levels two and three. Carnivorous mammals include the coyote and spotted skunk. Carnivorous birds include the American kestrel and Cooper's hawk. Carnivorous reptiles include the eastern yellowbelly racer and western diamondback rattlesnake.

Some species in the figure (or closely related species) are discussed in EPA's 1993 Wildlife Exposure Factors Handbook and are noted with an asterisk in the figure. These species include the deer mouse, least shrew, bobwhite quail, American kestrel, eastern yellowbelly racer, and coyote.

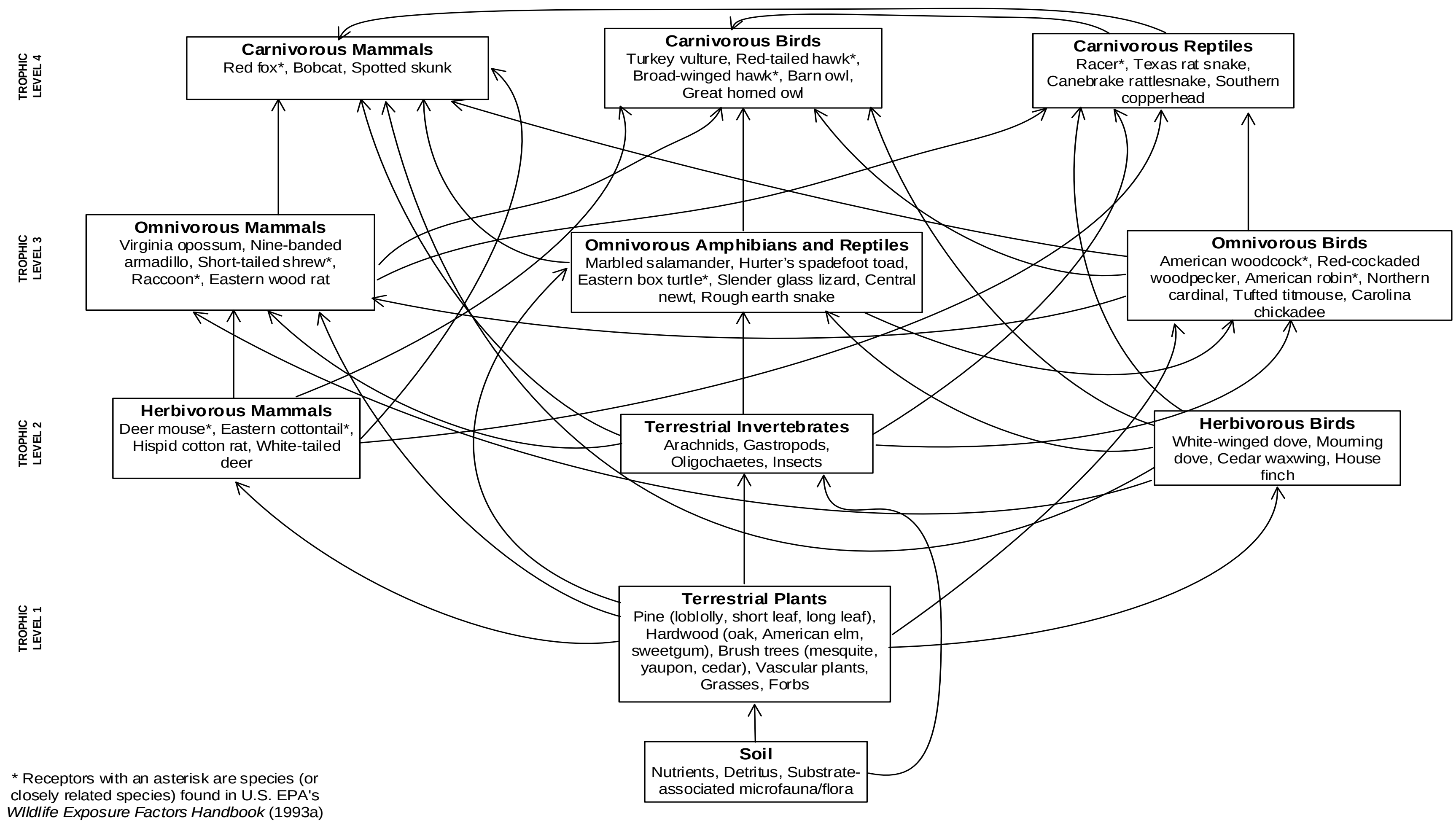


Figure A.2. Tallgrass-prairie food web.

Figure A.3 shows a shortgrass prairie food web, where soil is the basis for the food web and contains nutrients, detritus, and microorganisms. There are four trophic levels that are supported by soil.

The first is represented by terrestrial plants including forbs, mesquite, and grasses.

Trophic level two includes herbivorous mammals, terrestrial invertebrates, and herbivorous birds that would all ingest terrestrial plants in trophic level one. Herbivorous mammals include such species as the pocket mouse and prairie vole. Terrestrial invertebrates include oligochaetes and arthropods. Herbivorous birds include species like the mourning dove and sparrow.

Trophic level three is represented by omnivorous mammals, amphibians and reptiles, and birds. Omnivorous species ingest a variety of plants and tissues of other species. Omnivorous mammals include the least shrew and armadillo. The omnivorous amphibians and reptiles include various toads, ornate box turtle, and the Texas spotted whiptail. Omnivorous birds include the bobwhite and barn swallow.

Trophic level four is represented by carnivorous mammals, birds, and reptiles. Carnivorous species prey upon other species found in trophic levels two and three. Carnivorous mammals include the red fox and bobcat. Carnivorous birds include the American kestrel and burrowing owl. Carnivorous reptiles include the Great Plains rat snake and bull snake.

Some species in the figure (or closely related species) are discussed in EPA's 1993 Wildlife Exposure Factors Handbook and are noted with an asterisk in the figure. These species include the least shrew, bobwhite, ornate box turtle, Texas spotted whiptail snake, red fox, and American kestrel.

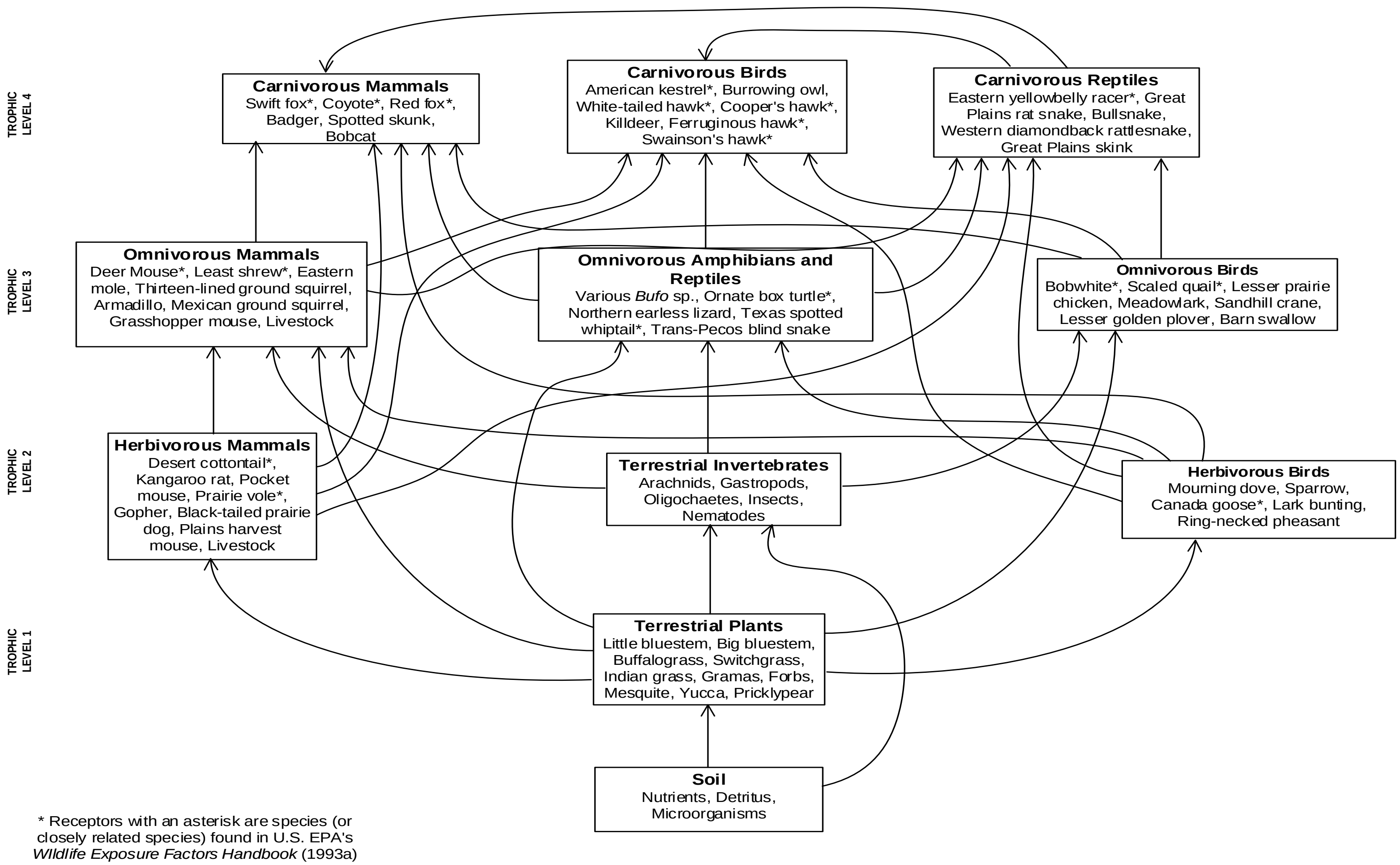


Figure A.3. Shortgrass-prairie food web.

Figure A.4 shows a shrub-scrub food web, where soil is the basis of the food web and contains nutrients and detritus. There are four trophic levels that are supported by soil.

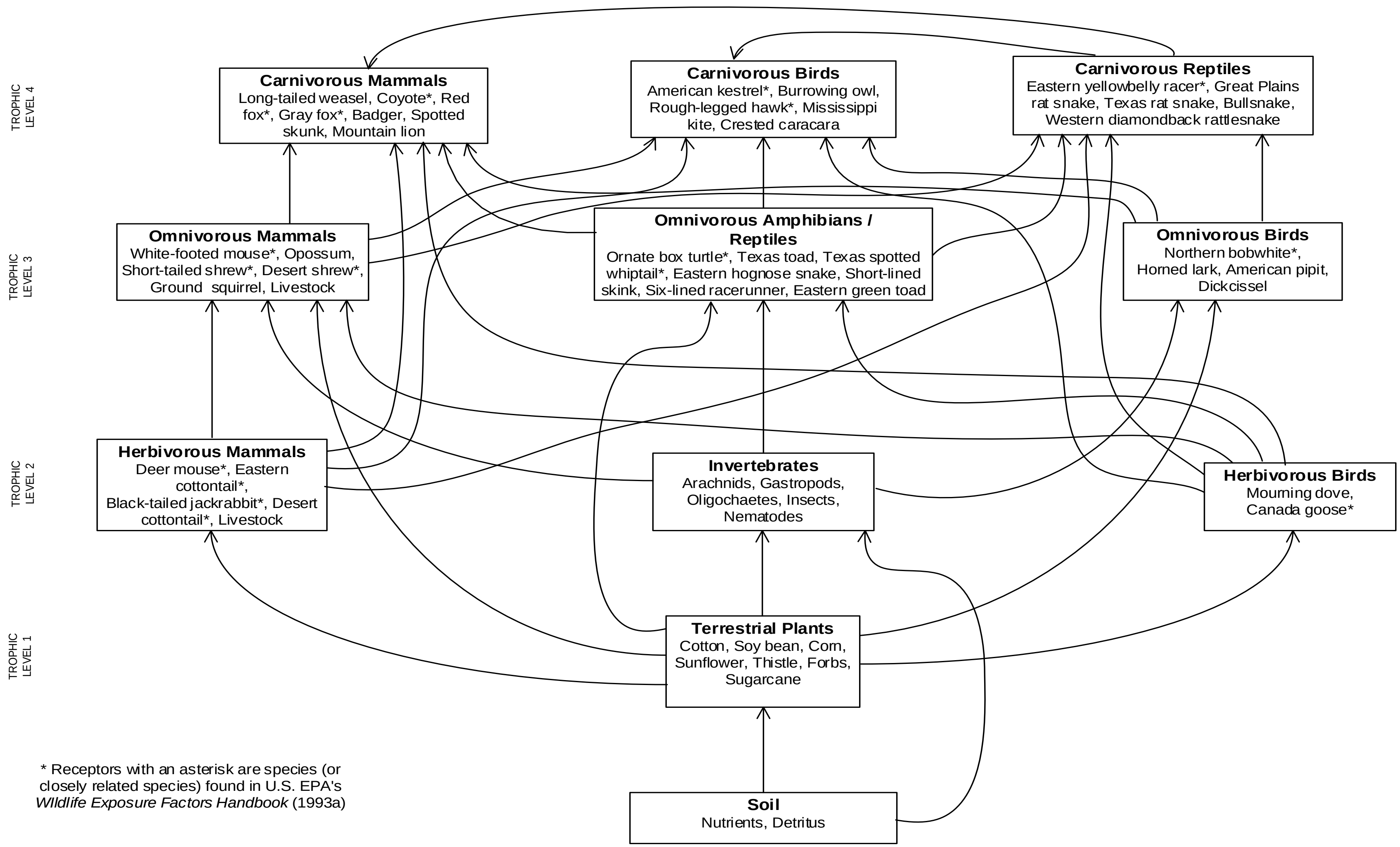
The first is represented by terrestrial plants including cotton, sunflower, and sugarcane.

Trophic level two includes herbivorous mammals, terrestrial invertebrates, and herbivorous birds that would all ingest terrestrial plants in trophic level one. Herbivorous mammals include such species as the black-tailed jackrabbit and eastern cottontail. Terrestrial invertebrates include oligochaetes and arthropods. Herbivorous birds include species like the mourning dove and Canada goose.

Trophic level three is represented by omnivorous mammals, amphibians and reptiles, and birds. Omnivorous species ingest a variety of plants and tissues of other species. Omnivorous mammals include the white-footed mouse and southern short-tailed shrew. The omnivorous amphibians and reptiles include the eastern green toad and the Texas spotted whiptail. Omnivorous birds include the northern bobwhite and western kingbird.

Trophic level four is represented by carnivorous mammals, birds, and reptiles. Carnivorous species prey upon other species found in trophic levels two and three. Carnivorous mammals include the red fox and coyote. Carnivorous birds include the American kestrel and burrowing owl. Carnivorous reptiles include the bull snake and western diamondback rattlesnake.

Some species in the figure (or closely related species) are discussed in EPA's 1993 Wildlife Exposure Factors Handbook and are noted with an asterisk in the figure. These species include the white-footed mouse, southern short-tailed shrew, northern bobwhite, red fox, coyote, and American kestrel.



* Receptors with an asterisk are species (or closely related species) found in U.S. EPA's *WIldlife Exposure Factors Handbook* (1993a)

Figure A.4. Shrub-scrub food web.

Figure A.5 shows a desert-arid food web, where soil is the basis of the food web and contains nutrients, detritus, and microorganisms. There are four trophic levels that are supported by soil.

The first is represented by terrestrial plants including yucca, mesquite, and forbs.

Trophic level two includes herbivorous mammals, terrestrial invertebrates, and herbivorous birds that would all ingest terrestrial plants in trophic level one. Herbivorous mammals include such species as the mule deer and desert cottontail. Terrestrial invertebrates include insects and arachnids. Herbivorous birds include species like the mourning dove and house finch.

Trophic level three is represented by omnivorous mammals, insectivorous reptiles, and omnivorous birds. Omnivorous species ingest a variety of plants and tissues of other species. Omnivorous mammals include the white-footed mouse and desert shrew. The insectivorous reptiles include various lizards such as the Texas horned lizard. Omnivorous birds include the bobwhite and cactus wren.

Trophic level four is represented by carnivorous mammals, birds and reptiles. Carnivorous species prey upon other species found in trophic levels two and three. Carnivorous mammals include the swift fox and mountain lion. Carnivorous birds include the red-tailed hawk and greater roadrunner. Carnivorous reptiles include the western coachwhip and western diamondback rattlesnake.

Some species in the figure (or closely related species) are discussed in EPA's 1993 Wildlife Exposure Factors Handbook and are noted with an asterisk in the figure. These species include the desert cottontail, white-footed mouse, desert shrew, bobwhite, swift fox, red-tailed hawk, and western coachwhip.

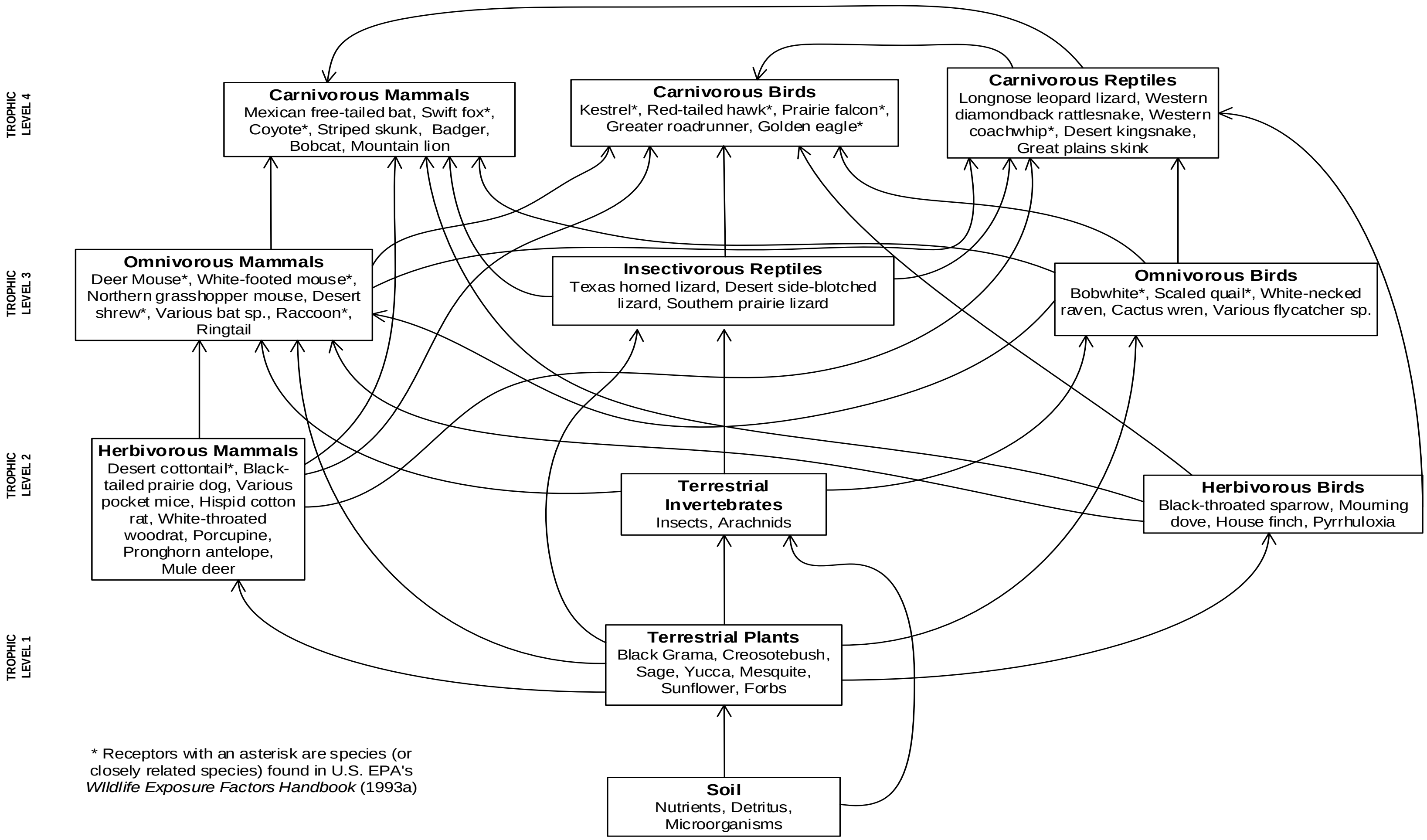


Figure A.5. Desert-arid food web.

Figure A.6 shows a freshwater-systems food web, where freshwater and sediment represent the basis of the food web and contain nutrients, detritus, bacteria and fungi. There are four trophic levels that are supported by water and sediment.

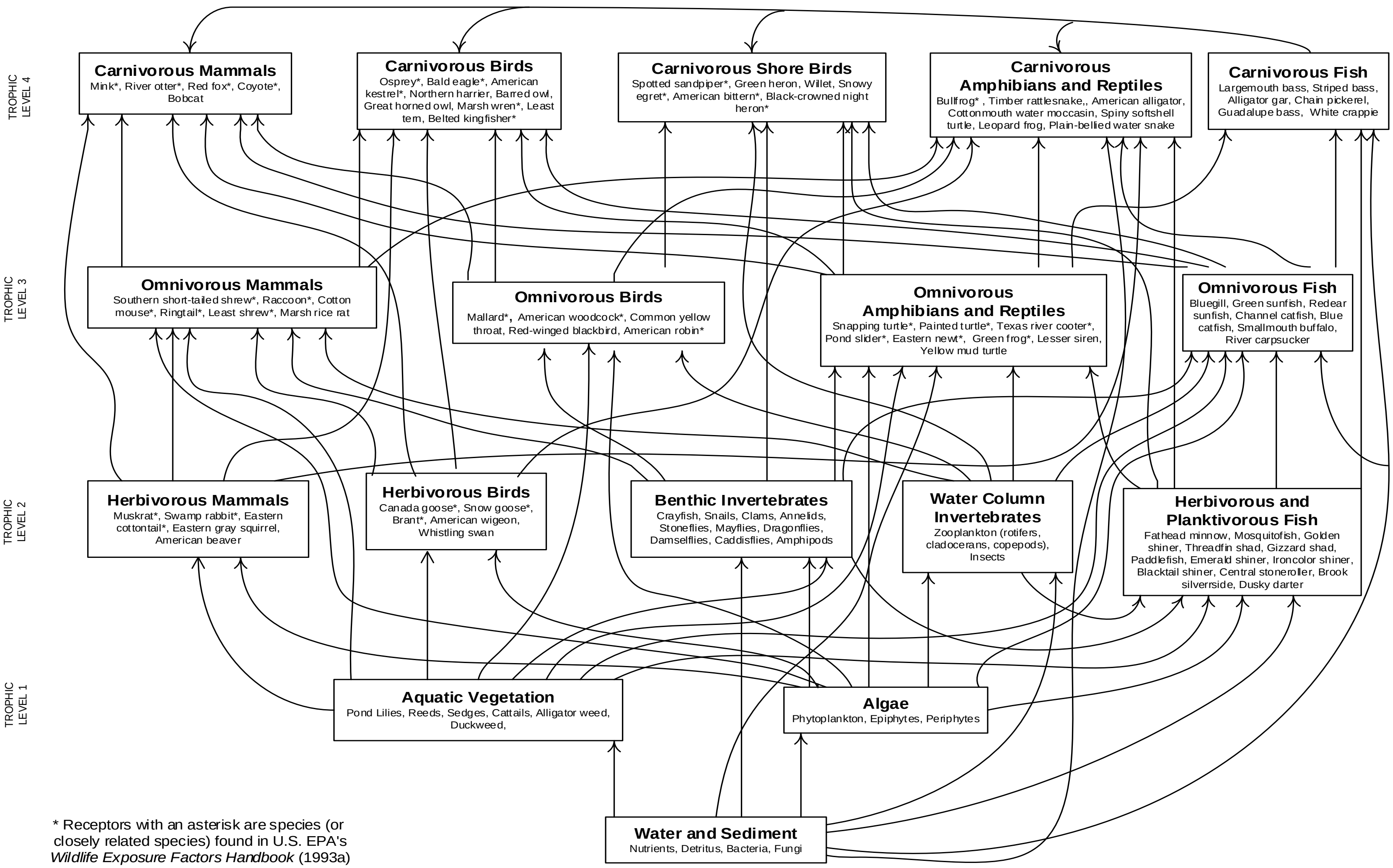
The first trophic level is represented by aquatic vegetation such as cattails, duckweed and algae.

Trophic level two includes herbivorous mammals and birds, benthic invertebrates, water column invertebrates and herbivorous and planktivorous fish. Herbivorous mammals include species such as the muskrat and swamp rabbit. Herbivorous birds include the Canada goose and snow goose. Benthic invertebrates include crayfish and snails. Water column invertebrates include zooplankton and insects. Herbivorous and planktivorous fish include the fathead minnow and mosquito fish.

Trophic level three is represented by omnivorous mammals, birds, amphibians and reptiles, and fish. Omnivorous species ingest a variety of plants and tissues of other species. Omnivorous mammals include the southern short-tailed shrew and raccoon. Omnivorous birds include the mallard and red-winged blackbird. Omnivorous amphibians and reptiles include the eastern newt and snapping turtle. Omnivorous fish include bluegill and sunfish.

Trophic level four is represented by carnivorous mammals, birds, shore birds, amphibians and reptiles, and fish. Carnivorous species prey upon other species found in trophic levels two and three. Carnivorous mammals include the mink and river otter. Carnivorous birds include the osprey and belted kingfisher. Carnivorous shore birds include the spotted sandpiper and green heron. Carnivorous amphibians and reptiles include the bullfrog and plain-bellied water snake. Carnivorous fish include largemouth bass and alligator gar.

Some species in the figure (or closely related species) are discussed in EPA's 1993 Wildlife Exposure Factors Handbook and are noted with an asterisk in the figure. These species include the muskrat, swamp rabbit, Canada goose, snow goose, southern short-tailed shrew, raccoon, mallard, snapping turtle, eastern newt, mink, river otter, osprey, belted kingfisher, spotted sandpiper, green heron, and bullfrog.



* Receptors with an asterisk are species (or closely related species) found in U.S. EPA's *Wildlife Exposure Factors Handbook* (1993a)

Figure A.6. Freshwater-systems food web.

Figure A.7 shows an estuarine-systems food web, where saline water and sediment represent the basis of the food web and contain nutrients, detritus, bacteria, and algae. There are four trophic levels that are supported by water and sediment.

The first is represented by aquatic vegetation such as cordgrass and algae.

Trophic level two includes herbivorous mammals and birds, benthic and water column invertebrates, and herbivorous and planktivorous fish. Herbivorous mammals include species such as the muskrat and swamp rabbit. Herbivorous birds include the Canada goose and snow goose. Benthic invertebrates include polychaetes and grass shrimp. Water column invertebrates include copepods and other zooplankton. Herbivorous and planktivorous fish include killifish and sheepshead minnow.

Trophic level three is represented by omnivorous mammals, birds, crustaceans, reptiles, and fish. Omnivorous species ingest a variety of plants and tissues of other species. Omnivorous mammals include the marsh rice rat and raccoon. Omnivorous birds include the mallard and herring gull. Omnivorous crustaceans include blue and stone crabs. Omnivorous reptiles include sea turtles. Omnivorous fish include spot and catfish.

Trophic level four is represented by carnivorous mammals, birds, shore birds, reptiles, and fish. Carnivorous species prey upon other species found in trophic levels two and three. Carnivorous mammals include the mink and river otter. Carnivorous birds include the bald eagle and marsh wren. Carnivorous shore birds include the spotted sandpiper and black-crowned night heron. Carnivorous reptiles include the American alligator and cottonmouth water moccasin. Carnivorous fish include flounder and red drum.

Some species in the figure (or closely related species) are discussed in EPA's 1993 Wildlife Exposure Factors Handbook and are noted with an asterisk in the figure. These species include the muskrat, swamp rabbit, Canada goose, snow goose, marsh rice rat, raccoon, mallard, marsh wren, mink, river otter, bald eagle, spotted sandpiper, and black-crowned night heron.

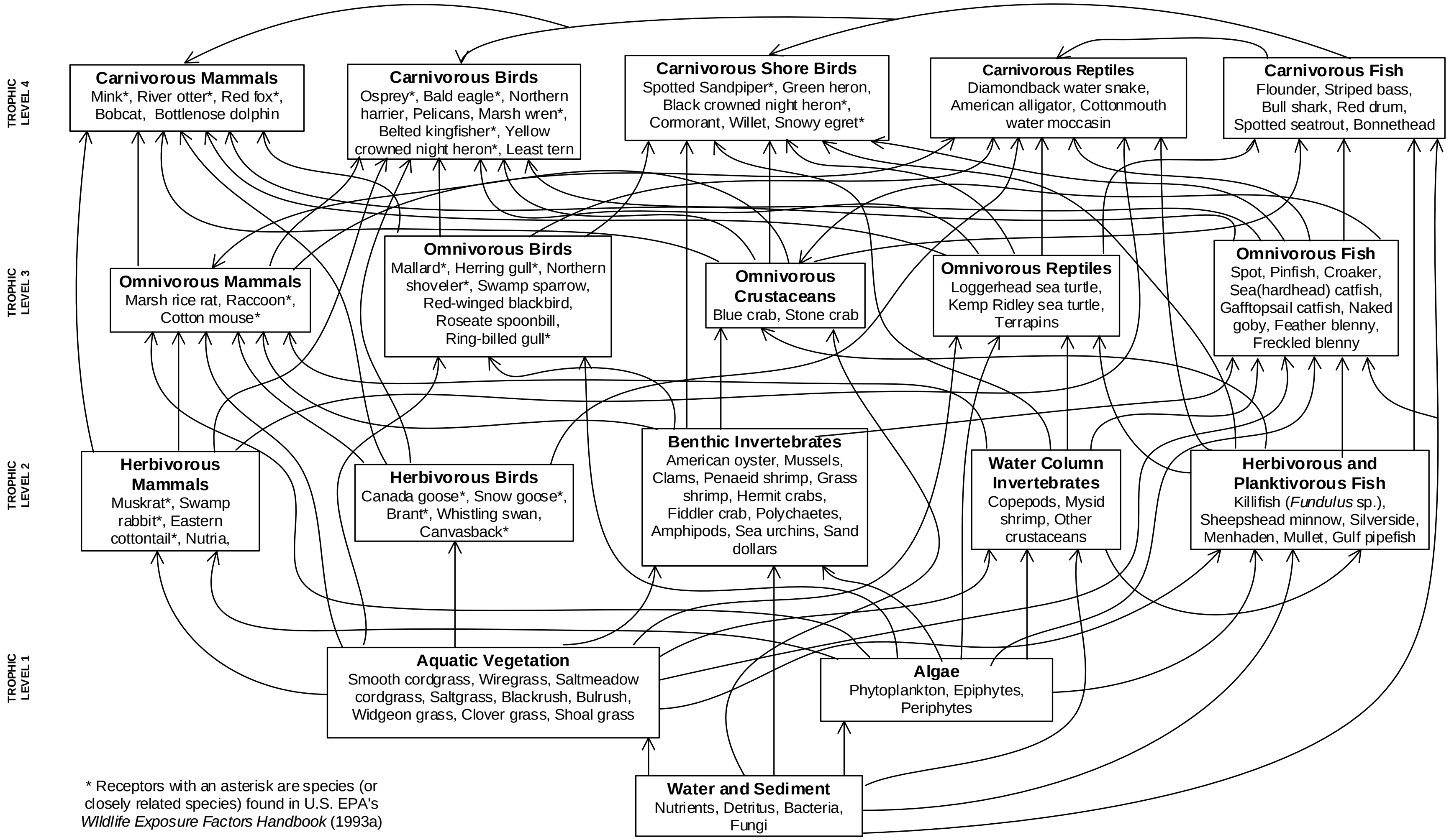


Figure A.7. Estuarine-systems food web

Appendix B: Selected Measurement Receptors for Evaluation of Minor Habitats

The measurement receptors in this appendix were selected from the species in the example food webs presented in **Appendix A** and can be used for evaluating ecological risk at affected properties with reduced habitats. These receptors are widely distributed across the state and are among the most commonly evaluated in ERAs. As stated in **6.2.3**, these species are incorporated into the Ecological PCL Database collectively, appearing as “Minor Habitat” and as individuals in the seven major habitats. Minor sub-habitats also occur, appearing as “Minor Habitat—Terrestrial” and “Minor Habitat—Aquatic.”

Since these measurement receptors represent those species that may forage in reduced habitats, not every feeding guild occurring in a major habitat is included. For example, a predominantly piscivorous mammal (e.g., mink) feeding guild is not included because a minor aquatic habitat (e.g., intermittent stream or small stock pond) could not reliably supply the necessary resources. Regardless of whether the affected property’s habitat is major, site specific, or minor, the person should ensure that sensitive species with the greatest potential for exposure are always included in the ERA. A brief discussion of each species’ life-history information appears below, and basic exposure inputs for these Minor Habitat species appear in Table B.1 at the end of this appendix.

Minor Habitat—Terrestrial

American Robin

The American robin (*Turdus migratorius*) is a good selection as the measurement receptor for the omnivorous-bird feeding guild based on:

- The robin serves an important function in seed dispersion for many fruit species, making it a valuable component of the ecosystem. It is also an important prey item for higher trophic-level predators.
- Its small home range, relatively high incidental soil ingestion, and diet of earthworms, snails, other invertebrates, seeds, and fruit result in the potential for a high degree of exposure to COCs.
- The robin occupies a variety of habitats, including forests, wetlands, swamps, and habitat edge where forested areas are broken with agricultural and range land.
- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

American Woodcock

The American woodcock (*Scolopax minor*) is a good selection as the measurement receptor for the omnivorous-bird feeding guild based on:

- The woodcock serves as an important prey item for higher trophic-level predators, such as hawks, owls, and weasels.

- Its high incidental soil ingestion and heavy diet of earthworms and invertebrates, found by probing the soil with its long prehensile-tipped beak, result in the potential for a high degree of COC exposure, particularly from lead, cadmium, and other heavy metals.

- The woodcock inhabits primarily woodlands and abandoned agricultural fields, particularly those with rich and moderately-to-poorly-drained loamy soils that support abundant earthworm populations.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Bobwhite Quail

The bobwhite (*Colinus virginianus*) is a good selection as the measurement receptor for the omnivorous-bird feeding guild based on:

- The bobwhite plays an important role in seed dispersion for many plant species and is an important prey item for snakes and small mammals. If habitat conditions permit, its numbers will increase rapidly, providing an additional food source for many predators. It is also valuable in controlling insect populations during certain times of the year.

- Its diet is mainly seeds and invertebrates although, in the winter, green vegetation can dominate the diet. It has a high potential for exposure through ingestion and dermal contact with soil during dust bathing.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Deer Mouse

The deer mouse (*Peromyscus maniculatus*) is a good selection as the measurement receptor for the omnivorous-mammal feeding guild based on:

- The deer mouse is preyed upon by owls, snakes, and small carnivorous mammals, making it a very important prey item. It also plays an important ecological role in seed and fruit dispersion for many types of vegetation. In addition, its burrowing activities influence soil composition and aeration.

- It is distributed statewide but is uncommon in the eastern, coastal, and southern parts of Texas. It lives in underground burrows, brush piles, or crevices among rocks.
- Its small home range, incidental soil ingestion, and its diet chiefly of seeds, fruits, bark, roots, herbage, and soil arthropods result in the potential for a high degree of exposure to COCs.
- Due to its burrowing and dietary habits, there is a high potential for direct and indirect exposure.
- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Desert Shrew

The desert shrew (*Notiosorex crawfordi*) is a good selection as the measurement receptor for the insectivorous-mammal feeding guild based on:

- The desert shrew is preyed upon by owls and snakes, making it a very important prey item.
- Desert shrews are found in the more arid, western and southern parts of the state but do not appear to be restricted to any habitat, as they have been found in cattail marshes, in beehives, under piles of cornstalks, among yuccas, in wood-rat nests, and beneath piles of brush and refuse.
- This shrew feeds largely on both larval and adult insects, including crickets, cockroaches, grasshoppers, moths, and beetles, and on centipedes and carrion. Its relatively high percentage of soil ingestion and its small home range provide a high potential for direct and indirect exposure.
- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Eastern Cottontail Rabbit

The eastern cottontail (*Sylvilagus floridanus*) is a good selection as the measurement receptor for the herbivorous-mammal feeding guild based on:

- The eastern cottontail is preyed upon by hawks, barn owls, opossums, coyotes, foxes, and small weasels, making it a very important prey item. This animal also plays an important ecological role in seed dispersion for many types of vegetation.
- It is active mostly in the twilight hours and at night, when it ventures to open pastures, meadows, or lawns to forage on grasses and forbs

primarily, but also on twigs and bark. It spends most of its day in beds in thickets and underground burrows. Due to its burrowing and dietary habits, there is a high potential for direct and indirect exposure.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weights, TRVs) also supports its selection as a measurement receptor.

Least Shrew

The least shrew (*Cryptotis parva*) is a good selection as the measurement receptor for the omnivorous-mammal feeding guild based on:

- Because of the least shrew's abundance and high population density, it is an important prey item for higher trophic-level predators and makes up a large portion of the diet of owls, hawks, and snakes. It also influences lower trophic-level populations through predation.

- This shrew is the smallest mammal in Texas, occurring in grasslands in eastern and central portions of the state, westward in the Panhandle to the New Mexico line, and to Val Verde County along the Rio Grande.

- It feeds on snails, insects, sow bugs, other small invertebrates, and carrion. Its main diet of invertebrates and its burrowing behavior result in a high potential of direct and indirect exposure to COCs.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Mourning Dove

The mourning dove (*Zenaida macroura*) is a good selection as the measurement receptor for the herbivorous-bird feeding guild based on:

- The mourning dove serves an important function in seed dispersion for many grain and grass species, making it a valuable component of the ecosystem. It is also an important prey item for higher trophic-level predators.

- It is the most widespread wild dove in Texas and the most heavily hunted bird in the country. It occurs primarily in open country, scattered trees, and woodland edges, feeding on the ground in grasslands, agricultural fields, backyards, and roadsides. Seeds, including cultivated grains, peanuts, wild grasses, weeds, and herbs make up most of its diet, although it will occasionally eat berries.

- It forages on the ground, and in heavily hunted areas may wind up eating fallen lead shot. Studies have found this problem is worst around fields specifically planted to attract the doves, and that about 1 in 20 doves wind up eating lead shot.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Nine-Banded Armadillo

The nine-banded armadillo (*Dasypus novemcinctus*) is a good selection as the measurement receptor for the invertivorous-mammal feeding guild based on:

- The armadillo is widespread in Texas, found in all but the western Trans-Pecos portion of the state in a variety of habitats, including forest, brush, scrub, and grasslands.

- It burrows in the ground for insects and other invertebrates and plays an important role in soil recycling and aeration through burrowing and tunnel excavation. Its burrowing and its subsequently high incidental soil ingestion result in a high potential for direct and indirect exposure to COCs.

- Many other wildlife species use and benefit from abandoned armadillo burrows, including rabbits, opossums, mink, cotton rats, striped skunks, burrowing owls, and some snakes.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Northern Cardinal

The Northern cardinal (*Cardinalis cardinalis*) is a good selection as the measurement receptor for the omnivorous-bird feeding guild based on:

- The cardinal serves an important function in seed dispersion for many fruit species, making it a valuable component of the ecosystem. It also provides economic benefits by eating insect pests such as boll weevils, cutworms, and caterpillars and is an important prey item for higher trophic-level predators.

- Cardinals eat mainly seeds and fruit, but also eat beetles, crickets, katydids, leafhoppers, cicadas, flies, centipedes, spiders, butterflies, and moths.

- The cardinal occupies a variety of habitats, including dense shrubby areas such as forest edges, overgrown fields, hedgerows, backyards, marshy thickets, mesquite, re-growing forests, and ornamental landscaping. Cardinals nest in dense foliage and look for conspicuous, fairly-high perches for singing. It is a year-round resident in most of the state.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Raccoon (Semiaquatic and Terrestrial)

The raccoon (*Procyon lotor*) is a good selection as the measurement receptor for the omnivorous-mammal feeding guild based on:

- The raccoon plays an important role in seed dispersal and is preyed upon by bobcats, coyotes, foxes, and great horned owls. It has a high potential for exposure due to its foraging in both semiaquatic and terrestrial habitats and direct contact with media.

- The raccoon is primarily an inhabitant of broadleaf woodlands, although it is rather common in the mixed-pine forests of southeastern Texas. It seldom occurs far from water, which seems to have more influence on its distribution than does any type of vegetation.

- It is strictly nocturnal. Its fondness for water is well-known and, except in seasons when fruits, nuts, and corn are maturing, it does most of its foraging near or in bodies of water.

- The den is usually a large hollow tree or hollow log in which the animal spends the daylight hours sleeping and in which it also rears its young. The raccoon is an opportunistic feeder but primarily eats fleshy fruits, nuts, acorns, corn, grains, insects, frogs, crayfish, and eggs. In summer and early autumn raccoons develop a fondness for adult and larval wasps and their stored foods. In winter, they concentrate in the river bottoms and subsist largely on acorns and crayfish.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Red Fox

The red fox (*Vulpes vulpes*) is a good selection as the measurement receptor for the carnivorous-mammal feeding guild based on:

- As a top predator, the red fox has a high potential for exposure due to bioaccumulation though the food chain. It is a valuable component to ecosystem structure in regulating the abundance, reproduction, distribution, and recruitment of lower-trophic-level prey.

- Most of its diet consists of small rodents, rabbits, birds, wild fruits and berries, and insects. It is distributed across eastern and central Texas to the Trans-Pecos, inhabiting mixed woodlands interspersed with farms and pastures, coastal areas, and crevices in rocky outcrops.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Red-Tailed Hawk

The red-tailed hawk (*Buteo jamaicensis*) is a good selection as the measurement receptor for the carnivorous-bird feeding guild based on:

- The red-tailed hawk's position as a high trophic level predator makes it a valuable component of terrestrial food webs through its regulation of populations of lower-trophic-level prey species.

- It is widely distributed across Texas among a diversity of habitat types ranging from woodlands to pastures. Its diet includes small mammals (such as mice, shrews, voles, rabbits, and squirrels), birds, lizards, snakes, and large insects.

- Red-tailed hawks have shown sensitivity to many COCs that disrupt reproduction or egg development.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Southern Short-tailed Shrew

The southern short-tailed shrew (*Blarina carolinensis*) is a good selection as the measurement receptor for the omnivorous-mammal feeding guild based on:

- The southern short-tailed shrew is preyed upon by hawks, owls, snakes, foxes, weasels and skunks, making it very important to the health of the ecosystem.

- Short-tailed shrews occur in the eastern quarter of the state in forested areas and their associated meadows and openings. Adequate cover and food appear to be more important in determining their presence than type of soil or vegetation.

- It is a small, mostly invertivorous mammal feeding on arthropods and earthworms. Its burrowing behavior results in a high potential for direct and indirect exposure to COCs and influences soil composition and aeration.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Texas Rat Snake

The Texas rat snake (*Elaphe obsoleta lindheimeri*) is a good selection as the measurement receptor for the carnivorous-reptile feeding guild based on:

- The Texas rat snake is a very common nonvenomous snake found throughout the eastern half of the state occurring in shortgrass, tallgrass, and coastal prairies, shrub-scrub, and upland forests. It is a large snake capable of growing more than 6 feet in total body length.
- Its diet includes small mammals (such as rats, mice, shrews, voles, rabbits, and squirrels) and birds and their nestlings. It is an effective and efficient predator of disease-spreading rodents.
- Its position as a high trophic-level predator makes it a valuable component of terrestrial food webs through its regulation of populations of lower-trophic-level prey species.

Virginia Opossum

The Virginia opossum (*Didelphis virginiana*) is a good selection as the measurement receptor for the omnivorous-mammal feeding guild based on:

- The opossum is beneficial for helping to control the overpopulation of snakes, rodents, insects, and ticks.
- The opossum is mostly solitary and strictly nocturnal, venturing forth to feed shortly after dark. It feeds on a variety of foods, including rats, mice, young rabbits, birds, insects, crustaceans, frogs, fruits, and vegetables.
- Opossums are primarily inhabitants of deciduous woodlands but are often found in prairies, marshes, and farmlands. Hollow trees and logs are preferred sites, but opossums will den in woodpiles, rock piles, crevices in cliffs, under buildings, in attics, and in underground burrows. Since they are not adept at digging burrows for themselves they make use of those excavated by other mammals.
- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

White-Footed Mouse

The white-footed mouse (*Peromyscus leucopus*) is a good selection as the measurement receptor for the omnivorous-mammal feeding guild based on:

- The white-footed mouse plays an important role in seed dispersal and is an important food source for raptors, snakes, and other mammals, including cats, weasels, and foxes.

- It is a medium-sized, short-tailed mouse distributed across the state and has a very small home range. It feeds on nuts, seeds, fruits, beetles, caterpillars, and other insects. Due to its burrowing and dietary habits, there is a high potential for direct and indirect exposure.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Minor Habitat—Aquatic

Belted Kingfisher

The belted kingfisher (*Ceryle alcyon*) is a good selection as the measurement receptor for the carnivorous-shorebird feeding guild based on:

- As a higher-trophic-level predator, the belted kingfisher plays an important role in the ecosystem by regulating lower trophic-level prey populations and influencing species composition.

- It is found statewide along rivers, streams, lakes, ponds, coasts, and estuaries. It looks for prey from a perch that overhangs water, such as a bare branch, telephone wire, or pier piling. Its diet consists mostly of fish near the surface or in shallow water; however, it also ingests some invertebrates.

- It nests in burrows in earthen banks, generally near suitable fishing areas. Due to its dietary habits and relatively small body weight, there is a high potential for exposure, particularly when fish take up COCs from sediment and water.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Great Blue Heron

The great blue heron (*Butorides virescens*) is a good selection as the measurement receptor for the carnivorous-wading bird feeding guild based on:

- As a higher-trophic-level predator, the great blue heron plays an important role in the ecosystem by regulating lower trophic-level prey populations and influencing species composition

- The Great blue heron is the largest heron in North America and lives in both freshwater and saltwater habitats, and forages in grasslands and agricultural fields, where it stalks frogs and mammals. It is capable of living in almost any wetland environment found in its range. It can be found in mangrove swamps, flooded meadows, lake edges, riverbanks, ocean shorelines, fresh and saltwater marshes and is a year-round resident in Texas.

- The great blue heron eats mostly fish but will eat nearly anything within striking distance, including fish, amphibians, reptiles, small mammals, insects, and other birds. It is known to stalk voles and gophers in fields, capturing rails at edges of marsh, and eating many species of small waterfowl.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Green Heron

The green heron (*Butorides virescens*) is a good selection as the measurement receptor for the carnivorous-shorebird feeding guild (particularly when there are no perches for a kingfisher) based on:

- As a higher-trophic-level predator, the green heron plays an important role in the ecosystem by regulating lower trophic-level prey populations and influencing species composition

- The green heron lives along wooded water margins and its general distribution is limited by the availability of wetlands. It frequents both saltwater and freshwater, showing great flexibility in habitat choice. Favored habitats are mangrove-lined shores and estuaries, and dense, woody vegetation fringing ponds, rivers and lakes. It nests in wooded and swamp patches, over water or in plants near water.

- The green heron mainly eats fish and invertebrates but is an opportunistic forager with a broad prey base, depending on the availability of species. It is known to exploit superabundant food resources, such as breeding frogs. Its invertebrate prey includes: leeches, earthworms, dragonflies, damselflies, water bugs, grasshoppers, and crayfish. Some of the many fish it eats are: minnows, sunfish, catfish, perch, eels, and, in urban areas, goldfish. Other vertebrates it eats are rodents, lizards, frogs, tadpoles, and snakes.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Marsh Rice Rat

The marsh rice rat (*Oryzomys palustris*) is a good selection as the measurement receptor for the omnivorous-mammal feeding guild based on:

- The marsh rice rat plays an important role in seed dispersal and is a major food item for many predators, including raptors, cats, weasels and snakes. The marsh rice rat has a high potential for exposure due to its aquatic diet and direct contact with media.

- It is found in eastern Texas west to Brazos County and south to Cameron County. It inhabits marsh and wetland areas where it feeds on crabs, insects, fruits, snails, carcasses, and aquatic plants, although the types of food vary with season and availability. It is semiaquatic and does not hesitate to swim or dive to escape capture.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weights, TRVs) also supports its selection as a measurement receptor.

Marsh Wren

The marsh wren (*Cistothorus palustris*) is a good selection as the measurement receptor for the omnivorous-bird feeding guild based on:

- The marsh wren consumes large numbers of benthic and aquatic invertebrates, thus regulating their populations, which makes it a valuable component of the ecosystem. Its main predators are snakes and turtles which prey heavily upon the eggs.

- It is common throughout Texas, inhabiting freshwater, brackish, and saltwater marshes. Its diet consists mainly of benthic and aquatic invertebrates (though it may eat snails and spiders), rendering it susceptible to accumulation and toxicity of bioaccumulative COCs.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Plain-Bellied Water Snake

The plain-bellied water snake (*Nerodia erythrogaster*) is a good selection as the measurement receptor for the carnivorous reptile feeding guild based on:

- As a higher-trophic-level predator, the nonvenomous plain-bellied water snake plays an important role in the ecosystem by regulating lower trophic-level prey populations and influencing species composition.

- It is widespread in Texas and is almost always found near a permanent water source, like a lake, stream, pond, or other slow-moving body. Its diet consists mainly of fish, but it also consumes salamanders and frogs.

- Its mouth has a white interior, resulting in it being misidentified frequently as the venomous cottonmouth. It hides in trees over streams. When feeding, it anchors itself to vegetation and holds its mouth open to catch fish and other prey that pass by. Although an efficient predator, it is often prey for many kinds of birds, mammals, turtles, snakes, and large fish.

Spotted Sandpiper

The spotted sandpiper (*Actitis macularia*) is a good selection as the measurement receptor for the invertivorous-carnivorous shorebird feeding guild based on:

- The spotted sandpiper plays an important role in the ecosystem by regulating lower trophic-level prey populations and influencing species composition.
- It inhabits a wide variety of habits usually associated with water or marsh. It requires open water for bathing and drinking, semi-open habitat for nesting, and dense vegetation for breeding.
- It has a high potential for exposure through ingestion of benthic invertebrates and significant portions of incidental sediment.
- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Yellow-Crowned Night Heron

The yellow-crowned night heron (*Nyctanassa violacea*) is a good selection as the measurement receptor for the invertivorous-carnivorous-shorebird feeding guild based on:

- The yellow-crowned night heron plays an important role in the ecosystem by regulating lower trophic-level prey populations and influencing species composition.
- It roosts in trees in wet woodlands, swamps, and low coastal shrubs. It frequents both saltwater and freshwater. Favored habitats are mangrove-lined shores and estuaries, and dense, woody vegetation fringing ponds, rivers, and lakes.
- It feeds primarily on freshwater and saltwater crustaceans, including marsh, fiddler, ghost, mud, and blue crabs. In inland areas, it feeds almost exclusively on crayfish. This results in a high potential for exposure through ingestion of benthic invertebrates and significant portions of incidental sediment.
- It is found year-round along the crustacean-rich Texas coast, but can also breed inland by feeding on crayfish in streams. The timing of the breeding season depends on when crabs emerge in the spring, which depends on local temperatures.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weight, TRVs) also supports its selection as a measurement receptor.

Table B.1. PCL Database exposure inputs for Minor Habitat measurement receptors.

Species	Body Wt. (kg)	Dietary Comp. (percent)	Food Ingestion Rate (kg/kg-d)	Soil or Sediment Ingestion (percent)	Home Range (acres) [hectares]
American Robin	0.0773	50 earthworms; 50 fruits	0.242	5.2	1.04 [0.40]
American Woodcock	0.169	90 earthworms; 10 arthropods	0.133	10.4	25.9 [10.48]
Belted Kingfisher	0.148	85 fish; 15 invertebrates	0.158	2.0	3.11 [1.26]
Bobwhite Quail	0.180	75 vegetation; 25 arthropods	0.0723	9.3	8.9 [3.60]
Deer Mouse	0.018	60 vegetation; 40 arthropods	0.207	2.0	0.084 [0.03]
Desert Shrew	0.004	90 arthropods; 10 earthworms	0.18	7.0	0.73 [0.30]
Eastern Cottontail	1.2	100 vegetation	0.0615	6.3	7.0 [2.83]
Great Blue Heron	2.23	100 fish	0.036	3	14 [5.67]
Green Heron	0.227	75 fish; 25 benthic invertebrates	0.0381	3.0	14 [5.67]
Least Shrew	0.0055	90 arthropods; 10 earthworms	0.196	7.0	0.5 [0.20]
Marsh Rice Rat	0.051	70 benthic invertebrates; 30 vegetation	0.120	2.0	0.73 [0.30]
Marsh Wren	0.0106	100 benthic invertebrates	0.221	7.3	0.134 [0.05]

Species	Body Wt. (kg)	Dietary Comp. (percent)	Food Ingestion Rate (kg/kg-d)	Soil or Sediment Ingestion (percent)	Home Range (acres) [hectares]
Mourning Dove	0.120	100 vegetation	0.141	9.3	40 [16.19]
Nine-Banded Armadillo	5.75	100 soil arthropods	0.01414	17.0	8.0 [3.24]
Northern Cardinal	0.0448	50 arthropods; 50 vegetation	0.162	9.3	1.6 [0.65]
Plain-Bellied Water Snake	0.20	100 fish	0.00805	5.9	13.34 [5.40]
Raccoon Semiaquatic	5.411	90 benthic invertebrates; 10 fish	0.0358	9.4	1558 [630.50]
Raccoon—Terrestrial	5.411	80 vegetation; 10 soil arthropods; 10 small mammals	0.0358	9.4	1558 [630.50]
Red Fox	4.535	75 mammals and birds; 20 vegetation; 5 arthropods	0.0298	2.8	2564.5 [1037.82]
Red-Tailed Hawk	1.138	100 small mammals	0.0316	2.8	1722 [696.87]
Southern Short-Tailed Shrew	0.009	85 arthropods; 15 earthworms	0.163	7.0	2.4 [0.97]
Spotted Sandpiper	0.0425	100 benthic invertebrates	0.220	18	5.0 [2.02]
Texas Rat Snake	1.729	100 small mammals	0.00656	2.8	57 [23.07]
Virginia Opossum	5	50 arthropods; 40 vegetation; 10 small mammals	0.0278	9.4	11.4 [4.61]

Species	Body Wt. (kg)	Dietary Comp. (percent)	Food Ingestion Rate (kg/kg-d)	Soil or Sediment Ingestion (percent)	Home Range (acres) [hectares]
White-Footed Mouse	0.0224	60 vegetation; 40 arthropods	0.132	2.0	0.25 [0.10]
Yellow-Crowned Night Heron	0.605	100 crustaceans	0.0307	3.0	11.0 [4.45]

Sources: Davis and Schmidly (1994), Schmidly and Bradley (2016), Tennant (1998), U.S. EPA (1993a, 1999).

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Appendix C: Assessing and Minimizing Impacts to Protected Species

C.1 Protected-Species Statutes

When sampling and remediating in ecological habitat at the affected property, it is important to avoid or minimize impacts on wildlife, especially to threatened, endangered, or otherwise protected species—e.g., birds under the federal Migratory Bird Treaty Act (MBTA) of 1918, which protects nearly all native bird species. State and federal endangered species laws protect a variety of plant, wildlife, invertebrate, and fish species across a wide variety of habitats. Other federal and state statutes restrict activities that can be conducted in areas inhabited by threatened, endangered, or otherwise protected species. These potentially include:

- the Endangered Species Act (1973, amended 1978, 1982, 1988)
- the Marine Mammal Protection Act (1972)
- the Convention on International Trade in Endangered Species of Wild Fauna and Flora (1975)
- 31 TAC Chapters 65.171–76 and 69.1–9

In addition, sampling and remediation activities that have an adverse impact on ecological habitat may increase natural-resource damages and associated liabilities.

C.2 Determining the Potential Presence of Protected Species

The easiest way to assess the potential for impacts on a protected species is to gather information on the species and its habitats that may be present at the affected property. The TPWD lists the state's protected wildlife species at <tpwd.texas.gov/huntwild/wild/rehab/protected/>. An evaluation of potentially present protected species may require surveys to assess the property to confirm their presence or the availability of suitable habitat.

Before sampling, the person should at a minimum evaluate these two TPWD sources for information pertaining to sensitive resources:

- Rare, Threatened, and Endangered Species of Texas by County (<tpwd.texas.gov/gis/rtest/>). This database briefly describes the habitat requirements for each listed species. After a survey of the affected property, a qualified individual should be able to determine the likelihood that any protected species could occur at the site by comparing the database information with the site characteristics. In any event, an evaluation to determine the presence of protected species on the affected property is typically conducted as part of a Tier 2 SLERA.

4. The Texas Natural Diversity Database (<tpwd.texas.gov/huntwild/wild/wildlife_diversity/txndd/>). The TXNDD contains location-specific information on protected species, natural communities, and other significant features of conservation concern to the TPWD. This information can be obtained by submitting an e-mail request to the program as described on its website. The TPWD's response will include TXNDD records, reports, and geographic information system-compatible shape files of recorded locations for protected species and other rare resources on the topographic quadrangle of the affected property and surrounding area. The TPWD cautions that use and interpretation of the information on protected species are the responsibility of the recipient. A qualified biologist should read and understand the data limitations and apply the information accordingly.

If federally-listed species may be present, the U.S. FWS should also be contacted for additional site-specific data.

C.3 Sampling or Remediating in Ecological Habitat

When sampling or performing remediation activities in ecological habitat, site personnel should incorporate best management practices specifically designed to minimize disturbance of wildlife. For example, if these activities necessitate the removal of vegetation, personnel should avoid or minimize impacts to large contiguous tracts of vegetation (e.g., dense brush) to prevent or reduce fragmentation of habitat that provides food, cover, nesting, and loafing sites for wildlife. With landowner approval, any cleared woody vegetation should be stacked into piles to provide cover for wildlife. If possible, cleared or disturbed areas should be reseeded with locally adapted native grasses or other native ground coverings. The use of introduced species such as Bermuda grass (*Cynodon dactylon*) for revegetating is strongly discouraged.

The TPWD and TCEQ recommend that state-listed or federally listed wildlife species encountered at the affected property should be allowed to leave the area on their own; human contact should be avoided altogether. It is important that activities at the site not take place near any areas used for nesting, loafing, or rearing young. Protected species may only be handled by persons with a scientific collection permit obtained through the TPWD or the U.S. FWS. Also, if protected terrestrial plants or soil invertebrates are found on public land, wildlife-management agencies should be contacted. The person should notify and consult with the TPWD if they encounter state-listed species. If they observe a federally listed species, they should notify the U.S. FWS of the sighting, as it has wider regulatory jurisdiction over these species. If the listed species could be adversely affected by site activities, the person should also submit an endangered-species consultation letter to the appropriate U.S. FWS field-services office for review of activities at the site.

C.3.1 Migratory Bird Treaty Act

The MBTA at <www.fws.gov/laws/lawsdigest/migtrea.html> prohibits the intentional and unintentional taking of migratory birds, including their nests and eggs, except as permitted by the U.S. FWS. To comply with the MBTA, the

U.S. FWS recommends that any vegetation clearing be conducted outside the nesting season. Under the MBTA, the peak nesting season is March through August, although some species nest much earlier (e.g., eagles begin nesting in November and December). If sampling or remediation activities that result in clearing or trampling of vegetation must occur during the nesting season, it is recommended that a qualified biologist survey the vegetation at the affected property for nests beforehand. If active nests are identified, they should be avoided until the young have fledged or the nests have been abandoned. The U.S. FWS further recommends that, for activities requiring removal of vegetation, a buffer of vegetation (50 meters for songbirds and more than 100 meters for wading birds) remain around the nest until young have fledged or the nest is abandoned. If a nest must be disturbed, consult the regional MBTA permit office at <www.fws.gov/birds/policies-and-regulations/permits/regional-permit-contacts.php> to ensure compliance.

C.3.2 Less-Mobile and Rare Species

Many protected reptile species are highly mobile and can usually avoid being affected by sampling or remediation activities. However, they can lose their agility during cold periods and cannot easily leave an area. Some species, such as the state-listed Texas tortoise (*Gopherus berlandieri*), are generally less mobile, so remedial or sampling activities should be modified to prevent injury or impacts to these species. Impacts to rare species should be avoided to help prevent them from becoming listed. Rare species are included on the TPWD county lists and in the Texas Wildlife Action Plan <tpwd.texas.gov/publications/pwdpubs/pwd_pl_w7000_1187a/>, a comprehensive wildlife-conservation strategy.

C.3.3 Injury of a Protected Species

If a protected species is injured during sampling or remediation, it is best to contact a permitted wildlife rehabilitator, the TPWD, and the U.S. FWS. Information on injured (and orphaned) wildlife as well as a list of wildlife rehabilitators (by county) is available online at <tpwd.texas.gov/huntwild/wild/rehab/>.

C.4 Risk Assessment and Management Considerations

When protected species have been documented, or their habitats identified on an affected property, several considerations should be made during risk assessment and management. Where the estimated risks are already considered unacceptable to a protected species, the person should consult with the TCEQ and the Natural Resource Trustee representatives to determine if near-term actions are needed to alleviate exposure of wildlife to contaminated media. Such short-term actions may include hazing (e.g., via lasers, streamers, and scare cannons) or other methods that would prevent or reduce exposure of wildlife receptors to the COCs by temporarily discouraging them from entering the affected property. Actions of this kind will require close coordination with the Trustees to ensure that wildlife is not harmed (see **C.3**), and that the methods used are the most appropriate.

Where the potential remedial actions may be more detrimental to the protected species than the risk associated with continued exposure to COCs in the PCLE zone, the person may consider undertaking an ESA, as described in 30 TAC 350.33(a)(3)(B) of the TRRP rule and in **14.2**. The ESA can be a useful approach to ecological risk management when working in close partnership with the TCEQ and the Trustees. Undertaking an ESA may not be appropriate or allowed in all situations; therefore, discussions and consultations with the TCEQ and the Trustees are critical.