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Determining Representative Concentrations of Chemicals of Concern for Ecological Receptors

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Prepared by
Remediation Division

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November 2013



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Determining Representative Concentrations of Chemicals of Concern for Ecological Receptors

Overview

Objectives	To familiarize readers with the approach for determining representative concentrations, defining exposure areas, and evaluating potential hot spots for ecological exposure pathways.
Audience	Regulated community and environmental professionals.
References	<p>The Texas Risk Reduction Program (TRRP) rule, together with conforming changes to related rules, is contained in Title 30, Texas Administrative Code, Chapter 350 (30 TAC 350).</p> <p>Find links for the TRRP rule and preamble, Tier 1 PCL tables, and other TRRP information at <www.tceq.state.tx.us/goto/trrp>.</p> <p>TRRP guidance documents undergo periodic revision and are subject to change. Referenced TRRP guidance documents may be in development. Links to current versions are at <www.tceq.state.tx.us/goto/trrp/guidance.html>.</p> <p>To be used in conjunction with the TCEQ ERA Guidance (RG-263, <i>Conducting Ecological Risk Assessments at Remediation Sites in Texas</i>), as revised and updated.</p>
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ACRONYMS and ABBREVIATIONS

AOC	area of concern
APAR	Affected Property Assessment Report
ASTM	American Society for Testing and Materials
AUF	area use factor
BAF	bioaccumulation factor
BSAF	biota-sediment accumulation factor
\bar{C}_{gw}	groundwater representative concentration (or exposure point concentration)
COC	chemical of concern
DDT	dichlorodiphenyltrichloroethane
DQO	data quality objective
ECOTOX	U.S. EPA's Ecotoxicology database
EPC	exposure point concentration
ERA	ecological risk assessment
<i>ERAG</i>	<i>Conducting Ecological Risk Assessments at Remediation Sites in Texas</i> (TCEQ publication RG-263)
ERED	environmental-residue-effects database
ESA	ecological services analysis
HQ	hazard quotient
HSI	habitat suitability index
IQR	interquartile range
LC ₅₀	lethal concentration, 50 percent
LOAEL	lowest observed adverse effect level
MBTA	Migratory Bird Treaty Act
MDL	method detection limit
MQL	method quantitation limit
MW	monitoring well
NAPL	non-aqueous phase liquids
ND	non-detect
NELAC	National Environmental Laboratory Accreditation Conference

NOAA	National Oceanic and Atmospheric Administration
NOAEL	no observed adverse effect level
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PCL	protective concentration level
PCLE	protective concentration level exceedance
POE	point of exposure
RCRA	Resource Conservation and Recovery Act
SedGW	groundwater-to-sediment PCL
SedGW _{eco}	groundwater-to-sediment PCL protective of ecological exposure pathways
SLERA	screening level ecological risk assessment (Tier 2)
SPM	solidified petroleum material
SSERA	site-specific ecological risk assessment (Tier 3)
^{SW} GW	groundwater-to-surface water PCL
^{SW} GW _{eco}	groundwater-to-surface water PCL protective of ecological exposure pathways
^{SW} SW	surface water PCL
SWMU	solid waste management unit
TAC	Texas Administrative Code
TCE	trichloroethene
TCEQ	Texas Commission on Environmental Quality
TDS	total dissolved solids
TMDL	Total Maximum Daily Load
TOXNET	U.S. National Library of Medicine's Toxicology Data Network
TPWD	Texas Parks and Wildlife Department
TRRP	Texas Risk Reduction Program
TRV	toxicity reference value
TSWQS	Texas Surface Water Quality Standards
TXNDD	Texas Natural Diversity Database
U.S. ACE	United States Army Corps of Engineers

U.S. EPA	United States Environmental Protection Agency
U.S. FWS	United States Fish and Wildlife Service
UCL	upper confidence limit
USGS	United States Geological Survey
WOE	weight of evidence
303(d) List	Federal Clean Water Act list of impaired waters

1

Introduction

This document contains guidance on determining representative concentrations of chemicals of concern for use in ecological risk assessments conducted under the Texas Risk Reduction Program. Specifically, it addresses:

- ensuring that data used in ERAs are representative and appropriate for each exposure medium
- defining the representative concentrations (called exposure point concentrations) for various ecological exposure pathways and routes
- defining exposure areas for ecological exposure pathways
- evaluating potential hot spots for ecological exposure pathways
- outliers and composite sampling related to ecological exposure pathways

This document is intended principally for persons conducting a Tier 2 screening level ecological risk assessment (SLERA), as specified in the TRRP rule at 30 TAC 350.77(c). Elements of this document (particularly the discussions of data for assessment for each medium) may be useful for persons preparing a Tier 3 site-specific ecological risk assessment (SSERA) as specified in the TRRP rule at 30 TAC 350.77(d).

1.1 Relationship to the TCEQ Ecological Risk Assessment Guidance

This guidance is not meant to replace the existing TCEQ ERA guidance (publication RG-263, *Guidance for Conducting Ecological Risk Assessments at Remediation Sites in Texas [ERAG]*, TNRCC 2001)¹ or the update to the *ERAG* (TCEQ 2006). This manual addresses aspects of ecological evaluations not discussed in detail in the *ERAG*. Persons conducting ERAs for TRRP sites should consult both this document and the *ERAG*. To avoid duplication and to assist the user, relevant sections of the *ERAG* are noted throughout this document.

1.2 Key Terminology

This document contains 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 are defined as follows:

¹ During the development of this publication, the *ERAG* is under revision. References to the *ERAG* throughout apply to the *ERAG*, as revised. Additionally, some topics are more current in the update to the *ERAG* (TCEQ 2006) rather than RG-263. These include, for example, the benchmarks and Table 3.1 (regarding bioaccumulative COCs). As long as the update is still in use, persons should ensure that they are using the most current reference of the two, depending on the topic in question.

affected property. The entire area (i.e., on-site and off-site; including all environmental media) which contains releases of COCs at concentrations equal to or greater than the assessment level applicable for residential land use and groundwater classification [30 TAC 350.4(a)(1)]. See also *assessment level*—the fundamental component of the definition of *affected property* as it relates to ecological exposure pathways.

assessment level. A critical protective concentration level (PCL) for a COC used for affected property assessments where the human-health PCL is established under a Tier 1 evaluation, except for the PCL for the soil-to-groundwater exposure pathway which may be established under Tier 1, 2, or 3, and ecological PCLs which are developed, when necessary, under Tier 2 and/or 3 in accordance with 30 TAC 350.77(c) and/or (d), respectively, (relating to ERA and development of the ecological PCL). The complete definition appears in the TRRP rule [30 TAC 350.4(a)(3)]. The most important concept to bear in mind is that the affected property boundaries are determined by the assessment level for a given COC, and the assessment level is based on the lower of the human-health and ecological PCLs. **2.1.1** provides further discussion of assessment levels with emphasis on the conundrum that there are no up-front ecological Tier 1 PCLs (similar to the TRRP human-health “look up” tables). There is one exception, and that is the application of the “de minimis” exclusion criterion in Tier 1 [see 30 TAC 350.77(b) of the TRRP rule and **2.2**, below] where the size of the affected property is determined solely with human-health PCLs. In this instance, the assessment level is the lower of the human-health Tier 1 total soil combined PCL and the human-health soil-to-groundwater PCL appropriate for the groundwater classification at an affected property.

attractive nuisance. A localized ecological condition created by human actions that attracts one or more species of biota, yet is detrimental to their health and/or existence. An attractive nuisance occurs when habitat and COC-containing media and/or food are readily available and the location is attractive to ecological receptors.

chemical of concern (COC). Any chemical that has the potential to adversely affect ecological or human receptors due to its concentration, distribution, and mode of toxicity. The term is often used interchangeably with “contaminant.” The complete definition is provided in the TRRP rule [30 TAC 350.4(a)(11)].

ecological risk assessment (ERA). 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 [30 TAC 350.4(a)(28)].

exposure area. Discussed throughout the document for the various receptor and exposure-pathway combinations. For an exposure medium (e.g., sediment or surface water), the exposure area is generally the area within the affected property throughout which receptors or community organisms may reasonably be assumed to move, and where contact with the exposure medium—direct or indirect (from ingestion of food or prey)—is likely at all locations. For sediment and surface water, the default exposure area is the entire affected property. For

soil, the default exposure area is *ecological habitat*—that portion of the affected property soils that does not meet the Tier 1 exclusion criteria at 30 TAC 350.77(b). For all three media, the affected property (or *ecological habitat for soils*) may be divided into smaller exposure areas, as a result of a receptor's specific natural history needs or because of variations in exposure caused by natural or physical anthropogenic effects. In some cases, exposure areas are delineated by the bounds of program-defined operable units within the affected property. These operable units may be larger or smaller than the available or required habitat of species of interest. This definition is specific to ecological exposure pathways. It should not be confused with the definition in the rule [30 TAC 350.4(a)(33)], which is specific to human-health exposure pathways.

Natural Resource Trustees. The state and federal agencies designated by law to act on behalf of the public as trustees of natural resources (e.g., water, air, land, wildlife). In Texas, the Natural Resource Trustee agencies are: TCEQ, Texas Parks and Wildlife Department (TPWD), the Texas General Land Office, the U.S. Department of Commerce (represented by the National Oceanic and Atmospheric Administration), and the U.S. Department of the Interior (represented by the U.S. Fish and Wildlife Service) [30 TAC 350.4(a)(58)].

person. An individual, corporation, organization, government or governmental subdivision or agency, business trust, partnership, association, or any other legal entity [30 TAC 350.4(a)(62)]. Throughout this document, it denotes the regulated entity or environmental consultant that is performing the ERA.

point of exposure (POE). The location within an environmental medium where a receptor will be assumed to have a reasonable potential to come into contact with COCs. The point of exposure may be a discrete point, plane, or an area within or beyond some location [30 TAC 350.4(a)(66)].

protective concentration level. The concentration of a COC which can remain within the source medium and not result in levels which exceed the applicable human-health risk-based exposure limit or ecological PCL at the POE for that exposure pathway [30 TAC 350.4(a)(68)].

representative concentration. The concentration calculated to represent ecological exposure conditions. In this guidance, this concentration is used to evaluate the potential for ecological exposure from sampled environmental media (soil, sediment, surface water, and groundwater). The representative concentration is generally assumed to represent an average² or typical level of exposure, expressed as a chemical concentration that a receptor may experience over an exposure area and time period consistent with the exposure pathway. In Tier 2 SLERAs, representative concentrations are usually concentrations measured in environmental media. Normally, a single value (i.e., the representative concentration) is needed for calculations associated with risk assessments. Herein, the term is synonymous with *exposure point concentration* (EPC).

² This is meant to be a general term and not necessarily the arithmetic mean.

Tier 1 Exclusion Criteria Checklist. Specified at 30 TAC 350.77(b) of the TRRP rule, the checklist is intended to aid in determining if further ecological evaluation is necessary at an affected property that is the subject of a TRRP response action. Exclusion criteria are those conditions at an affected property that preclude the need for a formal ERA due to incomplete or insignificant ecological exposure pathways as a result of the nature of the affected property setting or the condition of the affected property media [see 30 TAC 350.4(a)(32)]. This checklist (or a Tier 2 or 3 ERA, or a combination of these) must be completed for every affected property subject to the TRRP rule. This is referred to as the Tier 1 Checklist throughout this document.

1.3 Organization of the Guide

Each chapter (after this introduction) in this document is focused on a particular ecological exposure medium (soil, sediment, surface water, or groundwater). Each begins with a section titled “Data for Assessment,” which is not intended to duplicate existing guidance related to sampling, analysis, and data quality objectives (DQOs). Rather, it discusses what is appropriate and makes sense, to ensure that data used in ERAs are representative of, and appropriate for, each exposure medium. Each chapter then discusses various ecological exposure pathways specific to each exposure medium, gives guidance on determining exposure areas and EPCs, discusses special considerations for threatened and endangered species, and addresses the evaluation of potential hot spots. References cited are listed at the end of the document. The primary document is accompanied by five appendixes. These contain more details regarding:

- use of composite samples to support an ERA
- outlier tests
- examples of sediment data groupings
- specific calculation of a groundwater EPC as a source medium for ecological exposure pathways in surface water and sediment
- assessing and managing impacts to protected species while sampling and performing remediation activities

The various sections of the document are cross referenced throughout. Section (subsection, etc.) numbers are given in bold with only the number displayed.

1.4 Acknowledgments

This document was developed over several years with input and contributions from a multi-stakeholder work group made up of individuals from the TCEQ, environmental consulting firms, the regulated community, and Natural Resource Trustee personnel from multiple agencies. The TCEQ appreciates the contributions of these individuals, organizations, and agencies.

2

Soil Exposure Pathways

This section discusses the evaluation of soil exposure pathways for ecological receptors. 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, crops and livestock should be evaluated as potential receptors.³

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 at 30 TAC 350.77(b), the statistics used to estimate exposure concentrations, and the presence and evaluation of elevated concentrations (hot spots) of COCs.

2.1 Data for Assessment

This discussion is to ensure that the soil data set used to establish ecological exposure concentrations are representative and appropriate, such that the data accurately reflect the affected property's potential risks to ecological receptors. This section is not intended to replace existing TRRP guidance or the *ERAG* regarding the overall soil investigation design, sampling methods, and assessment approaches. The reader is encouraged to review these and other guidance documents. Additionally, Appendix A discusses the appropriateness of compositing soil samples for use in an ERA. Here, the focus is on those overarching assessment issues most important to evaluating ecological exposure to soils. Typical problem areas for soil assessments are highlighted. Detailed discussions of DQOs and quality assurance are not presented, given that extensive guidance on these topics currently exists and is widely available (e.g., United States Environmental Protection Agency [U.S. EPA] guidance on DQOs).⁴

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) regarding data collection proposed for use in soil exposure assessments. This could result in the development of an optional sampling work plan or discussion of the use and applicability of property-specific data collected from previous investigations at the property. The intent of early dialogue is to ensure that only those relevant and appropriate data are used to support the risk assessment. This would include a general discussion of how the

³ The *ERAG* specifically lists crops and livestock as representative receptors in Table 3-5. *ERAG* 3.9.3 gives direction on the need for, and evaluation of, ecological risks associated with crops and livestock.

⁴ Numerous DQO references are available at the U.S. EPA Quality System website at <www.epa.gov/QUALITY/dqa.html>.

proposed data are suitable and consistent with the objectives of the evaluation. Early dialogue with the TCEQ staff also promotes project efficiencies by minimizing comment exchange.

The remainder of this section discusses important factors for determining data acceptability for ecological exposure pathways for soil.

2.1.1 Adequacy and Appropriateness of Soil Data

Fundamental to any soil assessment is the characterization of the nature and extent of impacts on soil. Sufficient data should be collected to identify sources of contamination, potential migration pathways, and the depth and area of contamination. Persons evaluating the adequacy of the scope of the soil assessment should be cognizant of the TCEQ's ecological benchmarks,⁵ the TRRP Texas-Specific Soil Background Concentrations [30 TAC 350.51(m)] and property-specific background concentrations (if applicable), and laboratory method-quantitation limits (MQLs).

An *assessment level*, defined at 30 TAC 350.4(a)(3) of the TRRP rule, is generally 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** the ecological PCLs are developed (when necessary) under Tier 2 or 3 [30 TAC 350.77 (c–d)]. If the soil assessment levels are based on human-health soil PCLs alone, the evaluation may be inadequate for protecting ecological receptors.

Because ecological PCLs are not known at the time of the initial assessment (i.e., there is no lookup table), the *ERAG* (1.5.1) states that a person can choose to assume an assessment level based on human-health exposures (e.g., Tier 1 residential or commercial-industrial) or on ecological benchmark values, but warns that if the ecological PCLs developed in the ERA are lower than the previously assumed assessment level, additional sampling may be necessary to ensure proper delineation of COCs on the property. To ensure the assessment levels will not need to be subsequently lowered, the *ERAG* recommends persons assume an initial assessment level equal to the MQL. Here the TCEQ recommends using the standard available 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 (MDL), 30 TAC 350.54(e)(3) allows the agency to require a demonstration that a lower MQL is 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.

⁵ 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 screening level is lower than the value for plants and invertebrates.

If the affected property has failed the Tier 1 Checklist (see **2.2**) for the soil exposure pathway, meaning that the pathways from contaminated soil to ecological receptors are complete and significant, it is important to select a conservatively appropriate ecological assessment level. 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 as indicated in the previous paragraph, to avoid additional data collection to delineate contamination below the ecological assessment level.

Persons should also consider the location of ecological habitat, the likelihood of ecological receptors being present, and the quality of the habitat at the affected property when planning the soil assessment. Often, too few soil samples are collected in ecological habitat areas. Paucity 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, greater characterization or a more focused evaluation may be necessary for sites with higher quality habitat.

It is critically important to adequately characterize the nature and extent of contamination as it relates to ecological habitat and, subsequently, the appropriate EPC (discussed in **2.4.3**) 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 ecological EPC.⁶ Therefore, communication with the TCEQ ERA staff is recommended as persons plan 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 may result in an erroneous conclusion.

2.1.2 High-Biased and Low-Biased Data Distribution

Soil assessments evaluated for TRRP typically employ judgmental (i.e., biased) sampling as opposed to a random geospatial sampling regime. TRRP allows judgmental samples, as long as 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

⁶ TRRP affected properties will vary greatly in size, habitat, receptors, and COC distribution. Given this, the TCEQ is not suggesting a minimum sample number. The agency assumes that, for most TRRP sites, the person is not using some type of sampling design software to indicate the location and number of samples based on a desired level of performance and acceptance criteria. Absent such software, the first priority is to collect the number of samples needed to ensure that the nature and extent of soil impacts have been addressed. Additionally, 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.

known or potential source areas. 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 introduction of biases (high or low) should be discussed in the uncertainty analysis within the ERA. Alternatively, systematic sampling approaches (e.g., grid or random designs) may be considered.

Soil sample locations outside the boundaries of the affected property or the ecological exposure area should generally not be included in the calculation of the soil EPC. This statement may seem confusing, especially where soil concentrations exceed ecological benchmarks, yet are lower than the soil residential assessment level for human health. 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 **1.2** and **2.1.1**). In keeping with the TRRP rule at 30 TAC 350.51(l)(1), an EPC (see **2.4.3**) 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 affected property data, 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.

2.1.3 Soil Depth

For ecological exposure pathways, the TRRP rule denotes soil in the zone extending from ground surface to 0.5 feet in depth as *surface soil*, and soil in the zone between 0.5 feet and 5 feet in depth as *subsurface soil* [30 TAC 350.4(a)(86, 88)]. Conversely, for human-health exposure pathways, the TRRP rule defines *surface soil* as the soil zone extending from ground surface to 15 feet in depth (residential), and from ground surface to 5 feet in depth (commercial-industrial) [30 TAC 350.4(a)(88)]. *Subsurface soil* (for human-health exposure pathways) is defined as the portion of the soil zone between the base of surface soil and the top of the groundwater-bearing unit or units [30 TAC 350.4(a)(86)]. The different soil depths for human and ecological exposure pathways are illustrated in Figure 2.1.

Samples collected to evaluate human-health pathways are typically inappropriate for ERAs. In the inappropriate cases where human-health soil samples are used to address ecological exposure pathways, consider whether use of this greater depth interval may dilute out surface soil concentrations where the majority of the contamination is in the first half foot. Persons could compare soil concentrations for samples collected in the same bore hole at separate depths, or the highest measured concentrations for each COC for samples collected in the first half foot, with those collected at deeper depths. This would be appropriate for demonstrating that data from deeper soil intervals are a conservative representation of the actual surface soil concentrations at an affected property.

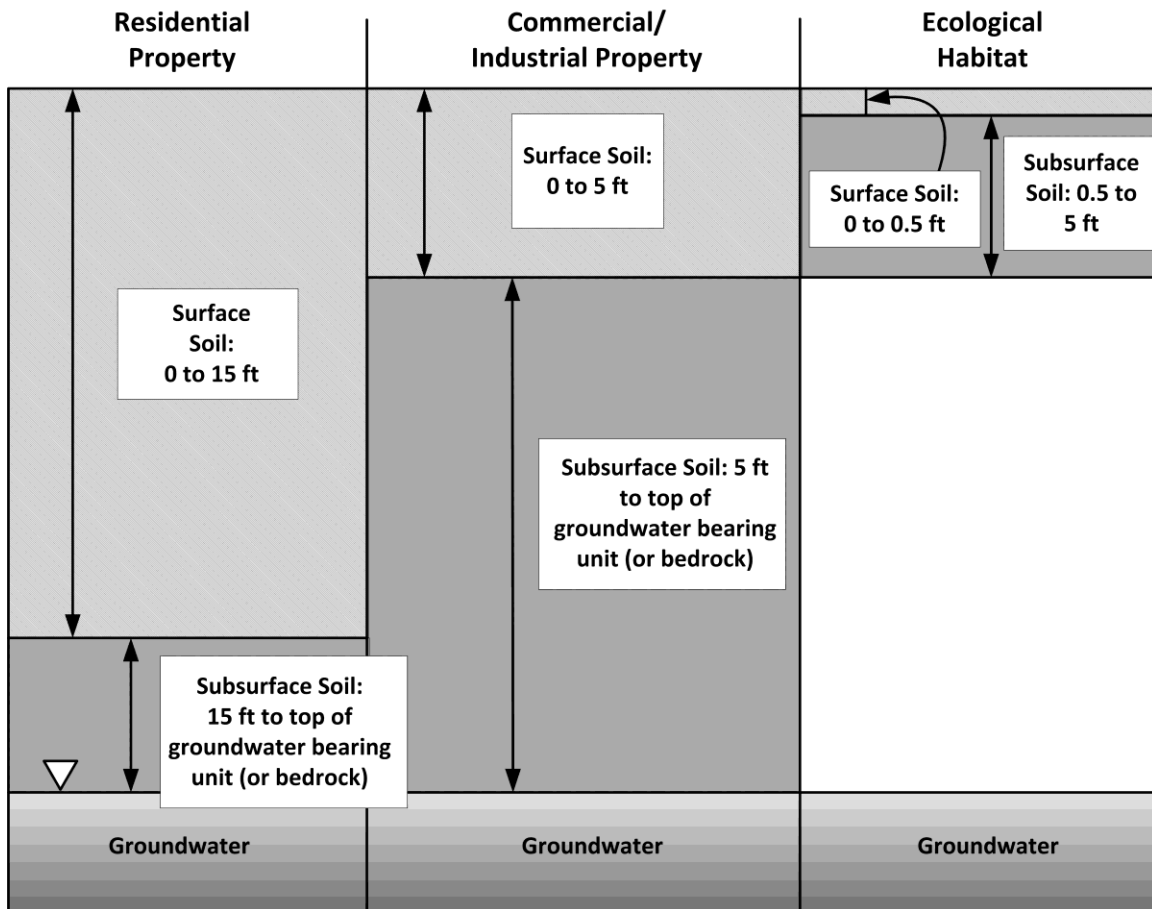


Figure 2.1. Surface and subsurface soil depths for human-health (residential and commercial-industrial) and ecological exposure pathways.

Use caution when mixing data from different depths, since property-related COC concentrations frequently vary with depth. Normally only surface soil data should be used in the ERA, unless there is a property-specific reason for considering deeper soil. For specific properties, 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.

2.1.4 Analytical Considerations

The accuracy and precision of analytical methodologies play a significant role in determining the suitability of soil data for use in a risk assessment. Data must meet the specifications in 30 TAC 350.54 and *Review and Reporting of COC Concentration Data* (RG-366/TRRP-13; TCEQ 2010c). 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 (NELAC) for the matrices, methods, and parameters of analysis. The analytical methods used should have

MQLs below the effect thresholds and soil benchmarks. TRRP [30 TAC 350.54(e)(3)] requires a standard available analytical method that provides a MQL below the necessary level of required performance for assessment and demonstration of conformance with critical PCLs. If that is not possible, persons should select the standard available analytical method that provides the lowest possible MQL for a given COC. This is especially critical for bioaccumulative COCs in soil such as polychlorinated biphenyls (PCBs), dioxins and furans, pesticides, and organochlorine compounds.

Historical soil data may prove useful for current assessments, particularly as the information may be used to develop new sampling plans. Historical data may certainly be included for qualitative discussions related to ecological exposures. However, more formal integration for quantitative risk assessment necessitates caution as it must meet the specifications in 30 TAC 350.54 and TRRP-13 if it will be used in the quantitative risk assessment to characterize ecological exposure conditions. Note that some provisions of TRRP-13 (such as the laboratory review checklist and the detectability-check sample) do not apply to data generated before February 2003. These provisions are discussed in detail in TRRP-13. Historical data not meeting the specifications in 30 TAC 350.54 and TRRP-13 cannot be used in the quantitative risk assessment. Consider the representativeness of historical data for characterizing current ecological exposure conditions—for example, that COC concentrations could change significantly with time. Soil data collected prior to a major physical disturbance (such as a removal) that are not representative of current conditions at the affected property should not be used in the ERA.

Laboratories typically prepare soil samples for analysis by removing rocks, pebbles, and plant debris. Normally, a representative portion of the sample is homogenized, and a subsample is collected from that portion for analysis. Persons should consult with their analytical laboratory to understand the exact nature of any sample manipulations. Appropriate questions could include how samples are homogenized and how subsamples are collected. Persons are advised to request standard operating procedures for preparing and processing samples from their laboratory.

2.1.5 Sample Sieving

Sieving is physically sorting a soil sample using screens of predetermined size to obtain uniform particle sizes. For a variety of reasons, persons commonly ask if soil sieving is appropriate at sites being evaluated for potential risk to ecological receptors. Possible reasons for soil sieving include:

- exclusion of large rocks and gravel
- selection for the more bioavailable fraction
- selection of coarse particles that may be used by birds as grit
- to better reflect the primary intent of environmental sampling (analysis of exposure media to support risk assessments)

Heterogeneity of materials in soil can influence COC concentrations, and thereby increase analytical variability. Following collection of a soil sample, vegetation (i.e., sticks, roots, leaf litter, and grasses) should be removed. Rocks and gravel should also be removed and discarded as they do not usually retain contaminants nor are amenable to laboratory analysis. Beyond this, 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. Various scenarios are discussed in the following paragraphs.

Metals and bioaccumulative COCs tend to concentrate in fine soil particles (Acosta et al. 2009; Ruby et al. 1999). Depending on the DQOs for the study, sieving and analysis of the fine fraction specifically could be beneficial for evaluating potential risks associated with this more readily bioavailable fraction. However, wildlife receptors arguably also contact the bulk soil, including larger particles. Where persons choose to sieve soil samples to specifically evaluate the finer fractions, the TCEQ recommends analysis of a representative number of samples in both the sieved and bulk forms (with rocks and plant debris removed) to address the uncertainty associated with this sample manipulation.

Soil sieving may also be useful to remove solidified petroleum material (SPM) when the primary intent of sampling is the assessment of the exposure media, rather than waste characterization. SPM, sometimes called *oil cake*, is heavily weathered crude oil resembling asphalt in appearance, consistency, and physical characteristics. The TCEQ recommends that large fragments of SPM (i.e., those retained by a No. 7 [2.8 mm] sieve) should be removed from the soil sample, and characterized separately as waste. This material may be the subject of separate risk assessment, removal, waste handling, or remediation-management decisions, depending on the nature of the constituents found. A similar approach can be used to assess soil containing visible fragments of slag, coal, coke, sulfur cake, etc. There may be other instances wherein sieving in the field is appropriate, such as for analyses of energetic materials. Because of the numerous decisions that can affect the quality of soil data, involve the TCEQ in the development of the sampling strategy. Any soil sieving should be clearly described in the risk assessment and discussed in the uncertainty analysis as appropriate. Moreover, the potential for larger fractions to be an ongoing source of exposure should also be addressed or discussed with the TCEQ (or both).

In the TCEQ's experience, sieving is most common when assessing soils at former small-arms firing ranges. The TCEQ's suggested approach for these types of sites, where lead is the primary COC and risk driver, is discussed in the remaining paragraphs. For ERAs, the soil fraction smaller than 2 mm is most commonly used in firing-range studies, and is thought to contain the most bioavailable lead fraction (Kaufman et al. 2007). Often the smallest soil fractions are characterized by the highest lead concentrations. For instance, in a study of lead shot in soil, the highest concentrations of lead were measured in soil particles passing through a 0.075 mm sieve (Duggan and Dhawan 2007). Additionally, many birds that use grit retain particles in the 2.8 to 0.5 mm size range (Peddicord and LaKind 2000).

For human-health exposure, U.S. EPA (2003) offers detailed advice on the appropriateness and size class for soil sieving associated with lead-impacted sites. This guidance is adapted herein to provide an approach for sieving soil samples that will be analyzed to support an ERA. The TCEQ recommends that soil samples be sieved twice—first with a No. 7 (2.8 mm) sieve to remove bulk debris, and then with a No. 60 (0.250 mm) sieve. The portion of the sample that passes through the No. 7 sieve, but is retained on the No. 60 sieve, is the coarse fraction. This fraction can also be used to represent the portion that may be intentionally ingested as grit by avian receptors, if that is an exposure route of concern. The portion passing through the No. 60 sieve is the fine fraction. This fraction is assumed to be most bioavailable and of the greatest potential for contributing to exposure via direct contact and incidental ingestion. The portion passing through the first sieve (No. 7) may be referred to as the “total” sample (i.e., coarse and fine fractions). Absent the grit-specific evaluation, the total soil concentration may be appropriate for predicting risks for future exposure scenarios. These larger fragments are an ongoing source of COCs to the biologically-available fraction as they weather over time.

Similarly, materials from firing range soils in the larger fraction (i.e., greater than 2.8 mm) may be an ongoing source of metals to the biologically relevant fraction. The uncertainty analysis should discuss the relative bioavailability and the potential that these larger fractions may be an ongoing source of contamination. Any plans to remove larger bullet and shell fragments from surficial soils should also be discussed in the uncertainty analysis.

2.1.6 Soil vs. Sediment

As described in more detail in the *ERAG* (3.9.2.6) and TCEQ (2005), 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 sediment associated with intermittent streams when water is present. In this case, an exposure modifying factor can be used to reflect the duration of exposure depending on the length of dry and wet cycles. Persons may evaluate the more conservative exposure scenario (i.e., that more likely to result in the higher hazard quotient [HQ] for a given COC for most receptors), if adequate justification is provided for not quantitatively evaluating the less conservative scenario. Persons may also provide property-specific information (e.g., personal communications with the landowner, site visits by the risk assessor or related staff, and historical flow data) that the stream is usually dry or flowing, together with an evaluation of the predominant scenario. The types of potentially exposed receptors will vary depending on the amount and duration of water present in the water body. This variability should be accounted for in the exposure evaluation.

2.2 Exclusion-Criteria Checklist

Exclusion criteria refers to those conditions at an affected property that preclude the need to conduct a formal ERA because there are incomplete or insignificant ecological exposure pathways due to the nature of the affected property setting or the condition of the affected property media [30 TAC 350.77(b)]. As stated in the *ERAG*, the purposes of the Tier 1 Checklist are to characterize the ecological setting of the affected property and to determine the existence of complete and potentially significant ecological exposure pathways.

The Tier 1 Checklist provides a tool for evaluating the potential ecological exposure to affected soil, groundwater, or surface water-sediments. The checklist is designed for use at 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 and endangered species). Instead, general affected property conditions are evaluated to determine whether affected media are present at locations or, in the case of soils, over a sufficient area that is attractive to ecological receptors such that significant exposure could occur.

Three of the four exclusion criteria in the Tier 1 Checklist concern the soil exposure pathway. The first of these addresses the affected-property setting and discusses the concept of *disturbed ground*—a location that is predominantly urban or commercial-industrial (and thus is characterized by human presence and activities) where any habitat that may have once existed has been altered, impacted, or reduced to such a degree that it is no longer likely to be used by ecological receptors for foraging or shelter. The next exclusion criterion evaluates the potential for exposure to contaminated soil. It focuses on soil COCs within the first 5 feet beneath ground surface where exposure is not prevented by a natural or artificial barrier. The final soil exclusion criterion evaluates the significance of the soil exposure through the concept of a *de minimis* land area of 1 acre or less.

As discussed in TCEQ (2005), it is appropriate to apply the results of the first two soil exclusion criteria to the conditions and questions associated with the *de minimis* criterion. This is to ensure the person only evaluates ecological habitat within the affected property that has the potential for wildlife exposure to COCs. Applying the criteria to the affected property in the order in which they are presented in the Tier 1 Checklist avoids the evaluation of non-ecological habitat, as discussed in the example in Common Issue #30 (see TCEQ 2005). The concept of evaluating only the ecological habitat (i.e., excluding disturbed ground) in an ERA was previously addressed in **2.1.1** and is also discussed in **2.4.2.2**.

2.3 Plant and Invertebrate Exposure to Soil

2.3.1 Assessment Considerations for Plant and Soil Invertebrates

2.3.1.1 Plant and Soil Invertebrate Populations

Populations of plant and soil invertebrate communities are important ecosystem components in that they are an energy and nutrient link between soil and upper trophic level receptors. In addition, plants (e.g., grasses, shrubs, and trees) afford 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 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 the habitat and foraging areas of wildlife that depend on them are frequently large enough to compensate for any localized losses in food or shelter. However, there are some property-specific 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. Persons 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' habitat, shelter, and forage. Also, if protected terrestrial plants or soil invertebrates can be potentially exposed to COCs on public 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.

Although potential risks to plants and soil invertebrates can be evaluated by comparing the highest measured COC concentrations with the soil screening benchmarks in Table 3.4 of the *ERAG* (or TCEQ 2006), those benchmarks are not solely used to evaluate risks to these receptors. Instead, the TCEQ assumes that the plant and invertebrate screening values are also protective of wildlife receptors, except where bioaccumulative COCs are present in soil (see **2.4** for guidance concerning the evaluation of potential risks to wildlife from COCs in soil).

2.3.1.2 Exposure Areas for Plants and Soil Invertebrates

Should a direct assessment of plants or soil invertebrates be deemed necessary, persons should consider the unique exposure characteristics of these organisms. For example, unlike most wildlife receptors, plants and soil invertebrates tend to be relatively sessile and may be confined to specific areas of an affected property

with little potential for lateral or vertical movement. As such, plant and soil invertebrate exposure areas directly and inseparably overlie their rooted or confined location and their exposure is generally constant. Accordingly, the plant and soil invertebrate exposure areas for consideration are wherever the organisms and affected soil overlap (i.e., only soils which do not meet the Tier 1 exclusion criteria at 30 TAC 350.77(b)).

Apart from the lateral distribution of plants and soil invertebrates, another important consideration is the depth of the biologically-active zone where plant roots and invertebrates may be found. As already discussed in **2.1.3**, the zero to five foot soil interval is considered as the POE for ecological soil exposure pathways. Although the majority of the root biomass is usually found within the first foot (and shallower for most grasses), burrowing soil invertebrates (and wildlife) in search of food and shelter may be found at greater depths within the biologically active zone. Moreover, COC concentrations in roots are highly variable and markedly decrease with depth (as do the concentrations of COCs in surface soil). Therefore, persons may determine rooting depth on a property-specific basis. This will help to fine-tune the exposure regime of any affected property COCs. Averaging exposure across the entire depth interval (i.e., surface to 5 feet) may underestimate the actual exposure. It should be noted that *surface soil*, defined as the soil zone extending from ground surface to 0.5 feet, is the default depth interval for the ERA. Subsurface soil (0.5 to 5 feet) is only evaluated where burrowing wildlife receptors may be at risk and where direct evaluation of terrestrial plants and invertebrates is warranted, as described in the previous section.

2.3.2 Exposure Point Concentrations for Plants and Soil Invertebrates

Because plant and soil invertebrate communities are sessile, the link between COC sources in soil and direct exposure or uptake of COCs is better defined than for more mobile receptors that may average exposure over the entire affected property. Nevertheless, the approach of using the 95 percent upper confidence limit (UCL) as the soil EPC for wildlife (see **2.4.3**) is appropriate for evaluating risks to plant and soil invertebrate communities, because potential impacts are evaluated at a population level for the entire affected property, which reflects community structure and ecological functional changes that could affect upper trophic level receptors. That is, the principal evaluation is to ensure that populations of upper trophic level receptors are not impacted due to a loss of food, shelter, or habitat. The exception would be where the affected property includes public lands that support protected plant or soil invertebrate species. In that case, more local evaluation of soil COCs may be needed, depending on the consultation with wildlife-management agencies. For example, if only a few specimens of a federally listed plant species are potentially at risk at an affected property, a wildlife management official may relocate these to an unimpacted area. In this case, the 95 percent UCL approach (as the EPC) could still be used.

2.4 Exposure of Wildlife Receptors to Soil

2.4.1 Purpose and Rationale

Soil is a key media of concern in terrestrial ecosystems because it directly and indirectly supports wildlife in Texas. 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.

Consistent with the importance of soil as an exposure medium, developing technically defensible approaches for soil evaluation should aim at adequate protection of wildlife from exposure to COCs at affected properties. Choices of appropriate receptors and ecological scale (the organism, population, or community) will be necessary. Discussion with the TCEQ staff before field activities may be necessary to save time and resources.

Unless the affected property can be used by threatened or endangered species, the ultimate goal of the soil investigation, assessment, and remediation stipulated by the TCEQ is the protection of wildlife populations. Methods and measures should benefit this ecological scale, unlike the threatened and endangered species, which require individual protection by federal and state law.

From the affected property assessment and remediation perspective, one of the key tools used to gauge the potential for risk to wildlife is the toxicity reference value. As discussed in the *ERAG* (3.9.5), TRVs are typically derived from toxicity studies (using a sensitive species) that are evaluated for population scale and relevant responses (such as growth, reproductive success, fecundity, offspring impacts, and mortality). When applied in a Tier 2 SLERA or a Tier 3 SSERA, the calculation of HQs using TRVs of this nature is intended to afford protection to wildlife at the population scale of biological organization. In contrast, for threatened or endangered species, protection should be ensured at the organism or individual scale.

Typically, birds and mammals dominate risk assessments for terrestrial biota. However, reptiles and amphibians may also be included as they are commonly found in Texas and may include sensitive or representative species. A qualitative or quantitative evaluation of amphibians and reptiles, depending on available information on toxicology and life history, should also be included if they are expected to occur at the affected property. A more rigorous evaluation is required where a threatened or endangered reptile or amphibian species may occur at the affected property. Additional discussion is provided in Common Issue #13 (see TCEQ 2005).

2.4.2 Assessment Considerations for Wildlife Receptors

2.4.2.1 Wildlife Populations

A *biological population* is a group of interbreeding organisms or individuals of the same species inhabiting a geographically restricted location at the same time. This group typically contains a number of individuals spanning a range of ages,

body sizes, adaptations, acclimations, and sensitivities to environmental and toxicological stresses. COCs and other stressors affect individuals and the consequences of these effects may be expressed in the population overall. However, a healthy population has a certain amount of resilience to loss; sensitive or heavily exposed individuals may perish, but resistant or adapted individuals may survive to ensure the propagation of the population. This resilience to loss is the rationale most often given for selecting populations as the endpoint for the protection of species that are neither threatened nor endangered.

A *local population* is defined as a group of individuals within an investigator-delimited area smaller than the geographic range of the species and often within a smaller area than that occupied by a population (Wells and Richmond 1995). The concept of a local population probably applies to most contaminant studies for waste sites where effects on populations are considered (Albers et al. 2000). For purposes of this guide, a *local population* is defined as all members of a population within an exposure area (defined in **2.4.2.2**) including transient or migratory species that may occupy or use the exposure area.

For assessing wildlife populations, a feeding guild approach is recommended by the TCEQ: 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 (often the most sensitive) species is selected for exposure assessment. Results for this indicator species (the *measurement receptor*) are intended to be descriptive (and protective) of all populations of species contained in that guild. Refer to the *ERAG* (3.6.2) for additional discussion of feeding guilds.

2.4.2.2 Exposure Areas for Wildlife Populations

For purposes of this guide, the *exposure area for soil* is defined as the *ecological habitat* 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). The generic approach presented in this guidance document, however, is to assume the entire *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 represents a receptor's *exposure area*, and this entire area 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 represent the exception rather than the norm. This aspect is discussed in more detail in **2.4.3.2**. Once the exposure area has been defined, the information and assumptions that support the identification of the exposure area should be included in the risk assessment discussion.

Habitat is defined as any physical area whose resources and conditions allow, or may allow, wildlife to live, forage, and reproduce for extended periods of time (i.e., be able to support long-term populations). *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.

2.4.2.3 Data Quality to Support the Exposure Assessment

To ensure adequate exposure assessment of wildlife, data for the affected property soil must first meet basic requirements for quality and accuracy (refer to **2.1** for more details). As specified in the *ERAG*, the TCEQ considers the biologically active zone in soil to extend to 5.0 feet in depth. However, most wildlife species are not exposed to soil depths beyond 0.5 feet (i.e., subsurface soil). Therefore, unless property-specific information may trigger it (i.e., burrowing or burrow-dwelling receptors are observed or anticipated), the exposure assessment for wildlife should normally focus on surface soil (0–0.5 feet).

In addition to relevant depth, a key item to look for in wildlife exposure assessment includes the ecological relevance of the sampling locations for past or future investigations. Frequently, characterizations of the nature and extent of contamination that focus on human-health concerns have too few soil samples collected from ecological habitats to adequately assess the potential for wildlife exposure. Therefore, additional sampling in ecological habitat areas may be needed. Key to any assessment is a demonstration that the nature and extent of soil COCs have been adequately determined, with a corresponding sample density appropriate to the concentration gradient.

In determining the soil EPCs within an affected property, the person must demonstrate that the samples selected for calculating the EPCs conservatively estimate the exposure to which receptors of concern will be exposed within the habitats of the affected property. For simple sites with uniform or heterogeneous conditions and habitat, an ideal sampling design would include data collection in a grid pattern uniformly across ecological habitats within the affected property, and the sample number would provide sufficient statistical power for determining reliable and defensible EPCs. However, property-specific characteristics (i.e., small site, variable habitat) may render it inappropriate to collect uniform grid samples. Whatever sampling design is selected before data collection, or whichever existing samples are selected for an assessment, the person must justify the selection of data. The ultimate goal is to ensure that sufficient and appropriate soil data are available to correctly evaluate the affected property's potential risks to wildlife receptors.

2.4.3 Exposure Point Concentrations for Wildlife Receptors

2.4.3.1 Introduction

The determination and use of an EPC, or representative concentration (a term used in the *ERAG*), is not an immediate step in the Tier 2 SLERA process. Rather,

the list of appropriate COCs is first narrowed down by a two-step series of comparisons described below.

1. *A comparison to a background concentration.* In this step, as detailed in the *ERAG* (1.5.2), a COC can be eliminated from further consideration at the assessment phase if its highest measured concentration is lower than either the property-specific soil background concentration or the Texas median background soil concentration for metals cited in the TRRP Rule and the *ERAG*. If the concentrations of COCs are higher than their background concentrations, then those COCs are carried forward to the next step.
2. *A comparison of the highest measured concentrations of non-bioaccumulative COCs in soil to the current ecological benchmarks provided by the TCEQ.* If these concentrations are higher than their respective soil benchmarks, then the COCs are retained for further evaluation in the Tier 2 SLERA process. All bioaccumulative COCs greater than the background concentrations used for comparison in the assessment phase are also retained for further evaluation. See the *ERAG* (3.4) for information about identifying and assessing bioaccumulative COCs.

Take care during the initial screening of COCs to ensure that the highest measured concentrations used are not outliers. The *ERAG* (1.5.2) states that the maximum concentration should “... be compared to the medium-specific ecological benchmark values (see Section 3.5) unless it is demonstrated that it can be considered an extreme outlier of a particular exposure medium.” See Appendix B for descriptions of the preferred methods for defining and identifying outliers.

These two steps typically result in a reduced list of COCs to carry forward into the Tier 2 SLERA. After the second step [comparison to the ecological benchmark value—i.e., 30 TAC 350.77 (c)(1)], the representative concentration that will be used as the EPC (i.e., the 95 percent UCL) in 30 TAC 350.77 (c)(5) and (6) is computed for the non-bioaccumulative COCs that exceeded the ecological benchmarks and the bioaccumulative COCs that exceeded background concentrations.

2.4.3.2 Data Used to Determine the Exposure Point Concentration

The term *EPC* (synonymous with the TCEQ’s “representative concentration”), according to U.S. EPA guidance, generally represents the average level of exposure—expressed as a concentration—that a receptor may experience over an exposure area during an extended period of time. Therefore, the EPC should reflect a conservative estimate of the true average value. In the initial exposure assessment, selection of an EPC for a particular exposure area in a Tier 2 SLERA conservatively assumes that wildlife receptors live and feed throughout the exposure area, and that their full life cycles are completed in the exposure area. Consistent with **2.4.2.2**, the EPC is computed from soil concentration data within the exposure area, regardless of the measurement receptor’s home range.

Some wildlife receptors have a home range larger than the exposure area. In these cases, area use factors (AUFs) may be included in the refined exposure

assessment to minimize the potential overestimation of true risks. Where the home range of a particular wildlife receptor is smaller than the exposure area, the TCEQ does not expect calculation of a series of EPCs to represent each hypothetical home range within the exposure area, assuming an evaluation of potential hot spots as discussed in **2.4.4**. Figure 2.2 contains a flowchart for estimating an EPC for a wildlife-receptor exposure area. The chart emphasizes the need for soil samples from within the ecological habitat that comprises the exposure area, to better represent the average exposure to wildlife receptors therein.

As indicated in **2.4.2.2**, the normal assumption is the entire ecological habitat within the affected property soils will be used in the determination of the EPC. In some cases it may be appropriate to subdivide the ecological habitats and calculate the EPCs for specific habitats and receptors (i.e., unique exposure areas). As stated earlier, this is the exception, rather than the norm. Some examples include:

- when protected species or their habitat exists within the affected property (i.e., the habitat where the protected species feeds is conservatively and appropriately evaluated, since these receptors are often habitat-limited and the essential foraging area at the affected property may be smaller than the total ecological habitat)
- when there are distinctly different habitats within the affected property ecological habitat and receptors of concern within each habitat are unique and expected to be different
- when ecological habitat within the affected property exists in patches that are not contiguous and the distance between patches is significant (see below)
- when a model based on the habitat-suitability index (HSI) is used to demonstrate habitat preference
- when soil samples were not collected within all patches of habitat and the patches are not contiguous
- when risk management decisions are expected to result in multiple and distinctly different remedial actions (e.g., a portion of the affected property is addressed through an expeditious removal while any remediation of the remaining affected property is deferred until approval of the Affected Property Assessment Report [APAR])

When division of the ecological habitat is contemplated for any reason, persons should ensure the data set is sufficiently robust to calculate an EPC. Further, persons should provide sufficient discussion and justification for subdividing the ecological habitat data set for a particular receptor-exposure pathway.

Figure 2.3 depicts an example scenario illustrating the questions that may come up when determining what data are appropriate for calculating an EPC. In this example, only portions of the affected property are classified as habitat (i.e., the answer to Subpart B of the Tier 1 Checklist is “yes” for some portions), the patches of habitat are not contiguous, and samples are not available from

within each habitat patch. In this scenario, there are several possible options for estimating exposure. These include:

- using only the data from within ecological habitats to calculate a single EPC for the combined habitat patches
- using only the data from within ecological habitats and calculating an EPC for each habitat patch, after making the case that the receptor in question is not likely to move and forage among two or more habitat patches
- averaging the nearest sample concentrations outside a habitat patch that does not contain samples (patch A) and assigning that average concentration to the habitat patch (and then selecting one of the first two choices above)
- using area-weighted averaging (e.g., kriging or Thiessen polygons) of sample concentrations outside the habitat patch that does not contain samples to compute estimated concentrations within the habitat patch (and then selecting one of the first two choices above)

Continuing with the previous example, the TCEQ recommends collection of samples from all ecological habitats within an affected property and computation of a single EPC for the combined habitat (i.e., the exposure area). Here again, when division of the ecological habitat is contemplated for any reason, persons should ensure the data set is adequate to calculate an EPC. Whichever of these options or other options is selected, sufficient explanation and justification of the decision should be provided in the ERA. The TCEQ may request additional sample data if the approach represents an unacceptable level of uncertainty or does not afford an appropriate measure of conservatism. The following paragraph discusses the application of AUFs when data are not available for all patches of habitat.

When patches of habitat within an affected property are not contiguous and portions of the available habitat are not sampled, or when portions of contiguous habitat are not sampled, the area of the entire available habitat within the affected property should be summed in calculating an AUF. Under either of these instances, estimating the total available habitat to which receptors are exposed as anything less than the total available ecological habitat will result in an AUF that is lower than appropriate for a screening assessment.

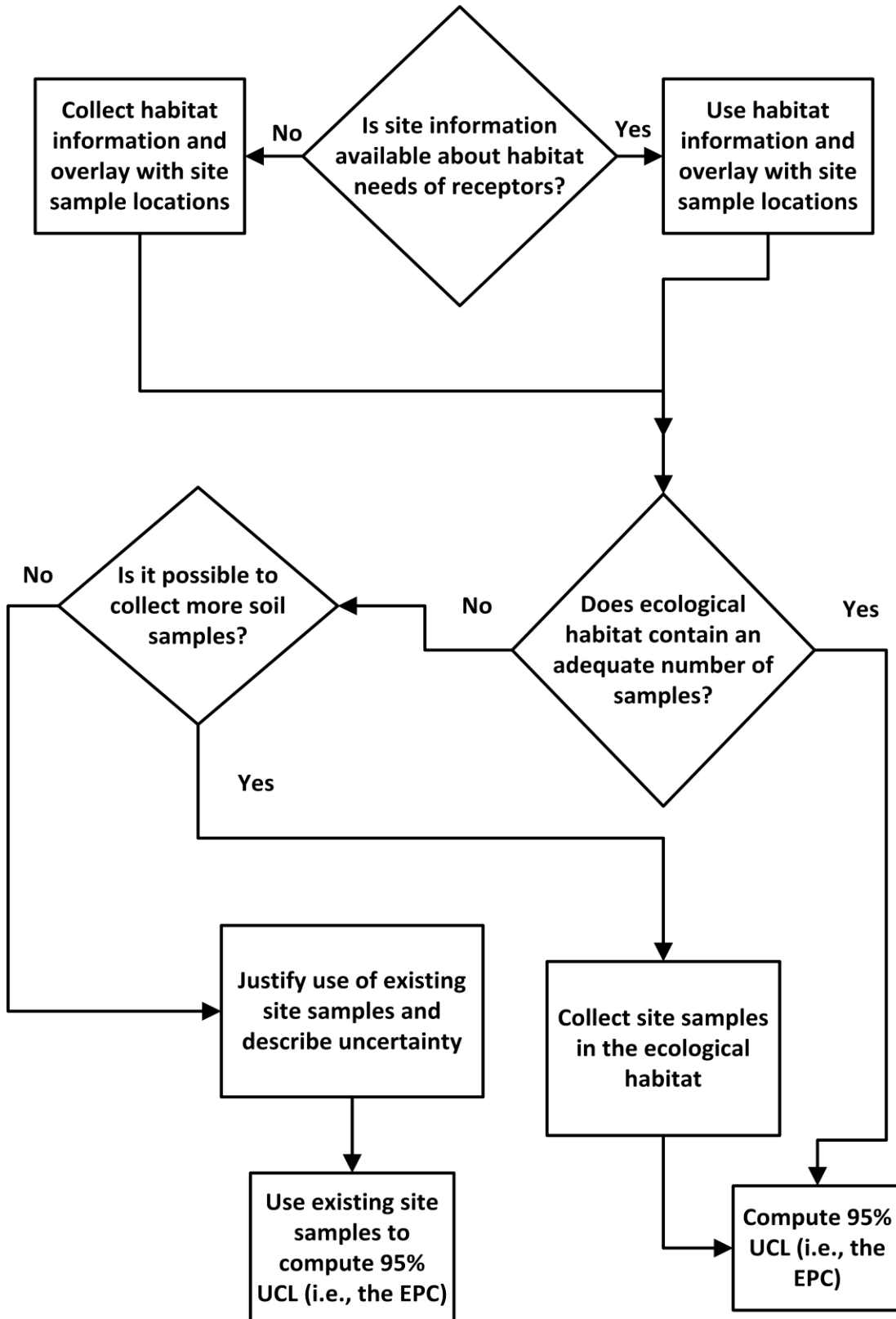


Figure 2.2. Flowchart for evaluating the adequacy of soil data in ecological habitat.

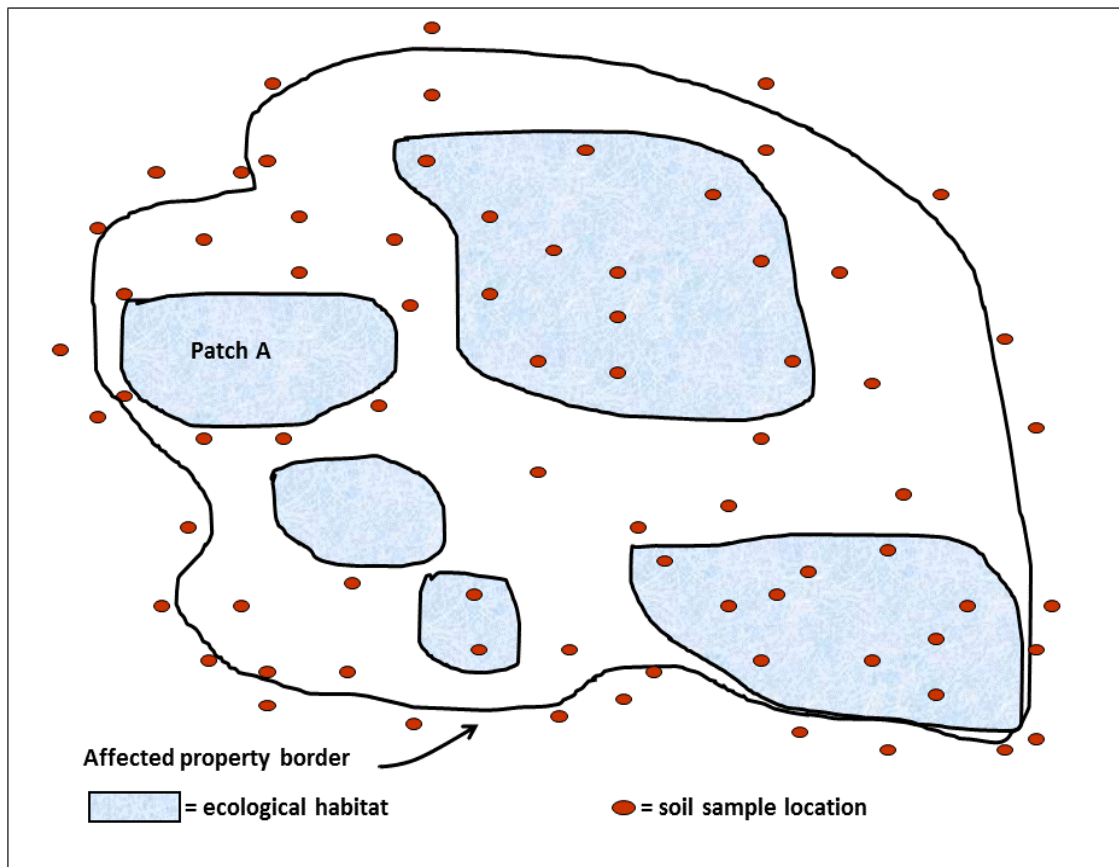


Figure 2.3. Example soil sample distribution for determining the soil EPC.

There are two exceptions:

- The person can give a reasonable explanation supporting the assumption that the receptor in question is not likely to move or forage among two or more separated habitat patches.
- The person can demonstrate that exposure within some patches of habitat is incomplete or exposures within distinct patches are unique, such as when the HSI model is employed.

For these or any other exceptions, persons should adequately justify alternate approaches for calculating an AUF.

2.4.3.3 Recommended Statistical Estimator for the Exposure Point Concentration

The *ERAG* (1.5.2, 3.9.2) suggests that the 95 percent UCL of the arithmetic mean (95 percent UCL) can be the preferred value for the representative concentration (also known as the EPC) for wildlife, but also defers to the approaches described in this guidance document. The *ERAG* (8.0) defines the 95 percent UCL 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, the use of 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 for wildlife receptors since the goal is to protect wildlife receptors at a population scale, and not individually (except for threatened and endangered, or state-sensitive species). An arithmetic or geometric mean should not be used in lieu of the 95 percent UCL.

The 95 percent UCL is most easily computed using readily available software. More sophisticated software packages (e.g., the U.S. EPA’s ProUCL—U.S. EPA 2010b) will compute a variety of 95 percent UCL statistics from a single data set and will recommend an appropriate level of quality for the input data. The statistical appropriateness of a given UCL statistic should be carefully considered, given that factors such as sample size, data variance, site features, and receptor type may all have some bearing on the type of proposed UCL statistic.

If most of the computed 95 percent UCL concentrations exceed the highest measured concentration (particularly true for small data sets or data sets with a large percentage of non-detect values), then persons may need to evaluate the appropriateness of the data set for calculating an EPC. Persons may also need to consider collecting additional samples from the exposure area to minimize variability and improve the quality of the data set (e.g., allow the use of statistics to compute 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. Defaulting to the highest measured concentration may not be protective when the sample size is very small because the true mean for the exposure area may be higher than this highest measured value (U.S. EPA 2002). Statistically based approaches for handling non-detected results⁷ and generating an EPC from such data sets may be appropriate if the technical basis is detailed. Any estimates based on limited data are likely to be highly uncertain and should be used, if at all, with extreme caution. Thus, it is important to collect enough samples in accordance with the DQOs for a site.

A separate hot-spot analysis (see following text in **2.4.4**) should be performed to identify unusually high COC concentrations relative to other sample locations. Comparisons with a PCL on a point-to-point concentration basis⁸ 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

⁷ See 30 TAC 350.51(n) of the TRRP rule for direction on the appropriate proxy value for non-detected results.

⁸ 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.

to a point-to-point comparison), the SLERA must also consider if COC hot spots are present in the exposure areas. Hot spots, as defined in the next subsection, would not be included in the calculation of the 95 percent UCL.⁹ All hot spots must be addressed in the discussion of risk management (see **2.4.4.4**).

2.4.4 Evaluating Soil Hot Spots for Wildlife Exposure

2.4.4.1 Introduction

The presence of hot spots at an affected property can be important in the assessment and management of wildlife risks. The purpose of a hot-spot evaluation is to identify any risks to wildlife receptors 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 UCL as the EPC) across larger areas. As stated previously in **2.4.1**, wildlife receptors for soil consist of birds, mammals, amphibians, and reptiles.

A hot spot generally refers to an area containing substantially elevated COC concentrations and associated elevated risk relative to other areas present (see **2.4.4.2** for a more detailed definition). The standard ERA evaluates the COC concentrations over an exposure area larger than a hot spot to evaluate potential risk based on the assumption that wildlife exposure to soil COCs will be equally distributed across that area. However, this assumption may not be protective if either: (i) a smaller area of soil with elevated COC concentrations poses a risk of acute toxicity, or (ii) a smaller area of soil with elevated COC concentrations is located in an area that contributes disproportionately to the receptor's chronic exposure (e.g., in a high-quality feeding area). In either of these cases, the area of elevated COC concentration would 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 (defined in **2.4.2.1**).

The identification and early treatment of hot spots (e.g., removal) can be useful in addressing particular 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 upfront 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 of a risk assessment.

2.4.4.2 Definition of a Hot Spot

The TRRP rule [30 TAC 350.51(1)(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,

⁹ This assumes a response action will address the hot spot.

1999) allows 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 current *ERAG* describes a hot spot as:

... A discrete area of substantially elevated COC concentrations relative to the surrounding area. What constitutes a hot spot depends in part on the concentration, toxicity and other properties of the COC, the medium in which it is detected, the extent of the area with elevated COC concentrations, and the biological characteristics such as receptor home range. Hot spots applicable to one feeding guild may not be applicable to other feeding guilds in a particular food web.

In other words, a hot spot is not just an area of substantially elevated concentrations relative to surrounding areas; it is also a function of relative risk to the measurement receptor in question. As a result, if an area of elevated COC concentration is identified, the goal of the hot-spot evaluation is to determine if this area: (1) poses a risk of acute toxicity, or (2) is a preferential exposure (feeding, nesting, or breeding) area relative to the surrounding habitat.

2.4.4.3 Recommended Procedures for Identifying Hot Spots

INTRODUCTION

The TCEQ has not identified any specific trigger for a hot-spot analysis for wildlife receptors. Simply put, a hot-spot evaluation may be necessary for some exposure pathways, in addition to the HQ evaluation in which the 95 percent UCL is the EPC for a wildlife receptor. This section describes the situations warranting more concern and also discusses possible approaches for identifying and evaluating a potential hot spot.

The person preparing a risk assessment should determine if a hot-spot evaluation is needed. If so, the evaluation should be presented in the uncertainty analysis. If a hot-spot evaluation is not warranted, a short justification should be presented. The TCEQ will evaluate the adequacy of the hot-spot analysis (or the justification for not performing one) and comment as necessary if clarification is needed. The TCEQ will also evaluate the conclusions of the hot-spot analysis and the associated risk management recommendation, as appropriate.

As indicated in **2.4.3.3**, the use of the 95 percent UCL as the EPC assumes that a wildlife receptor has equal and random access to all parts of an exposure area. This assumption ignores the foraging preferences as well as the relative spatial positions of the wildlife receptor and impacted soil and food. Moreover, some receptors may not have equal access to an exposure area or may follow fixed feeding routes that may overlap hot spots. For most wildlife receptors, the exposure to hot spots is minimized due to the mobility of the wildlife species. Therefore, the necessity of evaluating hot spots for wide-ranging receptors will be the exception rather than the norm.

For purposes of this hot-spot discussion, *small-ranging* receptors are those with a home range less than or equal to one hectare, and *wide-ranging* receptors are

those with a home range greater than one hectare (approximately 2.5 acres). For many risk assessments, the hot-spot evaluation may be a fairly simple discussion. The complexity of the analysis will depend on the wildlife receptors in question, the spatial configuration of the sample locations exhibiting elevated concentrations of COCs, and the corresponding habitat and foraging needs of the receptors. Once identified, a hot spot may not necessarily require remediation (see 2.4.4.4). However, this does not negate the need to determine if a hot spot exists; risk management and risk assessment, although intertwined, are separate processes.

This section presents approaches a person may choose when performing a hot-spot evaluation for wildlife exposure to impacted soil. This guidance is intended to present reasonable approaches for a typical Tier 2 SLERA. Although a Tier 3 SSERA may include an evaluation of soil hot spots, this guide focuses on a typical Tier 2 SLERA due to the property-specific considerations inherent in a Tier 3 SSERA. Although this guidance discusses a variety of approaches for evaluating hot spots, the TCEQ does not expect that all of these approaches will be used for any given site. The complexity of the evaluation should be dictated by the receptors and habitat in question relative to the footprint and concentration of the COCs in question, and the degree of uncertainty associated with the overall evaluation. In the discussion of hot-spot evaluations, considerations for small-ranging receptors appear first (both simple and more complex evaluations), followed by suggestions for wide-ranging receptors. There is also a discussion of hot-spot risks for threatened and endangered species, followed by risk management for wildlife hot spots. Figure 2.4 shows the different approaches for evaluating potential hot spots for wildlife exposure to soil.

Numerically, a hot spot may be identified simply as a disproportionately elevated sample concentration or group of sample concentrations relative to the surrounding area. As indicated in the previous section, the need to address a potential hot spot is ultimately a function of its associated risk. For example, even if an area of soil exhibits substantially elevated COC concentrations relative to the surrounding area, the concentrations may be below the level of concern toxicologically and ecologically for the receptor in question. Conversely, if the impacted area overlaps the foraging range of a receptor, and COCs are present at a level resulting in an unacceptable risk (e.g., COCs are present at acutely toxic concentrations or there is a preferential or constrained exposure scenario), this would indicate a hot spot that should be addressed.

PRACTICAL EVALUATION OF DATA FOR SMALL AND WIDE-RANGING RECEPTORS

Paramount to a hot-spot evaluation is an initial evaluation and presentation of the COC concentrations on a map. Visualization of sample locations exhibiting elevated concentrations of COCs can be helpful in determining if these data points are spatially discrete and distinct from surrounding areas, or if the elevated concentrations are grouped together. Persons should consider spatial patterns of any elevated concentrations relative to the exposure area generically, and specifically for any receptor with unique habitat and forage needs. If specific habitat information is available, data visualization should be used to determine

if the sample locations exhibiting elevated concentrations cluster together in islands of preferred or critical habitat, or wildlife corridors. Data visualization may also be useful in identifying any data gaps in the spatial coverage of soil samples.

Elevated sample locations in production areas, manufacturing areas, or areas that would meet the definition of disturbed ground in the Tier 1 Checklist [30 TAC 350.77(b), Subpart B] should not be considered possible hot spots as they are not ecological habitat. Review **2.1.1**.

Persons may elect to use a statistical outlier test to identify potential hot spots. If statistical outliers are identified and the elevated concentrations can be attributed to an error (e.g., lab or sample collection error, data-entry error, transcription error), the erroneous data should be removed from the data set for determination of the EPC and identification of hot spots. Clearly identify any data points removed and discuss them in the submission to the TCEQ (e.g., ERA uncertainty discussion, APAR). Where data are removed from the exposure-area data set, consider if additional sampling and analysis are needed to ensure a statistically robust data set with adequate sample density to evaluate the applicable exposure pathways. The remaining outliers should be viewed as potential hot spots for further evaluation.

The TCEQ suggests (but does not require) a statistical outlier test to identify potential hot spots, particularly where there are abundant data for the affected property. If the data set is too small to perform a statistical outlier test, consider whether it is robust enough for calculating a 95 percent UCL or performing an adequate ERA. Although the data set may be small, the TCEQ acknowledges that, in some cases, potential hot spots may be readily apparent such that the need for more soil data is obviated. More guidance regarding the evaluation of outliers appears in Appendix B.

SIMPLE QUANTITATIVE EVALUATION FOR SMALL-RANGING RECEPTORS

Receptors with a small home range may require a fairly simple quantitative evaluation. The information on soil contamination, receptor home range, size of the area that appears to have elevated COC concentrations (as compared to the overall exposure area), etc., should all be evaluated collectively in a weight-of-evidence (WOE) fashion. While there are no hard rules, one cannot simply dismiss a potential hot spot, or forgo the stepwise evaluation, even when the area of elevated concentrations is relatively small in proportion to the overall exposure area.

At this stage of the evaluation, sample locations exhibiting elevated COC concentrations should have been denoted on a map, and any erroneous data identified and removed from the hot-spot evaluation. Accordingly, any remaining outliers are potential hot spots necessitating further evaluation. As there is no quantitative threshold for identifying a hot spot, persons should consider the following approaches in formulating a hot-spot discussion for small-ranging receptors. As noted earlier, the stepwise approach the TCEQ recommends is

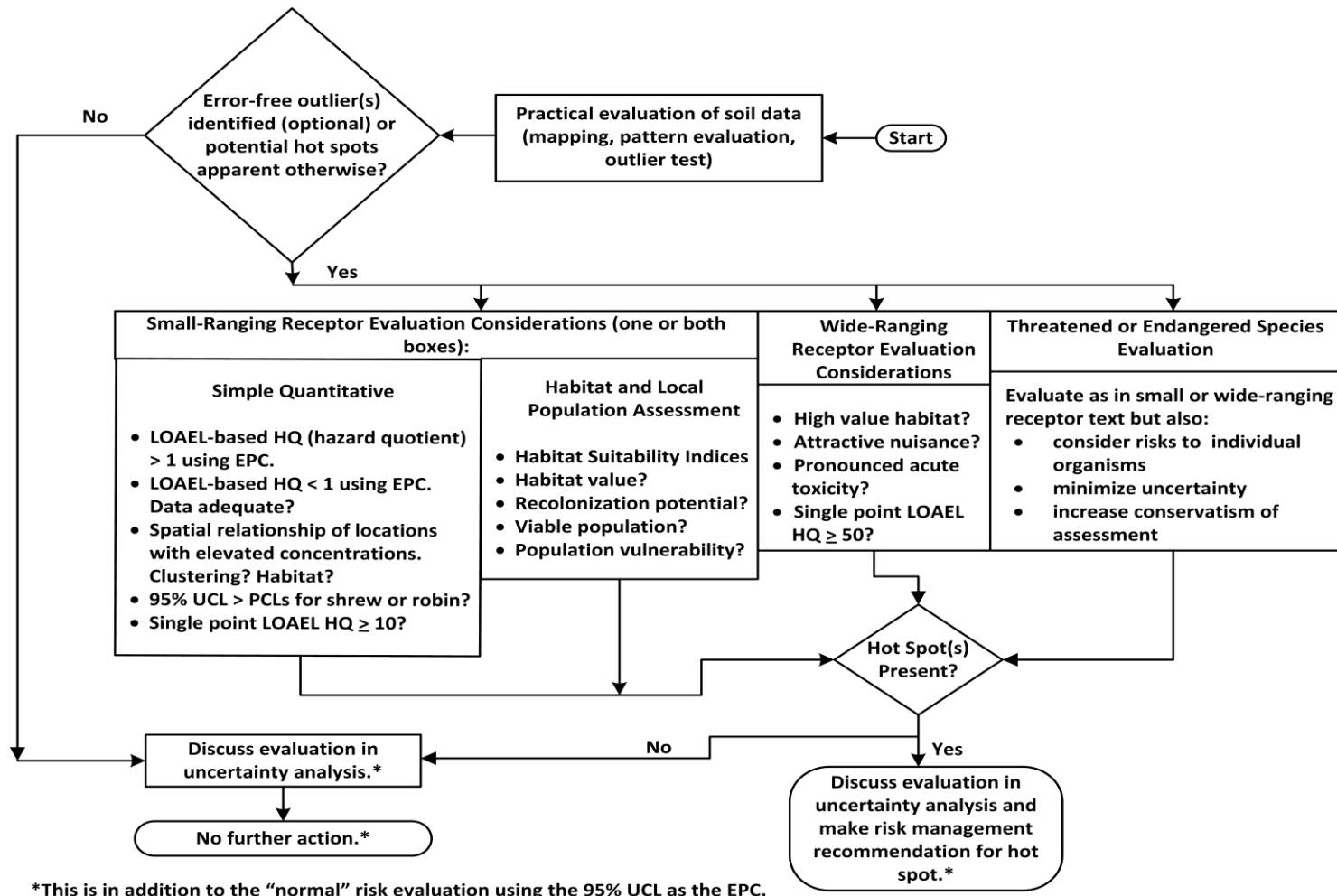


Figure 2.4. Approaches for evaluating potential hot spots for wildlife exposure to soil.

intended to be taken whole; elements are not to be used individually in an attempt to justify the absence of a hot spot. For example, it is not appropriate to automatically dismiss a potential hot spot exhibiting a relatively high HQ based on the lowest observed adverse-effect level (LOAEL) simply because the cluster of sample locations exhibiting elevated COC concentrations represents a small portion of the overall exposure area. It may be that the hot spot, while small, could overlap with much of the home range of a particular receptor, or occur in an area of the affected property conducive to the foraging and nesting of a particular species. Because it is impossible to predict all the situations that may be encountered in affected properties, individuals are encouraged to work closely with the TCEQ staff in developing their discussion of this issue.

LOAEL-Based Hazard Quotient > 1. Where the EPC (i.e., 95 percent UCL) for a wildlife exposure pathway results in a LOAEL-based HQ greater than 1 for a particular COC, a hot-spot evaluation is necessary. Unless the sample data indicates uniformly elevated soil concentrations, one or more data points are presumably driving the risk. These sample locations must be identified and evaluated to support an appropriate risk management recommendation that will address the elevated HQ. This evaluation may result in a decision to obtain more soil data to reduce any uncertainty associated with the sample density and property-specific habitat relative to the foraging and home-range characteristics of the associated receptors. This evaluation could also result in a decision to remediate the apparent hot spots such that the LOAEL-based HQ does not exceed 1. Additional discussion of risk management alternatives appears in **2.4.4.4.**

LOAEL-Based Hazard Quotient < 1. Where the EPC (i.e., 95 percent UCL) for a wildlife exposure pathway results in a LOAEL-based HQ less than 1 for a particular COC, a hot-spot evaluation is generally not needed—because the TCEQ believes the likelihood is low that hot spots will be problematic in this scenario. However, sometimes a hot-spot evaluation may nevertheless be needed. For example, if the LOAEL-HQ approaches 1, the sample density is low relative to the home range of the receptor, data are highly variable, or wildlife receptors appear to be stressed, there may very well be a need for a hot-spot evaluation.

A more careful evaluation of the risks in this scenario is also warranted if threatened or endangered species are known to occur at the affected property or the habitat is potentially suitable for a threatened or endangered species to use. In that case, evaluate the affected-property data to ensure adequate sample density given the home range and habitat preferences of the protected species. If the data set is determined to be inadequate, collect more soil data to address this uncertainty or conduct a limited biological survey to verify habitat availability and actual or potential site use (or do both).

Spatial Relationship of Sample Locations with Elevated Concentrations. Areas where sample locations exhibiting elevated COC concentrations cluster together could be a hot spot for small-ranging receptors. If these sample locations do not cluster together, and there does not seem to be any pattern in their distribution, there is less concern that these sample locations

represent a potential hot spot for small-ranging receptors. However, this statement assumes that the nature and extent determination was adequate and the sample density is appropriate to the receptor in question. If the spacing between the locations of elevated concentrations is large (e.g., 300 feet or more) relative to the home range of the receptor, more sampling is appropriate to define the extent of the potential hot spot. The TCEQ's home-range threshold is 1 hectare for small-ranging receptors, and 1 hectare (as a square area of soil) is roughly 300 feet long on each side.

As a general rule, if a cluster of sample locations exhibiting elevated COC concentrations does not represent more than 10 percent of the receptor's exposure area, the area is unlikely to be a hot spot. The rationale is that a relatively small proportion of the receptor's overall habitat within the exposure area on the affected property is likely to be affected. The small proportion (10 percent or less) that may be a potential hot spot is not expected to cause adverse impacts to the receptor at the local population level. However, if the elevated concentrations cluster together in islands of wildlife habitat or along wildlife corridors, this may very well indicate a hot spot necessitating remediation, since loss of habitat could have a negative impact on area ecology.

Consideration of a particular receptor's willingness or ability to move through non-habitat areas will be pertinent to this discussion where there is knowledge of the receptor's behavior. Similarly, consider whether a cluster of elevated concentrations has created or is co-located with an attractive nuisance. As always, the person may collect additional soil data or perform a limited biological and habitat survey to reduce the uncertainty associated with a suspected cluster of elevated concentrations that would otherwise be identified as a hot spot.

95 percent UCL in Excess of Default PCLs for the Shrew or Robin.

The TCEQ is developing conservative default PCLs¹⁰ that are intended to protect a large number of wildlife receptors typically evaluated in Tier 2 SLERAs. The PCLs for the shrew and the robin are conservative owing to these animals' high food ingestion rate relative to body weight, small home range, and invertivorous diet. These are characteristics that tend to increase exposure to soil COCs. A 95 percent UCL for a soil COC that exceeds a PCL for one of these receptors, given their small range, may indicate hot spots or more widespread contamination.

Single-Point LOAEL-Based Hazard Quotient ≥ 10 . Where the LOAEL-based HQ is highly elevated for a single point in soil, persons should evaluate the potential for a hot spot. The TCEQ recommends an HQ of 10 for this comparison. This approach should be considered a tool and the recommended HQ of 10 is not a defined threshold. For instance, a HQ of this magnitude becomes more problematic if the data set is limited and the sample density is poor. Accordingly, additional sampling and assessment near that sample location is recommended

¹⁰ These PCLs, as well as a discussion of their development, were not available when this manual was finalized. Check the TCEQ's ERA Program Web page <www.tceq.state.tx.us/remediation/eco/eco.html> for more recent information.

(depending on the receptor in question) to determine if there is a risk of acute exposure or if the location coincides with preferential habitat.

Thus far, this discussion has presented several relatively simple approaches for evaluating potential hot spots for small-ranging receptors. Persons could use these approaches to determine if a hot spot does exist, discuss the findings in the uncertainty analysis, and move on to a risk-management recommendation for any hot spots as appropriate (see 2.4.4.4). The primary question is whether a potential hot spot exists in habitat that would be preferentially used by a receptor or is located such that it restricts the receptor's movements. If neither is true, and concentrations are below acutely toxic thresholds, the potential hot spot is not a concern for ecological risk and should not require remediation. To support this determination, persons should present a general discussion of the receptor's habitat needs specifically for foraging, any corresponding site-specific details regarding the habitat availability at the affected property, and literature references to support the discussion. If the potential hot spot and habitat preferences overlap, the evaluation may stop here with a risk management recommendation (2.4.4.4), or continue further as described in the next section.

ASSESSING HABITAT AND LOCAL POPULATION OF SMALL-RANGING RECEPTORS

Persons may elect to use the approaches outlined above as a starting point for a more detailed assessment of the potential impacts on a local population of small-ranging receptors. Again, the key reason for the hot-spot evaluation is to avoid acute exposure or disproportionate chronic exposure that may otherwise go undetected where the receptor's exposure is averaged over some area using a 95 percent UCL. In either of these cases, the area of elevated COC concentration would be deemed a hot spot due to the elevated risk.

A more complicated evaluation should attempt to address the more difficult question regarding the potential risks for a local population if a hot spot occurs within the habitat for a particular small-ranging receptor. Long-term, intensive, and multidisciplinary studies necessary to understand the significance of impacts on populations would arguably fall in the realm of a Tier 3 SSERA. Given that the guidance herein is intended to primarily target submissions of Tier 2 SLERAs, the goal of the following discussion is to suggest various approaches for a meaningful and protective hot-spot evaluation that would fall short of a typical Tier 3 SSERA. Effects of COC hot spots on populations of small-ranging receptors depend not only on the exposure and sensitivity to the COC, but also on life-history characteristics, population structure, population density, interactions with other species, species mobility, and landscape structure, particularly where recovery via recolonization is considered. These basic concepts of wildlife ecology are reflected to some degree in the paragraphs that follow. All of these topics are intended to suggest ways a person could discuss whether a potential hot spot should be detrimental to a local population of a small-ranging receptor. No one approach is preferred by the TCEQ, and they can be used individually or in combination.

Habitat Suitability Indexes. Numerous methods and metrics are available for measuring ecological condition or valuing habitat. These methods can be used

to support an appropriately robust discussion of the importance of the potential habitat loss represented by a COC hot spot, or a discussion of the relative unsuitability of the habitat offered by the hot-spot location. The associated models typically reference several literature sources in an effort to consolidate scientific information on species-habitat relationships. One such tool is the habitat-suitability index developed by the U.S. Fish and Wildlife Service (see U.S. FWS 1981; Schamberger et al. 1982; USGS 2010). The HSI is a numerical index that uses measurements of important habitat characteristics for a species to produce a value between 0.0 (unsuitable habitat) and 1.0 (optimum habitat) based on the assumption that key environmental variables are related to habitat carrying capacity or the population size that can be supported by the available resources of the habitat. The habitat use information forms the foundation for the HSI. Short of complete model analysis, the habitat use information may support the hot-spot evaluation.

Discussion of Habitat Value in General. A general discussion of habitat value can be used to argue that a potential hot spot should or should not be addressed further for a given pathway. Habitat loss due to a chemical stressor becomes more significant if a critical area becomes diminished. However, persons should not contend that a potential hot spot is not of concern at a location demonstrating low habitat value if the diminished habitat can be specifically attributed to the elevated COCs in soil.

Various metrics for valuing habitat include:

- taxa richness
- number of sensitive species
- complexity of habitat structure
- presence of invasive or nonnative species
- presence of rare species or communities
- presence of an ecological corridor
- proximity to water
- ecological constraints (risk of predation, intensity of competition, and physical accessibility of resources)

More detailed discussion of most of the foregoing habitat value metrics is available in Efrogmson et al. (2008). Assessing habitat quality quantitatively manner can be difficult.

Potential for Recolonization from Adjacent Habitats. The hot-spot discussion may consider the potential for recolonization of wildlife from adjacent habitats. Here persons should consider the dispersal capabilities of the receptor in question relative to the distance between viable habitat patches. A related consideration is the persistence of any COCs in view of the exposure pathway being evaluated. Implicit to the assumption that a local population can be restored with individuals from the nearby habitats is an assertion that these

organisms will not be similarly affected by the potential hot spot in question. This discussion should be supported by appropriate literature citations specific to the mobility of the wildlife receptor, the ability of the receptor to traverse non-habitat areas, and any relevant information regarding the population and habitat for the source area.

Evidence of a Viable Population. A criticism of the standard approach to ERAs is that a site appears to be thriving despite elevated concentrations of COCs in soil (e.g., Tannenbaum 2005). Persons may attempt to discuss the relative insignificance of a potential hot spot from this point of view. A casual snapshot of the affected property based on limited field observations will not likely be acceptable to the TCEQ as evidence of a healthy ecosystem. Recognizing population monitoring (censusing) often requires many field measurements over more than one year; persons may present a discussion that strikes a balance between these extremes.

Discussion of the Population Vulnerability. COCs have the potential to contribute to the decline in terrestrial vertebrate populations directly by causing breeding failures or death, or indirectly, by reducing the food supply or altering habitat (Fox 2000). The life-history strategy for a particular receptor (r- vs. K-strategist)¹¹ may affect its vulnerability to a particular COC and exposure pathway. Persons may undertake this type of evaluation to contend that a population characteristic causes the local population to be more or less vulnerable to a potential hot spot. Characteristics of populations that increase the ecological significance of a stressor should be considered where possible. High-risk populations are characterized by:

- low density
- unstable age or stage structure
- low genetic diversity
- high natural mortality
- isolated population
- low dispersal ability

(Maltby et al. 2001)

¹¹ The terms *r-selection* and *K-selection* are used by ecologists to describe the growth and reproductive strategies of various organisms. According to this theory, organisms fall somewhere within an r to K continuum, depending on environmental pressures at the time. In general, an organism that is particularly well adapted to an exponential increase in population size is known as an *r-strategist*. r-Strategists typically live in unstable, unpredictable environments and are characterized by high reproductive rates. Offspring survival is low. K-strategists typically survive and prosper at or near carrying capacity. They occupy more stable environments, and are larger and have longer life expectancies. They produce fewer progeny, but place a greater investment in each. They typically grow slowly, live close to the carrying capacity of their habitat, and produce a few progeny each with a high probability of survival. See Reznik et al. 2002 for a general overview.

Other characteristics influencing population vulnerability include:

- age at first reproduction
- number of offspring usually produced in a life span (effect on a population growth rate will be greater for COCs that reduce juvenile survivorship or fecundity for species that reproduce once)
- rate of survival until first reproduction
- time between broods
- dispersal capacity
- specialized breeding or nest-site requirements
- unique behaviors
- territorial behavior
- patchy distribution
- feeding guild (i.e., insectivory, carnivory, scavenging)

(see De Lange et al. 2006; De Lange et al. 2009; Fox 2000)

Support the discussion of population vulnerability with appropriate literature citations specific to the life history of the wildlife receptor being assessed.

GENERAL EVALUATION OF POTENTIAL HOT SPOTS FOR WIDE-RANGING RECEPTORS

As stated above, a hot-spot evaluation for a wide-ranging receptor should be the exception rather than the norm. This discussion presents various scenarios where a hot-spot analysis would be appropriate. Where applicable, this analysis should be included in the uncertainty analysis of the risk assessment. Certainly a potential hot spot should not occur within or be associated with high value or unique wildlife habitat areas such as roosting and nesting areas, old-growth forests, and the soils surrounding and within playa lakes. Additionally, a potential hot spot would be of concern if the location provides better quality habitat or foraging base than is available in the remainder of the wide-ranging receptor's home range. In this case, it could function as an attractive nuisance and result in greater exposure to elevated COC concentrations. Another exposure situation of concern is where prey's vulnerability to predators is increased due to impaired behavior or mortality as a result of contamination associated with a hot spot. Examples include the pesticide-induced "insect rain" described by Stehn et al. 1976, and impaired predator avoidance of fish exposed to mercury (e.g., Webber and Haines 2003; Weis and Weis 1995). Finally, any hot spot is of concern that is so acutely toxic that limited interactions could lead to morbidity or death.

Similar to the discussion of small-ranging receptors, where the LOAEL HQ is particularly elevated (e.g., 50 or greater) for a single point soil concentration, persons may choose to use this as an indication of a potential hot spot. Arguably, this is likely to be inconsequential to the receptor population. However, an HQ of

this magnitude or greater becomes more problematic if the data set is limited and the sample density is poor relative to the mobility and foraging habits of the receptor in question. In that case, additional soil sampling and field evaluation around the sample location are advisable (depending on the receptor in question) to determine if there is a risk of acute exposure, or if the location coincides with preferential habitat.

CONSIDERATION OF HOT-SPOT RISKS FOR THREATENED OR ENDANGERED SPECIES

Potential risks to threatened or endangered species are necessarily considered at the level of an individual rather than the population. The premise behind this strategy is that a compromised population is less capable of tolerating the loss of individuals compared to 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 level of effort put forth in the hot-spot analysis may need to be increased where the measurement receptor in question is a protected species or its surrogate. For example, where COCs include volatile organic chemicals, explicit consideration of the inhalation pathway may be warranted if a burrowing receptor (protected species or surrogate) may forage at the affected property. Additionally, any uncertainty associated with the adequacy of the sample density, keeping in mind the ecology of the receptor, may necessitate more soil data or a field survey of the habitat. Furthermore, 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 in soil.

2.4.4.4 Risk Management for Soil Hot Spots for Wildlife Exposure

Risk managers should consider all of 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.

A hot spot can always be considered a *protective concentration level exceedance zone* [as defined by the TRRP rule at 30 TAC 350.4(a)(69)], although the hot spot and its associated PCLE zone are usually smaller in volume and are apart and distinct from any other larger areas of contamination that are not considered hot spots. However, a PCLE zone usually denotes a more widespread area of contamination where the COC concentrations are elevated above critical PCLs, but to a lesser degree. As previously mentioned, a true hot spot is smaller than the affected property. Because of this limited horizontal and vertical extent, and significantly elevated COC concentrations, hot spots are often associated with removal and backfill or capping as a response action. However, because of its larger volume and often less substantial PCL exceedances, a non-hot spot PCLE zone may be amenable to additional remedy options, such as an ecological services analysis (ESA) and monitored natural attenuation or some form of ecological compensation. An ESA could be proposed as the remedy for a hot spot. However, based on the nature of the exposure as discussed above, it is likely the

Natural Resource Trustees would recommend a more traditional remedial approach as appropriate.

As noted in the *ERAG*, determining what constitutes a hot spot for wildlife exposure and possible appropriate response actions are ultimately risk-management decisions specific to the property. In this context however, the person can suggest to the TCEQ what response actions are appropriate and can provide a rationale. It is also possible that additional sampling or more property-specific analyses are recommended by the person that may ultimately refine the potential risk-management alternatives.

The approach for evaluating hot spots for wildlife exposure may be iterative. Initially, additional sampling may reduce data gaps and influence the type of risk management actions that might be appropriate or reconcile situations where uncertainty remains high with an existing data set. In some cases, additional sampling may not be needed, but further data evaluation and discussion with the TCEQ staff may be the more appropriate step. A facility may elect to address a hot spot up front to minimize future investigation or liability. This may expedite subsequent evaluations of the remaining areas of soil contamination.

Keep in mind the following:

- 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 was in error, to establish the area of a hot spot, to establish a more appropriate sampling density, or to address a specific exposure pathway).
- Persons 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.
- 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.

3

Sediment Exposure Pathways

This section addresses the evaluation of sediment exposure pathways for ecological receptors. For these purposes, 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).

Although the benthic invertebrate community will usually be the group of receptors most susceptible to contaminated sediments, in some instances wildlife receptors will be the most at risk. The most common of these instances occurs when the water body will not support a viable benthic community, as discussed in Section 3.6.1 of the TCEQ's (2006) update to the *ERAG*. In this case, wildlife receptors with a comparatively high proportion of incidental sediment ingestion 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. This 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, 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.

This section is intended to provide additional clarity and perspective beyond the existing TRRP guidance and the *ERAG* for evaluation of sediment exposure pathways. The reader is further encouraged to obtain additional guidance directly available from the U.S. EPA on ERA methods.¹²

3.1 Data for Assessment

The discussion is not intended to replace existing TRRP guidance or the *ERAG* on the overall sediment investigation design, sampling methods, and assessment approaches. The reader is encouraged to review these and other guidance documents (e.g., TCEQ 2012b; U.S. EPA 2001; Mudroch and Azcue 1995; Radtke 2005). This discussion focuses on those overarching assessment issues critical for evaluating ecological exposure to sediments. Typical problem areas for sediment

¹² For example, see the U.S. EPA's page on Superfund risk assessment: <www.epa.gov/oswer/riskassessment/risk_superfund.htm>.

assessments are highlighted herein. Data-quality objectives and quality assurance are not discussed in detail, given that extensive materials on these topics are widely available (see for example, U.S. EPA 2006c). Additionally, Appendix A discusses the appropriateness of compositing sediment samples for use in an ERA.

The TRRP requires [30 TAC 350.51 (a–b)] 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) regarding data proposed for use in sediment exposure assessments. 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.

The remainder of this section discusses key considerations in the determination of data acceptability for assessing ecological exposures to sediment.

3.1.1 Adequacy and Appropriateness of Sediment Data

Fundamental to any sediment assessment is the characterization of the nature and extent of COC concentrations within the sediment. Sufficient data should be collected to identify sources, the extent of contamination, and potential migration pathways. Too often sediment assessments are arbitrarily truncated for reasons that have no basis in the adequacy of the data set. If the data set is inadequate, or if the sediment-assessment levels are based on human-health soil PCLs or human-health contact-recreation sediment PCLs, sediment exposure pathways may not be fully evaluated and the analysis may not be protective of ecological receptors. Persons evaluating the adequacy of the scope of the sediment assessment should be aware of the TCEQ's ecological benchmarks, property-specific background concentrations, and laboratory MQLs.

Collect sediment samples in areas relevant to all exposure pathways in question. For instance, if risks to wading birds are being evaluated, sediment data from deeper portions of the water body should not be included in the assessment for that particular feeding guild. Other practical considerations include the timing (season, tidal stage, and flow severity) of sediment sampling, the influence of other (not site-related) anthropogenic impacts, and sampling equipment (e.g., cores vs. grab samplers). Take care to minimize the loss of fine-grained sediments during sample collection and processing. Fine-grained sediments typically have higher organic carbon content (and are therefore more likely to reflect higher concentrations of COCs) relative to larger sediment particles (e.g., sand and gravel).

3.1.2 Sediment vs. Soil

As described in more detail in the *ERAG* (3.9.2.6) and TCEQ (2005), the TRRP rule [30 TAC 350.4(a)(79)] denotes the material lying below surface waters, including intermittent streams, as *sediment*. It is appropriate to evaluate

ecological exposure from both the dry stream bottom and from sediment associated with intermittent streams when water is present. Persons should evaluate exposure of land-based ecological receptors when the stream bottom is dry, and should perform normal surface water–sediment evaluations when the stream bottom is wet. Where persons treat a single location as both sediment and surface soil due to intermittent inundation, the intermittent exposure can be accounted for using appropriate exposure-modifying factors (i.e., exposure frequency and duration). Alternatively, persons may evaluate one exposure scenario or the other, but not quantitatively evaluating the remaining scenario requires justification. Additionally, persons may assume soil background concentrations for ephemeral streams where perennial pools do not occur, and there is adequate justification provided to evaluate the stream bottom as soil only. Surface water is defined in the Texas Surface Water Quality Standards (TSWQS) (see 30 TAC 307) and the definition is also discussed in 4.

3.1.3 Sediment Depth

Sediments within the top 4 inches (10 centimeters)—the sediment interval normally evaluated in an ERA—are often considered the biologically active zone. However, that is not always the case. Selection of a specific sediment sample depth should be supported with a discussion of any biota observed (i.e. sediment invertebrates) within the biologically active zone for a particular sample location. Consideration of remedial alternatives and physical mechanisms such as deposition and erosion (e.g., scouring), may dictate sampling at deeper depths. Review the *ERAG* (1.5, 3.9.2.6) for additional discussions related to affected property assessment and the POE for ecological receptors. The TRRP defines the sediment POE for human health as within 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.

3.1.4 Analytical Considerations

The accuracy and precision of analytical methodologies play a significant role in determining the suitability of sediment data for use in a risk assessment. Data must meet the specifications in 30 TAC 350.54 and *Review and Reporting of COC Concentration Data* (RG-366/TRRP-13; TCEQ 2010c). 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 Program for the matrices, methods, and parameters of analysis.¹³ The analytical methods used should have MQLs below the effect thresholds and sediment benchmarks. TRRP [30 TAC 350.54(e)(3)] requires that persons select a standard available analytical method that provides an MQL below the necessary level of required performance for

¹³ For more information about the Texas Laboratory Accreditation Program, visit <www.tceq.texas.gov/field/qa/env_lab_accreditation.html>.

assessment and demonstration of conformance with critical PCLs. If that is not possible, select the standard available analytical method that derives the lowest possible MQL for a given COC. This is especially critical for bioaccumulative COCs in sediment such as PCBs, dioxins and furans, pesticides, and organochlorine compounds. 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 MDL, 30 TAC 350.54(e)(3) allows the agency to require a demonstration that a lower MQL is 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.

Historical sediment data may lend information useful for current assessments, particularly as it may be used to develop new sampling plans. Historical data may be included for qualitative discussions related to ecological exposures. However, input into the quantitative risk assessment requires caution as it must meet the specifications in 30 TAC 350.54 and TCEQ publication RG-366/TRRP-13, if the historical data will be used in the quantitative risk assessment to characterize ecological exposure conditions. Note that some provisions of TRRP-13 (such as the laboratory-review checklist and the detectability-check sample) do not apply to data generated before February 2003. These provisions are discussed in detail in TRRP-13. Historical data not meeting the specifications in 30 TAC 350.54 and TRRP-13 cannot be used in the quantitative risk assessment. The representativeness of historical data for characterizing current ecological exposure conditions should be considered. For example, COC concentrations could change significantly with time as a result of sedimentation or due to source removal. Sediment data collected prior to any major physical disturbance (such as capping or dredging) should not be used in the ERA.

3.1.5 Use of Sediment Pore-Water Data

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. 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* 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. 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 2005 and U.S. EPA 2008b). Pore-water data can confirm predictions of equilibrium-partitioning theory.

Sediment pore-water sampling and analysis may be most appropriate to aid in the assessment of releases of impacted groundwater to surface water and sediment, particularly where data are desired to supplement groundwater data from interface wells. The dynamic biogeochemical processes often present at the

groundwater-surface water interface can generate strong vertical solute concentration gradients and alter the chemistry of discharging groundwater. This pathway is discussed in more detail in 5. Consult with hydrogeologists familiar with the affected property to select appropriate sample locations to best represent the areas of groundwater discharge.

A large variety of techniques are available for collecting sediment pore water, but *in situ* sampling techniques are preferred. Documents that contain a good discussion of methodologies for collecting pore water include U.S. EPA (2001), Carr and Nipper (2003), and Chapman et al. (2002).

Pore-water concentrations are typically compared to water quality criteria or screening values and not to bulk sediment PCLs. The TCEQ does not recommend any specific approach for determining an EPC for pore water. Persons should propose and provide a rationale for an approach to evaluate sediment pore water data. This could include statistical averaging or a point-to-point comparison, depending on the exposure pathway.

3.1.6 High-Biased and Low-Biased Data Distribution

Sediment assessments evaluated for TRRP typically employ judgmental sampling as opposed to a random (sometimes geospatial) sampling regime. TRRP allows judgmental samples, as long as the resulting estimated representative concentration is demonstrably not biased low [30 TAC 350.51 (1)(1)]. Where possible, sampling should target depositional areas dominated by fine-grained sediments. Professional judgment is needed to ensure that data are collected in a manner that most appropriately represents the true population concentration relative to potential ecological-exposure conditions. Any possible introduction of biases should be discussed in the uncertainty analysis.

Sediment-sample locations outside of 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 sediment EPC.¹⁴ 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.¹⁵ It may be inappropriate to include sediment sample locations that do not appear to be affected by the TRRP release in question, such as locations at the fringe or perimeter of the sampled area. Additionally, if sediment samples are being collected to assess the

¹⁴ See the discussion in 3.3.4 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.

¹⁵ Since the affected property represents the entire area that contains releases of COCs at concentrations equal to or greater than 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.

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

3.2 Exposure of the Benthic Invertebrate Community to Sediment

3.2.1 Exposure Areas for Benthos

3.2.1.1 General Discussion of Benthic Invertebrate Communities

Individual species within benthic invertebrate communities spend much or all of their life cycle residing within or immediately on top of sediments. Since these organisms are sessile or largely sedentary, they are likely to reside within relatively small confines for significant periods.

The overall goal of any assessment and resulting risk management action is to be protective of the benthic macroinvertebrate community as a group as opposed to individual organisms. This concept is discussed in the *ERAG* (3.6.1). Species-specific benthic invertebrate evaluations are not typically performed, except under special circumstances, such as when threatened or endangered species are present. An *ecological community* is generally defined by ecologists as a group of populations composed of numerous species with similar geographical and physical requirements, such as temperature, media composition, and light regime. The similarity of requirements dictates that these species are found together. Communities themselves usually do not have clear spatial designations. Despite these uncertainties, the *ERAG* requires protection at the community level, and so a spatial area must be defined to constitute (at a practical level) both the community and exposure area under evaluation.

Be aware that the *ERAG* specifies instances where a benthic assessment is not required. The TCEQ recognizes that conditions exist where the benthic invertebrate community may be diminished for reasons unrelated to releases of COCs from an affected property. For these water bodies (e.g., intermittent water bodies without perennial pools, or those that are concrete lined 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 eliminate the need for an evaluation of risks to higher trophic level organisms (wildlife) that may forage in these or nearby water bodies. This is a common misconception. Review the *ERAG* (3.6.1) and updates for more discussion of the benthic PCL exclusion.

Like other ecological pathways, there may be reasons to divide the affected property sediments into smaller exposure areas for the benthic community if impacted sediments occur over a large area. This would result in the determination of unique EPCs for the various benthic exposure areas as opposed to averaging across the entire affected property sediment data set. This discussion is intended to clarify when it may be appropriate to subdivide the data set for separate exposure areas. In TCEQ's view, subdividing will be the exception rather than the norm. Variations in exposure caused by anthropogenic effects

(e.g., releases or discharges from locations not part of the affected property assessment) and variations in benthic habitat should largely govern the selection of these benthic exposure areas in sediment. Persons should establish scientifically credible rationales for making decisions to subdivide. Similarly, persons should present a reasonable rationale for not subdividing a sediment data set if the circumstances appear to conflict with the guidance that follows.

3.2.1.2 Physical Features

If significant variations in physical features exist within a given area, the potential role these variations play in separating different benthic invertebrate communities should be considered. Differences in physical features could result in clear physical demarcations, such as those created by dams. Physical features may affect COC concentration gradients, and consequently alter the exposure regime for the benthic community. Other physical features that could be used to distinguish benthic exposure areas include the presence of tributaries or other significant hydrologic inputs, such as localized outfalls or groundwater influences. Below are additional examples of differences in physical features that may be used to guide the determination of exposure areas:

- riffle and pool habitats in a stream
- shellfish and seagrass beds
- cove areas in a lake or bay
- differences in communities at different water depths and in different substrates (different sediment compactions, differences in grain size)
- physical habitat fragmentation (e.g., from roads or saltwater intrusion barriers)
- differences in bottom substrate due to historical dredging or construction
- significant differences in canopy cover or watershed type
- differences in flow due to irrigation-water returns and effluent discharges

These are examples for consideration and discussion. It is not the case that, anytime any of these circumstances occurs at an affected property, the area should automatically be subdivided into different exposure areas. If such features do not in fact result in expected or observed differences in communities or exposure, there may not be a need to divide the area. However, even when such biological differences do not exist, there may be overriding risk management or practical considerations for making certain subdivisions.

3.2.1.3 Potential Spatial Distribution of COCs or Significant Differences in Sediment Chemistry

Anthropogenic and natural attributes of an impacted surface water system can affect the ecological risks associated with impacted sediments. Case by case, determine if the impacted sediments should be divided into different exposure

areas to account for characteristics that would affect risk differently in each area. Consider the nearby presence of:

- areas of upwelling or downwelling groundwater (including areas being evaluated for impacted groundwater releases to sediment)
- industrial and municipal wastewater and stormwater outfalls
- mining discharges and runoff
- discharges of cooling water
- oil and gas exploration
- large differences in bottom salinity

Two example scenarios (freshwater creek and estuarine bay) that describe data groupings based on particular circumstances appear in Appendix C.

3.2.1.4 Management of Risk

There may be risk management or operational reasons for subdividing sediment data into separate benthic exposure areas. If there is an upfront risk management decision that an area of sediment will be remediated due to elevated COC concentrations, the benthic exposure area should be adjusted. This could dramatically lower the overall risk in the remaining area. Similarly, if an area will be dredged for some other reason (e.g., navigation or dock and harbor maintenance) it can be removed from the benthic exposure area provided there is a clear indication (approved plan, documentation of completion of construction) that the activity has occurred or will occur. If it is uncertain when the dredging may occur, discuss this in the risk management section of the ERA, and evaluate sediment exposure with and without inclusion of the area planned for dredging. Further, there may be programmatic reasons to subdivide the benthic exposure area. For example, this is particularly true for Superfund sites or Resource Conservation and Recovery Act (RCRA) sites that may divide up different areas of impacted sediment into units or areas of concern (AOCs). Where subdivision is necessary for programmatic reasons, it may be necessary to expand the ERA to include an evaluation of more comprehensive, site-wide ecological risks.

3.2.2 Exposure Point Concentrations (EPCs) for Benthic Invertebrates

As indicated in the *ERAG* (1.5.2), the highest measured sediment concentrations (for each COC) are initially compared (Required Element 1) to the sediment benchmark values in Table 3.3 of the *ERAG* (or TCEQ 2006) as part of the Tier 2 SLERA. Where the highest measured concentrations exceed sediment benchmarks (and background),¹⁶ the COC is retained for further evaluation.

¹⁶ COCs present at or below background should be eliminated from further consideration [see 30 TAC 350.71(k)(2)(D)].

After this benchmark screening step, the EPC for the benthic invertebrate community may be estimated using a 95 percent UCL of the COC concentrations in the benthic exposure area. This is consistent with the discussion in the *ERAG* (3.9.2), which broadly states that appropriate statistical methods (e.g., 95 percent UCL of the arithmetic mean) may be used to compute EPCs from COC concentrations in the exposure medium for all steps after the initial benchmark screening.

The TCEQ has selected the 95 percent UCL as the preferred EPC for the benthic invertebrate community since the goal is to protect benthic organisms at a community level, rather than individually. The 95 percent UCL is a conservative estimate of the true mean and accounts for uncertainty in COC concentrations throughout the exposure area. The EPC term, according to U.S. EPA guidance, represents the average exposure experienced by a receptor over an exposure area during an extended period of time. Therefore, the EPC should be a conservative estimate of the true average value (such as an appropriate 95 percent UCL of the mean) and not the highest observed concentration. If most of the computed 95 percent UCL concentrations exceed the highest measured concentration (particularly for small data sets or data sets dominated by non-detect values), then persons may need to evaluate the appropriateness of the data set. They may also need to consider collecting additional data from the exposure area to minimize variability and improve the quality of the data set (e.g., allow the use of statistics to compute a reliable 95 percent UCL). Alternatively, the highest measured COC concentration can be used to represent the EPC. Note that defaulting to the highest observed concentration may not be protective when sample sizes are very small because the highest observed concentration may be smaller than the true statistical population mean (U.S. EPA 2002).

The TCEQ understands that sediment habitat characteristics—such as substrate type, patchiness, and heterogeneity—influence the benthic invertebrate assemblage and, thus, affect benthic exposures. The TCEQ maintains that estimating the average exposure for a community of different species of benthic organisms within the exposure area is an appropriate assessment method for a Tier 2 SLERA. Use of the highest measured COC concentration as the EPC for the benthic invertebrate community is not necessarily reasonable, because it likely overestimates the exposure of much of the benthic community within the identified exposure area, assuming a robust data set.

Comparisons with a PCL on a point-to-point concentration basis is relevant when the sample size is too small to use statistical methods to estimate an EPC, or when the protection goal involves a highly valuable species, population, or community (either economically or by regulation). Since little is usually known about the characteristics of the exposed benthic community in a typical Tier 2 SLERA, the use of an average concentration (i.e., 95 percent UCL) is the best approximation of overall exposure. A straight average or geometric mean should not be used in lieu of the 95 percent UCL. Persons may choose to perform

point-to-point comparisons¹⁷ to evaluate potential risks to benthos rather than use the 95 percent UCL as the EPC.

Where a 95 percent UCL or any other appropriate average is selected as the EPC for benthic exposure (as opposed to a point-to-point comparison), the SLERA must also consider if COC hot spots are present because the potential effects of these hot spots could be diluted by the use of 95 percent UCLs. Hot spots, as defined in the next section, would not be included in the calculation of the 95 percent UCL. Additionally, as discussed in **3.2.3.2**, all hot spots must be addressed in the risk management discussion.

3.2.3 Addressing Hot Spots for Benthic Invertebrates

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;” however, the rule does not define hot spot.

For evaluating risk to the benthic community, a *hot spot* is defined as a discrete area of elevated COC concentrations in sediments that present a substantially unacceptable risk to the benthic community. This definition is similar to that outlined in the *ERAG* (3.9.2.7) with additional emphasis on risk. However, do not focus on this definition alone. It is intended to be an initial description, whereas the real test of the presence of a benthic hot spot is the evaluation described below. Once all the appropriate factors described on the following pages are evaluated according to the WOE approach, determining if a suspect area is a hot spot for the benthic invertebrate community should be possible.

The initial goal of the hot-spot evaluation will be to ensure that a statistical presentation of the sampling data (e.g., 95 percent UCL) will not mask or dilute areas of elevated sediment concentrations that may otherwise pose a potential risk to the benthic community, or cause risk from the remaining portions of the exposure area to be overestimated. The overall goal of the benthic hot-spot evaluation is to facilitate an effective risk management recommendation for potential risks to the benthos, given the uncertainties associated with the assessment of this community. Rather than prescribe a spatial or concentration threshold, this exercise is intended to foster a more focused evaluation of the sediment data so risk assessors will know a hot spot when they see it. Once each areas of greatest risk is identified, risk management strategies (e.g., removal, capping, ESA, Tier 3) can be proposed, and these areas may be removed from further calculation of the 95 percent UCL for the benthic exposure area in question.¹⁸ One potential outcome is a recognition that uncertainty is high, which may result in a determination that more sediment sampling and analysis are needed. This can be an iterative process; ideally any initial and

¹⁷ See the explanation of *point-to-point comparisons* in **2.4.3.3**.

¹⁸ This statement pertains to the ERA only. As part of an ecological services analysis, persons may need to evaluate a 95 percent UCL or other averaging statistic with and without the hot spots included.

subsequent analyses will be included in the hot-spot evaluation submitted with the Tier 2 SLERA.

The following evaluative process should be used for identifying hot spots for the benthic community:

1. Identify sample locations that exhibit COC concentrations greater than the default (midpoint) benthic PCL for that COC (see *ERAG* 3.13.2 for discussion of the midpoint PCL).
3. Examine locations with elevated COC concentrations (i.e., greater than the midpoint PCL) with regard to the factors in **3.2.3.1**. Use professional judgment to determine if the magnitude of the COC concentrations or the number and proximity of the sample locations that exceed a benthic PCL are sufficient to classify an area as a hot spot for the benthic community.
4. Present a cogent, science-based discussion regarding the presence or absence of hot spots on the affected property for the benthic exposure pathway.

The TCEQ will determine the appropriateness of these discussions on a property-specific basis. The hot-spot evaluation should be presented in the uncertainty analysis. The TCEQ will evaluate the adequacy of the analysis (or the justification for not performing one) and comment as necessary if more detail or clarification is needed. The TCEQ will also evaluate the conclusions of the analysis and the associated risk management recommendation (see **3.2.3.2**), as appropriate. Figure 3.1 is an overview of the different approaches for evaluating potential hot spots for exposure of benthic invertebrates to sediment.

3.2.3.1 Factors for Evaluating Benthic Hot Spots

The following factors should be considered in the hot-spot discussion. Depending on the affected property, some factors will probably be more important than others, and these should reflect a corresponding degree of analysis. It may not be necessary to discuss all factors. In the development of these factors, the TCEQ does not prefer or endorse any particular weighting but rather recommends that a potential hot spot be evaluated case by case in consideration of all factors that are presented for discussion. The TCEQ recognizes that the risk assessor may recommend that one factor carry more weight than another; however, this is subject to agency concurrence upon review. Consideration of the spatial relationship of the elevated data points, along with a map that illustrates the potential hot spots (next paragraph), should always be part of the evaluation.

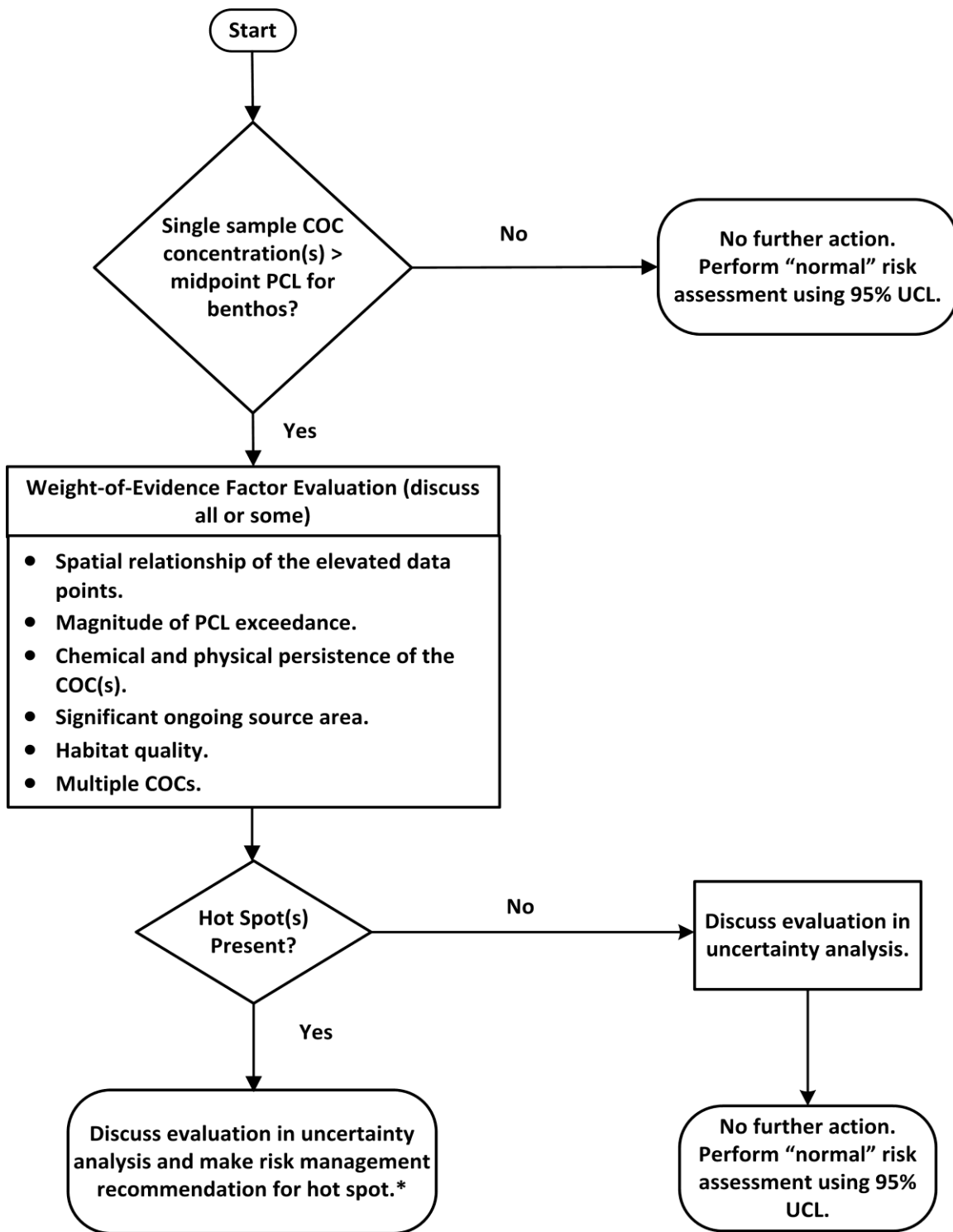
Spatial relationship of the elevated data points. Consider the sampling density relative to the sample locations with elevated concentrations of COCs. Present sediment sample locations and concentrations on a map. Highlight sample locations with COC concentrations greater than a benthic default PCL. Assume that the area a sample represents extends to half the distance to the next sample point, unless a different type of geospatial analysis is employed. Consider whether additional sampling is necessary to better delineate any potential hot spots.

Persons should also consider if contiguous sample locations exhibit elevated concentrations of COCs and consider the size of the area exhibiting elevated concentrations of COCs relative to the size of the affected property and the spatial characteristics of the water body. In general, the area of elevated concentrations should not be so large as to preclude movement of benthic invertebrates, including larval stages, across a water body. If there is a reasonable explanation for particular sample locations exhibiting higher concentrations (e.g., proximity to source area, depositional area), it should be considered in the evaluation. The depth of the potential hot spot should not exceed the depth of the biologically active zone unless there is concern that deeper sediments will become surficial sediments due to natural or anthropogenic events (e.g., scouring, dredging), and there is reason to believe that deeper sediments are impacted. The *ERAG* (3.9.2.6) further discusses the benthic POE, including the biologically active zone.

Magnitude of PCL exceedance. Consider the magnitude of the PCL exceedance (e.g., greater than the second effects level defined in the *ERAG*) relative to the toxicity of the COC in question. This factor alone should not be used as a reason to conclude that a locale is or is not a hot spot for benthos. Rather, all the factors should be discussed, weighing the evidence to arrive at a hot-spot decision. It may be useful to supply a contour map that compares the sediment concentrations with the midpoint PCL.

Chemical and physical persistence of COCs. Persons should consider the persistence of the COCs in the potential hot-spot areas relative to the surrounding area. Given the probability that benthic biota will recolonize formerly impacted areas, persons should discuss the likelihood that sediment COCs will remain in place given the expected persistence of the COCs in question, their likely breakdown products, and their ability to attenuate given the expected physicochemical conditions of the sediment environment at the affected property. A discussion of COC half-life (and corresponding references) is appropriate. Persons should also consider the relative concentrations of the COCs in question given the timing of the release in question (if known).

Significant ongoing source area. Persons should consider if the location of elevated COC concentrations in sediment constitutes a possible ongoing source area, such that the area of elevated COC concentrations could increase or be maintained. Inherent with this consideration is the nature and extent of the contamination as well as the hydrological properties of the surface water body, the mobility of the COCs and their breakdown products, the likelihood that sediment COCs could impact surface water, the probability of natural attenuation of the COCs, and the expected sedimentation rates for the water body in question. Note that if the source area comprises non-aqueous phase liquid, a NAPL evaluation is required. See *Risk-Based NAPL Management* (RG-366/TRRP-32, TCEQ 2013) for further discussion.



*This is in addition to the “normal” risk evaluation using the 95% UCL as the EPC.

Figure 3.1. Overview of different approaches for evaluating potential sediment hot spots for benthic invertebrate exposure.

Habitat quality. Consider and describe the quality of habitat available for benthos in the area of elevated sediment concentrations. This could include invasive vegetation as well as the physical morphology and water quality characteristics of the water body such as sediment grain size, vegetation, flow regime, and fluctuations in dissolved oxygen, temperature, and salinity. Indicate whether the habitat present is expected to be generally attractive or repellent to benthos relative to the surrounding areas. Avoid simply stating that a particular affected property is in an industrial-urban water body in an attempt to dismiss a potential hot spot absent a more detailed evaluation and site assessment. Although habitat degradation due to industrial and urban development is acknowledged, avoid this line of reasoning where the degradation is more likely attributable to the release in question than to other stressors. Further, be mindful that the impacted habitat can recover following implementation of a remedy. Certainly a hot spot should be indicated if the locations with elevated sediment concentrations occur in critical habitats or shellfish beds.

Suggested references typically used for assessing benthic habitat include Gibson et al. 2000; Fritz et al. 2006; and TCEQ 2007b. These references are for convenience; persons are not expected nor required to use them to complete the hot-spot evaluation.

Multiple COCs. Persons should consider whether any given sample locations, particularly contiguous locations, demonstrate elevated concentrations for multiple COCs or their typical breakdown products. Potential additive, synergistic, or antagonistic toxicity interactions should be considered where appropriate.

3.2.3.2 Risk Management for Benthic Hot Spots

By definition, hot spots present an unacceptable risk to the benthic community. Therefore, if hot spots are identified within the benthic exposure area, persons should recommend appropriate risk management practices. (See the *ERAG 5* for further discussion.) 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 determination and the resulting 95 percent UCL should be used as the EPC for the benthic invertebrate community.

3.3 Exposure of Wildlife Receptors to Sediment

3.3.1 Purpose and Rationale

Sediment is a key medium in aquatic ecosystems because it directly and indirectly supports wildlife in Texas. Being one of the primary exposure media, sediment serves as a principal depository and carrier of anthropogenic contaminants released into the environment, to which wildlife may be exposed via direct contact, ingestion, and food chain transfer.

Consistent with the importance of sediment as an exposure medium, implementing technically defensible approaches for sediment evaluation in Texas should be aimed at adequate protection of wildlife from exposure to COCs at

affected properties. In doing so, persons should recognize the need to select appropriate receptors and ecological scale (i.e., at the organism, population, or community scale). The TCEQ recommends discussions with its staff prior to field activities to save time and resources.

Sediment as an exposure medium can be found in numerous settings:

- within rivers, creeks, streams, and ditches
- within ponds and lakes
- within wetlands or low-lying areas that are permanently or intermittently flooded
- within tidal bays, estuaries, rivers, bayous, and channels

The ultimate goal of the sediment investigation, assessment, and remediation stipulated by the TCEQ is the protection of wildlife populations, and individuals of 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.

Typically, birds and mammals dominate risk assessments for aquatic-based wildlife receptors. A qualitative or quantitative evaluation of risks to amphibians and reptiles, depending on available toxicological and life-history information, should also be included in the ERA if they are expected to occur at the affected property. Amphibians and reptiles that are commonly found in Texas may include sensitive and representative species that may frequent areas where they may be exposed to COCs in sediments. A more rigorous evaluation is required where a protected reptile or amphibian species may occur at the affected property.

The TCEQ recognizes that health effects data for these classes, unlike birds and mammals, are sparse for many COCs. Toxicology information for amphibian and reptile exposure to COCs may be available from Pauli et al. (2000), Gardner and Oberdörster (2006), Sparling et al. (2010), or an online literature search from a database such as ECOTOX <cfpub.epa.gov/ecotox/> or TOXNET <toxnet.nlm.nih.gov>.

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 sediments (or soils). 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 high trophic level predators (animals that feed on other predatory animals), which makes them potentially sensitive to bioaccumulative COCs. Dermal exposure to COCs for amphibians is expected to be more significant than that for reptiles, which have a relatively impermeable skin. Amphibian exposure to COCs due to

transport across the skin may be the most significant route of exposure overall (Smith et al. 2007).

The natural histories for amphibian species indicate that they are potentially exposed to multiple media. Many species of amphibians lay their eggs in water, and the larvae live immersed in water until metamorphosis to the adult stage. The larvae of frogs and toads (tadpoles) have gills for part of their development, and therefore they have a potential exposure route of absorption of water across the gill membrane. Thus, exposure to surface water should also be assessed for amphibians, particularly where sediment COCs may likely partition to surface water. This is discussed in more detail in **4.3.1**.

Extensive discussion of exposure areas and EPCs for wildlife exposed to soil appears in **2.4.2–2.4.3**. Previous discussions (**2.4.4.1–2.4.4.3**) also addressed the evaluation of potential hot spots in soil for wildlife receptors. There is no need to repeat these concepts; the corresponding discussions for sediment specify where earlier discussions of soil are appropriate for sediment exposure pathways for wildlife. Conversely, the text will indicate approaches that differ from (or supplement) those recommended for soil.

3.3.2. Assessment Considerations for Wildlife Receptors

3.3.2.1 Wildlife Populations

The concepts of populations, local populations, and feeding guilds previously discussed for soil exposure pathways are equally relevant for sediment exposure pathways. Review **2.4.2.1**.

3.3.2.2 Exposure Areas for Wildlife Populations

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 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 sediments. A wildlife receptor may use 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 particular wildlife receptor will be smaller than the entire area represented by the affected property sediments. The generic approach presented herein, however, is to assume that all of the affected property sediments make up a receptor's exposure area, and this entire area should be used determining the EPC. A key challenge to resolve upfront 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 a particular exposure pathway. Since ecological PCLs protective of wildlife are not usually known at the time of the initial assessment, the *ERAG* (1.5.1) discusses the selection of an initial assessment level so that the affected property for ecological receptors can be effectively delineated.

To recap, a subdivision of the affected property sediments according to property-specific or receptor-specific characteristics is the exception rather than the norm. For these exceptions, the exposure area of affected property sediments should be delineated based on the receptor's natural history. COC concentrations within these unique exposure areas would be averaged to compute the EPC for that receptor. Examples of situations that should lead to the creation of exposure areas potentially smaller than the area of the affected property sediments are discussed in more detail in **3.3.3.2**. Once the exposure area has been defined, the information and assumptions that support the identification of the exposure area should be included in the risk assessment discussion.

The concepts of habitat, home range and foraging range related to soil exposure pathways are essentially the same for sediment as the exposure medium. Review **2.4.2.2**.

3.3.2.3 Data Quality to Support the Exposure Assessment

To ensure adequate exposure assessment of wildlife, data for the affected property sediments must first meet basic requirements for quality and accuracy (refer to **3.1** for more details). Important considerations for sediment sample depth are already detailed in 3.1.3. Additionally, consideration of erosion (e.g., scouring of surficial sediments) and potential remedial measures (e.g., dredging) may dictate collection of deeper sediment samples, particularly where it is important to evaluate potential ecological risks associated with deeper sediments that may become exposed. For water bodies that are characterized by stretches of exposed bedrock, sediment sampling should target depositional areas downstream.

Is the substrate sediment or soil? Flowing creeks and streams have a transitional riparian area that can be either soil or sediment depending on affected property conditions, so persons performing risk assessments should be cognizant of these variable exposure conditions. Wildlife receptors foraging in these areas may be exposed to both media depending on the current flow regime, and the duration and depth of exposure to either media. This variability should be addressed in the conceptual site model for the risk assessment and in the uncertainty analysis. The discussion should reflect reasons why the approach for a riparian area is appropriately conservative. Another consideration is that many streams in Texas are intermittent during dry months. Stream substrate that resembles soil may in fact be treated as sediment and soil depending on the wet-dry cycles for the water body in question. See the *ERAG* (3.9.2.6) for a more detailed discussion. Accordingly, persons should consider all of the potential exposure scenarios.

As always, it is imperative that there is an adequate nature and extent characterization, and that the sampling approach targets depositional areas dominated by fine particles (see **3.1.6**). This is particularly important for sediment assessments, since COCs may be transported long distances from potential sources. Satisfying these criteria ensures that potential ecological risks associated with sediment COCs are suitably characterized. Additionally, the person should ensure that sufficient and appropriate sediment data are available

to calculate a meaningful EPC and to correctly evaluate the affected property's potential for risk to wildlife receptors.

3.3.3 Exposure Point Concentrations for Wildlife Receptors

3.3.3.1 Introduction

The concepts previously discussed for soil exposure pathways are equally relevant for sediment exposure pathways. Review **2.4.3.1**. The discussion explains that a COC can be eliminated from further consideration at the assessment phase if the highest measured concentration is lower than either the property-specific soil background concentration, or the Texas median background soil concentration cited in the TRRP rule and the *ERAG*. Since the TRRP rule does not specify background COC concentrations for sediment, any background concentration for sediment will necessarily be property-specific.

3.3.3.2 Data Used to Determine the Exposure Point Concentration

Similar to the discussion for wildlife receptors exposed to soil, the term *EPC* (synonymous with the TCEQ's "representative concentration") generally represents the average level of exposure, expressed as a concentration, which a receptor may experience over an exposure area during an extended period of time. Therefore, the EPC should be estimated by using a conservative estimate of the true average value. The EPC for wildlife receptors exposed to sediment is computed from sediment concentration data within the exposure area, regardless of the measurement receptor's home range. Some wildlife receptors may have home ranges larger than the exposure area, and in these cases AUFs may be included in the exposure computation to address potential overestimation of true risks. As with the discussion of soil, where the home range of a particular wildlife receptor is smaller than the exposure area, the TCEQ does not expect calculation of a series of EPCs to represent each hypothetical home range within the exposure area. Rather, a single EPC represents the entire exposure area. However, this expectation assumes an evaluation of potential hot spots as discussed in **3.3.4**.

As indicated in **3.3.2.2**, the normal assumption will be that the entire affected property sediment data sets will be used to determine the EPC. Where it is appropriate to define an exposure area that is a subset of the affected property sediments for a particular receptor, there may be a single exposure area, or multiple exposure areas that are geographically separated. If it is likely, based on knowledge of the wildlife receptor's natural history information, that the receptor is able to use the separated exposure areas (i.e., for feeding or other behaviors that would lead to exposure), then the concentrations from all of these exposure areas should be combined to compute the EPC. As a result, unique EPCs for the single or multiple wildlife exposure areas will be determined as opposed to the normal practice of averaging across the entire data set for affected property sediments. Reasons for defining exposure areas that are subsets of the affected property sediments include:

- when protected species¹⁹ or their habitats exist within the affected property (therefore it is necessary that the habitats where the protected species feed are appropriately evaluated to ensure adequate protection)
- when the area of sediment used by the receptor (e.g., a diving or wading bird) is limited by water depth (see discussion that follows)
- when significant differences in physical features exist within a given area (e.g., differing reaches of a stream or watershed as tributaries join a main stem, physical habitat fragmentation, habitat differences that dictate prey availability)
- when risk management decisions are expected to result in multiple and distinctly different remedial actions (e.g., a portion of the affected property is addressed through an expeditious removal while the remainder undergoes the complete APAR process before any remedies are considered)
- when there are programmatic reasons to subdivide the affected property—e.g., Superfund or RCRA sites that divide up different areas of impacted sediment into operable or solid waste management units (SWMUs), or AOCs²⁰

Water depth can influence exposure to sediments; sediments covered by more than a few feet of water are inaccessible to non-diving animals (but may be available to benthic and fish communities as discussed later in this document). Water depth will constrain foraging by shorebirds, with larger species (those with longer necks, bills, and legs) feeding in deeper water than smaller species (Isola et al. 2000, and Bancroft et al. 2002). Dabbling ducks (in the genus *Anas*) such as the mallard, gadwall, and teal prefer water less than 12 to 18 inches deep (University of Maryland, n.d.; Yarrow 2009). In contrast, diving ducks (*Aythya* spp. and *Oxyura* spp.) make use of deeper water. For instance, canvasbacks (*Aythya valisineria*) generally forage for vegetation and invertebrates in water at depths of 0.5 to 2.0 meters and occasionally > 5 meters (Mowbray 2002). The foraging area may also be influenced by the bottom slope of the water body and sediment penetrability. In summary, for some waterfowl it may be appropriate to define an exposure area that is a subset of the affected property sediments. Where the waterfowl ecology dictates subdivision of affected property sediments, the SLERA should discuss the bird ecology and present relevant references. The evaluation should also discuss the exposure area size relative to tidal cycles or seasonal variability in water depth, as necessary. Sediments that are only intermittently covered represent a point of exposure, but if this is seasonal it may not coincide with a receptor's presence. Similarly, the analysis should consider

¹⁹ These receptors are often habitat-limited and the essential foraging area at the affected property may be smaller than the affected-property habitat.

²⁰ In addition to the calculation of ecological risks associated with the program-defined exposure areas, the SLERA may need to be supplemented with an evaluation that considers more comprehensive, site-wide ecological risks, particularly for receptors that may forage over multiple programmatically-defined areas.

the receptors likely to be present and the site hydrology when determining exposure area.

There may also be reasons to assume different sized sediment exposure areas for varying exposure pathways for a single receptor. For instance, consider a bird that ingests both benthic invertebrates and fish. The bird may be individually exposed to a smaller area of sediment while feeding on invertebrates than the area over which the fish range. Hence the exposure area for incidental ingestion of sediments and exposure to sediment from benthos (as prey) could be modeled differently than that assumed for the exposure to sediment from fish (as prey). Another hypothetical example is a receptor that only forages in brackish or saltwater habitats, but some of whose prey uses nearby freshwater ponds.

When subdivision of the sediment data set into separate exposure areas is contemplated for any reason, persons should ensure that the resulting data sets are sufficiently robust to calculate an EPC for each exposure area. Furthermore, persons should include sufficient discussion and justification for subdividing the data set for a particular receptor exposure pathway. The TCEQ acknowledges that a subdivision of the data set for a particular receptor may result in a less or more conservative EPC compared with the use of a value based on the entire affected property sediments. The TCEQ may request additional sample data if the approach represents an unacceptable level of uncertainty or does not afford an appropriate measure of conservatism.

3.3.3.3 Recommended Statistical Estimator for the Exposure Point Concentration

The concepts previously discussed for soil exposure pathways are equally relevant for sediment exposure pathways. Review **2.4.3.3**.

3.3.4. Evaluating Sediment Hot Spots for Wildlife Exposure

3.3.4.1 Introduction

The concepts previously discussed for soil exposure pathways are equally relevant for sediment exposure pathways. Review **2.4.4.1**.

3.3.4.2 Definition of a Hot Spot

The concepts previously discussed for soil exposure pathways are equally relevant for sediment exposure pathways. Review **2.4.4.2**.

3.3.4.3 Recommended Procedures for Identifying Hot Spots

INTRODUCTION

As with hot spots associated with soil COCs, the TCEQ has not identified any specific trigger for a hot-spot analysis for wildlife receptors exposed to sediment. Simply put, a hot-spot evaluation may be necessary for some exposure pathways, in addition to the HQ evaluation, in which the 95 percent UCL is the EPC for a wildlife receptor. The person preparing a risk assessment should determine if a hot-spot evaluation is needed. The hot-spot evaluation should be presented in the

uncertainty analysis. If a hot-spot evaluation is not warranted, a short rationale should be presented. The TCEQ will evaluate the adequacy of the hot-spot analysis (or the rationale for not performing one) and comment as necessary if more detail or clarification is needed. The TCEQ will also evaluate the conclusions of the hot-spot analysis and the associated risk-management recommendation, as appropriate. Figure 3.2 shows the different approaches for evaluating potential hot spots for wildlife exposure to sediment.

Unlike soil, the TCEQ does not recommend any classification of “small-ranging” or “wide-ranging” wildlife receptors based on home range in the evaluation of hot spots for sediment. With soil as the exposure medium, this distinction was important to the types of evaluations (individually or collectively) that the TCEQ recommends for evaluating potential hot spots. For wildlife that may be exposed to sediment (directly or indirectly), the recommended approach for evaluating hot spots is different. After the data-evaluation step discussed in “Practical Evaluation of Data for Wildlife Exposure to Sediment,” the TCEQ suggests that persons determine if a potential hot spot should be identified as more probable based on vulnerability. The TCEQ’s rationale is that various wildlife exposure pathways and receptors may be more susceptible to COC hot spots in sediment due to a receptor’s ecology, the habitat in question, or the COCs in question. These concepts are discussed in “Evaluating Vulnerability to COC Hot Spots in Sediment,” and include:

- sediment ingestion
- foraging strategy and prey preference
- home range and foraging area
- attractive-nuisance conditions
- patches of high-value habitat
- the presence of seeps and other springs

If a potential sediment hot spot coincides with one or more of the vulnerability scenarios described below, various aspects of the discussion of soil hot spots can also refine the hot-spot determination for sediment. The following concepts (from the discussion of soil hot spots in **2.4.4.3**) may be used to further evaluate and refine, individually or in combination, the hot-spot evaluation for wildlife exposure to sediments:

- LOAEL-based hazard quotient > 1
- LOAEL-based hazard quotient < 1
- Spatial relationship of sample locations with elevated concentrations
- Single-point LOAEL-based hazard quotient ≥ 10
- Assessing habitat and local population of small-ranging receptors

Regarding the last item listed, all topics presented could be used to evaluate potential hot spots for wildlife exposed to sediments. As with the soil discussion,

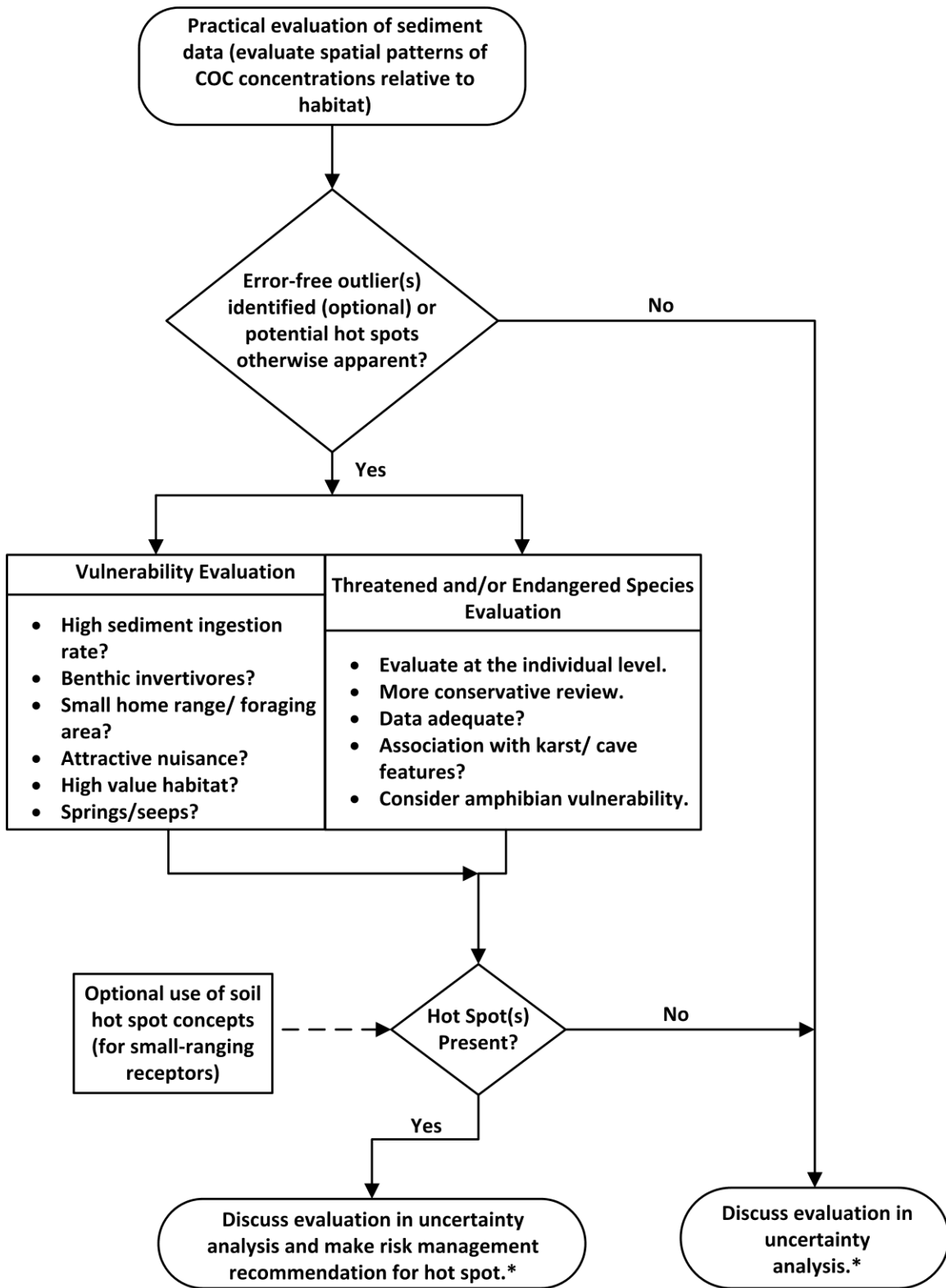
these types of evaluations should focus on the potential risks for a local population if a potential sediment hot spot occurs within the habitat for a receptor-habitat combination identified as potentially vulnerable. This type of assessment (last item), as discussed with soil hot spots, will typically be more rigorous than contemplated in an average Tier 2 SLERA.

PRACTICAL EVALUATION OF DATA FOR WILDLIFE EXPOSURE TO SEDIMENT

Paramount to a hot-spot evaluation is an initial evaluation and presentation of the COC concentrations on a map. Visualization of sample locations exhibiting elevated concentrations of COCs can be helpful in determining if these data points are spatially discrete and distinct from surrounding areas, or if the elevated concentrations are grouped together. Consider spatial patterns of any elevated concentrations relative to the exposure area generically, and specifically for any receptor with unique habitat and forage needs. If specific habitat information is available, data visualization should be used to determine if the sample locations exhibiting elevated concentrations cluster together in islands of habitat or in wildlife corridors. Data visualization may also be useful in identifying any data gaps in the spatial coverage of sediment samples.

Persons may elect to use a statistical-outlier test to identify potential hot spots in sediment. If statistical outliers are identified and the elevated concentrations can be attributed to an error (e.g., lab or sample-collection error, data-entry or transcription error), the erroneous data should be removed from the ERA dose and HQ calculations, determination of the EPC, and identification of hot spots. Any data points removed should be clearly identified and discussed in the submission to the TCEQ (e.g., ERA, APAR). Where data are removed from the exposure area data set, persons should consider if additional sampling or analysis is needed to ensure a statistically robust data set with adequate sample density to evaluate the applicable exposure pathways. The remaining outliers should be viewed as potential sediment hot spots warranting further evaluation.

A statistical-outlier test to identify potential hot spots is a suggested tool—not a requirement—particularly where data for the affected property are abundant. If the data set is too small to perform a statistical-outlier test, consider whether the set is robust enough to allow calculation of a 95 percent UCL or to support an adequate ERA. Although the data set may be small, the TCEQ acknowledges that, in some cases, potential hot spots may be readily apparent such that the need for more sediment data is obviated. More guidance regarding the evaluation of outliers appears in Appendix B.



*This is in addition to the “normal” risk evaluation using the 95% UCL as the EPC.

Figure 3.2. Evaluating potential hot spots for wildlife exposure to sediment.

EVALUATING VULNERABILITY TO COC HOT SPOTS IN SEDIMENT

Consider whether a particular wildlife receptor may be more vulnerable to local population impacts (from hot spots) due to its sediment-ingestion habits, home range (or foraging area along a stream), foraging strategy or prey preference, or other characteristics. These concepts are discussed below.

Sediment Ingestion. Receptors with particularly high rates of sediment ingestion (e.g., sandpipers, plovers, raccoons) may be more vulnerable to COC hot spots in sediment, since sediment ingestion will account for a large proportion of the receptor's overall COC dose. Additionally, some COCs may be present in higher concentrations in the sediment than in the food of a receptor. Receptors that dig or probe in the sediment in search of food will inadvertently ingest sediment as they forage. Deliberate ingestion of sediment by wildlife (for nutrients, grit, and nest construction) could also increase a receptor's risk of exposure to sediment hot spots.

Foraging strategy and prey preference. Benthic invertivores may be more vulnerable to sediment COC hot spots since certain benthic species are relatively sessile and may readily uptake some sediment COCs. Additionally, receptors that feed on sediment invertebrates are also exposed to increased concentrations of sediment that may be in the gut or on the body surface of the prey.

Home range and foraging area. Receptors with a small home range or foraging range may be more susceptible to COC hot spots in sediment. Examples include the spotted or stilted sandpiper, the marsh wren, and the killdeer. Sample locations exhibiting elevated COC concentrations that cluster together could represent a hot spot for receptors with limited range. Persons may choose to collect additional sediment data or perform a limited biological and habitat survey (or both) to reduce the uncertainty associated with a suspected cluster of elevated concentrations that would otherwise be identified as a hot spot. Keep in mind that, in contrast with terrestrial receptors associated with soil, territory and home range for receptors associated with sediments are often dependent on the length and width of shoreline where sediments occur. In these cases, home range is often expressed as a stream length (e.g., for the belted kingfisher in U.S. EPA 1993) rather than a contiguous land area (expressed in hectares or acres). Nevertheless, there is a width component so home range can be estimated as a narrow polygon or rectangle depending on the distance a receptor may forage from the shoreline.

Attractive nuisance. A potential hot spot will be of concern if the location functions as an attractive nuisance (e.g., a shallow pond in an otherwise arid landscape) for receptors and could result in higher exposure to elevated concentrations of COCs in sediment. Wastewater discharges may also function as an attractive nuisance if aquatic life and wildlife that prey or forage on aquatic species are attracted to areas of increased nutrients, oxygenation, or temperature associated with the release. Another concern would be a situation where predators are attracted to an exposure area because prey are more readily available due to impaired behavior or mortality as a result of contamination

associated with a hot spot. Examples include reduced antipredatory behavior of tadpoles exposed to heavy metals (Lefcort et al. 1998) and fish that are exposed to mercury (e.g., Webber and Haines 2003; Weis and Weis 1995).

Patches of High-Value Habitat. Consider whether a potential sediment hot spot could be associated with high-value or unique wildlife-habitat areas such as roosting and nesting areas or playa lakes.

Seeps and Other Springs. Natural springs and seepage areas are important aquatic resources. They can be reliable sources of drinking water and minerals, and may serve as habitat for prey or forage. An area of impacted sediment that overlaps seepage areas or springs may be a hot spot, as wildlife can be attracted to the location. Also consider the potential for a sediment hot spot around known areas of contaminated groundwater upwelling or seepage.

HOT-SPOT RISKS FOR THREATENED OR ENDANGERED SPECIES

Potential risks to a threatened or endangered species are necessarily considered at the level of an individual organism rather than the population. The premise behind this strategy is that a compromised population is less capable of tolerating the loss of individuals compared to a healthy population, and it is a violation of the Endangered Species Act to harm or otherwise “take” a threatened or endangered species or damage critical habitat. Accordingly, the conservatism of the TCEQ review will be greater and the level of effort put forth in the hot spot analysis may need to be increased 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 collecting more sediment data or conducting a field survey of the habitat (or both). Furthermore, the TCEQ may impose additional safety factors and conservative assumptions to prevent deaths and reproductive effects to listed species.

Sediment-affected properties that provide habitat for threatened and endangered species that may experience a higher level of sediment exposure (e.g., shorebirds) should be evaluated carefully for risks attributable to sediment hot spots. Persons should also be cognizant of depositional areas within creeks, seeps, or springs that are associated with karst features or caves, as these features often are habitat for various listed vertebrate species.

A number of listed 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 2013; Gunnar 2002). Potential sediment hot spots 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 sediment hot spots 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.

3.3.4.4 Risk Management for Sediment Hot Spots for Wildlife Exposure

Contaminated sediment management is a complex and multivariate operation, involving dynamic systems and large uncertainties that often dominate decision making. Although risk management of sediment hot spots is not immune to these challenges, decisions regarding hot spots may simplify the overall complexity. Because the size of a hot spot is most likely restricted (it does not usually comprise the entirety of the affected property), identification of sediment hot spots can help prioritize areas needing remediation and identify the type of remediation needed.

As noted in the risk management discussion of soil hot spots, determining what constitutes a hot spot for wildlife exposure and appropriate response actions are ultimately risk management decisions that are specific to the property. In this context, however, the person can suggest to the TCEQ what response actions are appropriate and can give a supporting rationale. For example, in some cases it may be appropriate to suggest a traditional type of remediation (e.g., removal or capping), while in others it may be more appropriate to evaluate the potential ecological impacts associated with various remedial actions and pursue an ESA. The ESA and any required compensatory ecological restoration must be performed in cooperation with, and with approval from, the Natural Resource Trustees for Texas (see Section 5.3 of the *ERAG* for more details). Additional sampling or more property-specific analyses may be recommended to ultimately refine the potential risk management alternatives.

The approach for evaluating sediment hot spots for wildlife exposure may be iterative. Initially, there may be a need for further sampling to reduce data gaps. These new data may influence the type of risk management that is appropriate or reconcile situations where uncertainty remains high with an existing data set. In some cases, additional sampling may not be needed, but further data evaluation and discussion with the TCEQ staff may be the more appropriate step. A facility may elect to address a hot spot upfront to minimize future study and investigation or liability. This may facilitate a more efficient subsequent evaluation of the remaining areas of sediment contamination.

Keep in mind the following guidelines:

- Determining the appropriate response action for hot spots is ultimately a risk-management decision specific to the property.
- The response action for a hot spot may differ from the action for the remainder of the affected property.
- The response action must take into account 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 hot-spot evaluation may be iterative. It may initially be used to define 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 address a specific exposure pathway).
- Persons 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 conceptual site model and sediment dynamics at the affected property is crucial to ensure that the remediated hot spot will not become recontaminated by new releases of COCs.
- 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 as long as there is no potential for recontamination from the affected property.

3.4 Exposure of Fish Receptors to Sediment

3.4.1 Purpose and Rationale

Potential risks to fish as receptors can be an important element of an ERA for impacted sediments. The fish community is a key component of freshwater, estuarine, and marine ecosystems. 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 key to the state's recreational and commercial fisheries. Threatened and endangered fish species can occur throughout the state and should be conservatively evaluated at the individual level where they may be present at a particular 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 forage-fish species. 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 affected property COCs. 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.²¹ The COC screening process for the sediment-to-fish pathway appears in **3.4.2**.

This discussion assumes that sediment data alone will be used to evaluate the sediment-to-fish exposure pathway, as is the case in a typical Tier 2 SLERA. Most often, sediment concentration data will be coupled with bioaccumulation factors or biota-sediment accumulation factors (BAFs or BSAFs) to estimate a tissue residue concentration. Unlike the typical approach for wildlife, a dose-based evaluation (expressed in mg/kg-day based on an ingested dose) of potential risks to fish is not the norm, as these types of TRVs are not readily available for fish. In a few cases, other endpoints for benthic fish species (sediment concentration thresholds to preclude deformities, lesions, and tumors) are available and can be proposed for use in a Tier 2 SLERA. Other tools (e.g., property-specific fish-tissue data, fish toxicity tests, and community analysis) are available for assessing risks to fish (from sediment), but these types of evaluations are more typically carried out within a Tier 3 SSERA, and therefore are not discussed here. Figure 3.3 summarizes the central questions for determining if the sediment-to-fish exposure pathway should be evaluated.

3.4.1.1 Sites to Be Evaluated

The TCEQ suggests that the sediment-to-fish pathway should be specifically evaluated at sites where the affected property sediments occur within a water body that meets the definition of a sustainable fishery. As presented in the TSWQS [30 TAC 307.3(a)(67)], *sustainable fisheries* include perennial streams and rivers with a stream order of three or greater; lakes and reservoirs with an area of at least 150 acre-feet or 50 surface acres; and all bays, estuaries, and tidal rivers. Additionally, all designated segments listed in Appendix A of the TSWQS (30 TAC 307.10) are presumed to have sustainable fisheries. Although Segments 1006 and 1007 of the Houston Ship Channel and its tidal tributaries do not have a designated aquatic life use, these water bodies should be evaluated as sustainable fisheries as this is the policy adopted by the TCEQ in the development of discharge permits under the Texas Pollutant Discharge Elimination System (TCEQ 2012a, as amended). Smaller water bodies that do not meet the definition of a sustainable fishery should be evaluated for the sediment-to-fish pathway only if there is a potential for protected fish species to occur at the affected property.

The TCEQ acknowledges that individual fish and fish communities can and do exist in water bodies that are not sustainable fisheries. For these, it is reasonable to assume that an assessment of sediment COCs for the benthic-invertebrate exposure pathway will be protective of the fish community.

²¹ Fish exposure to COCs in surface water is discussed in **4.2**.

3.4.1.2 Fish to Be Evaluated

Bottom-feeding fish, fish in direct contact with sediments, and upper trophic level fish that could be impacted by bioaccumulative COCs should be evaluated for the sediment-to-fish pathway. These types of fish are considered to be most at risk from exposure to sediment COCs. Examples of bottom feeding fish include catfish, drum, suckers, buffalo, croaker, striped mullet, and carp. Examples of fish that spend most of their lives in direct contact with sediments include flatfish such as flounder, bay whiffs, and tonguefish. Examples of upper trophic level fish that are exposed indirectly to sediment COCs through the food chain include largemouth bass, striped bass, blue catfish, flathead catfish, longnose gar, alligator gar, red drum, spotted sea trout, sand sea trout, and southern flounder.

3.4.2. Initial Screen for Evaluating the Sediment-to-Fish Exposure Pathway

An initial screen for evaluating the sediment-to-fish pathway is the use of the midpoint value between the primary benchmark and second effects level for benthic invertebrates. This midpoint value is described in the *ERAG* as the midpoint PCL for benthos (see *ERAG* 3.13 or TCEQ 2006). As in screening for the sediment-to-benthic invertebrate pathway, bioaccumulative COCs (see Table 3.1 of the *ERAG* or TCEQ 2006) 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. As in the process for benthic invertebrates, all COCs without a midpoint PCL specified in the *ERAG* should be retained for further evaluation. Because of the uncertainty of using screening values developed for benthos for evaluating the sediment-to-fish pathway, the midpoint PCL should not be used as a PCL for the pathway, but only to define what COCs warrant additional evaluation. For polycyclic aromatic hydrocarbons in sediment, only the total PAH midpoint PCL should be used for this screening step, rather than the individual PAH midpoint PCLs.

This evaluation is particularly important for COCs that bioaccumulate in fish. Here dietary exposure may be dominant; COCs that are not detected in water or are present at slightly elevated concentrations in sediment may result in high body burdens in fish. This pathway will usually be evaluated based on a modeled or measured fish-tissue concentration compared with an effects level.

For non-bioaccumulative COCs, it is assumed that the sensitivities of sediment-dwelling organisms to COCs are similar to 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 1991; 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.

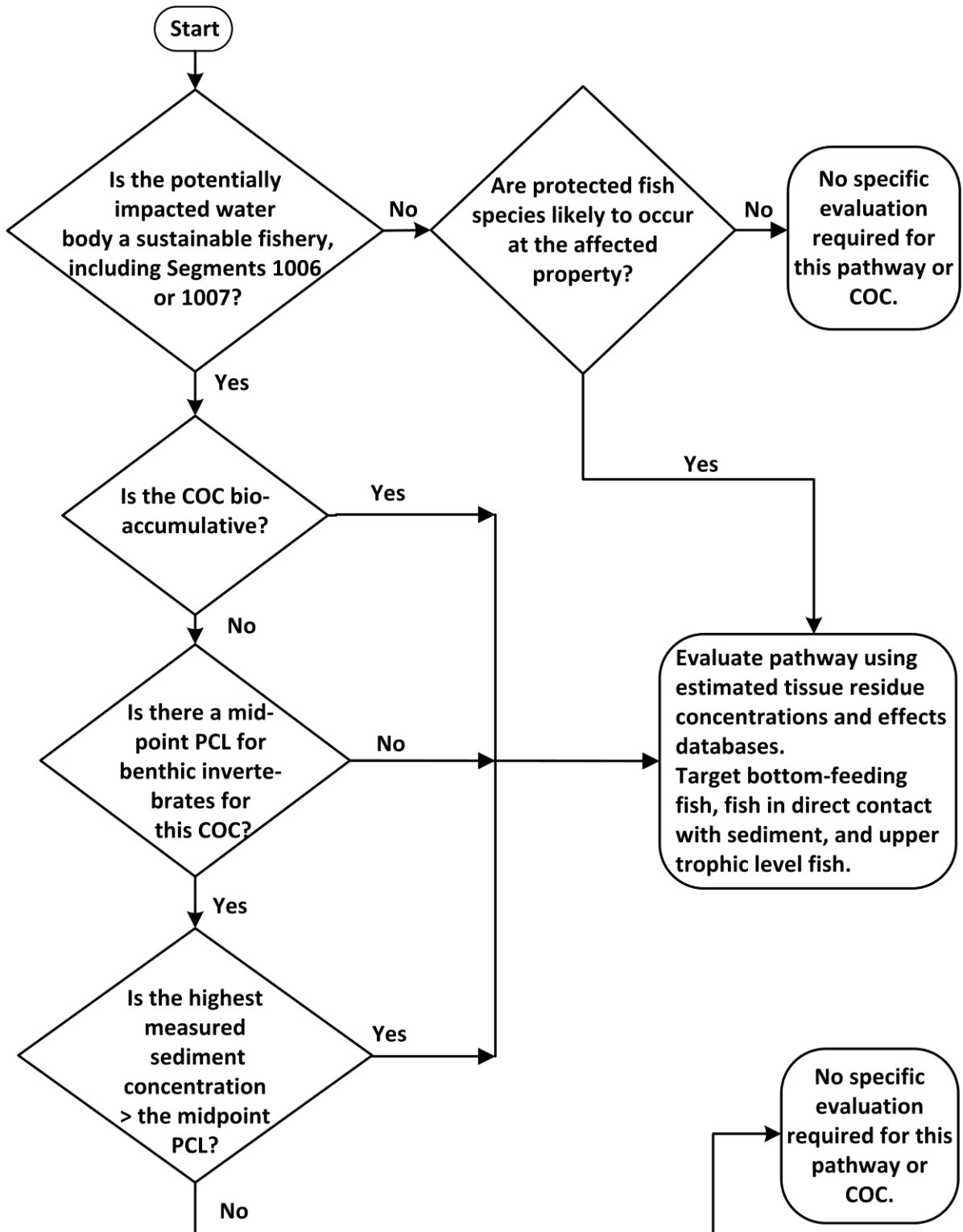


Figure 3.3. Evaluation of the sediment-to-fish pathway.

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.

3.4.3. Effects Databases for Evaluating the Sediment-to-Fish Exposure Pathway

Limited sediment guidelines are available for evaluating the potential risks of COCs in sediment to fish. Some studies have established relationships between COCs in sediment and the health of fish related to effects such as histopathological disorders, liver lesions, DNA damage, and reproductive abnormalities (e.g., Harshbarger and Clark 1990; Horness et al. 1998; Johnson et al. 2002). These sediment guidelines can be used as part of a WOE evaluation to evaluate potential risks associated with sediment COCs. By this we mean that these guidelines should not be the only tool used to assess potential risks associated with these groups of COCs.

As indicated already, the sediment-to-fish pathway is typically evaluated using estimated tissue residue concentrations based on sediment concentrations coupled with BAFs or BSAFs. 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). 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 specific to different tissue types (e.g., whole body, organs, eggs, larvae) and much 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. Typical effects databases used for this type of evaluation include the comprehensive database developed by Jarvinen and Ankley (1999), the ERED developed by the U.S. Army Corps of Engineers and the U.S. EPA (n.d.); and PCBRes, developed by the U.S. EPA (2009a). The database developed by Jarvinen and Ankley (1999) is available on-line in a searchable format as the Toxicity Residue Database maintained by the U.S. EPA (2009c). Additionally, COC-specific thresholds for fish-tissue residue have been developed for mercury and DDT (Beckvar et al. 2005), PCBs for juvenile salmonids (Meador et al. 2002), selenium for freshwater fish (U.S. EPA 2004), and dioxins (Steevens et al. 2005). Depending on the COC, these effects databases may provide 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 preferred. This is consistent with the discussion of TRVs in the *ERAG* (3.9.5). Persons 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 paucity of available data that link toxicity responses to tissue residues
- the variability and uncertainties associated with the use of BAFs and BSAFs
- the 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
- the 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 filet 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 tissue residue approach as a risk assessment tool appears in a number of papers (e.g., Meador et al. 2008; Barron et al. 2002; Sappington et al. 2011; McCarty et al. 2011; McElroy et al. 2011). Despite the uncertainties in using effects information based on tissue residue concentrations, this information remains the primary tool available for evaluating the sediment-to-fish pathway for Tier 2 SLERAs.

3.4.4. Assessment Considerations for Fish Communities

3.4.4.1 Communities vs. Individuals

The overall goal of any assessment and resulting risk management action is to be protective of the fish community as a group, as opposed to individual organisms. Species-specific evaluations are not typically performed, except under special circumstances, such as when threatened or endangered species are present. The *fish community* is loosely defined as a group of interacting fish species that occupy the same area.

For the purpose of assessing the fish community, fish selected as measurement receptors should include benthic fish and upper trophic level fish as appropriate for the affected property and the COCs in question (see discussion in **3.4.1**).

3.4.4.2 Exposure Areas for Fish

The *exposure area for fish* is defined as the area within the affected property sediments throughout which any life stage of the fish community may reasonably be assumed to move, and where direct or indirect contact (from ingestion of food or prey) with sediment is likely at all locations. 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. This would result in the determination of unique EPCs for the various measurement receptors that are selected for evaluation, as opposed to averaging across the entire data set for the affected property sediments. As with other pathways, the decision to subdivide will be the exception. The generic approach presented herein, however, is to assume that all of the affected property sediments make up a receptor's exposure area, and this entire area should be used in the determination of the EPC. This is particularly true for fish compared with benthic invertebrates, as fish are much more mobile and are more likely to integrate their exposure over a wide area of sediments. See 3.4.5.1 for detailed examples of situations that should lead to the creation of exposure areas potentially smaller than the areal extent of the affected property sediments. Once the exposure area has been defined, the information and assumptions that support its identification should be included in the risk assessment discussion.

3.4.5. Exposure Point Concentration for Fish Communities

3.4.5.1 Data Used to Determine the Exposure Point Concentration

See 2.4.3.2. As with wildlife, selection of an EPC for a particular exposure area conservatively assumes that fish live and feed throughout the exposure area, and that of their life cycles, in whole or in part, are completed in the area. Consistent with 3.4.4.2, the EPC is computed from sediment-concentration data within the exposure area, regardless of the measurement receptor's home range. Many fish have home ranges larger than the typical footprint for affected property sediments. In these cases, AUFs may be included in the exposure computation to address potential overestimation of true risks. Additionally, migratory fish may be selected as measurement receptors (e.g., some upper trophic level species). As migrants, these fish are not ideal for evaluation of the sediment-to-fish pathway. If there is reason to evaluate them in a Tier 2 SLERA, the TCEQ suggests selecting fish species that demonstrate limited movements outside a relatively small home range, and high site fidelity and pronounced homing outside of the seasonal migration.

In some cases it may be appropriate to subdivide the affected property sediments and calculate the EPCs for specific exposure areas or receptors. Again, subdivision is the exception, not the norm. Some examples include:

- when protected fish species or their habitats exist within the affected property (i.e., the habitats where the protected species feed are conservatively and appropriately evaluated since these receptors, particularly for fish associated with spring systems or specific watersheds, are often habitat-limited and the essential habitat at the affected property may be smaller than the total affected property sediments)
- when significant differences in physical features exist within a given area (e.g., differing reaches of a stream or watershed as tributaries join a main stem, physical habitat is fragmented by roads, dams, or saltwater barriers)

- when site observation or local knowledge suggests sediment exposure to fish or to a specific fish species is not uniform across the affected property (e.g., fish or fish species are regularly observed congregating in one or more areas, such as within warm effluents releases or spring waters, in contrast with exposure across the affected property sediments)
- when risk management decisions are expected to result in multiple and distinctly different remedial actions (e.g., a portion of the affected property is addressed through an expeditious removal while the remainder undergoes the complete APAR process before any remedies are considered)
- when there are programmatic reasons to subdivide the affected property (e.g., Superfund or RCRA sites that divide up different areas of impacted sediment into operable units or SWMUs, or AOCs)²²

When division of the affected property sediments is contemplated for any reason, persons should ensure that the data set is sufficiently robust to calculate an EPC and sufficient discussion and justification for subdividing the data set for a particular receptor-exposure pathway should be provided.

3.4.5.2 Recommend Statistical Estimator for the Exposure Point Concentration

Similar to the discussion in **2.4.3.3** for soil exposure pathways, the TCEQ has selected the 95 percent UCL as the preferred EPC for assessing potential risks to fish-community receptors since the goal is to protect fish at a community scale, and not individually (except for threatened and endangered species). An arithmetic or geometric mean should not be used in lieu of the 95 percent UCL. A more detailed discussion regarding its use and limitations associated with some data sets appears in the discussion of wildlife exposure to soil (see **2.4.3.3**). A separate hot-spot analysis (see **3.4.6**, below) should be performed to identify unusually high COC concentrations relative to other sample locations where threatened and endangered fish species may be associated with affected property sediments.

3.4.6. Evaluating Sediment Hot Spots for Fish Exposure

For the sediment-to-fish exposure pathway, a specific hot-spot evaluation should be necessary only where a threatened or endangered fish species may be present at an affected property. The TCEQ recognizes that this limited focus differs from that presented for wildlife exposure. The TCEQ also acknowledges that the tools for evaluating risks to fish within the confines of a Tier 2 SLERA are limited, and to routinely require an additional hot-spot evaluation is an unnecessary step that may compound the uncertainty of the assessment. Because of their inherent mobility, most fish communities should be less susceptible to COC hot spots in

²²Where fish exposure areas align with specific AOCs or similar program-defined areas, the SLERA may also need to evaluate more comprehensive, site-wide ecological risks from sediment exposure, particularly for fish that may forage across multiple programmatically defined areas.

sediments than benthic macroinvertebrates and some wildlife receptors. The hotspot analysis for benthic invertebrates should generally be protective of the fish community overall.

However, the TCEQ believes this hot-spot evaluation is necessary and appropriate where a threatened or endangered fish species may be present at an affected property that includes freshwater habitat. Because potential risks to a threatened or endangered species are considered at the level of an individual organism rather than at the community level, this enhanced conservatism is warranted. Since most protected fish species are constrained by habitat (such as species associated with spring features), a hot spot could be a risk for the individual organism if it coincides with the organism's local environment. The TCEQ does not suggest a similar evaluation for marine-estuarine habitats because of the limited number of protected fish species that may occur along the Texas coast. For example, although the smalltooth sawfish (*Pristis pectinata*) could occur anywhere in the Gulf of Mexico, the current range of this species has contracted to peninsular Florida (NOAA Fisheries 2013). Additionally, eggs and larvae of most marine species are buoyant or pelagic and would not be exposed to sediment for prolonged periods as would their freshwater counterparts (Blaxter 1988). Most freshwater fish lay demersal eggs that sink to the bottom, are laid directly on the bottom, or are placed in nests or redds in the substrate. Hence, eggs and larvae (before so-called swim-up) can be directly exposed to sediment COCs during development.

Therefore, potential sediment hot spots should be evaluated carefully where listed fish species are possibly present at an affected property that includes freshwater habitat. The evaluation should consider small or unique watersheds (e.g., desert stream ecosystems), seeps, or other springs associated with karst features or caves as these locations are often habitat for listed species. A key consideration is whether the species is specifically dependent on a particular locale and whether this locale overlaps a sediment hot spot. For more mobile receptors, persons should discuss the size of hot spot relative to home range, residence time, and metabolism of the species in question. Other potential considerations for the receptor in question include:

- existence of toxicity data or residue-effects data for sediment COCs for the protected species or a toxicological surrogate
- life-stage sensitivities²³
- vulnerability and availability of food
- needs for spawning and rearing habitat (e.g., substrate type, flow, water depth)

Additionally, any uncertainty associated with the adequacy of the sample density, keeping in mind the ecology of the receptor, may necessitate more sediment data

²³ Does the potential hot spot occur in areas where nesting is expected? Are there effects data available for sensitive life stages such as eggs and larvae?

or a field survey of the habitat (or both). Selection of effects data and BSAFs or BAFs should be conservative. The TCEQ may impose additional safety factors and conservative assumptions to prevent deaths and reproductive effects among listed species.

The hot-spot evaluation should be presented in the uncertainty analysis. If the person determines that a hot-spot evaluation is not warranted, a short justification should be presented. The TCEQ will evaluate the adequacy of the hot-spot 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.

4

Surface Water Exposure Pathways

Before beginning a discussion of the ecological exposure pathways associated with surface water, persons should have a clear understanding of what is meant by “surface water.” Its definition and, by connection, what is regulated under TRRP can often be unclear. 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)(66)]:

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, 30 TAC 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.

In essence, 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. 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)

For the purposes of this guidance, 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 impacted surface waters are used for drinking, although that is not typically a major exposure pathway. Among the most significant considerations required to assess surface water exposure pathways are:

- the quality of the available data on surface water
- the nature and size of the exposure area
- whether the water body is freshwater, 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

These topics are elaborated upon more fully in subsequent sections.

4.1 Data for Assessment

The section is not intended to replace existing TRRP guidance or the *ERAG* related to design of surface water investigation, sampling methods, and assessment approaches. Additionally, the TRRP rule [30 TAC 350.51(k)] directs that persons collect and handle surface water samples in accordance with the requirements in TCEQ (2012), as amended, or use an alternative methodology approved by the executive director. The reader is encouraged to review this and other guidance (e.g., USGS, variously dated) for additional information. Appendix A discusses the appropriateness of compositing surface water samples for use in an ERA. TCEQ (2012) is a guide for personnel, TCEQ contractors, and other organizations that perform routine monitoring to support the agency's surface water monitoring program. Although this guidance should be useful and appropriate for surface water assessments at most TRRP sites, the scope, sampling objectives, and questions unique to a TRRP site may mandate the use of alternate protocols. This section centers on those overarching assessment issues that are key to evaluating ecological exposure to surface water. Typical problem areas for surface water assessments are emphasized.

The primary objective of surface water sampling and analysis is to determine whether site-related COCs have 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 Natural Resource Trustees) regarding data that are proposed for use in surface water exposure assessments. The intent 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, persons may (for example) develop an optional sampling work plan. The remainder of this section discusses key considerations in determining what data may be considered acceptable when assessing ecological exposures to surface water.

4.1.1 Routine Monitoring Parameters

When collecting surface water samples, the sampler should note particular water-body characteristics (e.g., general appearance and condition, surrounding vegetation and activities, biological activity, size, depth, and flow conditions) as this is important information 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 uncertainty discussion), as well as in the consideration of potential remedial options. Salinity levels in particular 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.

Record field measurements and observations in a logbook, and make the information available to the TCEQ, if requested. Additionally, retain details regarding the calibration, maintenance, and performance of field instruments and make them available if requested. In addition to the analysis of potential COCs in surface water, routine laboratory analyses may include pH, total suspended solids, and hardness (particularly if COCs include metals).

4.1.2 Sample Depth

For most TRRP sites, surface water samples collected 1 foot below the water surface are usually acceptable. The TSWQS specify that the numerical aquatic life criteria apply to samples collected at any depth, and that samples collected approximately 1 foot below the water surface are acceptable for assessing attainment of standards [30 TAC 307.9(c)(3)]. 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 the bottom or banks of the water body to evaluate potential ecological risks associated with the impacted groundwater. TRRP is very clear, however, that the monitoring point for the groundwater-to-surface water pathway is normally a groundwater monitoring well placed immediately up-gradient of the zone of groundwater discharge to surface water [see 30 TAC 350.51(f)]. See the related discussion of seep sampling in 5 and TCEQ (2007a).

4.1.3 Sampling Sequence

For flowing water bodies, surface water sampling should proceed from downstream to upstream locations to minimize the impact of sampling disturbances on water quality. Where surface water and sediment samples

are collected during the same sampling event, they should be collocated, and the water samples collected first.

4.1.4 Sample Timing, Flow Conditions, and Tidal Influences

High flow conditions should be avoided unless the intent of the sampling effort 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 2012a, as amended). However, exceptionally low flow should be avoided when assessing risks to aquatic life.

Small streams in Texas may experience intermittent flow in summer months and eventually run completely dry, while others maintain perennial pools when flow is interrupted. The TCEQ prefers scheduling surface water sampling to avoid exceptionally low flow when evaluating aquatic life exposure pathways. Since most TRRP assessment activities are limited to a narrow window of time, this may not be an option. In these cases the TCEQ recommends that surface water sampling be performed anyway, and the ERA should discuss the uncertainty of the data based on the non-ideal sampling time. If samples are collected when a water body demonstrates exceptionally low flows, compliance with acute criteria in almost every case, and compliance with chronic criteria for intermittent streams with perennial pools (i.e., not perennial streams), would be important considerations.

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.

Additionally, for both direct and indirect wildlife exposure pathways, sampling events may need to correspond with the time of the year that various wildlife receptors may forage or drink from the surface water body, particularly where COC concentrations are expected to vary throughout the year. In some cases, it may be necessary to sample a lotic surface water body when flow conditions are exceptionally low, as this may represent episodes of high COC exposure for wildlife receptors.

4.1.5 Metals in Surface Water

The aquatic life criteria for most metals (with the exception of mercury, selenium, and silver) are expressed in the dissolved form rather than the total recoverable form of the metal. Therefore, when evaluating compliance with the numeric 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. Dissolved concentrations can be estimated by filtration of samples before analysis (see discussion that follows), or by converting from measurements of total recoverable metals in accordance with

the Implementation Procedures, as amended (see TCEQ 2012a and TCEQ 2007a). The TCEQ prefers dissolved-metals data for surface waters, where appropriate, rather values derived from the mathematical conversion. Since a measurement of total recoverable metals is usually applied to ERA calculations for metal uptake through the food chain, both total-recoverable and dissolved metal analyses are preferred where surface water data will also be used to evaluate wildlife exposure pathways.

The water quality criteria and ecological benchmarks for many metals are very low (in the parts-per-trillion to low parts-per-billion range). Therefore, it is very important to use ultra-clean techniques in sample collection, handling, filtration, preservation, and analyses to avoid sample contamination and to enable low detection limits when evaluating surface water for trace metals. The sample collection procedure is known as the “clean-hands, dirty-hands” technique and is described in a number of documents (e.g., U.S. EPA 1996; TCEQ 2012b; USGS variously dated).

Speciation is an important consideration for metals such as mercury, selenium, arsenic, and chromium. The aquatic life criteria for chromium are specified for trivalent (Cr^{+3}) and hexavalent (Cr^{+6}) forms. However, the aquatic life criteria for the other metals mentioned are not currently speciated. Although it can be anthropogenic, trivalent chromium is often naturally occurring, environmentally pervasive, and a trace element in humans and animals. In contrast, hexavalent chromium is almost always generated by human activities from a number of commercial and industrial sources, including chrome plating, steel production, metalworking, tanning, paint and pigment manufacturing, glassmaking, cement manufacturing, and as a preservative in pressure-treating wood. Therefore, where hexavalent chromium could be associated with a TRRP site, it is important that the surface water analytes include this form of the metal rather than limiting the assessment to trivalent chromium. Additionally, if hexavalent chromium is included in the analyte list, it is important to remember that the holding time for this analyte is normally 24 hours. However, a 28-day holding time is possible with the use of ammonium sulfate buffer solution, as specified in EPA Method 281.6 (Guidelines ... 2007).

It may be appropriate to accompany measurements of hexavalent chromium with additional organic and inorganic analyses. Characterizing the geochemical conditions that influence and affect the reducing conditions that govern chromium speciation and stability could include analysis of divalent iron and divalent manganese, total organic carbon, and dissolved organic carbon.

4.1.6 Other Analytical Considerations

The accuracy and precision of analytical methodologies are significant in determining the suitability of surface water data for use in a risk assessment. Data must meet the specifications in 30 TAC 350.54 and in *Review and Reporting of COC Concentration Data* (RG-366/TRRP-13, TCEQ 2010c). Additionally, analytical data must be generated by a lab that the Texas

Laboratory Accreditation Program has accredited under the NELAC standard for matrices, methods, and parameters of analysis.

Because surface water PCLs and numeric criteria often approach the detection levels for many COCs, persons should be particularly careful in selecting the analytical method for surface water samples and the sample-handling procedures they use (e.g., see **4.1.5**). Analytical methods and associated detection limits should be at or below the applicable water quality criteria and surface water benchmarks. The TRRP rule [30 TAC 350.54(e)(3)] requires a standard available analytical method that provides a MQL below the necessary level of required performance for assessment and demonstration of conformance with critical PCLs. Where that is not possible, the rule further requires selection of the standard available analytical method that derives the lowest possible MQL for a given COC. This is especially critical for bioaccumulative chemicals in surface water such as PCBs, dioxins and furans, pesticides, organochlorine compounds, and some metals.

4.1.7 Sampling Seeps

Where impacted groundwater surfaces at seeps, it may be important to sample the seep water to evaluate this exposure route. This is discussed in **5.0** and TCEQ (2007a).

4.2 Aquatic Life Receptors

4.2.1 Introduction

An evaluation of risks to the aquatic community is a fundamental component of the ERA process at sites where COCs are released to an aquatic system. 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. Subsection **4.3.1** discusses surface water exposure for 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. This guidance is primarily written in consideration of risks associated with toxic COCs (e.g., metals, pesticides, chlorinated organic compounds) although **4.2.6** briefly discusses the consideration of conventional pollutants (e.g., chloride, sulfate, pH, total dissolved solids, and nutrients) that could also be surface water COCs in rare cases. 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).

4.2.2 Exposure Areas for Aquatic Life Receptors

Like other ecological exposure pathways, there may be reasons to divide the affected property into smaller exposure areas for aquatic life receptors, particularly where surface waters may be impacted over a large area. Variations in exposure caused by anthropogenic effects (e.g., releases and discharges from sources not part of the site assessment) 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. This subsection addresses when it may be appropriate to subdivide the data set for separate exposure areas. In the TCEQ's view, the decision to subdivide will be the exception rather than the rule. Persons should establish scientifically credible rationales for making decisions to subdivide the affected property into smaller exposure areas. Similarly, persons should present a reasonable rationale for not subdividing a surface water data set if the circumstances appear to conflict with the guidance that follows. Consider the following factors in determining whether an area should be evaluated as a whole, or whether the area should be subdivided in some way for the aquatic life exposure pathway. These are meant to be examples for discussion. This is not to say that any time any of these circumstances occur at an affected property the area should automatically be subdivided into different exposure areas. If such features do not result in expected or observed differences in communities or exposure, there is no need to divide the area. However, even absent such biological differences, there may be overriding risk management or practical considerations for dividing an affected property into different exposure areas.

4.2.2.1 Physical Features

If significant differences in physical features exist within a given area, consider the potential role those differences play in demarcating different aquatic life communities. Physical features could result in clear physical demarcations, such as those created by dams. Other physical features that could distinguish areas include tributaries or other significant hydrologic inputs, such as localized outfalls or groundwater influences. Additional examples of differences in physical features that may be used in determining exposure areas include:

- differing reaches of a stream or watershed as tributaries join a main stem
- coves in a lake or bay
- physical habitat fragmentation (e.g., roads, saltwater intrusion barriers)
- differences in flow (different portions of a water body are intermittent, intermittent with pool(s), or perennial)

4.2.2.2 *Spatial Distribution of COCs and Significant Differences in Water Chemistry*

A careful examination of the spatial distribution of COCs can offer insight on areas where risk may be more or less prevalent. With this in mind, there may be circumstances where certain groupings of data are preferred to quantify risk. In other words, it may not make sense to group data where the water-column chemistry and site COCs may cause profound differences in the exposure and composition of the aquatic community for reasons unrelated to the site or any releases in question. Consider the nearby presence of:

- industrial and municipal wastewater and stormwater outfalls
- mining discharges and runoff
- discharges of cooling water
- oil and gas exploration
- large differences in salinity, dissolved oxygen, or temperature

Where surface water data for the affected property have been subdivided according to separate exposure areas, compare the EPCs for the individual areas with the water quality criteria or equivalent values. This is discussed in **4.2.4**.

4.2.3 Consideration of Aquatic Life Use

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 the Houston Ship Channel Tidal (Segment 1006) and the 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.

4.2.4 Exposure Point Concentrations for Aquatic Life Receptors

Within the SLERA, the highest measured concentration for a given COC should first be compared with the surface water benchmarks to determine if a COC should be retained for further evaluation. If the highest measured concentration of any COC is less than the corresponding surface water benchmark, no further evaluation of that COC is required for the aquatic life exposure pathway. However, if the highest measured concentration of any surface water COC exceeds the surface water benchmark (and the COC is present above site-specific background concentrations), that COC should be retained for further evaluation of potential risks to aquatic life. Additionally, if a bioaccumulative COC (see Table 3.1 of the *ERAG* or TCEQ 2006) is present at concentrations above its background concentration, that COC should be evaluated further for potential risks to wildlife that may forage within the water body (see **4.3**). If a nonbioaccumulative COC is present at concentrations above its background concentration and the surface water benchmark, that COC should also be evaluated further for potential risks to wildlife that may forage within the water body (see **4.3**).

After comparing measured COC concentrations to benchmarks, the next step is a simple comparison of the surface water data's EPC to the appropriate water quality criterion (acute, chronic, or both) or equivalent value (for COCs without state or federal numeric criteria). The EPC for evaluating aquatic life exposure pathways deserves some discussion, as this subject brings to light questions regarding an approach typical of the 303(d) listing process rather than one typical of other ecological exposure pathways for TRRP. Briefly, the *303(d) List* is a list of impaired waters that are not meeting state water quality standards (uses and criteria) as defined by Section 303(d) of the federal Clean Water Act. Every two years, states must submit a list of impaired waters to the U.S. EPA for approval, and Texas must document the methodology used to add or delete waters from the existing list. The TSWQS [30 TAC 307.9(a), (e)(4)] generally defer to *TCEQ Guidance for Assessing and Reporting Surface Water Quality in Texas* (the *Surface Water Assessment Guidance*), as amended (e.g., TCEQ

2010b, 2008) for details concerning how surface water data are evaluated to assess standards compliance. This document is reissued every two years, after input from a stakeholder group, with each Texas Integrated Report. Based on historical data, the report describes the status of water quality in all Texas surface water bodies that were evaluated for a given period. It presents the assessment process, and identifies water bodies that are not meeting standards on the 303(d) List.

The approach to assessing compliance with the numeric criteria for aquatic life protection, as outlined in the *Surface Water Assessment Guidance*, changes periodically. The approach in this guidance is used to evaluate compliance with the TSWQS as part of 303(d) listing. Therefore, it is not practical or appropriate to use the approach outlined therein for evaluating the aquatic life surface water exposure pathway for a TRRP site. Ideally, this approach used in 303(d) listing uses data collected at a routine frequency over periods of more than one year from sample locations that are intended to be reasonably characteristic of major hydrologic portions of a water body. Surface water data of this type are not the norm for surface water assessments associated with most TRRP sites, because sampling events are usually limited to one or two deployments over a shorter period within a focused area of a water body. The TCEQ recommends the use of the 95 percent UCL to represent the EPC for assessing potential risks to aquatic life. If the surface water data set is small (less than 10 data points), that may warrant evaluation by a statistician to determine if statistics can be appropriately used. If not, then the highest measured concentration should be used to represent the EPC. See **2.4.3.3** within the discussion of wildlife exposure pathways for soil for details about the definition of the 95 percent UCL and the calculation of the EPC.

If most of the computed 95 percent UCL concentrations exceed the highest measured concentration in a data set (particularly true for small sets or sets with a large percentage of non-detect values), then persons may need to evaluate the appropriateness of the data set for estimating representative concentrations. Persons may also need to consider collecting additional samples from the exposure area to minimize variability and improve the quality of the data set (e.g., allow the use of statistics to compute a reliable 95 percent UCL). Alternatively, the highest measured COC concentration can be used as the representative concentration, although this should be done with caution.

4.2.5. Consideration of Hot Spots for Aquatic Life Receptors

Given the dynamic and transient nature of most COCs in surface water (particularly lotic systems) coupled with the inherent mobility of aquatic life, in most cases it is not necessary to perform a hot-spot evaluation for aquatic life receptors. A specific hot-spot evaluation should only be necessary where a threatened or endangered freshwater fish, amphibian, or invertebrate species (e.g., a threatened mollusk) may be present at an affected property. Because potential risks to threatened or endangered species are considered at the level of an individual organism rather than at the community or population level, enhanced conservatism is warranted. Since most protected fish, amphibian, and

invertebrate species are constrained by habitat (such as species associated with spring features), a hot spot could be a risk for the individual organism if it is coincident with the organism's local environment. The evaluation should consider small or unique watersheds such as seeps and spring-fed streams, as these locations often include habitat for listed species. A key consideration is whether a species is specifically dependent on a particular locale and whether this locale overlaps a surface water hot spot. Other potential considerations for the receptor in question include:

- existence of toxicity data for surface water COCs for the protected species or a toxicological surrogate (assuming no state or federal standard is available)
- life stage sensitivities²⁴
- vulnerability and availability of food
- needs for spawning or rearing habitat

Additionally, any uncertainty associated with the adequacy of sample density and timing, keeping in mind the ecology of the receptor, may necessitate collecting more surface water data or conducting a field survey of the habitat (or both). The selection of effects data should be conservative. Additional safety factors and conservative assumptions may be imposed by the TCEQ to prevent deaths and reproductive effects among listed species.

In many cases, particularly where the source of COCs is impacted groundwater, the hot-spot evaluation may default to an evaluation of impacted groundwater (see 5). The TRRP rule [30 TAC 350.51(f)] stipulates that the monitoring point for the groundwater-to-surface water pathway is normally a groundwater-monitoring well placed immediately up-gradient of the zone of groundwater discharge to surface water. In this case, groundwater data are evaluated as surface water data in the ERA.

The hot-spot evaluation for aquatic life receptors should be presented in the uncertainty analysis. If the person determines that a hot-spot evaluation is not warranted, he or she should present a short justification. The TCEQ will evaluate the adequacy of the hot-spot analysis and comment as necessary if more detail or clarification is needed. The TCEQ will also evaluate the conclusions of the analysis and the associated risk management recommendation, as appropriate.

4.2.6 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 affected property. TCEQ (2007a; RG-366/TRRP-24) and, to a lesser extent, TCEQ (2010c) discuss selection of the surface water PCLs and risk-based exposure levels for these

²⁴ Does the potential hot spot occur in areas where fish nesting is expected? Are there effects data available for sensitive life stages such as eggs and larvae?

types of conventional pollutants. The TCEQ recommends that persons use the 95 percent UCL as the EPC for conventional pollutants. It will consider other approaches (e.g., average, geometric mean, or percentile), with some supporting discussion, since these pollutants are uncommon as COCs and guidance elsewhere is limited.

4.3 Exposure of Wildlife Receptors to Surface Water

4.3.1 Purpose and Rationale

Surface water is a principal medium to be evaluated in aquatic ecosystems because it directly and indirectly supports wildlife receptors. COCs enter surface water by:

- direct discharge
- releases of impacted groundwater
- spills
- runoff of impacted soil
- air deposition
- sediment desorption

Surface water is a primary depository and carrier of anthropogenic contaminants in the environment to which wildlife may be exposed via direct contact, ingestion, and food chain transfer.

The ultimate goal of the surface water investigation and assessment is the protection of wildlife populations, and individuals of 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 harmed by 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 surface water concentrations.

Although this discussion is focused on aquatic-based 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 this 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 come into contact with 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 threatened or endangered 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. Toxicology information for amphibians and reptiles for COCs may be available from Linder et al. (2003a, 2003b), Gardner and Oberdörster (2006), Pauli et al. (2000), Schuytema and Nebeker (1996), Sparling et al. (2010), or an online literature search from a database such as ECOTOX <cfpub.epa.gov/ecotox/> or TOXNET <toxnet.nlm.nih.gov>. For amphibians in particular, significant effects data [e.g., lethal concentration, 50 percent (LC₅₀) endpoints] are available for evaluating exposure to toxicants in surface water.

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 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). According to Burkhart et al. (2003), water-contaminant guidelines may not always be protective of amphibians principally due to their physiology, development, and life strategies. However, for conducting an ERA, the numeric water quality criteria specified in the TSWQS and ecological benchmarks (i.e., Table 3.1 in the *ERAG* or TCEQ 2006) 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).

There is one exception. In the event that 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. Although some effects data (e.g., LC₅₀ endpoints)²⁵ are available for evaluating amphibian exposure to COCs in surface water, toxicological studies demonstrate that many amphibians are often more sensitive to various COC stressors (Birge et al. 2000) when compared with fish and aquatic invertebrates. Therefore, if non-amphibian effects data are used, an uncertainty factor of 0.1 should be applied to a chosen concentration endpoint.

Herein, **2.4.2–2.4.3** extensively discuss exposure areas and EPCs. Rather than repeat many of these concepts, the corresponding discussions for surface water will specify where the previous soil discussions are appropriate for surface water exposure pathways for wildlife. Conversely, the text will indicate approaches that are different (or supplemental to) those recommended for soil.

²⁵ An LC₅₀ is the concentration of a chemical that is lethal to 50 percent of the test organisms as a result of exposure for a given period of time.

4.3.2 Assessment Considerations for Wildlife Receptors

4.3.2.1 Wildlife Populations

The concepts of populations, local populations, and feeding guilds—previously discussed for soil exposure pathways are equally relevant for surface water pathways. Review **2.4.2.1**.

4.3.2.2 Exposure Areas for Wildlife Populations

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 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 particular wildlife receptor may be modeled as a subset of the water body or bodies represented by the affected property's surface water. The standard approach presented herein is to assume that all of the affected property surface water represents a wildlife receptor's exposure area, and that entire area should be used in determining the EPC for surface water COCs. For the non-standard scenario (see **4.3.3.2**), the exposure area of affected property surface water should be delineated based on the receptor's natural history, and COC concentrations within these unique exposure areas would be included to compute the EPC for that receptor. Examples of situations that should lead to the creation of exposure areas potentially smaller than the area of the affected property surface water are discussed in more detail in **4.3.3.2**.

The concepts of habitat, home range and foraging range related to soil exposure pathways will be essentially the same when surface water is the exposure medium. See **2.4.2.2**.

4.3.2.3 Data Quality to Support the Exposure Assessment

To ensure adequate exposure assessment of wildlife, data for the affected property surface water must first meet basic requirements for quality and accuracy (refer to **4.1** for more details).

4.3.3 Exposure Point Concentrations for Wildlife Receptors

4.3.3.1 Introduction

The concepts previously discussed for soil exposure pathways are equally relevant for surface water exposure pathways. Review **2.4.3.1**. That discussion explains, in part, that COCs can be eliminated from further consideration at the assessment phase if the highest measured concentration is lower than either their site-specific soil background concentration or the Texas-specific soil background

concentrations (for metals) cited in the TRRP Rule. Since the rule does not specify background COC concentrations for surface water, any such concentration for surface water will necessarily be site specific.

4.3.3.2 Data Used to Determine the Exposure Point Concentration

As with wildlife receptors exposed to soil, the EPC (the TCEQ's "representative concentration"), generally represents the average level of exposure (see **4.4.3.3** for a discussion of the recommended statistical estimator for the EPC), expressed as a concentration, which a receptor may experience over an exposure area during an extended time. Therefore, the representative concentration should be estimated by using a conservative estimate of the true average value. The EPC for wildlife receptors exposed to surface water is computed from surface water concentration data within the exposure area, regardless of the measurement receptor's home range. Some wildlife receptors may have home ranges larger than the exposure area, and in these cases AUFs may be included in the exposure computation to address the issue of potential overestimation of true risks.

As indicated in **4.3.2.2**, the normal assumption is that the entire affected property surface water will be used in determining the EPC. Where it is appropriate to define an exposure area that is a subset of the affected property surface water for a particular receptor, there may be a single exposure area, or multiple areas that are geographically separated. This designation of exposure areas smaller than the affected property might result in the computation of unique EPCs for the various wildlife exposure areas as opposed to averaging across the entire affected property surface water. Some examples of where computing unique EPCs may be necessary include:

- when protected species²⁶ or their habitats exist within the affected property (therefore the habitat where the protected species feed must be appropriately evaluated to ensure adequate protection)
- when significant differences in physical features exist within a given area (e.g., differing reaches of a stream or watershed as tributaries join a main stem, physical habitat fragmentation, habitat differences that dictate prey availability)
- when risk management decisions are expected to result in multiple and distinctly different remedial actions (e.g., a portion of the site is addressed through an expeditious removal while the remainder undergoes the complete APAR process before any remedies are considered)
- when there are programmatic reasons to subdivide the affected property (e.g., Superfund or RCRA sites that divide different surface water areas into operable units, SWMUs, or AOCs)

²⁶ These receptors are often habitat-limited and the essential foraging area at the affected property may be smaller than the total affected property habitat.

When division of the surface water data set is contemplated for any reason, persons should ensure that the data set is sufficiently robust to calculate an EPC. Further, persons should sufficiently discuss and justify subdividing the data set for a particular receptor or exposure pathway. Where the affected property is subdivided for programmatic reasons, it may be necessary to accompany the dose and HQ calculations based on the subdivided areas with an evaluation that considers more comprehensive, site-wide ecological risks, particularly for receptors that may forage over multiple areas.

4.3.3.3 Recommended Statistical Estimator for the Exposure Point Concentration

The TCEQ has selected the 95 percent UCL as the preferred representative concentration for wildlife receptors exposed to surface water. The concepts previously discussed for soil exposure pathways are equally relevant for surface water exposure pathways. Rather than repeat text here, the reader should review the text in **2.4.3.3**.

4.3.4 Evaluating Surface Water Hot Spots for Wildlife Exposure

As described more fully in **2.4.4.2**, hot spots are areas of elevated COC concentrations and elevated risk. For the surface water-to-wildlife exposure pathway, a specific hot-spot evaluation should rarely be necessary. Three scenarios that may warrant the evaluation of potential hot spots are discussed in the following paragraphs, along with the associated evaluation recommendations.

The first scenario is where the surface water (and possibly groundwater) associated with a playa lake is impacted, and is used by mammalian and avian wildlife as a water source. Because of their mobility, mammalian and avian wildlife in general should be less susceptible to COC hot spots in surface water unless circumstances constrain the receptor population to obtain water and food from a confined locale such as a spring, seep, or playa lake. Playas are often the primary or sole source of water for fauna in arid portions of Texas and therefore attract and concentrate wildlife.

The second scenario involves a spring-seep feature, with elevated COC concentrations. Seeps and other springs can be a valuable supply of water for terrestrial birds and mammals, particularly during droughts, and are one of the first areas where vegetation emerges in early spring or otherwise support a perennial plant community important to wildlife in arid portions of the state. In the panhandle and far West Texas, spring and seep areas may be open, snow-free locales during winter that are used by wildlife as feeding sites. Additionally, wildlife may be attracted to springs with a high mineral content to supplement their diets with essential elements such as sodium, calcium, or iron.

For each of these two scenarios, the areas of elevated COC concentrations should be identified geographically, sampled specifically (including physical characteristics), and treated in the ERA as possible hot spots. For avian and mammalian receptors, discuss the size and persistence (if it periodically dries up) of a potential hot spot relative to the home range and residence time of the

species in question. A key consideration would be the potential for acute toxicity if a wildlife receptor population would be forced to use a particular location for water or food for a limited period of time. Alternatively, if the water feature is an attractive nuisance, potential chronic toxicity should be evaluated.

The third scenario is where a threatened or endangered amphibian species is potentially exposed to an area of elevated COCs within an affected surface water body. Since most protected amphibian species are constrained by habitat (such as a species uniquely associated with spring features or localized watersheds), a hot spot could be a risk for the individual organism if it coincides with the organism's local environment. Therefore, potential surface water hot spots should be evaluated carefully where listed amphibian species may be present at an affected property at a freshwater location. The evaluation should consider small or unique watersheds, seeps, and other springs associated with karst features or caves, as these locations often provide habitat for listed species. A key consideration is whether the species is specifically dependent on a particular locale and whether this locale overlaps a potential surface water hot spot. Potential considerations for amphibian receptors include:

- existence of toxicity data for surface water COCs for the protected species or a toxicological surrogate
- the timing of COC presence and life stage of amphibians²⁷
- vulnerability and availability of food
- habitat needs for spawning and rearing

The evaluation should consider, as appropriate, the potential for exposure of these receptors to surface water hot spots, as the amount of available habitat in temporary wetlands and pools diminishes with fluctuating water levels. COCs that may slow development or growth could decrease survival of larvae and fitness of adults. 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 (e.g., Bridges and Semlitsch 2005) if the water body dries up before metamorphosis is complete.

The hot-spot evaluation should be presented in the uncertainty analysis. If the person determines that a hot-spot evaluation is not warranted, a short justification should be presented. The TCEQ will evaluate the adequacy of the hot-spot analysis and comment as necessary if more detail or clarification is needed. The TCEQ will also evaluate the conclusions of the hot-spot analysis and the associated risk management recommendation, as appropriate.

²⁷ Is egg and larval development coincidental with COC pulses (particularly where the source of the COC is impacted groundwater)?

5

Groundwater as a Source Medium for Surface Water and Sediment

This section presents guidance on 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). Before discussing the ecological exposure pathways associated with this interface, persons should have a clear understanding of what is meant by “groundwater” and “surface water.” *Surface water* has already been defined in 4. *Groundwater* is a zone of water-saturated subsurface soils and geologic material. In TRRP, groundwater is delineated into classifiable groundwater-bearing units, as described in *Groundwater Classification* (RG-366/TRRP-8, TCEQ 2010a). Groundwater ecological pathways include discharges of groundwater to:

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

For the purposes of 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–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 surface water at the groundwater–surface water interface
- benthic invertebrate communities living within sediments exposed to pore water
- uptake of COCs in pore water by aquatic macrophytes rooted in sediments
- 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

Determining PCLs for Surface Water and Sediment (RG-366/TRRP-24, TCEQ 2007a) discusses determining the groundwater–to–surface water PCL (^{SWG}) and groundwater-to-sediment PCL (^{Sed}GW). This chapter is intended to provide additional clarity and perspective beyond the discussions in TRRP-24 and the *ERAG*. The remaining sections specifically discuss groundwater as a source medium for surface water and sediment impacts. These include:

- considerations for existing groundwater-assessment data
- data selection in the presence of active remediation systems

- seasonality and the temporal nature of groundwater
- groundwater well placement to accurately characterize the groundwater–to–surface water–sediment interface
- sampling seeps and pore water
- the presence and evaluation of elevated concentrations (e.g., hot spots) of COCs at the groundwater–surface water–sediment interface
- determining the groundwater EPC for assessing various ecological exposure pathways

More detailed discussions of the groundwater–to–surface water–sediment pathways and relevant assessment techniques are available elsewhere (e.g., U.S. EPA 2008a; Brodie et al. 2007; and Environment Agency 2009).

5.1 Data for Assessment

The TRRP requires investigation for the presence of groundwater beneath a site at which a release has occurred. Detailed instructions for performing groundwater investigations and assessments appear in TRRP-8 (TCEQ 2010a) and the APAR. 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. The delineation of each COC plume includes COC isoconcentration contours depicting the full extent and magnitude of the dissolved groundwater plume. If time-series data are available, information on plume dynamics is also depicted. This and other information on groundwater and the aquifer is normally included in Section 5 of APAR submissions.

This section is not intended to replace TRRP-8 (TCEQ 2010a) when it comes to groundwater investigation design, sampling methods, and assessment approaches. Rather, this section focuses on assessment issues that are key to evaluating ecological exposure to surface water and sediment where impacted groundwater is the primary source medium. Ultimately, these data are used to support relevant ERAs and subsequent risk management decisions.

5.1.1 Groundwater Monitoring Network for Groundwater–to–Surface Water–Sediment Pathways

Standard methods such as those specified in TRRP-8 (TCEQ 2010a) should be used to evaluate groundwater as a source medium for surface water and sediment COCs. A monitoring array must be constructed for the groundwater–surface water interface for evaluating groundwater quality at the interface. The groundwater monitoring network must be constructed in such a way as to allow sampling of representative groundwater quality in the area of discharge.

Monitoring wells should be screened in the most transmissive zone of each applicable groundwater-bearing unit in a position most likely to intercept groundwater plumes up-gradient of the receptor locations they may affect. For

seeps and springs, monitoring well screened intervals should be placed as to intercept the same groundwater flow that discharges to those surface features.

5.1.2 Temporal and Seasonal Variation

Groundwater and receiving water bodies can be subject to seasonal fluctuations due to variability in the water table and surface water levels during cycles of dry and wet weather. Persons should determine the period of time when the groundwater contribution (flow or concentrations) to the receiving water body is greatest, and the evaluation of groundwater as a source medium to surface water or sediment should attempt to reflect this time period. Thus, groundwater or pore water samples should be collected during this period to represent this worst-case scenario for potential groundwater COC impacts to ecological receptors. As a conservative measure, the exclusive use of groundwater data from this period may be appropriate for evaluating potential risks to certain receptors, such as amphibians or threatened and endangered species, exposed at the groundwater–surface water interface, if these receptors are likely present during these time periods. Therefore, groundwater sampling should be planned accordingly to take into account seasonal variations in COC mass flux and time periods when more sensitive receptors or life stages are present. If more sensitive receptors are not expected in the receiving water at these times, then groundwater data collected across wet-dry cycles can be used to determine the groundwater EPC.

If multiple impacted groundwater plumes differing in volume or mass enter a water body, then it may be necessary or appropriate to determine unique EPCs for each COC in each plume using groundwater data from different wells and time periods. See 5.3.

5.1.3 Use of Existing Groundwater Monitoring Data

At sites for which time-series groundwater data exist, general deductions may be possible regarding future expectations for groundwater-plume behavior at the interface. Trend analyses on time-series data (such as the Mann-Kendall statistic—U.S. EPA 2009b) are useful for discerning trends in groundwater concentrations and facilitating monitoring decisions. For example, analytical results that indicate groundwater concentrations increasing over time should prompt an evaluation of the likelihood of a future exceedance of a groundwater–to–surface water PCL protective of ecological pathways ($^{SWGW}_{eco}$) or a groundwater-to-sediment PCL protective of ecological pathways ($^{SedGW}_{eco}$). Alternatively, decreasing groundwater concentrations over time may be used to facilitate decisions regarding termination of groundwater monitoring at the interface. Trend analyses can be applied to time-series intra-well monitoring data when single wells are used to represent groundwater concentrations, or can be applied to time-series discharge-averaged groundwater concentrations (i.e., the EPC discussed in 5.3) values.

Site data may show that groundwater concentrations in the core of the plume (up-gradient of the interface) significantly exceed or may exceed the $^{SWGW}_{eco}$ or $^{SedGW}_{eco}$ or may cause an exceedance of a surface water or sediment PCL. In these cases, the groundwater plume may be considered a future threat to the

interface, prompting consideration of a response action. Additionally, persons would be ill-advised to only rely on groundwater data from interface wells to evaluate potential risks to ecological receptors. Alternatively, where site groundwater data indicate no concentrations that exceed the $^{SW}GW_{eco}$, $^{Sed}GW_{eco}$, or the surface water PCL (^{SW}SW) throughout the groundwater plume, it may be presumed that the groundwater plume will cause no future concern at the groundwater–surface water interface.

Historical groundwater data may be presented in a risk assessment for qualitative discussions related to ecological exposures. However, more formal integration for quantitative risk assessment necessitates caution, as it must meet the specifications in the TRRP rule at 30 TAC 350.54 and *Review and Reporting of COC Concentration Data* (RG-366/TRRP-13, TCEQ 2010c), if it will be used in the quantitative risk assessment to characterize ecological-exposure conditions.

5.1.4 Using Groundwater Data Generated in the Presence of Active Remediation Systems

The effect of an active remediation system should be considered if there is a conclusion of “no ecological risk” for the groundwater–to–surface water–sediment pathways, based on groundwater data collected while the groundwater remediation system is in use. When the groundwater samples are collected from plumes that are affected by active remediation systems, there should be a risk management recommendation in the ERA to conduct groundwater monitoring to confirm that groundwater concentrations are stable or decreasing after the cessation of active groundwater treatment (e.g., with pump-and-treat systems).

Where a groundwater treatment system is in use when an ERA is submitted, there is substantial uncertainty regarding the future groundwater concentrations at the interface. Any increase in COC concentrations in groundwater at the groundwater–surface water interface above levels identified in the risk assessment will constitute a substantial change in circumstances as specified in the TRRP rule [30 TAC 350.35(d)]; potential ecological risks will need to be revisited, as well as any previously approved response actions and groundwater PCLs. As a risk management recommendation (and response action), the TCEQ may require future groundwater monitoring at applicable affected properties to address this uncertainty. Groundwater monitoring reports should clearly identify all appropriate groundwater–to–surface water monitoring wells (or seeps) to ensure compliance with the PCLs protective of ecological exposure pathways.

5.1.5 Multiple Groundwater Plumes

In cases where multiple groundwater plumes that are being managed separately (e.g., groundwater plumes from separate SWMUs or AOCs) contribute to the same receiving water body, each plume should be addressed separately. For example, an affected property may have multiple groundwater plumes with different response actions. Some plumes may have active treatment systems in place, and some may not. In both cases, it is necessary to establish unique EPCs for groundwater COCs for each plume. Where multiple plumes enter a water body, persons should ensure that groundwater concentrations are protective of

ecological receptors and that the combined risks from each plume are taken into account. Here, surface water (and sediment) monitoring may be appropriate to evaluate any cumulative effects of multiple plumes. Additionally, the TRRP rule [30 TAC 350.75(i)(4)(F)] states that persons may be required to take appropriate action to ensure that discharging groundwater plumes do not result in exceedances of surface water quality standards in significant areas of the potentially affected surface water body. This is addressed in more detail in TRRP-24 (TCEQ 2007a).

5.1.6 Sampling Pore Water²⁸

In some cases, sediment pore water samples can be collected in lieu of the normal practice of using groundwater sampled from a monitoring well network to evaluate the groundwater-to-surface water-sediment pathway. Although the water in the pore space is not exclusively groundwater, the idea is to collect samples representative of groundwater as it is flowing through the sediment rather than sample surface water that is in the sediment pores (or interstitial spaces). Pore-water sampling is not necessarily a requirement for evaluating the groundwater-to-surface water-sediment pathways. However, there may be reasons to collect pore-water samples, largely to reduce uncertainty associated with the groundwater concentrations at the interface:

- Pore water is likely more representative of the actual POE concentrations in sediment before dilution in the receiving water body.
- Biogeochemical processes can alter the chemical character of groundwater discharging to surface water, including the formation of daughter chemicals and the attenuation of groundwater COC concentrations.
- Monitoring wells may not be located appropriately to adequately represent groundwater quality being discharged to a receiving water body.
- Access issues may prohibit well installation at the interface.

Sample sequencing is an important consideration with surface water and sediment sampling to avoid cross-contamination. Sequencing for groundwater samples is not a significant issue except with the collection of pore water. For flowing water bodies, pore-water samples should be collected from downstream to upstream, as with sediment and surface water. Minimize sediment disturbance to avoid introducing sediment particles into the pore-water sample. When collected improperly, the sample may be more representative of the overlying surface water than the pore water itself. Surface water can infiltrate into a pore-water sample if the sample is collected too quickly or if the pore water is sampled at a time that does not reflect the temporal fluctuations of both the groundwater and the receiving water body. Several devices are available for collecting pore water, such as the Trident Probe (Chadwick et al. 2003), diffusion bags, pushpoint samplers, peepers, mini-piezometers, and other devices designed

²⁸ See definition in 3.1.5.

to maintain low flow and reduce infiltration of overlying surface water. Various methods and tools for the collection of pore water are summarized in Duncan et al. (2007), U.S. EPA (2008a, 2001), and Environment Agency (2009). Assuring that the samples are appropriately representative of exposure conditions can be complex, so coordination with the TCEQ staff is recommended.

5.1.7 Sampling Seeps and Other Springs

There is little difference between a seep and a spring. A *seep* is simply a spring with relatively low flow. Hydrologically, seeps and springs are essentially the same, the difference being the rate of discharge. A *spring* is a location where groundwater naturally emerges from the subsurface in a defined flow and in an amount large enough to form a pool or stream-like flow.

Where impacted groundwater surfaces at a seep or spring, it may be important to sample the water to evaluate this exposure route, particularly important in arid areas where ecological receptors depend on seeps or springs for drinking water. Seep or spring-water sampling may also be important where unique ecological assemblages or protected species are associated with the feature or when it is the most practical way to evaluate the quality of groundwater before it enters surface water. If groundwater surfaces at multiple seeps or springs, whether to sample all of these features would be a site-specific determination, depending primarily on the consistency of the groundwater concentrations across the seeps or springs, and the exclusivity of the exposure pathways and receptors at each seep or spring.

If a seep or spring is associated with a pool, the sampling point may vary depending on the objective. If it is to evaluate the water quality of the pool after the groundwater has emerged, then collect samples from the pool. Sampling personnel should ensure that debris and sediments are not introduced into the sample. If the objective is to evaluate the groundwater quality alone, it is preferable to directly sample the seep or spring at an emergence point (e.g., a fracture in the rock face, a hillside, the side walls of a drainage feature, banks at low tide) before the water enters the surface water body. Sampling methods more specific to the groundwater-surface water interface are described elsewhere (e.g., U.S. EPA 2008a, 2000). Sample timing is an important consideration where groundwater seepage volume and quality are seasonal or influenced by tidal cycles.

5.2 Appropriateness of a Groundwater-to-Surface Water Dilution Factor

The TRRP rule [30 TAC 350.75(i)(4)] states that persons may establish a surface water dilution factor when the concentration of a COC in groundwater at the zone of discharge to surface water exceeds the ^{SW}SW (surface water PCL) for any COC at the time the affected property assessment is conducted (with some limitations). Subsubsection 7.1.2.2 of TRRP-24 (TCEQ 2007a) details an approach for evaluating historical groundwater data to determine if a dilution factor can be applied to the surface water PCL. Briefly, if no COC concentrations measured at any ^{SWG}GW POE wells have exceeded their respective ^{SW}SW, then a

dilution factor is not allowed and the ^{SWG}W is equal to the ^{SW}SW (e.g., the dilution factor is 1). TRRP-24 explains how to evaluate historical groundwater data to determine if a dilution factor can be allowed. This determination should not be confused with the comparison of groundwater data to the $^{SWG}W_{eco}$ and $^{Sed}GW_{eco}$ PCLs.

5.3 Determining the Groundwater Concentration at the Groundwater-to-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. In such situations, it will be necessary to determine a representative groundwater concentration for use in the subsequent ERA. 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.

The most conservative representative groundwater concentration may be established from the highest measured COC concentration from the wells in the monitoring array. A site-specific groundwater EPC may be determined using a discharge-weighted approach (see 5.3.1 and Appendix D).

5.3.1 Determining a Discharge-Weighted Groundwater Exposure Point Concentration at the Groundwater-to-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. Since the ^{SWG}W calculation averages the groundwater discharge into the surface water system's discharge, the groundwater concentration variable also can be averaged. However, groundwater COC concentrations are not uniform across the groundwater plume along the interface, and groundwater monitoring networks along the interface usually are not spatially suitable to determine a representative groundwater concentration (C_{gw}) by arithmetic averaging. Therefore, discharge-weighted averaging is recommended for determining a groundwater EPC. See Appendix D for a detailed presentation of the recommended approach. The discussion in Appendix D presents the groundwater EPC as the C_{gw} variable in the discussion and equations. Appendix D also presents an example calculation for determining C_{gw} .

5.3.2 Groundwater Data to Use for Determination of the Groundwater Exposure Point Concentration

Because dissolved COC groundwater plumes are dynamic, groundwater concentrations at any given monitoring well are expected to be different from one monitoring event to the next. During the ERA, when the groundwater EPC is determined and a groundwater PCL is established as appropriate, the current groundwater data should be used for calculation of the EPC and for the purposes

of evaluating compliance with any groundwater PCLs protective of ecological exposure pathways. See also **5.1.3** for sites for which time-series groundwater data exist.

5.4 Groundwater Exposure Point Concentration for the Groundwater-to-Surface Water Pathway

See **5.3** and Appendix D for details of the methodology for determining the groundwater EPC across an interface. Aquatic life and wildlife receptors may move and forage throughout a water body. Coupled with the transient nature of the surface water environment, it is normally appropriate to determine a groundwater EPC across the plume interface (i.e., across wells) for use in the dose calculation for wildlife receptors and for comparison with state and federal water quality standards (in the case of aquatic invertebrates and fish).

However, if groundwater seeps into isolated pools or otherwise isolated water bodies used by threatened or endangered species, it may not be appropriate to calculate an EPC using groundwater data across the interface. Additionally, if site information indicates that groundwater discharges at discrete locations (e.g., preferential pathways such as paleo-channels or utility corridors) within a water body, or if some areas of discharge are known to be attractive to aquatic life or wildlife, it may be more appropriate to evaluate the groundwater data representative of that area only. For example, fish may be attracted to cooler, more oxygenated water in areas of groundwater discharge during summer months. For consideration of localized impacts represented by one or a small number of wells, the TCEQ recommends using the highest measured groundwater concentration as the EPC. That is also appropriate if localities within a water body serve as habitat for a threatened or endangered fish, invertebrate, or amphibian species.

5.5 Evaluation of the Groundwater-to-Sediment Pathway

The TRRP is very 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. Determine sediment EPCs for benthic and wildlife pathways as detailed in **3.2.2** and **3.3.3**. 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. Pore-water analysis may derive an additional measure of COC bioavailability for some receptors and COCs associated with groundwater (or sediment) (see, e.g., U.S. EPA 2007, 2008b). In these situations, pore water analyses may better indicate groundwater impacts to

sediment than bulk sediment analyses. Where sediment pore-water data are used to conservatively reflect groundwater impacts to sediment, persons should give a rationale for the pore-water sampling locations, and for an approach to evaluating 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.

Also consider whether discrete groundwater discharges cause bulk-sediment hot spots as discussed in **3.2.3**.

Appendix A

Use of Composite Samples in Ecological Risk Assessments

A.1 Introduction

Composite sampling is a technique that combines a number of discrete samples collected from a given exposure medium into a single homogenized sample for physical or chemical analysis. The composite sample can consist of individual samples collected at various locations, depths, times, or a combination thereof (i.e., vertical, lateral, or temporal).

Since information on chemical and physical extremes and variability may be substantially reduced compared to discrete sampling, the appropriateness of composite sampling is dependent upon the sampling objectives, site media characteristics, underlying data distributions, and statistical assumptions for the investigation (Brumelle et al. 1984; U.S. EPA n.d.). In general, averaging of composite sample data for use in ERAs is seldom appropriate, since the composite samples do not represent the variability among individual samples. A 95 percent UCL calculated using data from composite samples will always be lower than that calculated from discrete samples because of the lower variance obtained from composite samples (Mattuck et al. 2005). Although the computed EPC (i.e., a 95 percent UCL) lacks inclusion of the total variance between samples, composite sampling may derive a more representative EPC because more samples can be included across the area to be investigated. Nevertheless, averaging of composite sample results may be allowed if a measure of overall site exposure is sought, since such an average is often similar to one based on discrete sample results. Guidance on the acceptability of this approach for a specific site should be sought from the TCEQ staff.

Although composite sampling may offer greater site coverage at a decreased cost, information about the variability in the sample concentrations may be reduced, and hot spots may be masked (U.S. EPA n.d.). Conversely, certain types of composite sampling can increase the ability to detect hot spots by increasing the number of locations sampled (Ohio Environmental Protection Agency 2009). Composite sampling is generally more appropriate for sites when the distribution of COCs is expected or known to be random, and the variability is expected or known to be low. It may be difficult to demonstrate that both conditions are met at a particular site. Many sites have a nonrandom distribution of COCs in large areas of surface soil. Additionally, COC concentrations often vary greatly at sites (due to a wide scale of concentrations ranging from non-detects to high levels at hot spots).

For some situations, multiple composite samples are collected to obtain adequate volume or mass to support the necessary analytical requirements. Often the individual samples are collected in close proximity, such as with multiple casts of a sediment grab sampler. This type of composite sampling should not be

confused with compositing of samples over large areas. As discussed earlier, the TCEQ does not support the latter approach for most ERAs.

Composite sampling can improve the spatial coverage of an area without increasing the number of analytical samples. Areas with a known or expected biased distribution of COCs (e.g., stained soil, waste trenches, soil or sediment areas near a source or downwind or downstream of a source, and areas affected by known or suspected historical operational practices) should be sampled separately with either discrete samples or composite sampling restricted to within the biased area. In other words, don't combine samples across biased and unbiased areas.

If composite samples are used to evaluate any environmental media at a site, the size and shape of each sampling area should be discussed, and the subsample locations and depths should be well-documented along with the compositing technique (i.e., number of aliquots, sample processing, sample volumes) and proposed statistical analysis. This is particularly important where the sample data will be used to support an ERA. Additionally, discussions with the TCEQ ecological risk assessors and project managers are advised while the sampling plan is being developed.

U.S. EPA (n.d.) gives an overview of the use of composite samples to support an ERA. Much of the text in this introduction was derived from that source. The text that follows discusses the appropriateness and limitations of composite sampling for various media within the context of an ERA.

A.2 Use of Composite Samples for Soil Exposure Pathways

Except for incremental sampling where the sampling depth accurately reflects the surface soil interval, surface soil samples to support ERAs should generally not be composited. However, a depth-integrated composite sample (from a single core sample) may be used to analyze COCs in the subsurface soil interval (0.5 to 5 feet). The rationale here is that a burrowing animal is likely to receive its soil exposure across this depth interval. Similarly, food items (i.e., invertebrates and plant roots) may take up soil COCs across this depth interval. Samples that will be submitted for volatile COC analyses should not be composited due to the potential for COC loss during mixing.

Discrete subsamples should be of equal volume. The area over which they are collected and combined should be of similar size and shape (such as grid sampling) unless that is geographically impossible. Additionally, ensure that compositing leads to uniform sample mixing so that the laboratory can obtain representative subsamples for chemical analysis. In cases where soil is highly compacted or caked or has elevated plasticity due to high clay content, thorough mixing may not be achievable. Accordingly, field-based compositing is not recommended. Better mixing may be possible in a laboratory, but loss of mass of semi-volatile chemicals is possible if mechanical mixing is too vigorous. Therefore, it is generally inadvisable to composite soils that are particularly hard to mix.

Incremental sampling is a type of composite sampling that may be appropriate at contaminated soil sites such as those impacted by munitions or those with substantial land coverage. Incremental sampling is designed to provide a reliable estimate of the average COC concentration across a predetermined *decision unit*²⁹ by pooling a large number of individual increments, or samples, into a single analytical sample. For very large sites, it can substantially reduce analytical costs relative to discrete sampling. Some guidance and discussion is available (e.g., U.S. ACE 2009; Hawaii Department of Health 2009; Hawaii Department of Health 2011; U.S. EPA 2006a; State of Alaska Department of Environmental Conservation 2009; Ohio Environmental Protection Agency 2009; Interstate Technology and Regulatory Council 2012; Hadley et al. 2011). Incremental sampling was originally designed for characterization of large munitions sites, not for risk assessment. Nevertheless, it may be a reasonable assessment approach for some sites and COCs where there are appropriately defined decision units. If incremental sampling is being considered at a TRRP site, persons should contact the TCEQ ecological risk assessors and remediation project managers before any site-assessment activities that may generate soil data for the ERA.

A.3 Use of Composite Samples for Sediment Exposure Pathways

Ideally, chemical analyses should be conducted on discrete sediment samples collected from a single deployment of the sampling device at each location for the target depth interval. In practice, collecting multiple sediment subsamples (usually three to five) per location is often necessary when the proposed analyses (including chemical analyses, physical analyses, and toxicity testing) require larger volumes of sediment from the targeted depth than can be acquired in a single cast of the sampling device. Persons should coordinate with their contract laboratories while planning their assessment to determine required sample volumes. In these cases, the sampling device (e.g., a Ponar or Ekman dredge) should be deployed more than once at the same location, taking care to sample as close as possible to other casts there. In some cases, such as a cobble-bottom creek, it is only feasible to collect sediment with multiple scoops. Where multiple sediment samples collected within 1 foot of each other are composited, the analytical result of the composited sample can be used to determine the EPC (i.e., the 95 percent UCL), as appropriate. The sampling strategy should be noted in the risk assessment. The subsamples should generally be equal in size, evenly spaced, and from adjacent sampling locations.

Normally, sediment from the top aerobic layer (top 10 cm or less) is collected for use in an ERA. There may be reasons to sample deeper sediments to support risk management decisions (e.g., future dredging, possibility of scouring events, sediment deposition rate, presence of deeper-dwelling polychaetes). If sediment

²⁹ A decision unit is simply an area or volume that is targeted for characterization. The size of a decision unit is site-specific and represents the smallest volume of soil (or other media) about which a decision is to be made. Decision units, including the concept of an ecological exposure area decision unit, are discussed in detail in Interstate Technology and Regulatory Council (2012).

samples at greater depths are applicable to the ERA or are necessary for other reasons already mentioned, sediments collected from the targeted depth with each deployment of the sampling device (usually a core sampling device) can be combined with the other sediments collected from that depth at that station and homogenized to a uniform appearance by stirring after removal of large materials (e.g., woody debris, shells, rocks). Subsamples should then be taken from this composite sediment sample for chemical analyses, physical analyses, and toxicity testing. Vertical compositing should be limited to sediment depths in which exposure potential is uniform, depending on the exposure pathway.

Sediment samples collected for the analysis of volatile chemicals (e.g., total sulfides, volatile organic compounds) should not be composited or homogenized. These samples should be removed from the sampling device immediately after retrieval and placed in appropriate containers before homogenization and subsampling of the remaining sediment (that will be analyzed for other types of COCs).

Compositing is not recommended where combining samples could serve to dilute a highly toxic, but localized, sediment hot spot. Collect discrete samples from areas with known or expected elevated COC concentrations. Sediment samples from locations with different physical characteristics (such as grain size, organic content, and clay content) or different stratigraphic layers of core samples should not be composited. Similar to the recommendation for soil, ensure that compositing leads to uniform sample mixing and, where this cannot be achieved, samples should not be composited.

TCEQ (2012b) and U.S. EPA (2001) give additional details on the collection and homogenization of sediment composite samples.

A.4 Use of Composite Samples for Surface Water Exposure Pathways

For toxic COCs, 30 TAC 307.9(c)(3) of the TSWQS specifies that numeric aquatic life criteria apply to water samples collected at any depth. Thus, for water-column aquatic receptors, surface water samples used in the ERA may be depth-integrated composite samples or samples collected at approximately 1 foot below the water surface. Where wildlife exposure pathways are considered (e.g., mink or heron), depth-integrated composite samples are preferred, since prey may theoretically exist anywhere in the water column. Where the surface water body is well mixed, samples collected at approximately 1 foot below the water surface are acceptable.

Although depth-integrated composite samples are acceptable in some cases, surface water samples used in an ERA should not be composited across large areas. Large areas of surface water may include highly variable concentrations of contaminants (e.g., at an inlet, at the groundwater–sediment interface) that need to be identified in the ERA. In some cases, surface water composite samples collected over a given time period (flow-proportioned or composited over time) may be used to assess temporally variable events that are important to document

for the exposure pathway in question. These may include impacted soil runoff events, tidal cycles, varying groundwater seepage, and irrigation cycles.

A.5 Biological-Tissue Samples

Biological-tissue samples are often collected to support a Tier 3 SSERA. Examples include tissue from:

- fish
- small mammals
- benthic invertebrates
- soil invertebrates
- plants

In many cases, target tissues or organisms are small, and compositing is necessary to achieve a minimum analytical mass or to satisfy DQOs. Persons should coordinate with their contract laboratories while planning the assessment to determine the desired sample mass. TCEQ (2012b) guidance offers suggestions regarding the collection of fish- and shellfish-tissue composite samples for a variety of uses.

In general, if tissue samples are being composited to evaluate the COC concentrations in food or prey for a particular receptor or guild, tissue samples should ideally not be combined across species due to differences in bioaccumulation potential and the food size preference of the predators. If this is not possible, only combine species (from a particular size class) and tissue types (i.e., whole or certain parts) that a receptor or guild is expected to ingest. Further, depending on the pathway in question, consider whether COC concentrations in prey or food items of differing sex, development stage, age, lipid content, size, and trophic status are likely to differ at the affected property. In these cases, care should be taken to target certain prey or food items and to avoid compositing across these various categories.

The tissue-compositing scheme (e.g., number of aliquots, tissue type, species or taxonomic group, number of individuals, sample processing) should be documented in the ERA.

Appendix B

Outliers

Outliers are important to recognize in a data set because screening-level results initially depend on a comparison of the highest measured concentration to ecological benchmark values. If the highest measured concentration is the result of a measurement, entry, or sampling error, or is from a hot-spot anomaly location, then the initial screening-level results will be skewed. Similarly, a calculated representative concentration based on the entire sample data set may be unduly high because of the influence of a high-end outlier value.

An *outlier* is, in plain terms, a value that notably differs from other values in a given data set. In environmental data sets, outlier values tend to be in the high range. *Outliers*, as described in textbook statistics sources (e.g., Devore and Peck 1986), are values that are greater than 1.5 times the interquartile range (IQR) value on either end of the data distribution. The IQR value is the upper quartile value minus the lower quartile value. The upper quartile value is the median of the upper half of the sample data and the lower quartile value is the median of the lower half of the sample data.

Graphical representation of a data set, such as a histogram, box-and-whisker plot, normal probability plot, or quantile-quantile plot, will usually reveal outlier values and, sometimes provide additional insight. A histogram, for example, may reveal that the data set is actually composed of two distinct subpopulations that might be indicative of more than one contamination source. Examination of the data set through graphical representation is a good first step in searching the data set for outliers (and many other issues), because it works regardless of data set size and underlying distribution type.

A mild outlier is defined as greater than 1.5 IQR, but less than 3 IQR. An *extreme outlier*, of particular interest because it is mentioned in the *ERAG* (1.5.2), is defined as greater than 3 IQR (Devore and Peck 1986). A box-and-whisker plot will readily illustrate outlier values, even with small data sets, as it is not a statistical procedure sensitive to the distribution type.

As discussed in **2.4.4.3**, outliers related to errors (e.g., lab error, sampling error, data entry or transcription error, or matrix anomaly) should be removed from the data set and clearly documented in the APAR and ERA. Sources of error can be many, so this issue should be thoroughly investigated to find the possible cause for unusually high values. If errors are not a pertinent issue, outliers should be evaluated by consulting a sample location map. It may be apparent by examining the map that the outliers are related to hot spots at the site. Review of the sample location map and identification of a hot spot may lead to the decision to collect additional samples near the hot spot. Additional sample data would necessitate a new evaluation, including searching for outliers from the new data set. Hot spots may be remediated outside of the ERA process, if so desired (i.e., see **2.4.4.1**). These locations should be clearly documented in the APAR, including a discussion of the remedy and any confirmation sampling. A statistical test is not required for outliers to be removed from the data set, as long as the related media

samples are addressed (i.e., remediated as a hot spot) within or outside the ERA. The decision to remove outliers from the data set is a separate consideration from the identification of outliers and may necessitate input from the various stakeholders (regulators, Natural Resource Trustees, or responsible persons and their representatives).

If desired, the data set can be further evaluated for outliers by performing classical statistical outlier tests, provided that the data set is large enough or certain assumptions are met. Most outlier tests require the assumption of normality (log-transformed or not) for the data distribution, an assumption that can often be false for environmental data. For example, Dixon's test can be used on data sets as small as three values with an assumed normal distribution (minus the outlier) (U.S. EPA 2006b), but the assumption cannot be reliably evaluated by statistical procedures for such small data sets. Other tests include Grubbs' test (NIST/SEMATECH 2013) and Rosner's test (U.S. EPA 2010a), but there are many more that can be found in the literature. Any statistical analysis for outliers should be accompanied with a complete discussion, including the statistical tests used and their null hypothesis, literature references, assumptions and limitations related to the chosen tests, and printouts from the software used to perform the tests.

The decision may be made that an identified error-free outlier will remain a part of the data set and be used as the highest measured concentration to compare to the ecological benchmarks. If so, the ERA discussion should note that the highest measured concentration and possibly other high values have been identified as error-free outliers. Deliberation should be given to the decision to include an outlier in the subsequent step of computing the conservative average concentration (i.e., the representative concentration designated by the 95 percent UCL on the mean) for exposure analysis, because the representative concentration may be significantly higher when the outlier is included in the data set, depending on the size of the data set and the gap between the outlier and the next-highest value. If the error-free outlier is removed from the determination of the EPC, there should be an assumption that the sample location represents a potential hot spot for some ecological receptors and it **would require a response action**. Treatment of outlier concentrations should be discussed in the uncertainty section of the ERA report.

There is an assortment of sources to consult for guidance on evaluating outliers, including consensus organizations (ASTM 2008), governments (U.S. EPA 2006b; U.S. EPA 2010a; NIST/SEMATECH 2013), and most statistics reference books (e.g., Gilbert 1987).

Appendix C

Examples of Sediment Data Groupings

C.1 Freshwater Creek

Consider a freshwater creek that receives nonpoint surface water runoff from agricultural fields and paved roadways upstream of the stretch of the creek under evaluation. Assume that the area being evaluated is a 1-mile reach of the creek that may have been impacted by a release subject to TRRP. A former facility at the site mixed pesticides and cleaned engine parts with chlorinated solvents. Pesticides, metals and solvents are present in the soils and groundwater.

The potential mechanisms of COC releases to the creek are surface water runoff and groundwater discharges to surface water. The first half mile of the reach is channelized and all of the riparian vegetation has been actively cut back to accommodate increased flows caused by storms and releases from surface water drainage ditches. A major drainage that receives surface water nonpoint runoff from a large portion of the facility discharges into this channelized stretch. The second half mile of the reach is un-channelized and overgrown, as there has been no vegetation control. Trees have fallen into this portion of the creek and beaver dams have created riffle-run areas.

The creek was divided into three exposure area groupings for the risk evaluation:

1. channelized—upstream of the major drainage
2. channelized—downstream of the major drainage
3. the unchannelized portion

All three areas could have received releases from the TRRP site and nonpoint releases. This approach to the data grouping was appropriate for the following reasons:

- Based on field observations, the depth and type of sediment was consistent throughout the channelized area.
- Potential source areas existed along the channelized portion of the creek upstream of the major drainage that were not present downstream of it. Isolating the data into these two groups helped determine the source of COCs and was helpful in selecting locations for remedy application. From a habitat evaluation alone, however, the entire channelized portion of the reach could have been combined into one data group.
- Enough sediment samples were collected to allow for calculation of a 95 percent UCL EPC for each of the three areas.
- The un-channelized stretch presented a different habitat due to the riparian cover and the reduced energy flow attributable to the fallen trees and beaver dams. This area was diverse and ecologically active, as numerous wildlife species were observed. Additionally, the fallen trees provided nursery habitat for fish.

C.2 Estuarine Bay

At a hypothetical TRRP site adjacent to a Texas bay, stormwater is collected from the facility and discharges into the bay through a drainage ditch. The flow of the on-site surface water is managed by a series of gates associated with a levee system. The bay is tidally influenced, and brackish water will flow into the facility if the gates are not closed with incoming tides. The assessment has shown that the drainage ditches on-site have received COCs from on-site source areas. All of the on-site drainage ditches join the primary ditch that exits the facility and flows into the bay. Source areas originated from improper disposal of waste petroleum products. Site COCs include petroleum products, PAHs, PCBs, and pesticides that may have been released into the bay via the primary ditch.

For the SLERA, the on-site ditch system was evaluated as freshwater because of the documented and active control of the gates. Further, freshwater vegetation and freshwater invertebrates were found within the on-site drainage ditches. Given that some of the COCs are bioaccumulative, it was important to evaluate the nature and extent of potential releases to the bay. However, it was not economically feasible to sample bay sediments in a systematic grid pattern; therefore, an approach based on professional judgment was implemented, beginning where the primary freshwater ditch entered the bay and fanning out from this original point.

The data indicated that the COCs were rapidly dispersed into the larger bay area. Data were grouped into on-site freshwater sediment data and off-site estuarine sediment data. Data from the bay were further divided into (1) those locations in the bay near the mouth of the primary drainage ditch and (2) those locations where COCs may have dispersed throughout the bay from wave action. This approach to the data grouping was appropriate for the following reasons:

- Distinct EPCs (95 percent UCL) were calculated for the two areas in the bay since those two groupings of data represented different exposure scenarios.
- One area in the bay was influenced by freshwater input from the primary ditch and is less affected by wave action; the other grouping represents lower concentrations of sediment COCs over a large area.
- Field observations indicated that sediment accumulated at the freshwater discharge point within the bay, but sediments found away from the discharge point were transported by the wave action.
- Comparison of the two estuarine sediment data sets provided site-specific information on transport mechanisms within the bay.

Appendix D

Discharge-Weighted Representative Groundwater Concentration at the Groundwater–Surface Water Interface

For the calculation of ecological PCLs protective of surface water when groundwater is the source medium ($^{SWGW}_{eco}$) it is necessary to determine a representative groundwater concentration (C_{gw}). Since the $^{SWGW}_{eco}$ calculation uses a bulk value for groundwater discharge entering into and mixing with the surface water, a bulk representative groundwater concentration also can be used. However, groundwater COC concentrations are not uniform along the groundwater–surface water interface and groundwater monitoring well networks along the interface usually are not spatially suitable to determine a representative groundwater concentration by arithmetic averaging. Therefore, a discharge-weighted averaging procedure is recommended for determining a site-specific representative groundwater concentration. See Figure D.1 for a pathway schematic.

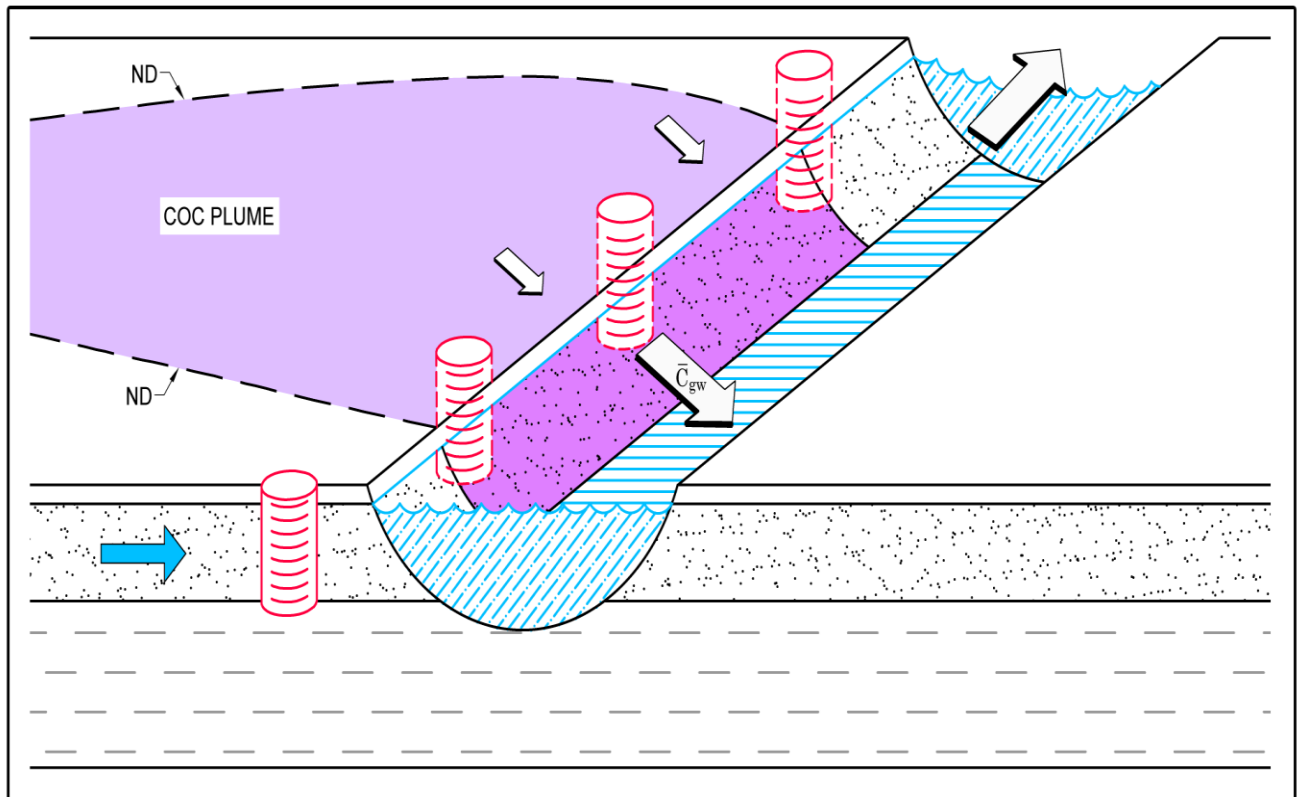


Figure D.1. Groundwater-plume seepage face at the surface water interface.

Since groundwater discharge to surface water can be treated as a bulk value (see above), a bulk groundwater concentration also can be used, or:

$$\text{Groundwater Representative Concentration} = \frac{\text{Total mass of COC}}{\text{Total volume GW}}$$

$$C_{\text{gw}} = \frac{m_{\text{COC}}}{V_{\text{gw}}} \quad [\text{EQ 1}]$$

EQ 1 permits the determination of C_{gw} by the summation of COC mass discharging from discretized areas along the groundwater–surface water interface. However, the groundwater monitoring network along the interface is linear and the groundwater volume in EQ 1 cannot be determined. But, groundwater discharge can be determined from data collected for the groundwater assessment (Section 5, APAR). Therefore, C_{gw} can be restated in terms of groundwater discharge as:

$$\text{Groundwater Representative Concentration} = \frac{\text{COC mass flux}}{\text{GW discharge}}$$

$$C_{\text{gw}} = \frac{\dot{m}_{\text{COC}}}{Q_{\text{gw}}} = \frac{\frac{m_{\text{COC}}}{t}}{\frac{V_{\text{gw}}}{t}} = \frac{m_{\text{COC}}}{V_{\text{gw}}} \quad [\text{EQ 2}]$$

where,

$$C_{\text{gw}} = \frac{\dot{m}_{\text{COC}}}{Q_{\text{gw}}} = \frac{\text{COC concentration} \times \text{seepage area} \times \text{specific discharge} \div \text{effective porosity}}{\text{Seepage area} \times \text{specific discharge} \div \text{effective porosity}}$$

or,

$$C_{\text{gw}} = \frac{\dot{m}_{\text{COC}}}{Q_{\text{gw}}} = \frac{C \times A \frac{q}{\eta_e}}{A \times \frac{q}{\eta_e}} = \frac{C \times A \times q \times \eta_e}{A \times q \times \eta_e} \quad [\text{EQ 3}]$$

where,

Specific discharge = aquifer hydraulic conductivity \times hydraulic gradient

$$q = K \frac{dh}{dl} \quad [\text{EQ 4}]$$

The determination of a *discharge-weighted* C_{gw} requires the summation of COC mass fluxes at each of n discrete seepage areas discharging affected groundwater to the surface water. There is one seepage area associated with each of the n monitoring wells. Combining EQ 2, EQ 3 and EQ 4 gives the summation:

$$\bar{C}_{gw} = \frac{\sum \dot{m}_{coc}}{\sum Q_{gw}} = \frac{K \frac{dh}{dl} \eta_e \sum_{i=1}^n C_i A_i}{K \frac{dh}{dl} \eta_e \sum_{i=1}^n A_i} \quad [\text{EQ 5}]$$

Since each discrete seepage area, A , is:

Area = aquifer saturated thickness discharging to surface water x distance along interface

then,

$$A = b \times d \quad [\text{EQ 6}]$$

For the case where hydraulic conductivity, hydraulic gradient, thickness of saturated zone and effective porosity have *constant values* along the groundwater-surface water interface of the groundwater plume, EQ 5 and EQ 6 can be rewritten:

$$\bar{C}_{gw} = \frac{\sum \dot{m}_{coc}}{\sum Q_{gw}} = \frac{K \frac{dh}{dl} b \eta_e \times \sum_{i=1}^n C_i d_i}{K \frac{dh}{dl} b \eta_e \times \sum_{i=1}^n d_i} \quad [\text{EQ 7}]$$

Where the aquifer parameters are *not constant* along the groundwater-surface water interface, EQ 7 is modified to accommodate the site-specific values in each interface-area cell as follows:

$$\bar{C}_{gw} = \frac{\sum \dot{m}_{coc}}{\sum Q_{gw}} = \frac{\sum_{i=1}^n C_i d_i K_i \frac{dh}{dl_i} b_i \eta_{ei}}{\sum_{i=1}^n d_i K_i \frac{dh}{dl_i} b_i \eta_{ei}} \quad [\text{EQ 8}]$$

Figure D.2 shows idealized discretized seepage areas and symbols.

The sum of the distances between wells along the groundwater–surface water interface, d_i , equals the width of the groundwater plume at the interface, as defined by the *non-detect (ND) COC isoconcentration contour* (see Figure D.2), for n number of wells:

$$\text{Width of plume} = \left(d_0 + \frac{d_1}{2} \right) + \sum_{i=1}^{n-1} \frac{d_{i+1} + d_i}{2} + \left(\frac{d_{n-1}}{2} + d_n \right) \quad [\text{EQ 9}]$$

The calculation of the discharge-weighted representative groundwater concentration for the condition in EQ 7 is:

$$\bar{C}_{\text{gw}} = \frac{K \frac{dh}{dl} b \eta_e \times \left[C_1 \left(d_0 + \frac{d_1}{2} \right) + \sum_{i=2}^{n-1} C_i \left(\frac{d_{i-1} + d_i}{2} \right) + C_n \left(\frac{d_{n-1}}{2} + d_n \right) \right]}{K \frac{dh}{dl} b \eta_e \times \left[\left(d_0 + \frac{d_1}{2} \right) + \sum_{i=2}^{n-1} \frac{d_{i-1} + d_i}{2} + \left(\frac{d_{n-1}}{2} + d_n \right) \right]} \quad [\text{EQ 10}]$$

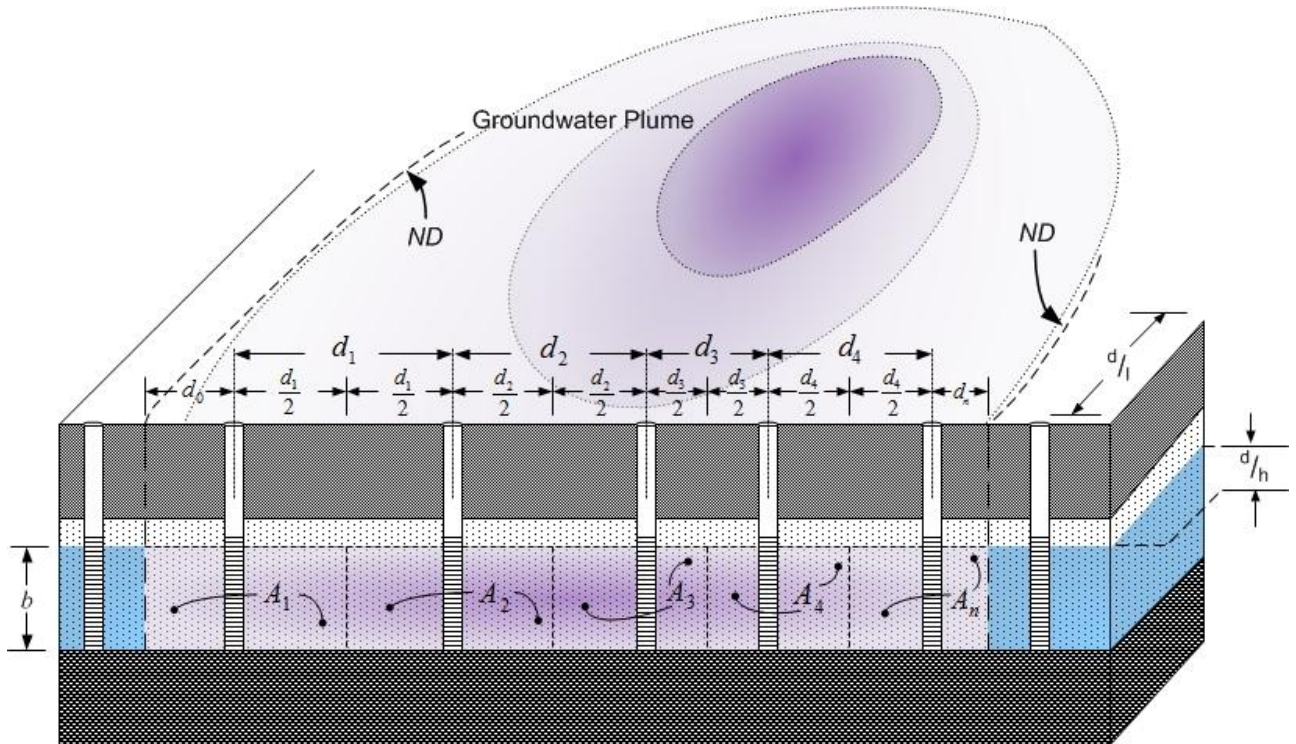


Figure D.1. Cross-section of the groundwater–surface water interface showing discretized discharge cells and distance variables used in equations.

or:

$$\bar{C}_{gw} = \frac{C_1 \left(d_0 + \frac{d_1}{2} \right) + \sum_{i=2}^{n-1} C_i \left(\frac{d_{i-1} + d_i}{2} \right) + C_n \left(\frac{d_{n-1} + d_n}{2} \right)}{\left(d_0 + \frac{d_1}{2} \right) + \sum_{i=2}^{n-1} \frac{d_{i-1} + d_i}{2} + \left(\frac{d_{n-1} + d_n}{2} \right)} \quad [\text{EQ 11}]$$

For the general solution with site-specific variation [EQ 8]:

$$\bar{C}_{gw} = \frac{C_{gw_1} K_1 \frac{dh}{dl_1} \eta_{e_1} \left(d_0 + \frac{d_1}{2} \right) b_1 + \sum_{i=2}^{n-1} \left(C_{gw_i} K_i \frac{dh}{dl_i} \eta_{e_i} \left(\frac{d_{i-1} + d_i}{2} \right) b_i \right) + C_{gw_n} K_n \frac{dh}{dl_n} \eta_{e_n} \left(\frac{d_{n-1} + d_n}{2} \right) b_n}{K_1 \frac{dh}{dl_1} \eta_{e_1} \left(d_0 + \frac{d_1}{2} \right) b_1 + \sum_{i=2}^{n-1} \left(K_i \frac{dh}{dl_i} \eta_{e_i} \left(\frac{d_{i-1} + d_i}{2} \right) b_i \right) + K_n \frac{dh}{dl_n} \eta_{e_n} \left(\frac{d_{n-1} + d_n}{2} \right) b_n} \quad [\text{EQ 12}]$$

When aquifer properties are observed to vary significantly along the groundwater–surface water interface, EQ 12 can be used to determine a representative groundwater concentration. A spreadsheet that facilitates the calculation of EQ 12 will be posted on the TCEQ website with this guidance.

Parameters

A (ft ²)	area of aquifer discharging to surface water
b (ft)	saturated thickness of aquifer discharging to surface water
C (mg/L)	COC concentration
C _{gw} (mg/L)	groundwater concentration
\bar{C}_{gw} (mg/L)	discharge-weighted representative groundwater concentration
d (ft)	distance between wells, or to edge of groundwater plume
dh/dl (ft/ft)	hydraulic gradient
dh (ft)	change in head between two points
dl (ft)	distance between two points where heads are measured
GW	groundwater
K (cm/s)	groundwater hydraulic conductivity
m (mg)	COC mass
ND	non-detect
q (cm/s)	groundwater specific discharge
Q _{gw} (ft ³ /s)	groundwater discharge from aquifer
t	unit time
V _{gw} (ft ³)	volume of groundwater

\dot{m} (mg/s) COC mass flux
 η_e (-) effective porosity

D.1 Example

A groundwater plume of dissolved trichloroethene (TCE) has been documented (in a hypothetical APAR) to be discharging into a stream channel that has incised the aquifer (see Figure D.1 for general schematic). The groundwater assessment (documented in APAR) defined the TCE concentration zonation of the TCE plume. An approximately linear groundwater monitoring network was installed along the groundwater–surface water interface that derives information about the range of TCE plume concentrations discharging into the stream at the monitoring points. The TCE concentrations are observed to vary along the interface between the two non-detect isoconcentration contours. Table D.1 summarizes the data determined during the site groundwater assessment in the APAR. See Figure D.2 for notation and locations.

Parameter	Symbol	Value
Number of wells inside groundwater plume	n	5
Distance: ND line to monitoring well 1 (MW-1)	d_0	20.0 ft
Distance between MW-1 and MW-2	d_1	70.0 ft
Distance between MW-2 and MW-3	d_2	55.0 ft
Distance between MW-3 and MW-4	d_3	40.0 ft
Distance between MW-4 and MW-5	d_4	50.0 ft
Distance from MW-5 to ND line	d_5	15.0 ft
Concentration in MW-1	C_{gw1}	10.0 mg/L
Concentration in MW-2	C_{gw2}	15.0 mg/L
Concentration in MW-3	C_{gw3}	20.0 mg/L
Concentration in MW-4	C_{gw4}	11.0 mg/L
Concentration in MW-5	C_{gw5}	8.0 mg/L
Hydraulic conductivity	K	2.5×10^{-4} cm/s
Hydraulic gradient	dh/dl	0.001 ft/ft
Effective porosity	η_e	0.33
Aquifer thickness (discharging)	b	5 ft

Table D.1. Site information for example \bar{C}_{gw} calculation.

D.1.1 First Tier (Conservative): Highest Concentration

Per 5.3, the highest concentration in the groundwater-surface water interface monitoring network should be used as the conservative groundwater concentration for use in calculations for the groundwater-to-surface water exposure pathway (in this example: $C_{gw3} = 20 \text{ mg/L}$).

D.1.2 Second Tier (Site-Specific): Discharge-Weighted Concentration

A site-specific discharge-weighted determination of the representative groundwater concentration discharging to the surface water body can be estimated using EQ 11:

$$\bar{C}_{gw} = \frac{C_1 \left(d_0 + \frac{d_1}{2} \right) + \sum_{i=2}^{n-1} C_i \left(\frac{d_{i-1} + d_i}{2} \right) + C_n \left(\frac{d_{n-1}}{2} + d_n \right)}{\left(d_0 + \frac{d_1}{2} \right) + \sum_{i=2}^{n-1} \frac{d_{i-1} + d_i}{2} + \left(\frac{d_{n-1}}{2} + d_n \right)}$$

In this example, aquifer parameters are constant along the interface and only concentrations and interwell distances vary.

D.1.2.1 Calculation of the Numerator (COC mass flux)

$$C_1 \left(d_0 + \frac{d_1}{2} \right) + \sum_{i=2}^{n-1} C_i \left(\frac{d_{i-1} + d_i}{2} \right) + C_n \left(\frac{d_{n-1}}{2} + d_n \right)$$

$$C_{gw1} \left(d_0 + \frac{d_1}{2} \right) = 10.0 \frac{\text{mg}}{\text{L}} \left(20.0 \text{ ft} + \frac{70.0 \text{ ft}}{2} \right) = 550.0 \frac{\text{mg}}{\text{L}} \times \text{ft}$$

$$\begin{aligned} \sum_{i=2}^{n-1} C_{gwi} \left(\frac{d_{i-1} + d_i}{2} \right) &= \sum_{i=2}^4 C_{gwi} \left(\frac{d_{i-1} + d_i}{2} \right) = C_{gw2} \left(\frac{d_1 + d_2}{2} \right) + C_{gw3} \left(\frac{d_2 + d_3}{2} \right) + C_{gw4} \left(\frac{d_3 + d_4}{2} \right) \\ &= 15.0 \frac{\text{mg}}{\text{L}} \times \left(\frac{70.0 \text{ ft} + 55.0 \text{ ft}}{2} \right) + 20.0 \frac{\text{mg}}{\text{L}} \times \left(\frac{55.0 \text{ ft} + 40.0 \text{ ft}}{2} \right) + 11.0 \frac{\text{mg}}{\text{L}} \times \left(\frac{40.0 \text{ ft} + 50.0 \text{ ft}}{2} \right) \\ &= 2,382.5 \frac{\text{mg}}{\text{L}} \times \text{ft} \end{aligned}$$

$$C_{gwn} \left(\frac{d_{n-1}}{2} + d_n \right) = C_{gw5} \left(\frac{d_4}{2} + d_5 \right) = 8.0 \frac{\text{mg}}{\text{L}} \left(\frac{50.0 \text{ ft}}{2} + 15.0 \text{ ft} \right) = 320.0 \frac{\text{mg}}{\text{L}} \times \text{ft}$$

Substituting in all values to determine the numerator:

$$C_1 \left(d_0 + \frac{d_1}{2} \right) + \sum_{i=2}^{n-1} C_i \left(\frac{d_{i-1} + d_i}{2} \right) + C_n \left(\frac{d_{n-1} + d_n}{2} \right) =$$

$$550.0 \frac{\text{mg}}{\text{L}} \times \text{ft} + 2,382.5 \frac{\text{mg}}{\text{L}} \times \text{ft} + 320 \frac{\text{mg}}{\text{L}} \times \text{ft} = 3,252.5 \frac{\text{mg}}{\text{L}} \times \text{ft}$$

D.1.2.2 Calculation of the Denominator

$$\left(d_0 + \frac{d_1}{2} \right) + \sum_{i=2}^{n-1} \frac{d_{i-1} + d_i}{2} + \left(\frac{d_{n-1} + d_n}{2} \right)$$

$$\left(d_0 + \frac{d_1}{2} \right) = \left(20.0 \text{ ft} + \frac{70.0 \text{ ft}}{2} \right) = 55.0 \text{ ft}$$

$$\sum_{i=2}^{n-1} \frac{d_{i-1} + d_i}{2} = \sum_{i=2}^4 \left(\frac{d_{i-1} + d_i}{2} \right) = \left(\frac{d_1 + d_2}{2} \right) + \left(\frac{d_2 + d_3}{2} \right) + \left(\frac{d_3 + d_4}{2} \right)$$

$$= \left(\frac{70.0 \text{ ft} + 55.0 \text{ ft}}{2} \right) + \left(\frac{55.0 \text{ ft} + 40.0 \text{ ft}}{2} \right) + \left(\frac{40.0 \text{ ft} + 50.0 \text{ ft}}{2} \right) = 155.0 \text{ ft}$$

$$\left(\frac{d_{n-1} + d_n}{2} \right) = \left(\frac{d_4 + d_5}{2} \right) = \frac{50.0 \text{ ft}}{2} + 15.0 \text{ ft} = 40.0 \text{ ft}$$

Substituting in all values to determine the denominator:

$$\left(d_0 + \frac{d_1}{2} \right) + \sum_{i=2}^{n-1} \frac{d_{i-1} + d_i}{2} + \left(\frac{d_{n-1} + d_n}{2} \right) = 55.0 \text{ ft} + 155.0 \text{ ft} + 40.0 \text{ ft} = 250 \text{ ft}$$

D.1.2.3 Final Calculation of Example \bar{C}_{gw} for the TCE Plume

$$\bar{C}_{gw} = \frac{C_1 \left(d_0 + \frac{d_1}{2} \right) + \sum_{i=2}^{n-1} C_i \left(\frac{d_{i-1} + d_i}{2} \right) + C_n \left(\frac{d_{n-1} + d_n}{2} \right)}{\left(d_0 + \frac{d_1}{2} \right) + \sum_{i=2}^{n-1} \frac{d_{i-1} + d_i}{2} + \left(\frac{d_{n-1} + d_n}{2} \right)} = \frac{3,252.5 \frac{\text{mg}}{\text{L}} \times \text{ft}}{250 \text{ ft}} = 13.0 \frac{\text{mg}}{\text{L}}$$

Appendix E

Assessing and Managing Impacts on Protected Species

E.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], 1918). The act 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)
- chapters 65 and 69 of the Texas Parks and Wildlife Code
- 31 TAC 65.171–76 and 69.1–9

In addition, sampling and remediation activities that have an adverse impact on the ecological habitat may increase natural-resource damages and associated liabilities.

E.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 www.tpwd.state.tx.us/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, persons should at minimum evaluate the two following TPWD sources for information pertaining to sensitive resources:

1. The list of Rare, Threatened, and Endangered Species of Texas by County www.tpwd.state.tx.us/landwater/land/maps/gis/ris/endangered_species/. This database provides a brief description of 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.

2. The *Texas Natural Diversity Database*. 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 <txndd@tpwd.state.tx.us>. 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 for each database and apply the information accordingly.

If federally listed species are potentially present, the U.S. FWS should also be contacted for additional site-specific data.

E.3 Sampling and 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 TCEQ recommends that if state or federally listed wildlife species are encountered at the affected property, they should be allowed to leave the area on their own and 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. Persons 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 the site activities.

E.3.1 Migratory Bird Treaty Act

The MBTA <www.fws.gov/migratorybirds/regulationspolicies/treatlaw.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 to avoid impacts to nesting birds. 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, the TCEQ recommends 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 vegetation removal, 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 MBTA permit office to ensure compliance.

E.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 <www.tpwd.state.tx.us/publications/pwdpubs/pwd_pl_w7000_1187a/>, a comprehensive wildlife-conservation strategy.

E.3.3 Injury of a Protected Species

If a protected species is injured during sampling or remediation, the TCEQ suggests contacting 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 <www.tpwd.state.tx.us/huntwild/wild/rehab/>.

E.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, persons 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 Natural Resource Trustees and resource agencies to ensure that wildlife are 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, a person may consider undertaking an ESA, as described in 30 TAC 350.33(a)(3)(B) of the TRRP rule. The ESA can be a useful approach to ecological risk management when working in close partnership with the TCEQ and the Natural Resource Trustees. Undertaking an ESA may not be appropriate in all situations and, therefore, discussions and consultation with the TCEQ and the Trustees may be helpful.

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