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Conducting Ecological Risk Assessments at Remediation Sites in Texas

DRAFT

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Remediation Division
TEXAS COMMISSION ON ENVIRONMENTAL QUALITY

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DRAFT

Prepared by
Remediation Division

RG-263
Revised January 2014

DRAFT: Conducting Ecological Risk Assessments at Remediation Sites in Texas

Overview

Objectives: Instruction in conducting ecological risk assessments at remediation sites in Texas.

Audience: Regulated community and environmental professionals

The regulatory citation for the Texas Risk Reduction Program (TRRP) rule is Title 30, Texas Administrative Code (30 TAC), Chapter 350.

> The TRRP rule, together with conforming changes to related rules, is contained in 30 TAC 350 and was initially published in the September 17, 1999 *Texas Register* (24 *Tex. Reg.* 7413–944). The rule was amended in 2007 (effective March 19, 2007; 32 *Tex. Reg.* 1526–79).

Find links for the TRRP rule and preamble, Tier 1 PCL tables, and other TRRP information at: <www.tceq.texas.gov/remediation/trrp/>.

TRRP guidance documents undergo periodic revision and are subject to change. Referenced TRRP guidance documents may be in development. Links to current versions appear at: <www.tceq.texas.gov/remediation/trrp/guidance.html>.

The TCEQ is committed to accessibility. If you are unable to access the information in any portion of this document, please contact the Technical Program Support Team at the phone number or e-mail address below.

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For mailing addresses, refer to: <www.tceq.texas.gov/about/directory/>.

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ABBREVIATIONS

Abbreviation	Description
AhR	aryl hydrocarbon receptor
APAR	affected property assessment report
API	American Petroleum Institute
ASTM	American Society for Testing and Materials
ATSDR	Agency for Toxic Substances and Disease Registry
AUF	area-use factor
AVS	acid volatile sulfide
BAF	bioaccumulation factor
BCF	bioconcentration factor
BLRA	baseline risk assessment
BSAF	biota-sediment accumulation factor
BTEX	benzene, toluene, ethylbenzene and xylene
BTF	biotransfer factor
CHRIS	Chemical Hazard Response Information System
COC	chemical of concern
CBR	critical body residue
DOI	U.S. Department of the Interior
dSAY	discounted service-acre-year
EF	exposure frequency
EMF	exposure-modifying factor
ERA	ecological risk assessment
ER-L	effects range—low
ESA	ecological services analysis
FCM	food chain multiplier
FR	*Federal Register*
GIS	geographic information system(s)

Abbreviation	Description
GW	groundwater
GWP	groundwater protection
HEA	habitat-equivalency analysis
HI	hazard index
HPAH	high-molecular-weight polycyclic aromatic hydrocarbon
HQ	hazard quotient
HSDB	Hazardous Substances Data Bank
K_{ow}	octanol-water partition coefficient
LC_{50}	lethal concentration (50 percent)
LD_{50}	lethal dose (50 percent)
LEL	lowest-effects level
LOAEL	lowest observed adverse-effect level
LPAH	low-molecular-weight polycyclic aromatic hydrocarbon
MOU	memorandum of understanding
MQL	method quantitation limit
MSCs	media-specific concentrations
MW	molecular weight
NAWQC	national ambient water quality criteria
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no observed adverse effect level
NRD	natural-resource damage
OECD	Organization for Economic Cooperation and Development
OHMTADS	Oil and Hazardous Materials Technical Assistance Data System
ORNL	Oak Ridge National Laboratory
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-*p*-dioxins

Abbreviation	Description
PCDF	polychlorinated dibenzofurans
PCL	protective concentration level
PCLE	protective concentration level exceedance
PHH	planar halogenated hydrocarbons
POE	point of exposure
PQL	practical quantitation limit
PRACR	post-response action care report
QSAR	quantitative structure-activity anaysis
RACR	response action completion report
RAER	response action effectiveness report
RAP	response action plan
RATL	reptile and amphibian toxicological literature
RCRA	Resource Conservation and Recovery Act
RRR	Risk Reduction Rule
SAI	Soil MSC based on inhalation, ingestion and dermal contact
SAR	structure-activity analysis
SCVs	secondary chronic values
SEM	simultaneously extracted metals
SETAC	Society of Environmental Toxicology and Chemistry
SIN	self-implementation notice
SLERA	screening-level ecological risk assessment
SSERA	site-specific ecological risk assessment
SQB	sediment quality benchmark
SW	surface water
T&E	threatened and endangered (species)
TAC	Texas Administrative Code
30 TAC 350	Title 30, Texas Administrative Code, chapter 350

Abbreviation	Description
TCDD	tetrachlorodibenzo-*p*-dioxin
TCEQ	Texas Commission on Environmental Quality
TEC	threshold effect concentration
TEF	toxicity equivalency factor
TEL	threshold effects level
TEQ	toxicity equivalent
TGLO	Texas General Land Office
TL	trophic level
TPH	total petroleum hydrocarbon
TPWD	Texas Parks and Wildlife Department
TRRP	Texas Risk Reduction Program
TRV	toxicity reference value
TSWQS	Texas surface water quality standards
UCL	upper confidence limit
UF	uncertainty factor
U.S. ACE	United States Army Corps of Engineers
U.S. EPA	United States Environmental Protection Agency

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The TCEQ gratefully acknowledges the Ecological Work Group whose members developed this guide. The Ecological Work Group is a multi-stakeholder committee of representatives from the regulated community, the Natural Resource Trustees, the EPA, environmental consultants, academia, legal, and agency staff.

1 INTRODUCTION

1.1 Purpose

This publication outlines the TCEQ's ecological risk assessment program and describes the interface between the ERA program with the Texas Risk Reduction Program (TRRP) Rule (30 TAC 350) and the Risk Reduction Rule (RRR) (30 TAC 335). This guide also discusses the interactions of the ERA process with ecological services analysis and the role that the Natural Resource Trustee agencies play in both under 30 TAC 7.124. This guide is not itself a rule or compilation; in the case of any apparent conflict, the rule itself governs. The purpose of this publication is to promote the development of consistent and technically defensible ERAs to be submitted under TCEQ remediation programs. It also includes technical advice and insight as to how the TCEQ may evaluate ERAs.

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

This document is intended to provide technical guidance for conducting ERAs at TRRP sites. Therefore, terminology specific to the TRRP rule and the science of ERAs is used throughout. Some terms are defined within the context of the relevant discussions and are denoted by italicized text. Others appear in the glossary near the end of this manual. Specifically note the use of "person," which has a special meaning under the TRRP rule. This document is divided into numbered chapters, sections, subsections, etc., referenced by figures in **bold** type. For example, "see **3.9.2**" means "see subsection 3.9.2."

1.2 Definition of an Ecological Risk Assessment

The U.S. Environmental Protection Agency's *Framework for Ecological Risk Assessment* (U.S. EPA, 1992a) sets forth a basic structure and a consistent approach for conducting ERAs, but is not intended for program-specific guidance. The *Framework*'s structure and approach have been expounded in this publication to aid in the development of consistent and technically defensible ERAs within the TCEQ's remediation programs. *Ecological risk assessment* is defined as a process that evaluates the likelihood that adverse ecological effects are occurring or may occur as

a result of exposure to one or more stressors (ibid.). The *Framework* further defines *stressor* as any physical, chemical, or biological entity that can induce an adverse ecological response; this publication only addresses chemical stressors—those subject to risk-management decisions at remediation sites. A risk cannot exist unless: the stressor has the ability to cause one or more adverse effects and it occurs with, or contacts, an ecological component (*receptor*) long enough and at a sufficient intensity to elicit the identified adverse effect. For the purposes of this guide, the primary functions of an ERA are to:

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

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

1.3 History of TCEQ ERA Guidance

In November 1996, the TCEQ (then the Texas Natural Resource Conservation Commission) published a draft ERA guide to foster consistency among ERAs within the agency's remediation programs, designed to interface with the TRRP rule which was under development at that time. It also facilitated consideration of ecological issues under then current agency rules, the Risk Reduction Rule (RRR). The final TRRP rule became effective on September 23, 1999 and had an implementation date of May 1, 2000. On November 8, 1999, the agency posted interim ERA guidance designed to replace the draft 1996 guidance. The interim guidance remained in effect until replaced with the draft guidance of August 28, 2000, followed by the December 2001 final version. The TCEQ released a January 2006 update revising the ecological screening benchmarks for surface water, soil, and sediment. This current guidance incorporates the update, reflects the agency's name change, updates the screening benchmarks, addresses the 2007 revisions to the TRRP rule, and reflects the TCEQ's commitment to accessibility.

[1] In this publication, *response* is equivalent to *response action* as defined in the TRRP rule (see glossary).

1.4 Applicability

This guidance applies to sites regulated by the TCEQ's Remediation Division that are being investigated under the TRRP or RRR. For the sake of convenience, the terminology herein conforms to that of the TRRP rule. Persons who elect to continue conducting response actions under the RRR may use this guide to evaluate risk to ecological receptors, where required. Such an action does not conflict with the applicability clause of the TRRP rule [30 TAC 350.2(m)], which mandates use of one chapter or the other (30 TAC 335 or 30 TAC 350) but not both.

1.4.1 ERA under the Risk Reduction Rule

The RRR specifies that environmental receptors should be evaluated and protected [30 TAC 335.556(b), 559(d)(4), 562(c)(3), 563(i)(2)(B), and 563(j)(3)]. The RRR applies to programs subject to 30 TAC 335 and gives three standards for closure:

- *Standard 1:* background or practical quantitation limits (PQLs)
- *Standard 2:* medium-specific concentrations (MSCs)
- *Standard 3:* baseline risk assessment

No ERA activities are required under RRR Standard 1. Under Standard 2 or 3, evaluating environmental receptors may be necessary. The Tier 1 Ecological Exclusion Criteria Checklist may be used to evaluate the need for, and scope of, further ERA activities under Standards 2 or 3. For sites requiring further ecological evaluation, a Tier 2 Screening-Level Ecological Risk Assessment (SLERA), a Tier 3 Site-Specific Ecological Risk Assessment (SSERA), or both may be performed.

If chemicals of concern (COCs) are present at concentrations that could pose a potentially unacceptable ecological risk, then media cleanup standards should be developed based on protection of ecological receptors, unless a remedy planned for protection of human health can be shown to eliminate the relevant ecological exposure pathways. Ecologically based MSCs should be compared to either Standard 2 MSCs or MSCs generated as part of a Standard 3 baseline risk assessment (BLRA) for protection of human health and groundwater. The lower of the two values should be used as the cleanup standard for the site. Ecological risk management under Standards 2 and 3 is discussed in **5.1**.

1.4.2 ERA under the Texas Risk Reduction Program Rule

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

An overall flowchart for the Texas Risk Reduction Program appears in Figure 1.1. Under this process, the person must implement a response action as needed to prevent human or ecological exposure to potentially harmful levels of COCs. Following discovery and notification of a COC release that is subject to these response-action requirements, the first step is an affected property assessment (30 TAC 350.51) to define the nature and extent of affected environmental media (i.e., soil, sediment, surface water, groundwater). Before, during, or after the affected property assessment, the person should complete the Tier 1 Exclusion Criteria Checklist to determine whether additional ecological evaluation is necessary. If so, the person may need to conduct a Tier 2 SLERA, a Tier 3 SSERA, or both to determine the applicable protective concentration levels (PCLs) for each affected medium (see Section 350.77). The development of human-health PCLs follows a fairly similar path (350.71).

To evaluate the need for a response, measured COC concentrations are compared to the lower of the human-health PCL or ecological PCL for each COC (the lower of the two is called the *critical PCL*). If measured COC concentrations exceed the critical PCL for any COC, the person may either refine the PCLs by going to the next tier in the risk analysis—assuming the current tier is 1 or 2 for human health or 2 for ecological—or implement a remedy pursuant to the TRRP requirements. However, if a planned response will eliminate the ecological exposure pathway or render it insignificant, or if human-health PCLs will be protective of ecological receptors, then no further ecological risk assessment is required, provided the person submits a reasoned justification [see Section 350.77(a) and **2.2**]. In addition, if the person can demonstrate that an expedited stream evaluation (see Section 350.77(a) and **2.6**) will determine that the surface water and sediment pathways are insignificant, then no further ecological risk assessment evaluation is required, provided there are no complete and significant soil exposure pathways.

Responses must conform to one of two options for performance standards, termed Remedy Standards A and B (30 TAC 350.31). Under Remedy Standard A, affected media must be removed or decontaminated to permanently reduce COC concentrations below critical PCLs (350.32). Under Remedy Standard B, removal, decontamination, or control measures may be applied to prevent exposure media exceeding critical PCLs (350.33). Under Remedy Standard B, use of such control measures may entail post-response care and associated financial assurance [350.33(g–n)]. With the exception of Class 1 groundwater resources (which require decontamination), the person may choose to implement either a Remedy Standard A or B. In a Remedy Standard B response targeted toward ecological concerns, the person may conduct an ecological services analysis to evaluate the net benefit of the response to ecological resources [350.77(f)(2)]. The action is complete once the applicable Remedy Standard A or B objectives have been satisfied (350.34). Ecological risk management under Remedy Standards A and B is discussed in **5.2**.

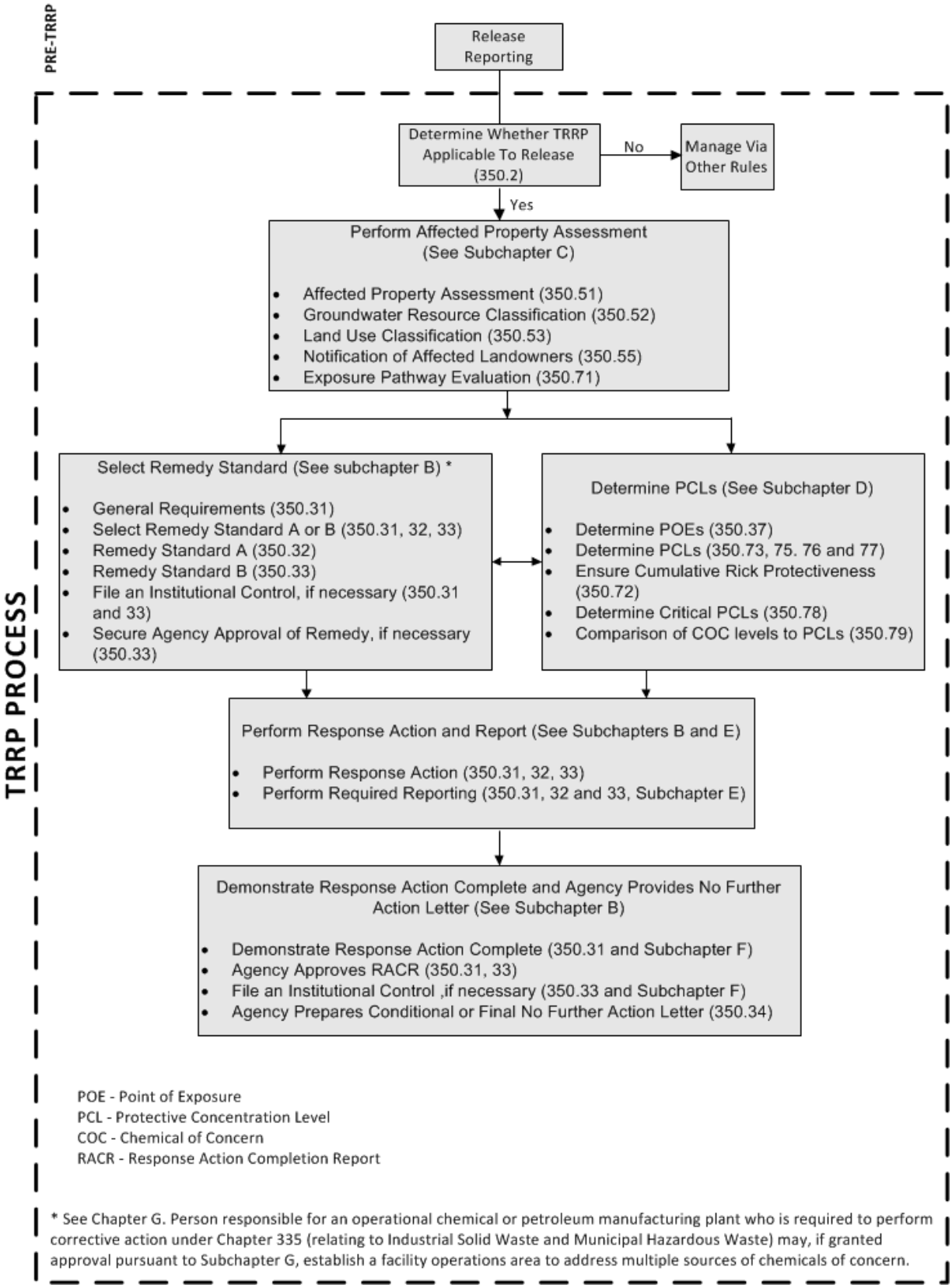


Figure 1.1. General overview of TRRP process.

As described above, within the overall TRRP process, the ERA is conducted to develop PCLs that protect against potential ecological exposures. As defined in the TRRP rule [350.4(a)(27)], the ecological PCL is a concentration of a COC within an exposure medium (e.g., soil, sediment, surface water, groundwater) which is protective of: (1) wider-ranging ecological receptors that may frequent the affected property and use less mobile receptors (e.g., plants, soil invertebrates, small rodents) as a food source, and (2) benthic invertebrates within waters in the state, where appropriate. The ERA should be conducted in a manner that results in the protection of ecological receptors subject to management by other state and federal agencies and consistent with these agencies' statutory authority. Under the TRRP ERA process, flexibility is provided for use of either screening-level or site-specific ecological PCLs, based on a tiered evaluation.

1.5 Affected Property Assessment

As detailed in Subchapter C of the TRRP rule [350.51(a)], the affected property assessment must:

- delineate the lateral and vertical extent of environmental media affected by the release,
- define the groundwater classification and current land use,
- characterize site geology and hydrogeology so as to predict COC fate and transport,
- identify potentially complete exposure pathways and the possible locations of relevant human or ecological receptors,
- evaluate the effectiveness of existing physical controls, and
- support remedy selection and notify affected landowners.

1.5.1 Assessment Level

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

For use in ERAs, the assessment level may be matched to PCL values developed under either Tier 2 or Tier 3 of the ERA process. However, in contrast to human health—which can derive conservative values for the assessment level from a table—ERAs require site-specific knowledge, even at the initial screening step in the SLERA. Because the ecological PCL is not known at the time of the initial assessment, there is a chance that an assessment level based only on human health could be higher than the

assessment level ultimately determined after ecological PCLs become available. The person may choose to perform the delineation to an assessment level based on human-health exposures (e.g., Tier 1 residential or commercial-industrial) or on ecological benchmark values and, after determining the ecological PCLs, collect additional samples if the ecological PCLs require a lowering of the assessment level. Alternatively, a person who wants to ensure that an assessment level will not require a downward adjustment could choose a level equal to the method quantitation limit of the COC.[2]

The required comparison of the COC concentrations against residential assessment levels entails the assumption of residential land use when calculating the human-health PCLs used in determining the assessment levels. The reference to residential assessment levels does not negate the requirement to include ecological PCLs in determining the assessment level, when the ERA process produces ecological PCLs. For each COC associated with the release, affected media must be delineated to an assessment level. The extent of the affected property is defined by the assessment level, and the assessment level incorporates human health and ecological PCLs, with the exception of the *de minimis* exclusion criterion in Tier 1 (see **2.3.3.4**), which defines affected property based on a comparison to human-health PCLs.

A related issue is the definition of *soil* for human-health and ecological exposures. “Surface soil” and “subsurface soil” are defined differently in the TRRP rule for ecological and for human-health risk assessments. Refer to Definitions (**8**) and the Tier 1 Exclusion Criteria Checklist (Attachment 1). Users of this guide should establish their sampling protocol in conformity with the definitions in the ecological checklist.

1.5.2 Affected Property: Representative Concentrations

A *COC* is any chemical with the potential to adversely affect ecological or human-health receptors due to its concentration, distribution, and mode of toxicity [350.4(a)(11)]. For any given release, the list of COCs is determined by the applicable remediation program; such determination should precede any data screening and should be the same for both human and ecological receptors. The identification of COCs should be consistent with *Target COC*s (TCEQ publication no. RG-366/TRRP-10).

COCs present at or below background levels in any media or meeting the criteria in 30 TAC 350.71(k) (see *COC Screening*, RG-366/TRRP-14) are eliminated from consideration during assessment and do not require further ERA analysis. Background concentrations may be based on

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

site-specific data or, for metals, on Texas median concentrations as discussed below. Where COCs are screened out, the person should make a reference to that effort in any subsequent ERA evaluation. However, as noted in **1.5.1**, 30 TAC 350.71(k)(1) does not allow elimination of COCs for the ERA by comparison to the residential assessment levels that do not consider ecological PCLs.

Texas median concentrations for metals in soils appear in Table 3.4. These concentrations are statewide median background values and should only be applied to the appropriate, corresponding affected property concentrations. For example, if the COC is hexavalent chromium, the total chromium background number cannot be used as a screening tool. Similarly, if the COC is methylmercury, it would be inappropriate to use the mercury number as a screen. Alternatively, site-specific background concentrations for COCs (e.g., metal species such as hexavalent chromium) in any medium may be used after being determined consistent with the TRRP rule and *Determining Representative Concentrations of Chemicals of Concern for Ecological Receptors* (RG-366/TRRP-15eco).

Either the highest COC concentration or statistical procedures can be used to characterize exposure concentrations in affected environmental media and compare representative exposure concentrations to applicable PCLs. Discussion of appropriate methods for characterization of exposure concentrations (including hot spots—see **3.9.2**) and comparison of these concentrations to applicable PCLs are available at 30 TAC 350.51 and 350.79, and in *Determining Representative Concentrations of Chemicals of Concern for Ecological Receptors*.

For the Tier 2 SLERA—for the initial screening—the maximum measured COC concentration in the exposure medium should be compared to medium-specific ecological benchmark values (see **3.5**). The maximum value can be evaluated an extreme outlier for the data set of a particular exposure medium. Extreme outliers are defined in *Determining Representative Concentrations of Chemicals of Concern for Ecological Receptors*. When the maximum value is an extreme outlier, then the next highest value (that is not an extreme outlier) is compared to the medium-specific benchmark value. However, for all subsequent iterations of the Tier 2 SLERA or Tier 3 SSERA, the person may apply statistical methods (e.g., 95 percent upper confidence limit of the arithmetic mean) consistent with *Determining Representative Concentrations of Chemcials of Concern for Ecological Receptors*.

1.5.3 Reporting

For sites under the RRR, the Tier 1 exclusion checklist may be submitted, when appropriate, with a demonstration of attainment of Risk-Reduction Standard 2 MSCs or with a human-health BLRA. If warranted, additional ERA activities—a SLERA, an SSERA, or both—may be incorporated into a BLRA document or presented as a stand-alone report. If a potentially unacceptable ecological risk is determined, then a cleanup standard

protective of ecological receptors for all COCs in the relevant media should be generated and compared with human-health standards.

According to 30 TAC 350.91, documentation of ERA activities within the TRRP, including exclusion criteria checklists, SLERAs, and SSERAs, should be included in the affected property assessment report (APAR). Supporting documentation, such as reasoned justifications (**2.2**) and additional ecological evaluations, such as expedited stream evaluations (**2.6**), should also be included. Under TRRP, Remedy Standard A may be self-implementing, and the APAR may be submitted with the response action effectiveness report or response action completion report (30 TAC 350.93 and 350.95, respectively). However, self-implementation is not recommended for affected properties failing the exclusion criteria checklist. Affected properties selecting Remedy Standard A based on ecological PCLs are encouraged to submit an APAR before implementing a remedial strategy. If Remedy Standard B is selected, the APAR is submitted with the response action plan. The RAP discusses the selected remedy and any necessary compliance sampling and monitoring (350.94). The person may also need to submit post–response action care reports. Each PRACR should include the results of any monitoring program, summarize the operation and maintenance of physical controls, and discuss any actions to correct a failure of controls (see 350.96).

To lessen the complexity and to give consistency to all of these TRRP reporting requirements, the TCEQ and other stakeholders have developed several standardized report forms: the affected property assessment report (TCEQ-10325/APAR); response action plan (10326/RAP); response action effectiveness report (10327/RAER); response action completion report (10328/RACR); and post response action care report (10329/PRACR). These forms are available through the TCEQ Web site at: <www.tceq.state.tx.us/goto/trrp-guidance>.

1.6 Overview of Tiered ERA Process

As described in the TRRP rule and illustrated in Figure 1.2, the TCEQ has developed a three-tiered approach to conducting ERAs. The person may elect to commence the ERA process at any of the following tiers:

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

Tier 2—Screening-Level Ecological Risk Assessment (SLERA): Under Tier 2, non-bioaccumulative COCs may be screened from further evaluation based on comparison to ecological benchmarks. If any COCs are not excluded on this basis, a conceptual exposure model is developed to characterize complete exposure pathways and representative receptors.

Exposures are compared to literature-based effects levels using conservative exposure assumptions that may be later refined with available site-specific information. Tier 2 ecological PCLs are derived for any retained COCs.

Tier 3—Site-Specific Ecological Risk Assessment (SSERA): Under optional Tier 3, ecological risk indicated from earlier tiers may be compared to site-specific "weight-of-evidence" information regarding the presence or absence of ecological effects. Such site-specific assessments may include analysis of tissue samples, toxicological testing of affected media, comparison of species diversity to reference areas, and other appropriate analyses. If effects are confirmed, these site-specific data may be employed to derive Tier 3 ecological PCLs for the relevant receptors and exposure media.

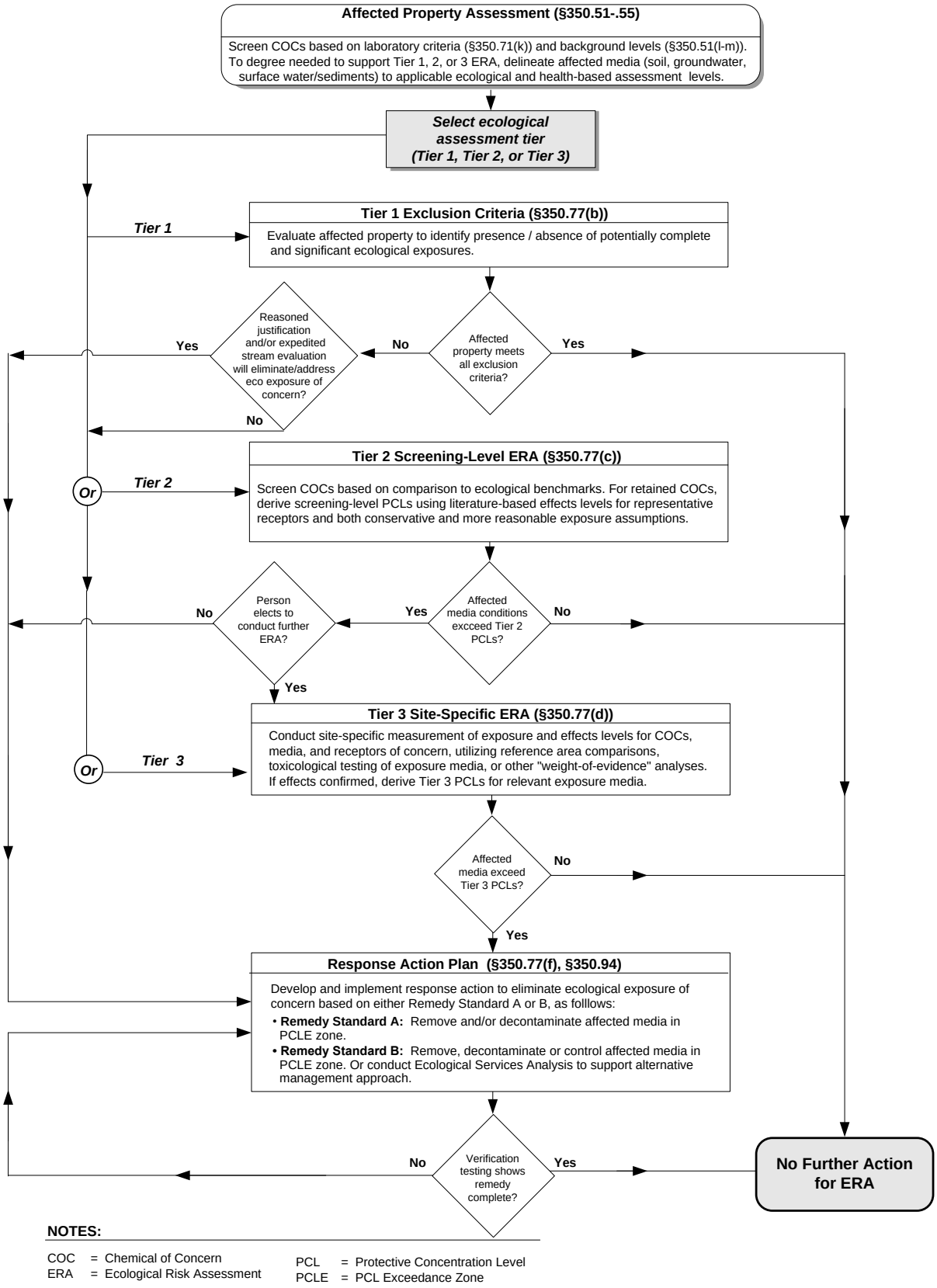


Figure 1.2. Overview of tiered ERA process.

2 TIER 1: EXCLUSION CRITERIA CHECKLIST

2.1 Purpose

The purposes of the Tier 1 checklist [(30 TAC 350.77(b)] are to characterize the ecological setting of the affected property and to determine the existence of complete and potentially significant ecological exposure pathways through the use of exclusion criteria. As outlined in Figure 2.1, *exclusion criteria* refers to those conditions at an affected property that preclude the need for a formal ERA because ecological exposure pathways are incomplete or insignificant due to the nature of the affected property setting or the condition of media at the affected property. The checklist attempts to make an early, non–resource-intensive determination of the presence of complete and significant ecological exposure pathways using these exclusion criteria.

The checklist is a standardized form consisting of mostly non-technical questions that could be completed by someone who is familiar with the affected property. It must be completed for all affected properties subject to TRRP [30 TAC 350.77(b)], unless a decision is made to begin the ecological evaluation at a higher Tier. Also, as stated earlier, sites subject to response actions under Risk Reduction Standards 2 or 3 (30 TAC 335) are required to evaluate and protect ecological receptors and are encouraged to begin their ecological evaluation by completing a Tier 1 checklist. The Tier 1 checklist is available at <info.sos.state.tx.us/fids/200900890-1.pdf>.

The completed Tier 1 checklist should identify any significant ecological exposure pathways that are complete or reasonably anticipated to be complete. If the affected property meets the exclusion criteria, then the person has fulfilled the ERA obligation and is not required to conduct a Tier 2 or Tier 3 ERA unless changing circumstances result in the affected property not meeting the exclusion criteria. If the exclusion criteria cannot be met, then the person may submit a reasoned justification for ending the ERA as discussed in **2.2**, conduct an expedited stream evaluation to also conclude the ERA as discussed in **2.6**, or perform a Tier 2 SLERA or a Tier 3 SSERA.

2.2 Use

Based on preliminary information regarding conditions of the affected property, the checklist may be applied to confirm the presence or absence of any potentially complete pathways for ecological exposure. Alternatively, if an ecological exposure is already known or suspected, the person may elect to proceed directly to Tier 2 or Tier 3. However, since the completion of the checklist may eliminate some ecological exposure pathways, it is advisable to begin all ecological evaluations at Tier 1 to better focus the assessment.

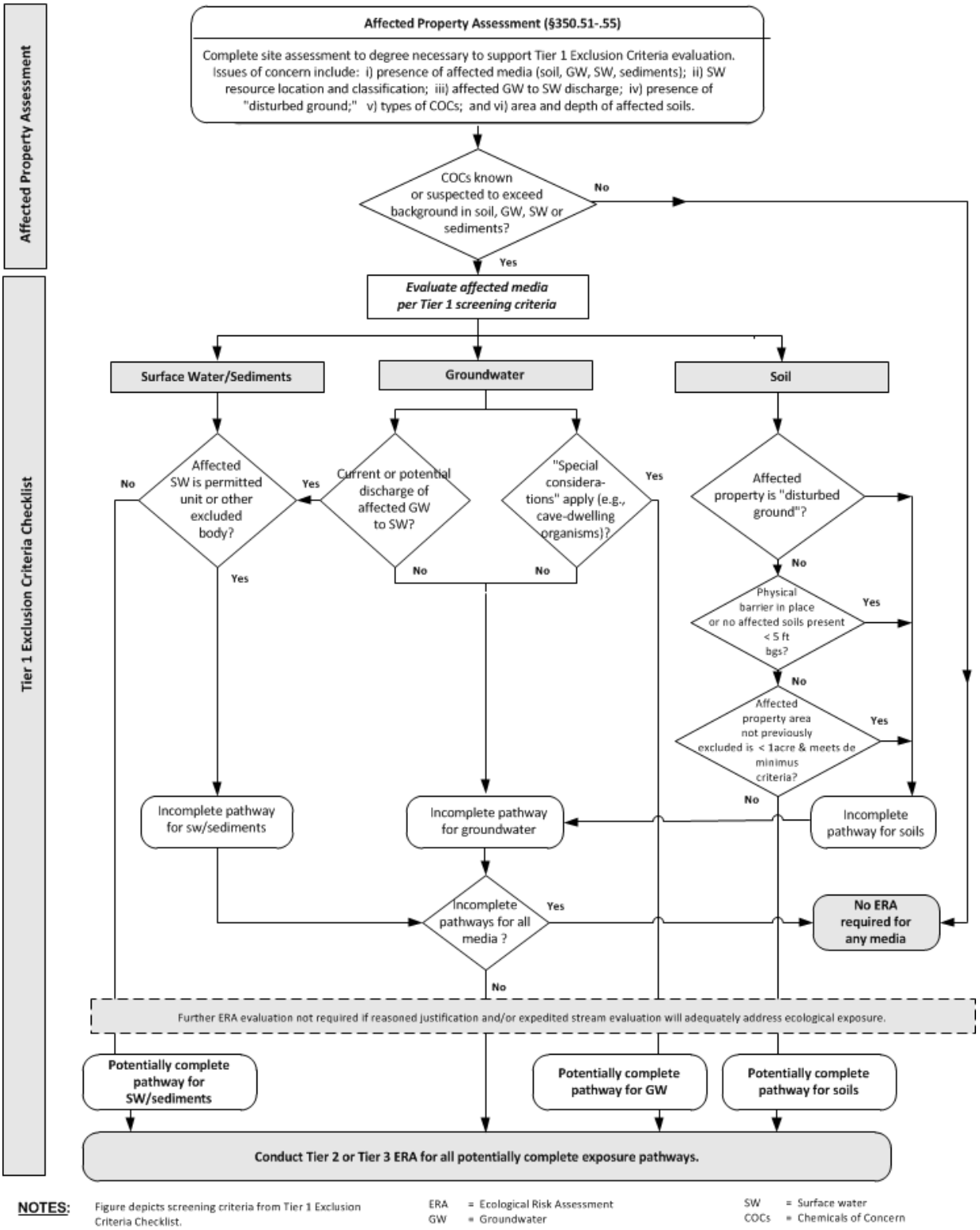


Figure 2.1. Tier 1 evaluation.

The Tier 1 checklist evaluates potential ecological exposure to affected soil, groundwater, or surface water and sediments. The checklist is designed for an early stage of the affected property assessment and, consequently, does not require detailed information on COC concentrations, the precise extent of affected media, or the specific ecological receptors (except for threatened or endangered species). Rather, as shown in Figure 2.1, general site conditions are evaluated to determine whether affected media are present at locations or, in the case of soils, over a sufficient area attractive to ecological receptors such that the receptors face significant exposure. With the exception of the *de minimis* exclusion discussed in **2.3.3.4**, the presence of COCs should not be based on a comparison to human-health PCLs. As described earlier (**1.5.1**), each COC in any affected media must be delineated to an assessment level. To avoid the having to collect additional samples, the person can choose an assessment level that is equal to the method quantitation limit of the COC, provided that the MQL is at or below the ecological screening benchmarks listed in Tables 3.2–3.4. Also (see **1.5.2**), COCs present below background levels in any media or meeting the criteria in 30 TAC 350.71(k) are eliminated from further consideration before comparison to ecological benchmarks and do not require further ERA analysis.

The checklist must be completed with existing information (i.e., any planned remediation may not be considered). However, prior to beginning a Tier 2 assessment, if it is believed that the implementation of response actions (e.g., covering the affected property with a cap) will effectively eliminate the ecological exposure pathways or render them insignificant, or if remediating to human-health PCLs will protect ecological receptors, then the person may develop a "reasoned justification" [see 30 TAC 350.77(a) and 350.91(b)(7)] for ending the ERA that explains how the planned remediation will eliminate or minimize the ecological exposure pathways. Accordingly, the person should submit the failed checklist and the reasoned justification in the APAR if the affected property is being addressed under TRRP, or as part of the specific remediation program requirements when the site is being addressed under the RRR. Furthermore, this response action[3] should be mostly limited to soil; however (ibid, **2.6**), an expedited stream evaluation may be used to address releases into certain types of water bodies in lieu of having to conduct a Tier 2 ERA.

2.3 Checklist Instructions and Tips

Although some instructions for completion of the Tier 1 checklist appear on the form itself, the information below may also be helpful. When including attachments that provide supporting information (e.g., correspondence with

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

wildlife-management agencies) or additional pages when more space is needed to answer a question, the person must ensure that these attachments are clearly identified. Also, supporting information may be referenced from other documents (e.g., an APAR) as long as the person clearly cites a reference (i.e., document name, date, section and page numbers). The following subsection headings correspond to particular headings within the checklist.

2.3.1 Preliminary Information

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

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

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

2.3.2 Affected Property Identification and Background

1. The information requested here may be repeated from other documents pertaining to the affected property or supplied as attached photocopies from those documents—or referenced with a clear citation. It is extremely important that the person supply or reference a map, aerial photo, or other photographs of the affected property, as these will greatly help the TCEQ evaluate the responses to the checklist questions.
2. After checking all relevant boxes under "Known or suspected COC location," indicate whether the COC location is *known* or *suspected* by circling the appropriate word or in the explanation section. Again, clearly cite any previously submitted information.
3. When completing this section, it is essential that the person have available the most current version of the Texas Surface Water Quality Standards (30 TAC 307), linked at <www.tceq.texas.gov/goto/tswqs>. The person must also realize that the answer to this question may not be the same answer for the question about the exclusion criterion in Subpart A. The purpose of this question is to ascertain the nearest surface water body (except for those excluded) that has received, or could receive, releases of COCs from the affected property. This includes waters that may be upgradient of the affected property or not even in the same watershed, but which could have been subject to airborne releases because they are located downwind of the affected property. The primary difference between this question and the criterion question in Subpart A is that this question serves to identify the nearest surface water body and does not focus on the likelihood of a complete exposure pathway to that water body, whereas the criterion question does focus on the completed pathway to any non-excluded surface water body (which may not be the nearest). In answering this question, the person need not be aware of all of the potential pathways of exposure from the affected property and

should simply identify the surface water body that is physically located nearest the affected property.

2.3.3 Part II. Exclusion Criteria and Supportive Information

2.3.3.1 Subpart A. Surface Water and Sediment Exposure

The purpose of this initial question is to identify those surface waters (and underlying sediments) that may be subject to further ecological evaluation because the pathways between them and the affected property are complete or are likely to become complete. This question also serves to identify waters and sediments that should be excluded from further evaluation.

According to the Texas Surface Water Quality Standards (30 TAC 307.3), surface water in the state is essentially all waters except those in authorized systems for treating waste. These "treatment system waters" are excluded from ERA consideration, unless the treatment system is no longer authorized. However, waste-treatment waters may be considered as sources of COCs if there is an "unpermitted" release from these waters into other Texas waters. Also, even permitted waste-treatment waters may be subject to other rules and regulations designed to protect ecological receptors—such as the Endangered Species Act or Migratory Bird Treaty Act. Furthermore, when a permitted outfall discharges into a water body, it is the treatment process waters preceding the outfall that are excluded from consideration, not the receiving water itself. For those rare occasions where the receiving water itself is designated as the permitted outfall, COCs regulated by the permit are excluded from evaluation. However, impacts on the water body associated with a release of unpermitted COCs from the affected property should be evaluated in the ERA.

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

If the affected property has had a release to surface water or sediment, it fails the checklist and will have to undergo additional ecological evaluation. However, that does not necessarily entail a Tier 2 assessment. As discussed in **2.6**, an expedited stream evaluation may be appropriate for the release of surface water or sediment, depending on the type of water body. In any case, the person should complete the remainder of the checklist to determine if there is a complete and significant soil exposure pathway.

If the soil pathway is incomplete or insignificant, further evaluations need only focus on the surface water or sediment exposure pathway.

2.3.3.2 Subpart B. Affected Property Setting

The affected property must qualify as attractive to ecological receptors before the person can consider answering the question about the setting of the affected property. Field observations and discussions with others who are also familiar with the affected property should be used to help determine attractiveness. If the affected property is attractive to ecological receptors (including threatened and endangered species), the person should bypass the question about the setting of the affected property and proceed to Subpart C.

If needed, information on threatened or endangered or otherwise protected species may be obtained from the following wildlife management agencies and, to a lesser extent, their websites:

Texas Parks and Wildlife Department (Austin)

Endangered Species (512) 389-8070

<www.tpwd.state.tx.us/huntwild/wild/wildlife_diversity/texas_rare_species/listed_species/>

<www.tpwd.state.tx.us/landwater/land/maps/gis/ris/endangered_species/>

U.S. Fish and Wildlife Service (Austin) 512-490-0057

<www.fws.gov/endangered/index.html>

<www.fws.gov/southwest/es/AustinTexas/>

Campbell (2003) is also a good source for information on life history and management of threatened and endangered species in Texas, though other, more recent, sources may be needed. Additional information on protected species may be found at local libraries or at city or county natural-resource programs—often essential, as state or federal lists may not reflect locally protected species. The person will need to document efforts to ascertain the presence of protected species potentially harmed by the affected property (e.g., by attaching copies of any written correspondence or records of oral communication with the wildlife-management and natural-resource agencies). These efforts are to be identified in the “explanation” space. The presence of protected species should not be determined solely from observation, but also upon their range and the availability of suitable habitat at the affected property. Obviously, if the affected property is not attractive to any wildlife due to a lack of habitat, there is no reason to ascertain the presence of protected species.

“Disturbed ground” primarily refers to a location that is predominantly urban or commercial-industrial in nature (and thus characterized by human presence and activities) where any habitat that may have once existed has

been altered, impacted, or reduced to a degree such that it is no longer conducive to use by ecological receptors. Regarding what constitutes "disturbed ground," closed "waste control units" [defined in the TRRP rule at 30 TAC 350.4(a)(91)] with engineered covers are considered disturbed ground, provided they are meeting their design specifications. On the other hand, agricultural crops and, more often, pastureland are not usually considered disturbed ground because they are both ecological receptors and potential habitat for other ecological receptors. Golf courses are usually considered disturbed ground and are mostly not attractive to ecological receptors. However, if the release or potential release is to a wooded, isolated portion of the golf course, ecological receptors may be present and thus warrant further investigation. Also, if the release or potential release is to waters or sediments within the golf course, the person will be required to conduct further evaluation of those media unless the waters (including water hazards) meet the Subpart A criteria (i.e., the waters qualify as a decorative pond that is not in contact with other surface water in the state and is not valuable habitat). Circumstances surrounding the presence of crops, pastureland, or golf courses associated with the affected property require case-by-case evaluation.

2.3.3.3 Subpart C. Soil Exposure

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

2.3.3.4 Subpart D. *De Minimis* Land Area

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

A sensitive environmental area is habitat that may require protection or special consideration because of the presence of particular ecological receptors and natural resources, or because legislatively conferred protection (national-monument status) has been conferred (U.S. EPA, 1999a). Examples of sensitive environmental areas are listed below; however, as this list is generic, the person should evaluate the presence of these areas at his or her own site. Some local areas not listed below could serve important habitat functions that may require consideration when completing the checklist, reflecting some knowledge of local wildlife-management priorities. The person should identify all sensitive environmental areas within one-quarter mile of the affected property.

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

- critical habitat for species designated as endangered or threatened by state or federal government—or any habitat known to be used by such species, or species proposed for such designation, or for which such designation is under review
- marine sanctuary
- state or national park
- designated state or federal wilderness area
- areas identified under the Coastal Zone Management Act
- sensitive areas identified under the National Estuary Program or Near Coastal Water Program
- critical areas identified under the Clean Lakes Program
- state or national monument
- national seashore recreational area
- national lakeshore recreational area
- state or national preserve
- state or national wildlife refuge
- unit of Coastal Barrier Resources System
- coastal barrier (undeveloped or partially undeveloped)
- state or federal land designated for protection of natural ecosystems
- administratively proposed state or federal wilderness area
- spawning areas critical for the maintenance of fish or shellfish species within river, lake, or coastal tidal waters
- migratory pathways and feeding areas critical for maintenance of anadromous fish species within river or reaches or areas in lakes or coastal tidal waters in which the fish spend extended period of time
- terrestrial areas used for breeding by large or dense aggregations of animals
- state or national river reach designated as recreational
- scenic or wild river so designated by state or federal government
- other state land designated for wildlife or game management
- other state-designated natural areas
- particular areas, relatively small in size, important to maintenance of aquatic life
- wetlands

Whether COCs in soil migrate primarily depends upon its fate and transport (see **3.8**). For example, a significant portion of pesticides remains in the soil

as a persistent residue bound to soil particles (Donker et al., 1994). Also, if the topography of the affected property is such that there is no surface water runoff (due to percolation or evaporation) or if the runoff is captured and treated and the area of the affected property up to the point of capture does not exceed 1 acre, it is likely that the area of the affected property will not increase.

If the affected property meets all four of the qualifying conditions, the person using the TRRP rule should apply human-health PCLs for the applicable land use to determine if the extent of the affected property for each COC is one acre or less. For application of the *de minimis* concept, the assessment level is the lower of the Tier 1 combined soil PCL ($^{Tot}Soil_{Comb}$) and the soil-to-groundwater PCL appropriate for the groundwater classification (GW_{Soil}). The soil-to-groundwater PCL may be established under Tier 1, 2, or 3. The Tier 1 human-health PCLs are available as PDF files or as Excel tables online at <www.tceq.state.tx.us/goto/pcl>.

The person using the RRR, again provided the site meets all four of the qualifying conditions, should use the extent of contamination in excess of Risk Reduction Standard 2 human-health MSCs for the applicable land use to determine if the extent of the affected property is one acre or less. For application of the *de minimis* concept under the RRR, the "assessment level" is the lower of the soil MSC based on inhalation, ingestion, and dermal contact (referred to as "SAI") and the soil MSC based on groundwater protection (GWP). The area exceeding the lower of the SAI and GWP must be 1 acre or less for each COC to qualify for the *de minimis* exclusion under the RRR. Updated RRR Standard 2 MSCs are available at <www.tceq.texas.gov/remediation/rrr.html>. A large affected property may not be arbitrarily divided up into 1 acre units for this exclusion criterion. Divisions of the affected property should be based on types of process areas, geography, or other appropriate delineations.

2.3.4 Part III. Qualitative Summary and Certification

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

2.4 Checklist Review and Response

The completed checklist should be submitted to the appropriate TCEQ project manager as part of the APAR if the affected property is being addressed under the TRRP or as part of the specific remediation-program requirements when the site is being addressed under the Risk Reduction Rule. The person will make the initial decision regarding the need for

further ecological evaluation. The project manager will review the checklist and may or may not concur with the person's decision. The project manager may consult with the agency's ERA staff. Completed checklists that indicate the presence of possible ecological exposures accompanied by a separate reasoned justification will be reviewed by TCEQ project managers and ecological risk assessors. The project manager will notify the person in writing regarding the approval or disapproval of the checklist and any reasoned justification.

In summary, there are a few things to remember about the Tier 1 checklist:

- The checklist can be used under both the RRR and TRRP.
- TCEQ project managers are responsible for review and approval or disapproval.
- Only existing information may be used—planned remediation cannot be considered.
- Knowledge of the particular ecological receptors present (with the exception of threatened and endangered species) and the concentrations of COCs that may affect them is not required.
- Cropland and pastureland are not usually considered "disturbed ground."
- Regarding affected property under the *de minimis* criterion for exclusion, the extent is based on human-health PCLs and it cannot be arbitrarily divided up into 1-acre units.
- Completed checklists that indicate the presence of possible ecological exposures, along with a separate reasoned justification for ending the ERA based on planned remediation, can be submitted in lieu of proceeding to Tier 2.
- Completed checklists that indicate a complete surface water–sediment exposure pathway, along with a separate expedited stream evaluation (**2.6**) for qualifying water bodies, can be submitted in lieu of proceeding to Tier 2.

2.5 Example

The information presented may not represent all the facts about the affected property needed to properly complete the checklist, but serves to illustrate particular points.

2.5.1 Background

The facility is a former chemical plant located in west Texas, occupies 10 acres, and is fenced. The affected property consists of 0.8 acres and is adjacent to similar, but unimpacted, open areas. The surface and subsurface soils contain organophosphate pesticides and arsenic, and COCs have been detected in the shallow groundwater. The groundwater seeps to the nearest surface water, which is an unnamed, artificial, intermittent drainage ditch located 500 ft to the south that runs for a half mile before entering into

Little Silver Creek, a tributary of the Colorado River. Acute water quality criteria are met in the ditch. The area topography is mostly flat with a slight slope to the north. Indications are that the affected property does not drain to the ditch and that runoff mostly evaporates or infiltrates into the site soil. The vegetation around the ditch has been mostly cleared. The area surrounding the affected property is commercial-industrial, but a Texas horned lizard was observed on suitable habitat within the facility property.

Subpart A. Surface Water–Sediment Exposure

The unnamed, artificial, intermittent drainage ditch is the nearest surface water in the state, but the affected property does not drain to it because of the topography. With the exception of flow during heavy rainfall and the seepage from the affected property, the drainage ditch is dry most of the time and therefore is not used consistently or routinely as valuable habitat. However, because of the seepage, the surface water–sediment pathway is complete and the affected property does not meet the exclusion criterion. As indicated in **2.3.3.1** and discussed in detail in **2.6**, that does not necessarily mean that a Tier 2 assessment is needed. The person should complete the remainder of the checklist to see if there is a complete or significant soil-exposure pathway. Proceed to Subpart B.

Subpart B. Affected Property Setting

The affected property does not meet the "not attractive to wildlife or livestock, including threatened or endangered species" condition because the horned lizard is a threatened species. Proceed to Subpart C.

Subpart C. Soil Exposure

COCs were found in both surface and subsurface soils and therefore are not "solely below the first 5 ft beneath ground surface." Proceed to Subpart D.

Subpart D. *De Minimis* Land Area

The affected property does not meet the initial condition because of the presence of a threatened species. As this is the last soil exclusion criterion, the affected property is not excluded from the soil exposure pathway. For informational purposes, evaluate the remaining conditions. Similar but unimpacted habitat is adjacent to the affected property. The affected property is surrounded by land that is designated commercial-industrial and therefore is not likely to be located within one-quarter mile of sensitive environmental areas. Given the low mobility of the COCs and the lack of overland flow, there is no reason to suspect that the affected property will increase in acreage. The affected property is less than 1 acre.

Conclusions

This affected property did not meet the exclusion criteria because of a complete surface water–sediment exposure pathway and the presence of a threatened species. The standard recommendation would be further

ecological evaluation. However, it is likely that the horned lizard was observed on the open, unimpacted portions of the facility where it was foraging for ants. Based on information obtained from the TPWD and research concerning the life history (including feeding and habitat preferences) of the horned lizard, the person could propose remediation at the affected property that would minimize impacts while reducing risk to the horned lizard. A reasoned justification explaining the proposed remediation along with the documentation of communication with the TPWD and the horned-lizard research could be submitted along with the completed checklist in lieu of proceeding to a higher tier of ecological evaluation for the soil exposure pathway. Application of the expedited stream evaluation to address the surface water–sediment exposure pathway is discussed in **2.6.2**.

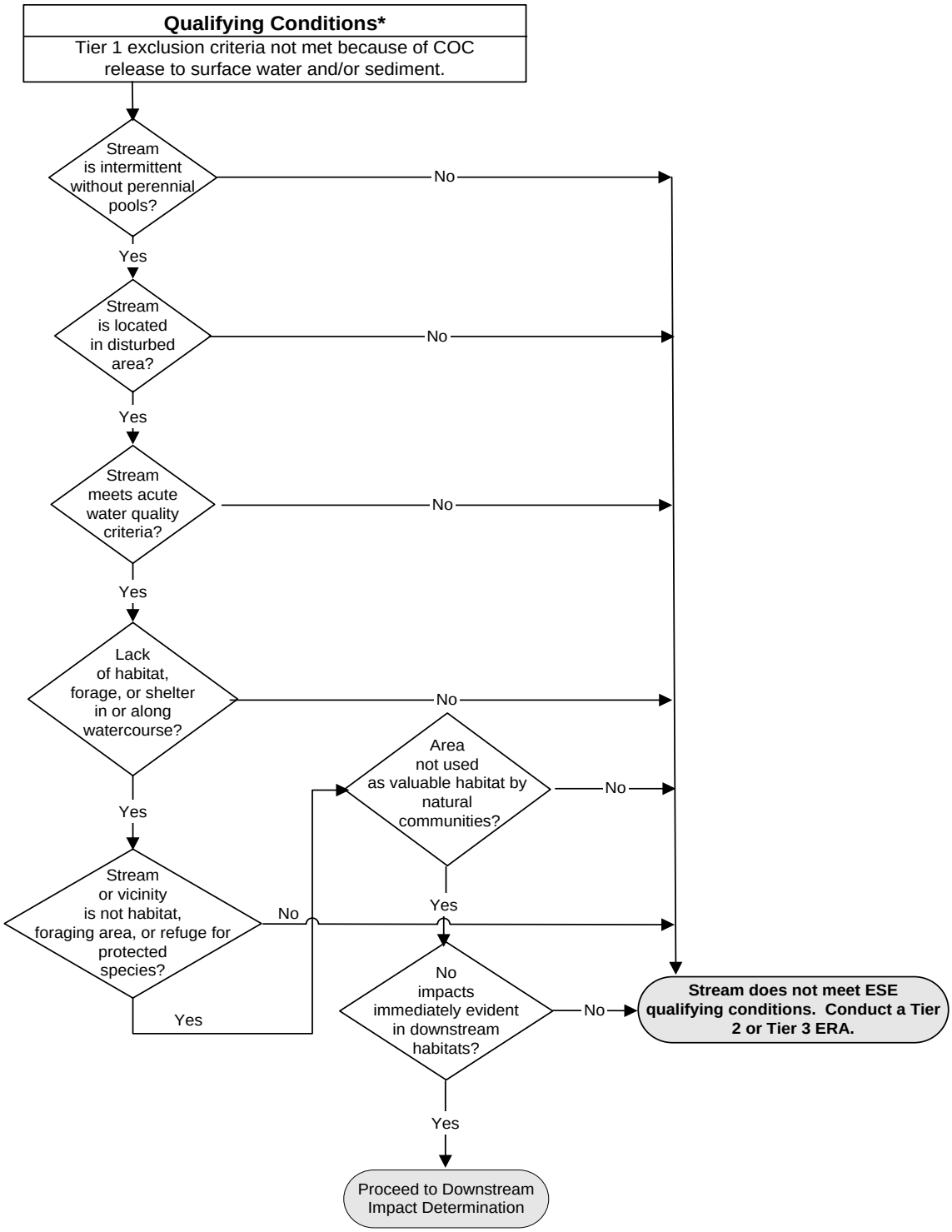
2.6 Expedited Stream Evaluation for Surface Water and Sediment Releases to Intermittent Streams without Perennial Pools

This section describes a process whereby an "expedited stream evaluation" [see 30 TAC 350.77(a) and 350.91(b)(7)] is used to determine that, although a COC release to surface water or associated sediment is a completed pathway, it may not be a significant pathway when the water body and its surroundings meet certain conditions. More specifically, the release must be into an intermittent stream without perennial pools that does not support a benthic community needing to be protected (see the conditions discussed in **3.6.1**). In addition, there should be no immediately apparent downstream impacts. If not meeting the surface water–sediment pathway criterion is the only reason the checklist was failed (i.e., the soil exposure pathway proved to be incomplete or insignificant), this expedited stream evaluation may be used to conclude the ERA. Although somewhat similar in rationale (see **2.6.1**), this evaluation is not part of the Tier 1 checklist contained in the TRRP rule. It is also not eligible under the reasoned justification clause discussed in **2.2** because of the potential for continuing ecological exposure downstream due to transport of COCs. However, it serves the same purpose as both of these other processes in that if the relevant criteria are met, a Tier 2 assessment is not needed. Expedited stream evaluation is depicted in Figures 2.2 and 2.3.

2.6.1 Comparison to the Tier 1 Checklist

The initial exclusion criterion described in **2.3.3.1** is the only one concerned with the surface water–sediment exposure pathway. It mandates additional ecological evaluation if there is a release to a qualified water body or the underlying sediment. There is no consideration of the environmental setting, habitat, or presence or absence of receptors associated with the water body. On the other hand, the three soil-exposure-pathway exclusion criteria do consider the setting, habitat (or lack thereof), and the likelihood of ecological receptors being present.

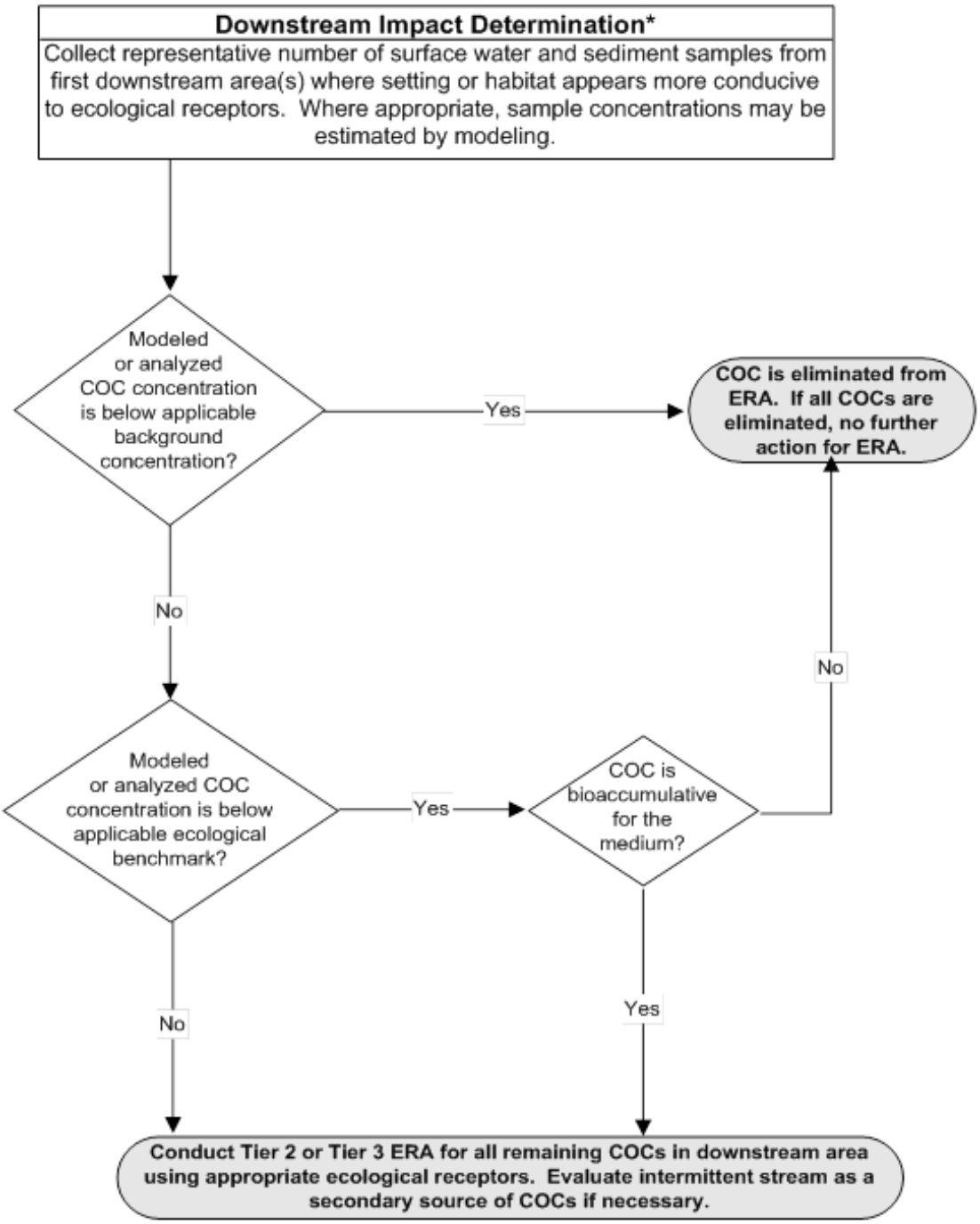
In order to more consistently apply these considerations across media, for some releases into particular types of water bodies (or associated sediments), a Tier 2 SLERA may not be necessary for the potentially impacted intermittent stream.



Notes:

* = Requires Supporting Photo Documentation
COC = Chemical of Concern
ERA = Ecological Risk Assessment
ESE = Expedited Stream Evaluation

Figure 2.2. Expedited stream evaluation: qualifying conditions.



Notes:

* = Requires Supporting Photo Documentation
COC = Chemical of Concern
ERA = Ecological Risk Assessment

Figure 2.3. Expedited stream evaluation: determining downstream impact.

However, these considerations alone are not enough to preclude an evaluation of risks to higher-trophic-level receptors that may become exposed to sediment and surface water COCs from transport to downstream water bodies where more suitable habitat may be available. As described below, after a particular water body has met the expedited stream-evaluation qualifications, a determination of potential downstream impacts from that body is required.

As stated in **2.1**, the Tier 1 checklist can be completed by nearly anyone who is familiar with the affected property. However, the expedited stream evaluation should be conducted by a qualified environmental professional, as a determination of instream (and likely off-property) impacts is needed.

2.6.2 Expedited Stream Evaluation Qualifications

According to the implementation procedures (TCEQ, 2011) as amended, an "intermittent" stream is one which has a period of zero flow for at least one week during most years and is considered "intermittent with perennial pools" when adequate pools persist that would be expected to provide habitat for significant aquatic-life use. Perennial-pool status is determined case by case using available data and best professional judgment. As discussed in **3.6.1**, an intermittent stream without perennial pools associated with the affected property may already qualify for not having the person develop a sediment PCL for protection of the benthic community. If the intermittent stream in question does not support a protectable benthic community, then this stream may also qualify for not having the person develop an ecological PCL for higher-trophic-level receptors because of a lack of associated habitat and receptors. As illustrated in Figure 2.2, to ascertain whether or not an expedited stream evaluation is appropriate, the following conditions must all be met:

- The stream is intermittent (dries up completely at least one week a year) without perennial pools. [Intermittent streams with perennial pools are defined at 30 TAC 307.3(a) and discussed in the implementation procedures (TCEQ, 2011).].
- The stream is located in a disturbed area (generally, a predominantly urban or commercial-industrial setting).
- The stream meets the acute water quality criteria specified in Table 1 of 30 TAC 307.6 or appropriate surrogate values if no criterion is specified.
- There is a lack of appreciable instream, edge, or riparian habitat, forage, or shelter in or along the watercourse.
- The watercourse or surrounding vicinity is not known to serve as habitat, foraging area, or refuge to threatened, endangered or otherwise protected species.
- The area is not consistently or routinely used as valuable habitat for natural communities including birds, mammals, reptiles, etc.
- No impacts are immediately evident in downstream areas where habitat is more likely to support wildlife.

As discussed in **2.6.4**, compliance with these conditions should be supported by photographic evidence. However, if all of these conditions are met, the stream needs no further evaluation unless, as discussed in the following section, more thorough downstream analyses reveal impacts. In that case, the stream may need to be evaluated as a potential secondary source of COCs. If any one of these conditions is not met, then the person will need to conduct a Tier 2 SLERA that includes the water body in question, as well as any downstream resources that may be impacted.

The example case in **2.5** is illustrative. The ditch is definitely intermittent and, because it is artificial, does not likely have perennial pools. The surrounding area is commercial-industrial and therefore disturbed. Water samples collected from the ditch indicate that acute criteria are being met. Because of the nature of the ditch and because the bank vegetation has been mostly cleared, there is no habitat to be used by potential receptors. The horned lizard observed at the facility of the affected property is being addressed via the reasoned justification. Assuming that there is no evidence of downstream impacts, this ditch qualifies for an expedited stream evaluation.

2.6.3 Determining Downstream Impacts

As presented in Figure 2.3, the expedited stream evaluation must include, as appropriate, the collection or estimation (via modeling) of downstream surface water and sediment samples, COC and conventional analyses of collected samples, and comparison of the reported or estimated concentrations to background concentrations or the ecological benchmarks listed in **3.5** (e.g., chronic surface water criteria), or both. Sampling is needed in the first downstream area or areas where setting or habitat appears more conducive to aquatic life (e.g., large pool) or wildlife (e.g., riparian vegetation). In the example case in **2.5**, appropriate locations in Little Silver Creek would need to be sampled. As discussed below, photo documentation of the sampling locations is recommended. Surface water and sediment samples should be colocated and collected from depositional areas (e.g., pools, point bars on the inside banks of streams) where possible and appropriate. The number of samples will depend on site-specific circumstances, considering spatial scale of the potential depositional area. However, when feasible, collecting at least five pairs of samples is recommended. When the COC release to the surface water is through a groundwater discharge, surface water concentrations should be derived according to the TRRP rule at 350.37(i) and 350.75(i)(4) and as discussed in *Determining PCLs for Surface Water and Sediment* (TCEQ publication RG-366/TRRP-24).

If the maximum affected property–related COC concentrations are below the greater of background or benchmarks, then the evaluation may be concluded. However, if concentrations are greater than both background and benchmarks, or if COC concentrations exceed background and are considered bioaccumulative (see Table 3.1), then the person will need to conduct a Tier 2 ERA. Although this approach appears similar to the first

required element of a traditional Tier 2 assessment (see **3.5**), it differs in that the concentrations compared to background or benchmark values will not be from samples collected from the intermittent stream, but rather from an appropriate downstream location where the likelihood that the concentrations would be below these values is greatly increased. If a downstream concentration does exceed the appropriate background or benchmark, then the intermittent stream may need to be evaluated as a secondary source of COCs.

2.6.4 Reporting and Review

Any decisions regarding the qualification and appropriate application of the expedited stream evaluation to a water body should be supported by photographic evidence. Therefore, in order to assist the TCEQ in its review, it is strongly recommended that the person supply photo documentation of the intermittent stream, the surrounding area, the downstream area, and the sampling locations.

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

3 TIER 2: SCREENING-LEVEL ECOLOGICAL RISK ASSESSMENT

3.1 Purpose and Use

The purposes of the Tier 2 SLERA are to scientifically eliminate COCs that do not pose an ecological risk and to develop PCLs for those COCs that do pose an unacceptable risk to selected ecological receptors. The Tier 2 SLERA serves to identify COCs, exposure pathways, and ecological receptors of concern based on application of default exposure assumptions and literature-based effects levels. Although the Tier 2 SLERA (Figure 3.1) has been designed to minimize effort, it will probably need to be conducted by an environmental professional. Those persons who undertake a Tier 2 evaluation will need to meet a number of required elements. However, within the required elements, there are four potential exit points from the ERA process before having to develop ecological PCLs. The list of required elements for a Tier 2 SLERA appears in **3.3**; a discussion of each element follows in **3.5–14**. Persons conducting a SLERA may exit the process, or at least eliminate particular COCs or media, if they meet the conditions of required elements (1), (6), (7), or (8).

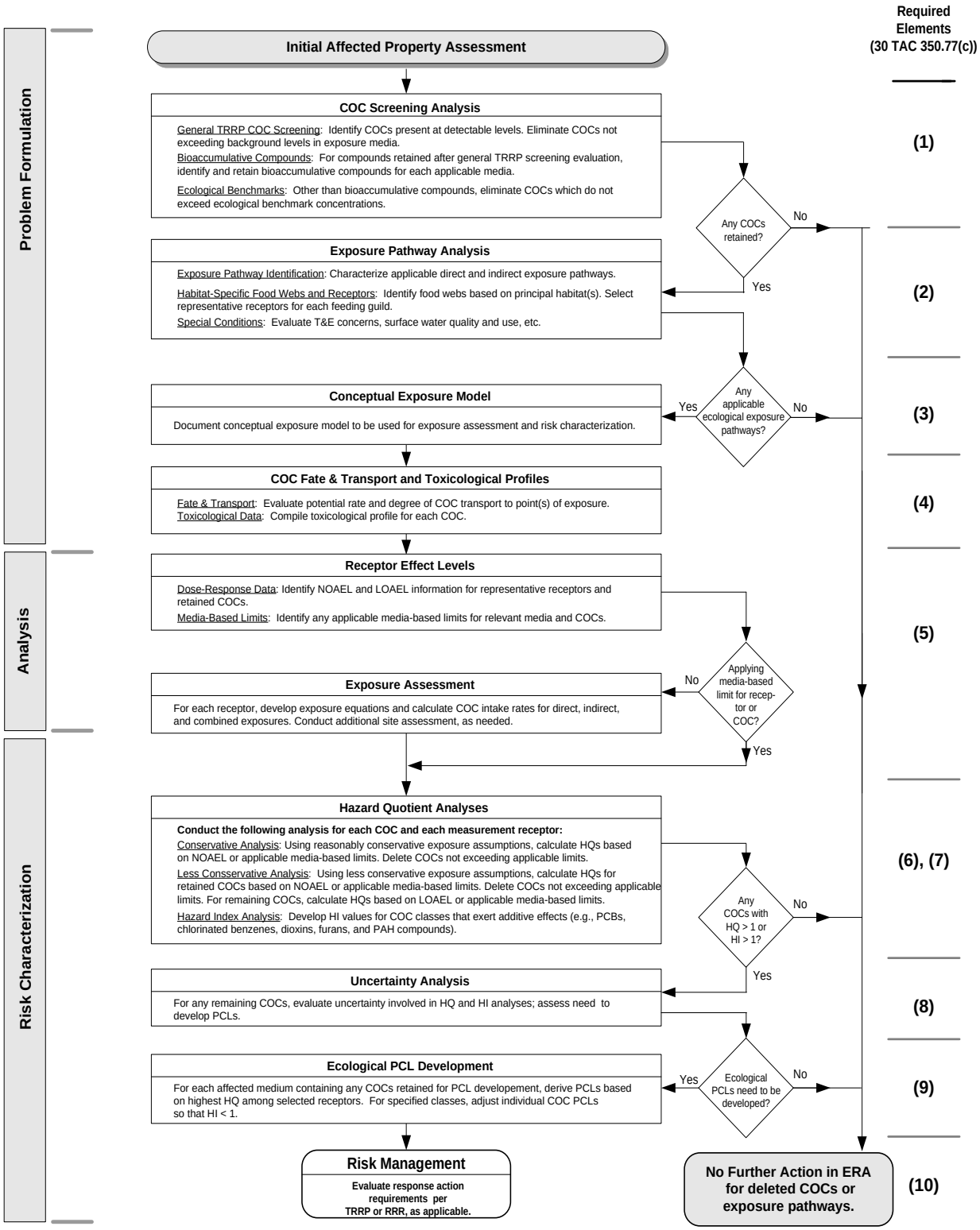
3.2 Phases of a Screening-Level Ecological Risk Assessment

The TRRP rule at 350.77(c) states that the screening-level ecological risk assessment should contain the three widely acknowledged phases of an ERA. These phases, as described in U.S. EPA (1992a) and as illustrated in Figure 3.2 are:

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

3.2.1 Problem Formulation

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



NOTES:

COC = Chemical of Concern
ERA = Ecological Risk Assessment
HI = Hazard Index
HQ = Hazard Quotient
LOAEL = Lowest Observed Adverse Effect Level
NOAEL = No Observed Adverse Effect Level
PAH = Polycyclic Aromatic Hydrocarbons
PCB = Polychlorinated biphenyls
PCL = Protective Concentration Level
RRR = Risk Reduction Rules (30 TAC 335)
SLERA = Screening-Level Ecological Risk Assessment
T&E = Threatened and endangered species
TRRP = Texas Risk Reduction Program (30 TAC 350)

Figure 3.1. Tier 2 screening-level ecological risk assessment (SLERA).

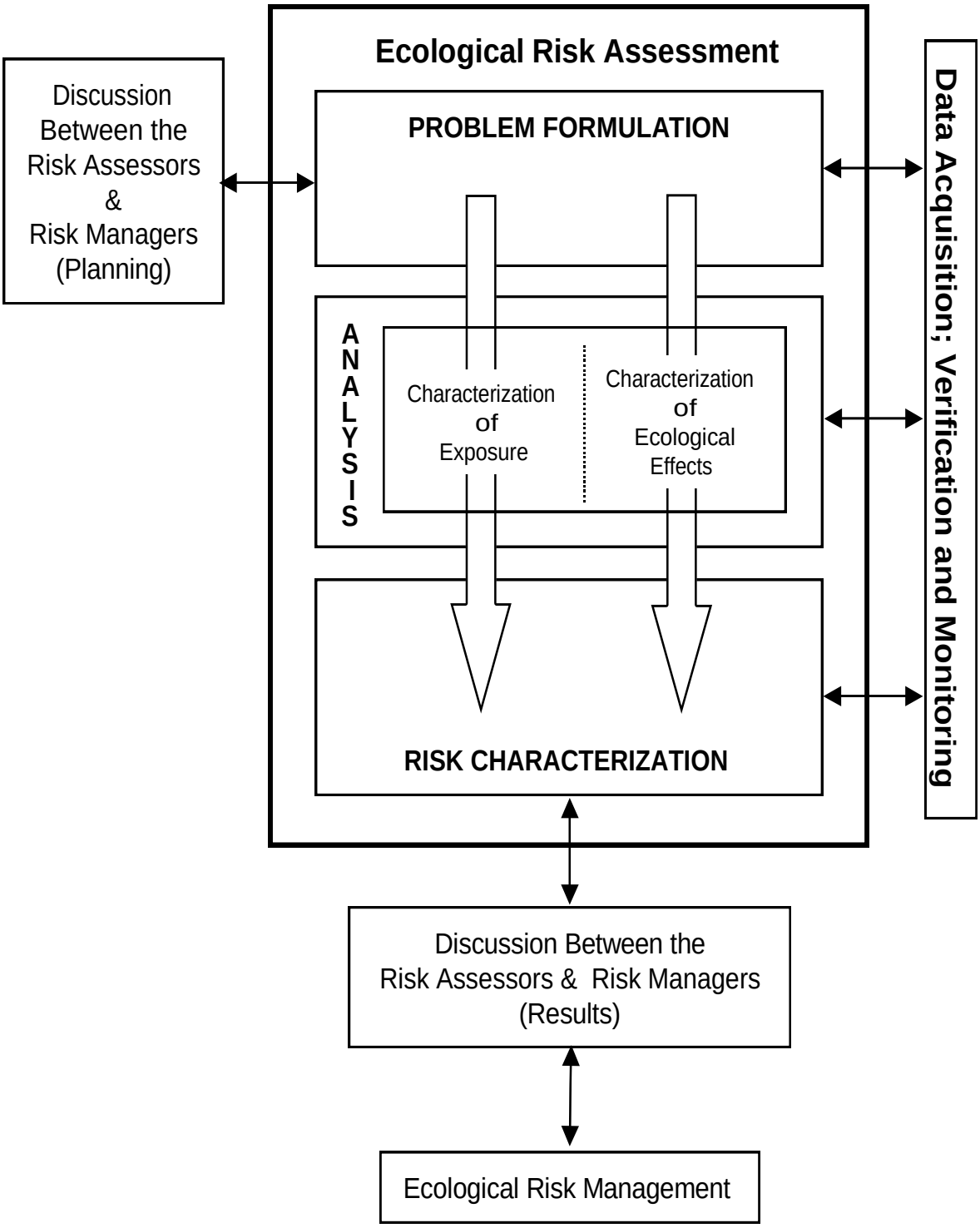


Figure 3.2. Ecological Risk Assessment Framework (modified from U.S. EPA 1992a).

Problem formulation addresses the following issues:

- environmental setting and COCs[4]
- COC fate and transport
- toxic mechanisms and potential receptors
- complete exposure pathways
- ecological endpoints

Problem formulation includes discussions between risk assessors and risk managers and other stakeholders to identify the stressor characteristics, ecosystems potentially at risk, and ecological effects to be evaluated. During problem formulation, assessment and measurement endpoints for the SLERA are identified, as described in **3.6.6–7**.

A product of problem formulation is a conceptual model for the SLERA that describes how a given stressor might affect ecological components of the environment. The conceptual model also describes questions about how stressors affect the assessment endpoints, the relationships among the assessment and measurement endpoints, the data required to answer the questions, and the methods that will be used to analyze the data (U.S. EPA, 1997a). As used in the context of this guidance, the "problem formulation" phase corresponds to required elements (1–4), which are listed in **3.3** and discussed in **3.5–8.**

3.2.2 Analysis

(modified from U.S. EPA, 1999a)

As illustrated in Figure 3.2, the analysis phase connects problem formulation with risk characterization through an examination of (1) the exposure of a measurement receptor (see **3.6.7**) to a COC, and (2) the ecological effects (toxicity) of that exposure. Exposure estimates emphasize contact and uptake into organisms, while effects estimates frequently entail extrapolation from test organisms to the organisms of interest.

Exposure is the contact of a receptor with a COC (e.g., ingestion). Exposures of ecological receptors to COCs released from affected properties are evaluated through consideration of exposure pathways. As discussed in **3.6.5**, all exposure pathways that are potentially complete should be evaluated, absent a reasonable justification. The existence of a potentially complete exposure pathway indicates that a receptor may contact a COC; it does not necessarily indicate that a receptor will be adversely affected. Exposure pathways considered herein include all direct uptake pathways of a COC from affected media (e.g., soil, sediment, and surface water) for lower-trophic-level receptors evaluated at the community level, and ingestion of a COC-laden organism (plant or animal food item) or media for higher-trophic-level receptors evaluated as class-specific feeding guilds.

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

Note that exposure pathways currently not addressed in this guide due to limited data include (1) inhalation and dermal exposure pathways for higher-trophic-level organisms, (2) ingestion via grooming and preening, and (3) foliar uptake and direct deposition uptake of COCs by plants.

The ecological effects of a COC are assessed by identifying toxicity reference values specific to a COC and the measurement receptor being evaluated (see **3.9.5**). As discussed in **3.10**, TRVs are subsequently set as the denominator for computing ecological hazard quotients during risk characterization. The TRVs used in risk characterization for lower-trophic-level communities are media specific, whereas TRVs for upper-trophic-level class-specific guilds are given in terms of dose ingested. As used herein, the analysis phase corresponds to required element (5) (**3.3, 3.9**).

3.2.3 Risk Characterization

In risk characterization, data on exposure and effects are integrated into a statement about risk to the assessment endpoints established during problem formulation (U.S. EPA, 1997a). Risk characterization is the final phase of risk assessment and includes two major components: *risk estimation* and *risk description* (U.S. EPA, 1992a). Risk estimation is an integration of the exposure assessment (see **3.9.2**) and the ecological effects or toxicity assessment (see **3.9.5**) to determine the potential risk to a community or feeding guild from exposure to a COC. Risk is estimated using the hazard-quotient method. Risk description describes the magnitude and nature of potential risk for each community and guild, based on the quantitative results of the risk estimation and calculated HQ values (U.S. EPA, 1999a) by summarizing the associated uncertainties and identifying a threshold for adverse effects on the assessment end points. To estimate potential ecological risk, an HQ should be calculated specific to each measurement receptor, COC, and exposure-scenario location evaluated in the risk assessment (ibid.).

According to U.S. EPA (1999a), risk description considers the magnitude and nature of potential risk for community and class-specific guild measurement receptors evaluated, and supplies information for the risk managers to evaluate the significance of an HQ value. Risk description also discusses the significance of the default assumptions used to assess exposure, because they affect the magnitude and certainty of the calculated HQ value. The resultant risk characterization should consider any major uncertainties and limitations associated with results generated in performing the screening-level risk assessment, because uncertainty can be introduced into a risk assessment at every step of the process.

The final outputs of the risk-characterization phase are COC concentrations in each environmental medium that bound the threshold for estimated adverse ecological effects given the uncertainty inherent in the data and models used (U.S. EPA, 1997a). The lower bound of the threshold will be based on reasonably conservative assumptions and no observed adverse effect level (NOAEL) toxicity values. The upper bound will be based on

observed impacts or predictions that ecological impacts could occur. This upper bound will be developed using justified less conservative exposure assumptions, site-specific data, and lowest observed adverse effect level (LOAEL) toxicity values. As used in the context of this guidance, the "risk characterization" phase corresponds to required elements (6–9), which are listed in **3.3** and discussed in **3.10–13**.

3.3 SLERA: Required Elements

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

1. Compare concentrations of non-bioaccumulative COCs at the affected property against established ecological benchmarks or use approved methodologies to develop benchmarks to determine potential effects and to eliminate COCs that pose no unacceptable ecological risk. If all COCs are eliminated at this point, the assessment ends (see **3.4–5**).
2. Identify communities (e.g., soil invertebrates, benthic invertebrates) and major feeding guilds (e.g., omnivorous mammals, piscivorous birds) and their representative species which are supported by habitats on the affected property for each exposure pathway that is complete or reasonably anticipated to be completed (see **3.6**).
3. Develop a conceptual model that graphically depicts the movement of COCs through media to communities and the feeding guilds (see **3.7**).
4. Discuss COC fate and transport and toxicological profiles (see **3.8**).
5. Prepare a list of input data including values from the literature (e.g., exposure factors, intake equations that account for total exposure, values for no observed adverse effect level [NOAEL] and lowest observed adverse-effect level [LOAEL], references), any available site-specific data, and reasonably conservative exposure assumptions, then calculate the total exposure to selected ecological receptors from each COC not eliminated according to required element (1). Present these calculations in tables or spreadsheets (see **3.9**).
6. Use an ecological-hazard-quotient methodology to compare exposures to the NOAELs in order to eliminate COCs that pose no unacceptable risk (i.e., NOAEL hazard quotient ≤ 1); however, when multiple members of a class of COCs are present which exert additive effects, an ecological-hazard-index methodology is also appropriate. If all COCs are eliminated at this point, the assessment ends (see **3.10**).
7. Justify the use of less conservative assumptions (e.g., a larger home range) to adjust the exposure and repeat the hazard-quotient exercise in required element (6), again eliminating COCs that pose no unacceptable risk based on comparisons to the NOAELs and adding another set of comparisons, this time to the LOAELs for those COCs indicating a potential risk (i.e., a NOAEL hazard quotient > 1); however, when multiple members of a class of COCs are present whose effects are additive effects, an ecological-hazard-index methodology is also

appropriate. If all COCs are eliminated at this point, the assessment process ends (see **3.11**).

8. Analyze the major areas of uncertainty associated with the screening-level ecological risk assessment, including a justification for not developing PCLs for particular COCs and pathways, if appropriate (e.g., NOAEL hazard quotient > 1 > LOAEL hazard quotient, an evaluation of the likelihood of ecological risk, a discussion of the half-life of the COCs, etc.). However, when multiple members of a class of COCs with additive effects are present, an ecological-hazard-index methodology is also appropriate. If all COCs are eliminated at this point, the ecological risk assessment process ends (see **3.12**).
9. Calculate medium-specific PCLs bounded by the NOAEL and the LOAEL used in item 7 for those COCs that are not eliminated as a result of the hazard-quotient exercises or the uncertainty analysis (see **3.13**).
10. Make a recommendation for managing ecological risk at the affected property based on the final ecological PCLs, unless proceeding under Tier 3 (this procedure may be included as part of the affected property assessment report, the self-implementation notice, or the response action plan) (see **3.14**).

3.4 Identification and Evaluation of Bioaccumulative COCs

Although not a separate required element under the TRRP rule, the identification of bioaccumulative COCs present at the affected property is nonetheless required for properly comparing concentrations in the affected property to ecological screening benchmarks under required element (1) (see **3.5**).

3.4.1 General Information

Bioaccumulative COCs tend to increase in concentration within some organisms relative to their concentration in environmental media and dietary sources due to sequestration in certain body tissues. Biomagnification (bioaccumulation in successive trophic levels of a food chain) can result in concentrations of COCs many times greater than those in environmental media. Bioaccumulation is an important aspect of ecological risk assessment because it can result in increased exposure to multiple trophic levels as compared with COCs that do not bioaccumulate. Also, bioaccumulative COCs can be present at a concentration in environmental media that is protective for direct exposure, but that can pose indirect risk to higher trophic levels. The ecological benchmarks selected for use in this publication (see **3.5**) evaluate direct exposure to specific media for selected receptors and are not expected to evaluate bioaccumulation concerns.

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

et al., 1997). In evaluating the potential for effects due to bioaccumulation, the person should be careful to select the appropriate measure of effect as some effects of bioaccumulation (e.g., eggshell thickness among birds) cannot be detected in standard laboratory toxicity tests (Franke et al., 1994). A variety of factors determines the site-specific potential for bioaccumulation, including biotic (e.g., feeding strategy, behavior, and physiology) and abiotic (e.g., chemistry of the environment) considerations. For example, polycyclic aromatic hydrocarbons accumulate in most benthic invertebrates because they have a minimal capacity for metabolism of PAHs, while fish are thought to rapidly metabolize and excrete PAHs.

3.4.2 Identification

The TCEQ has identified specific COCs (Table 3.1) that may pose substantial risk due to bioaccumulation. In identifying the organic COCs listed in the table, the agency has made use of lists of variants of bioaccumulative COCs by various authorities, including the U. S. EPA (1995), Environment Canada (1995), the United Nations Economic Commission for Europe (1998), and the North American Commission for Environmental Cooperation (1997). The presence of COCs on multiple lists was used to identify the predominant organic bioaccumulative COCs that were included in Table 3.1. Organic bioaccumulative chemicals were included for sediment, water, and soil, except for those COCs with a state or federal water quality standard that underwent additional analysis, as described below.

The listing of metals in Table 3.1 is based on media-specific uptake factors (accumulation factors). The listing of metals for soil and sediment exposure is based on the use of uptake factors for soil plants (Bechtel Jacobs Company, 1998a), soil invertebrates (Sample et al., 1998a), and benthic invertebrates (Bechtel Jacobs Company, 1998b). Uptake factors were reported as the ratio of the concentration of a given COC in biota to that in an abiotic medium. Typically, data from those references are based on field studies and the analytical methods were for “total” analysis; thus, it is not possible to determine the particular metal species responsible for the uptake factor. Each of these studies reports the 90th-percentile uptake factor for all data it includes (e.g., the cadmium 90th percentile is based on 120 individual uptake factors). Metals with a reported 90th-percentile uptake factor > 1 for soil plants or invertebrates are listed in Table 3.1 for soil exposure, and metals with a 90th-percentile uptake factor > 1 for benthic invertebrates were listed in Table 3.1 for sediment exposure. The significant exceptions are tributyltin and selenium, in which case professional judgment was used to apply the listing to sediments.

The listing of metals for surface water is based on bioconcentration factors obtained from U. S. EPA (1999a). Metals with a bioconcentration factor (BCF) greater than 1,000 for aquatic invertebrates or fish were initially listed in Table 3.1 for water exposure. However, those metals identified as bioaccumulative that have a state or federal water quality standard underwent additional analysis, which is described below. Thallium is the

sole metal listed in Table 3.1 for water exposure that does not have a water quality standard—it is listed based on a BCF > 1,000.

Bioaccumulative COCs listed in Table 3.1 also have ecological screening benchmarks listed in Tables 3.2–3.4 to aid in the understanding of the potential for varying impacts—or lack thereof—at different ecosystem trophic levels. As discussed in **3.5.1**, the benchmarks presented in Tables 3.2–3.4 do not directly address the food-chain transfer of bioaccumulative COCs. When the concentration of a bioaccumulative COC falls below a corresponding ecological screening benchmark, and the COC for that medium has been recognized as being a concern for bioaccumulation to higher trophic levels (see Table 3.1), then the person may be required to further evaluate the possible risks from exposure to that COC for that medium through the food chain. For example, a bioaccumulative COC may be evaluated for upper-trophic-level wildlife receptors (e.g., heron, raccoon, mink), but the screening-level benchmark may indicate that the sediment concentrations are not likely to adversely affect the benthic community.

There has been much debate within the multi-stakeholder ecological work group about the appropriateness of further evaluating surface water risks through the food chain when the COC concentration is at or below a state or federal benchmark. To address this issue, the work group evaluated the surface water–to–piscivore exposure pathway for those COCs with state-adopted water quality criteria for aquatic life. More than any other type of wildlife, the piscivore generally has the greatest potential of exposure to surface water COCs through food-chain uptake. Based on its widespread distribution, relatively lower body weight, and frequent inclusion in ERAs, the belted kingfisher was selected as a representative piscivore.

A preliminary evaluation of the metals was conducted by setting COC exposure-point concentrations equal to their respective chronic aquatic-life surface water criteria and by using conservative exposure factors (e.g., 100 percent area use). The results indicated that most of the metals would be protective of wildlife at their water quality criteria concentrations; only mercury and selenium should be further evaluated. A more comprehensive evaluation of these two metals and the organic COCs was then conducted by the Work Group.

Total exposure (diet plus ingested water) for the belted kingfisher was modeled using bioaccumulation factors or equivalents (e.g., BCFs × food-chain multipliers). The exposure assumed that 100 percent of the diet and ingested water came from areas with individual COC concentrations at the chronic surface water quality criterion for aquatic life (the lower value between freshwater and saltwater). These values and natural-history information for the belted kingfisher were plugged into the general dose equation (see **3.9.2**) and a hazard-quotient method (see **3.10**) was used to determine potential risk. Toxicity reference values (see **3.9.5**) were obtained from the available literature and adjusted, as appropriate, with uncertainty factors as described in **3.9.5.4**. The results of this evaluation

are reflected in Table 3.1; a discussion of how BAFs were determined is presented in **3.9.2.1.**

It was determined that when mercury, selenium,[5] DDT, or PCBs were evaluated at concentrations equal to their respective chronic aquatic-life criteria, bioaccumulative effects to the belted kingfisher are possible (i.e., these water quality criteria may not be protective of wildlife). Therefore, whenever these COCs are in water at the affected property, they will need to be analyzed for bioaccumulative effects through the food chain. In addition, site-specific circumstances may warrant a more diligent evaluation of bioaccumulative COCs in surface water depending on the food-chain dynamics, receptors, COC chemistry, etc.

Based upon the results of the previously described methodology and the kingfisher exercise, bioaccumulative COCs and the media in which they bioaccumulate are listed in Table 3.1. The methodology (identifying COCs requiring evaluation of risk to upper-trophic-level organisms, regardless of their environmental concentration in relation to screening benchmarks) is restricted to this table and does not reflect TCEQ policy on other issues, such as the use of 90th-percentile uptake factors.

The listing of bioaccumulative COCs for water in Table 3.1 does not impose or suggest a new or different water quality criterion at the affected property that would apply to the water column or to sediments. The evaluation of COCs is intended to address site-specific exposure pathways that might not be addressed by the statewide water quality criteria. This approach is analogous to that taken by other regulatory programs of the TCEQ. For example, wastewater permits sometimes require additional treatment of pollutants—beyond that needed to meet numerical water quality criteria—when this additional treatment is needed to address site-specific, water-quality concerns.

The TCEQ retains a site-specific ability to identify and require evaluation of additional compounds that may pose risk due to bioaccumulation. Even though Table 3.1 is the primary tool for identifying bioaccumulative COCs, it is not expected to identify all COCs that tend to bioaccumulate. The commission reserves the right to require evaluation of additional compounds that may pose risk due to bioaccumulation under site-specific conditions (e.g., COC distribution in environmental media, food-chain dynamics, and receptors) and considering COC-specific characteristics. Connell (1990) notes that current information does not allow absolute identification of specific COC characteristics necessary for bioaccumulation, but he does provide a general discussion of the often interrelated characteristics that allow bioaccumulation to occur. Characteristics of particular importance include: chemical structure, molecular weight,

[5] The piscivore evaluation indicated that the water quality standard for selenium may be protective in water (i.e., LOAEL hazard quotient < 1), but the ecological work group wanted it retained based on site experience and professional judgment.

molecular dimensions, stability, log K_{ow}, water solubility, and degree of ionization.

The multi-stakeholder work group agreed that COCs in sediment or surface water with log K_{ow} values between 3.8 (Feijtel et al., 1997) and 8.0 (Thomann, 1989) may be identified by the TCEQ as warranting evaluation for bioaccumulation. Furthermore, COCs with a molecular weight > 700 will be considered to have a reduced potential to bioaccumulate, regardless of their log K_{ow}.

In developing this guidance, the multi-stakeholder work group was not able to identify any specific mechanism or threshold to trigger the TCEQ's evaluation of potentially bioaccumulative COCs in soil not listed in Table 3.1. As stated earlier, the TCEQ will determine whether to require evaluation of COCs not listed in Table 3.1 case by case. There is no default requirement for the person to evaluate bioaccumulation potential for COCs not listed in Table 3.1. However, the agency prefers that any voluntary evaluations (quantitative or qualitative) be included in the ERA.

3.4.3 Evaluation of Risk

All COCs listed in Table 3.1, as well as those that are retained through the benchmark screening process described in **3.5**, are subject to further evaluation. Identification of COCs that bioaccumulate precedes the application of ecological benchmarks, and is used to retain COCs for food-chain analysis regardless of their concentration relative to ecological benchmarks. Bioaccumulative COCs that are present in media at a concentration below applicable benchmarks will not be evaluated further for direct exposure to those media. For soils to be retained for evaluation of risk to higher trophic levels, the metals listed in Table 3.1 must be present above background concentrations. The bioaccumulation evaluation, including any site-specific considerations (e.g., metallic species present) should be conducted during the food-chain analysis, as discussed in **3.6–11**. Justification is required to eliminate from food chain analysis those COCs listed in Table 3.1.

An important component of bioaccumulation potential relates to the particular metal species present. Many metals are only bioaccumulative in specific forms (e.g., oxidation state, elemental or organic compounds) and are more likely to bioaccumulate than other species of the same metal (e.g., lead acetate tends to bioaccumulate as opposed to elemental lead). Metals are assumed to be in a bioaccumulative form unless sufficient data are available to identify the species present so that their individual potential to bioaccumulate can be evaluated. Such evaluations need to consider environmental and biological transformation between metallic species in relation to bioavailability, including uptake and elimination rates. For instance, some COCs may have a very low media concentration of the bioavailable form because it is rapidly accumulated by organisms, which can give a false impression that it is not present at levels of concern. Additional discussion of metals appears in **3.9.6.1**.

Table 3.1. Bioaccumulative COCs.

Metals

CAS No.	COC	Applicable Media
7440-43-9	Cadmium	sediment, soil
7440–47-3	Chromium[a]	soil
7440-50-8	Copper[a]	sediment, soil
7439-92-1	Lead[a]	soil
7439-97-6	Mercury[c]	water, sediment, soil
744–02-0	Nickel[a]	sediment, soil
7782-49-2	Selenium[a,d]	water, sediment, soil
7440-28-0	Thallium[e]	water
688-73-3	Tributyltin[a]	sediment
7440-66-6	Zinc[a,b]	sediment, soil

Organochloride Pesticides

CAS No.	COC	Applicable Media
309-00-2	Aldrin[a]	sediment, soil
57-74-9	Chlordane[a]	sediment, soil
72-54-8	DDD[e,f]	water, sediment, soil
72-55-9	DDE[e,f]	water, sediment, soil
50-29-3	DDT[c,f]	water, sediment, soil
60-57-1	Dieldrin[a]	sediment, soil
72-20-8	Endrin[a,b]	sediment, soil
76-44-8	Heptachlor[a,b]	sediment, soil
1024-57-3	Heptachlor epoxide[a]	sediment, soil
8001-35-2	Toxaphene[a,b]	sediment, soil

Other Pesticides and PCBs

CAS No.	COC	Applicable Media
2385-85-5	Mirex[a]	sediment, soil
3980-114-4	Photomirex[a]	sediment, soil
1336-36-3	PCBs[c]	water, sediment, soil

Other Semi-Volatiles

CAS No.	COC	Applicable Media
N/A	Dioxins[e]	water, sediment, soil
N/A	Furans[e]	water, sediment, soil
118-74-1	Hexachlorobenzene[e]	water, sediment, soil
608-73-1	Hexachlorocyclohexanes[a]	sediment, soil
29082-74-4	Octachlorostyrene[e]	water, sediment, soil
87-86-5	Pentachlorophenol[g]	sediment, soil

The following footnotes pertain to how the belted-kingfisher food-chain evaluation exercise may affect the "water" listing of the COCs in Table 3.1 and the listing of pentachlorophenol for sediment and soil:

[a] Evaluated as part of the belted-kingfisher exercise but not retained for "water" because the LOAEL HQ did not exceed 1.

[b] NOAEL HQ of 1 was exceeded and may warrant food-chain evaluation depending on site-specific circumstances.

[c] Evaluated as part of the belted-kingfisher exercise and retained for "water" because the LOAEL HQ of 1 was exceeded.

[d] Water listing based on-site experience and professional judgment of the ecological work group.

[e] Not evaluated in the belted-kingfisher exercise because there is no state-adopted aquatic life water quality standard.

[f] DDT and its metabolites (DDD and DDE) need to be evaluated cumulatively (i.e., use hazard index approach) and are therefore all listed for water.

[g] Pentachlorophenol is listed for sediment and soil based on its log K_{ow} of 4.74, and Ecological Soil Screening Levels in U.S. EPA, 2007d, indicating the potential for risk to birds and mammals at soil concentrations significantly below levels protective of plants and soil invertebrates.

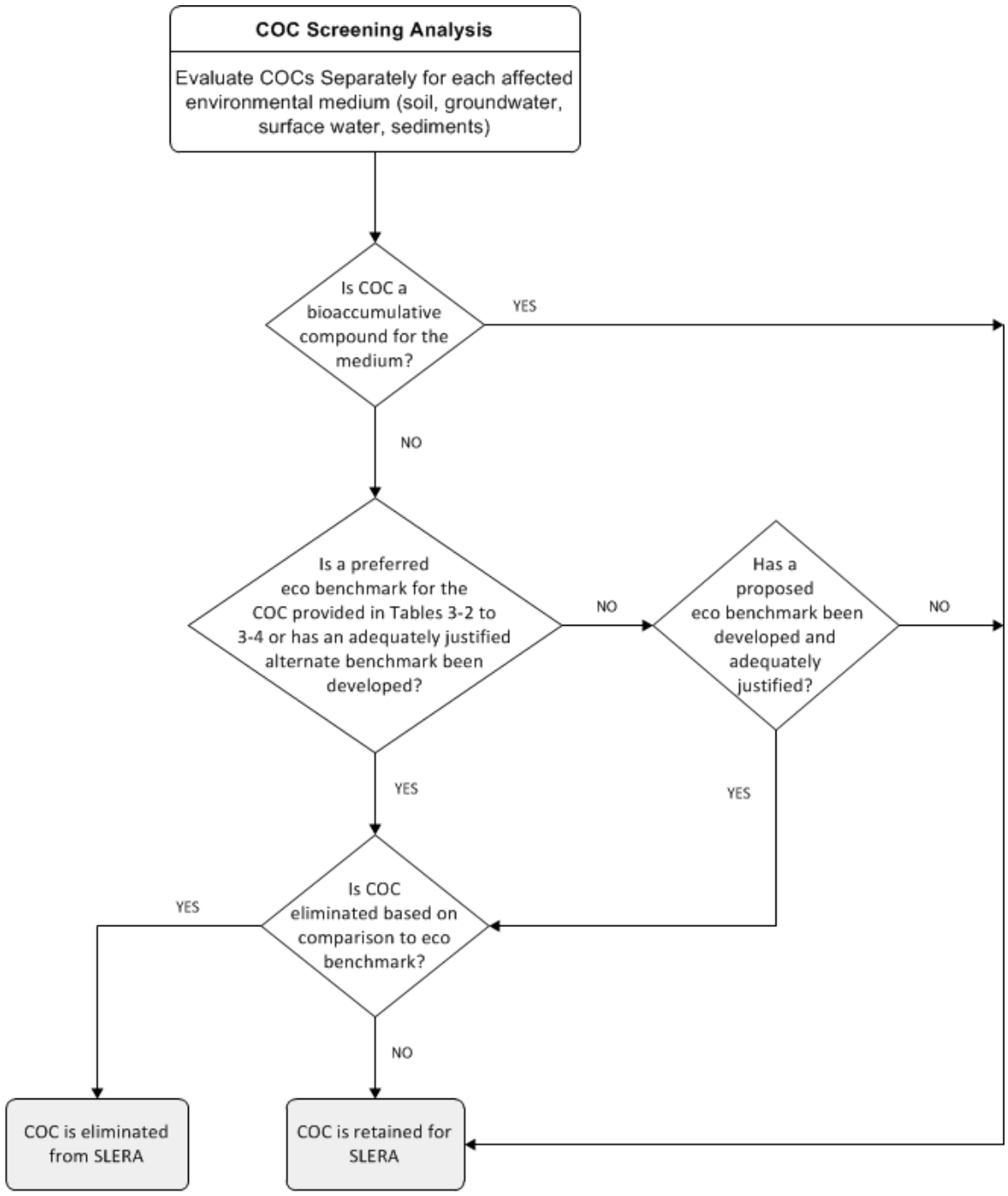
3.5 Ecological Screening Benchmarks

Comparison of affected property concentrations to ecological benchmarks is the first required element in a Tier 2 SLERA, as specified in the risk rule [30 TAC 350.77 (c)(1)]. This section presents the commission's preferred ecological screening benchmarks for water, sediment, and soil (see Tables 3.2–3.4). It also discusses the use of benchmarks (including alternatives to those specifically given herein), and discusses the derivation of benchmarks where none are specified for a particular COC and surrogate COCs for benchmark selection. A detailed discussion regarding the sources of these media benchmarks and others in the literature, their derivation, and particular advantages or limitations associated with various methodologies appears in Appendix A.

Benchmarks offer a simple approach for comparing COC concentrations in media at the affected property against values for media concentrations presumed safe to biota likely to be the most exposed (aquatic life, benthic and soil invertebrates, and plants). COCs present in media at sufficiently high concentrations to justify further evaluation in an ERA should be retained, whereas those that present little or no potential risk should be eliminated from further ecological review. Therefore, if a COC concentration is less than the benchmark for a particular medium, the COC may be excluded from further consideration, unless the COC is potentially bioaccumulative (see **3.4, 3.5.1**).

Benchmarks are intended to be conservative, and generally should not be used as triggers for remediation, or as cleanup goals. Similarly, the media benchmarks are not intended to be used in hazard quotient (HQ) calculations. As illustrated in Figure 3.3, if a COC is not bioaccumulative for the media in question **and** the COC is present below benchmark levels, further evaluation of that COC is unnecessary. Where no benchmark exists or can be readily derived for a COC in a particular medium (see discussion in **3.5.1**), the person would be required to further evaluate that particular COC in the ERA. As needed, the TCEQ may update the benchmark values in this guide if it deems the newer value or derivation process superior in quality and accuracy, and in response to rule and policy changes such as a change in the Texas Surface Water Quality Standards. Updates will likely appear at the TCEQ website.

For the Tier 2 SLERA, it is recommended that, for initial screening, the maximum measured COC concentration in the exposure medium be compared to the medium-specific ecological benchmark values, unless the maximum value can be demonstrably considered an extreme outlier for the data set of a particular exposure medium; in that case the next highest value (that is not determined to be an extreme outlier) will be used. Extreme outliers, along with an appendix for addressing outliers, are discussed in *Determining Representative Concentrations of Chemcials of Concern for Ecological Receptors* (TCEQ publication RG-366/TRRP-15eco). However,



NOTES:

COC = Chemical of Concern
SLERA = Screening-Level Ecological Risk Assessment

Figure 3.3. Tier 2 SLERA COC screening.

for all subsequent iterations of the Tier 2 SLERA or Tier 3 SSERA, more refined statistical methods (e.g., the 95 percent upper confidence limit of the arithmetic mean) may be applied (ibid.).

The benchmarks presented herein are based on a consensus generally agreed upon by the ecological work group. During selection, the work group primarily sought agreement on selecting qualifying sets of benchmarks (e.g., Region 4 surface water screening values; ER-Ls from Long et al., 1995), with the assumption that the whole set would be used rather than picking and choosing COCs individually from the various databases. However, in order to expand the list of COCs that have recommended benchmarks, individual values from other sources were also included. Preference for sets of benchmarks (or individual benchmarks) was based on the following characteristics:

- sets containing a large number of COCs
- sets developed transparently
- those appropriate for conservative screening
- benchmarks used in other TCEQ programs, including regulatory criteria
- benchmarks used in other state and federal ERA programs
- sets using data that include species relevant to Texas and the Gulf of Mexico
- sets that are relatively recent or meet current technical standards

3.5.1 Alternate and Proposed Benchmarks

In the context of this discussion, an *alternate* benchmark is one developed in lieu of the benchmark specified for a particular COC or medium in Tables 3.2–3.4. Where no benchmark is specified in those tables, the COC should be carried forward in the ERA unless a *proposed* benchmark can be developed or an appropriate surrogate chemical is used (discussion follows). With one exception, alternate ecological-screening benchmarks for water, sediment, and soil may be developed if the person adequately justifies their use. The exception concerns surface water benchmarks based on existing state or federal criteria (marked with a superscript *a* or *c* in Table 3.2). In this situation, alternate benchmarks may not be substituted. In the absence of any state or federal aquatic-life criterion for a bioaccumulative COC in water (see Table 3.1), the person may propose the use of the human-health value (if available) to see if that concentration is protective of wildlife. The person will need to present this evaluation in the ecological risk assessment.

As indicated in Figure 3.3, alternate and proposed benchmarks that are developed or cited by the user should be fully explained, including those discussed in Appendix A. The person should justify why there is a need for the alternate benchmarks, including why the guidance-specified benchmarks are not suitable, if applicable. The literature source and derivation steps used in obtaining the alternate or proposed benchmarks should be provided. The alternate benchmarks should be comparable or better than the guidance-specified benchmarks in their data quality, statistical power, toxicological effects level, and mode of exposure. Where

multiple alternate or proposed benchmarks of similar nature are available, persons may calculate a geometric mean for a particular COC. Durda and Preziosi (2000) have outlined a two-phased approach for evaluating the quality of ecotoxicological data to be used in benchmark development that may be useful in selecting alternate or proposed benchmarks. Similarly, Clark et al. (1999) provide a thoughtful discussion related to benchmark development and application. Information detailing the level of comparability (with the guidance-specified benchmark) should be included for each alternate media benchmark.

The use of marine benchmarks for inland waters and vice versa, including highly saline waters in West Texas, is discouraged for COCs in water, and is not recommended for COCs in sediment. The person should make every effort to use benchmarks that are appropriate for the surface water conditions in question, rather than immediately defaulting to the other benchmark. See additional discussion in **3.5.4**.

3.5.1.1 Derivation of Surface Water Benchmarks Using LC_{50} Data

The Texas Surface Water Quality Standards, or TSWQS [30 TAC 307.6(c)(7)] provide a mechanism for deriving numerical criteria where there are no standards **and** available data are insufficient to allow the use of EPA guidelines [e.g., *Guidelines for Deriving Water Quality Criteria for the Protection of Aquatic Life and Its Uses* (U.S. EPA,1985) and *Water Quality Standards: Establishment of Numeric Criteria for Priority Toxic Pollutants; States' Compliances,* 57 FR 60848 (U.S. EPA, 1992)]. Depending on the persistence and bioaccumulative nature of the COC in question, the TSWQS specify that a multiple of the LC_{50} for the most sensitive aquatic organism may be used.

- for non-persistent COCs, chronic criteria = (LC_{50}) (0.10)
- for persistent COCs that do not bioaccumulate, chronic criteria = (LC_{50}) (0.05)
- for COCs that bioaccumulate, chronic criteria = (LC_{50}) (0.01)

In selecting toxicity data to calculate a value, LC_{50} test results are preferred, and marine or freshwater (depending on the nature of the receiving water) species indigenous to Texas should be used whenever possible. Results from aquatic plant and algae toxicity tests will not usually be accepted (TCEQ, 2011). Additionally, flow-through tests with exposure times of 48 hours (for invertebrates) or 96 hours (for vertebrates) are preferred, but static test results can be used, particularly where the data indicate a higher sensitivity. If the toxicity-test data do not meet these conditions, the person should justify, as much as possible, the rationale behind the selection of toxicity-test data. Generally, the most conservative LC_{50} should be used that meets the selection preferences indicated. If more than one LC_{50} data point is available for a species, the geometric mean should be calculated and used. If the LC_{50} approach is used, the person should provide some rationale for the selection of all LC_{50} data points. The EPA ECOTOX database, available

online at <cfpub.epa.gov/ecotox/>, may be used as a source of aquatic (and terrestrial) toxicity data, although it is certainly not the only source. When possible the user should obtain the original paper associated with the ECOTOX reference, and should review the documentation and control codes assigned to the reference. The source paper should be indicated (and preferably reviewed), rather than simply citing the ECOTOX reference number alone.

Selection of the appropriate multiplier for the calculation is a function of persistence and of the tendency for that material to bioaccumulate. For calculating alternate or proposed criteria, COCs are considered persistent if the half-life in water or sediment is 60 days or greater, and COCs are considered bioaccumulative if the bioconcentration factor or bioaccumulation factor for the constituent (measured or estimated using regression analysis) is 1,000 or greater (TCEQ, 2011). BCFs or BAFs determined from laboratory or field studies using water-column invertebrates or fish are preferred over estimated values. A range of BCF or BAF values may be presented. The person should use half-lives or environmental-fate rate constants (e.g., volatilization, photolysis, hydrolysis, biodegradation) that are most appropriate for the surface water or sediment exposure pathway in question. A range of half-life values may be presented. The person should briefly justify the selection of the BCF or BAF and half-life information used in these decisions and cite the source. Appendix A gives an example derivation of an aquatic-life-protection value for chloroform using this approach.

3.5.1.2 Derivation of Sediment Benchmarks Using Equilibrium Partitioning

The ecological work group discussed the utility of the equilibrium partitioning (EqP) approach for developing sediment benchmarks, particularly for chemicals where other preferred benchmarks were unavailable from common and generally accepted sources. The EqP approach may be desirable for this purpose, because the chemical-specific partitioning coefficients necessary for the application of EqP are generally either known or fairly easily estimated, based on literature values. The approach (U.S. EPA, 1993h; Di Toro et al., 1991; Di Toro and McGrath, 2000) provides a means to calculate a bulk sediment COC concentration benchmark for a single nonpolar hydrophobic organic constituent by predicting the bioavailability of COCs sorbed to sediments.

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

organic-carbon partition coefficient K_{oc} (L/kg oc), and the mass fraction of organic carbon (kilograms oc per kilogram sediment) in sediment:

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

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

SQB (in µg/kg) = $K_p \times$ WQC, where WQC (µg/L) is the water quality final chronic value or similar value.

Where K_{oc} is unavailable, it can be estimated using the octanol-water partition coefficient (Di Toro et al., 1991):

$\log_{10} (K_{oc}) = 0.00028 + 0.983 \log_{10} (K_{ow})$.

These equations are included as general information. The person should use the log K_{ow} and log K_{oc} values specified in the TRRP rule at 30 TAC 350.73(f).

Finally, SQB = $f_{oc} \times K_{oc} \times$ WQC.

The principal advantages of this approach are that it allows for the derivation of COC-specific sediment benchmarks (and PCLs where appropriate; see **3.13.2**), and the methodology can be adapted to site conditions by adjusting the organic carbon parameter. EPA used this methodology to develop draft sediment quality criteria for acenaphthene, phenanthrene, fluoranthene, dieldrin, and endrin (U.S. EPA, 1993b–f) although the proposed criteria were never adopted. The EqP methodology is applicable for sediments with a f_{oc} value of 0.2–12 percent (U.S. EPA 1992b) and has been tested on nonionic organic compounds with log K_{ow} between 3.8 and 5.3 (U.S. EPA, 1997b). Although EqP values have been published for freshwater environments, the method also has been used for marine environments (Swartz, et al., 1990; Swartz, 1999; Boese et al., 1999).

Regarding the limitations of this approach, there is uncertainty with the value of the K_{ow} selected, how it was derived (same for f_{oc}), and its use as a predictor of K_{oc} because it is experimentally derived and organic matter in sediment can have different binding capacities [again, the person should use the log K_{ow} and log K_{oc} values specified in the TRRP rule at 30 TAC 350.73(f)]. The method assumes that organic carbon is a predominant sorption phase for nonionic organic COCs in sediments although characteristics other than organic-carbon concentration may also control sorption of nonionic organic COCs (grain size, dissolved organic carbon, mineralogy, source of the organic carbon, and molecular structure and surface area of organic carbon). The method also assumes that partitioning of the organic COC between organic carbon and the pore water is stable at equilibrium. Finally, the methodology assumes that the COC concentrations in interstitial water are acceptable predictors of adverse ecological effects—

in other words, it assumes that the sensitivities of benthic species and species used to drive the water quality criteria (primarily water column organisms) will be similar. This may be an inappropriate assumption for tube-dwelling organisms where exposure is primarily at the sediment-water interface, or for organisms where sediment ingestion is a primary exposure route (Kaag, et al., 1997).

Similarly, the methodology has not been extensively field validated and was primarily developed using acute-toxicity tests with single compounds, and its uncertainty has not been adequately evaluated. Since water quality criteria or similar benchmarks are used to derive the sediment benchmark using this approach, the methodology is limited by the availability of water quality criteria. Although descriptions of all the components and assumptions built into the EqP approach are beyond the scope of this guide, a full understanding of its intricacies will be needed to apply and defend its use outside of the discussion below on volatile compounds.

Although the agency is skeptical about the appropriateness of using EqP to derive sediment benchmarks to protect the benthic community—primarily because it does not address the sediment-ingestion exposure pathway—it does recognize that EqP may be more applicable to certain classes of COCs. Therefore, the ecological work group evaluated the use of EqP to derive sediment benchmarks for volatile organic chemicals (VOCs). The EqP process may be more appropriate for volatiles since these chemicals are expected to be more water soluble and available in the pore water than more hydrophobic chemicals that are bound to sediment particles (where ingestion of sediment would be a pathway of concern for benthic invertebrates). As presented below, the work group used the EqP equation modifications for volatiles suggested by Fuchsman (2003) and the TCEQ LC_{50}-based surface water screening values to calculate sediment benchmarks for these COCs:

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

Where:

SQB = sediment quality benchmark (mg/kg)

WQB = water quality benchmark (mg/L)

K_{oc} = organic carbon partition coefficient (unitless)

f_{oc} = fraction organic carbon (0.01 kg OC/kg sediment, TCEQ Publication RG-366/TRRP-24 default)

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

These EqP-derived sediment benchmarks for VOCs appear in Table 3.3. The person should remember that, even though surface water benchmarks are

used as input, the resulting sediment benchmark values are not effects-based for benthics.

The EqP estimation method explicitly assumes thermodynamic equilibrium, which is not necessarily true for sediments at all sites (SAB, 1990, 1992). The method, as typically applied, also does not account for the presence of dissolved organic carbon within the pore water, variations in sediment particle sizes (e.g., Di Toro, 1985; Weber et al., 1995) nor does it address issues such as irreversible adsorption, a characteristic of "aged" sediments (Karickhoff and Morris, 1985; Kelsey and Alexander, 1997; Hulscher et al., 1999; Kan et al., 1998). The EqP approach has also been criticized because it assumes that pore water is the only exposure pathway. Thus, the three main limitations of the EqP method are believed to be:

1. Its assumption of kinetic equilibrium.

2. Its failure to consider the presence of competing partitioning compartments (dissolved organic carbon, clays, biota).

3. Its failure to consider the additional exposure pathways to ecological receptors (e.g., sediment ingestion).

The first two limitations would lead to an overestimation of the pore-water chemical concentration protective of benthic invertebrates, that is, the concentration estimated to protect receptors would be unnecessarily elevated. The third limitation could result in an underestimation of the PCL for a sediment-associated chemical.

If applied to a well-characterized sediment bed and COC, the EqP method can more realistically evaluate site-specific relevant receptors and measurement endpoints. The method is highly conservative for most COCs (i.e., those with moderate K_{ow} and K_{oc}) and will tend to overestimate risks especially with high-K_{oc} chemicals. In addition, the EqP approach will result in a more realistic false positive error rate than other approaches. The EqP method also can work well for multiple chemicals (i.e., address co-occurring chemical stressors). Given that additive, synergistic, and antagonistic effects due to co-occurring chemicals have been extensively studied in water exposures (e.g., Sprague, 1985; McCarty, 1991) the method can and has been adapted to co-occurring chemicals (e.g., Fuchsman et al., 1999). Additional adjustments of the method also appear effective in predicting toxic effects due to very low K_{ow} or K_{oc} type chemicals (Fuchsman et al., 2000). For chemicals with high K_{oc}, where sediment ingestion may be an issue, partitioning can still be used to predict tissue burdens (**3.9.5.5**) for calculating risk potentials based on residues (e.g., McCarty, 1991; Abernathy et al., 1988; U.S. ACE, 1992b,c, 1996a,b). For COCs with a log $K_{oc} < 2$, it is important to use the mathematically rigorous form of the equilibrium-partitioning relationship (Fuchsman, 2003).

3.5.2 Use of Surrogate Chemical Data

Where a benchmark is not specified for a particular COC and there is insufficient information to derive a benchmark, the person may use toxicity

data or available benchmarks for surrogate compounds. The assumption is that the surrogate compound and the COC in question will have similar toxicity. In support of this assumption, the person should evaluate the similarity in chemical structure, chemical substitutions, molecular properties (e.g., lipophilicity, polarity), metabolites or breakdown products, mechanisms of action, similarity in effect at different exposure durations or test endpoints, metabolic pathways, and physicochemical properties (e.g., octanol-water partition coefficient, water solubility). A number of models based on molecular structure have been developed to estimate the ecological effects of chemicals. These models, termed *SAR* (structure-activity analysis) or *QSAR* (quantitative structure-activity analysis), can be used to derive surrogate measures of toxicity. The EPA has developed a series of QSAR models for aquatic toxicity which are available as the ECOSAR software (Clements and Nabholz, 1994). Other models are also commercially available, such as TOPKAT (Enslein et al., 1989). As inputs, these and other QSAR models typically require either the chemical structure or physical properties of a novel chemical, such as its octanol-water partitioning coefficient, water solubility, molecular weight, and chemical subgroups (e.g., aromatic rings, hydroxyls).

COCs in the same class of chemicals, groups of isomers, congeners, and close homologues are often used as surrogates. For example, high-molecular-weight PAHs could be grouped together and evaluated using the toxicity data from a PAH compound belonging to this group, such as benzo[*a*]pyrene. However the person should review any relevant toxicological endpoints associated with the surrogate selection to evaluate whether the candidate surrogate is appropriate given the selected affected property receptors and food web. For example, benzo[*a*]pyrene as a surrogate for the high-molecular-weight PAHs may generally be acceptable for most terrestrial mammals, but in some cases (e.g., an aquatic food web), other high-molecular-weight PAHs such as pyrene may be more appropriate due to the greater concern for a different endpoint (e.g., developmental effects). Proper surrogate selection (including toxicity reference-value endpoints) is critical, particularly where sensitive species may be present.

3.5.3 Surface Water Benchmarks

This subsection discusses the selected benchmarks for surface water (Table 3.2); more details regarding their selection appear in Appendix A. Figure 3.4 depicts the preferred hierarchy for selecting ecological screening benchmarks for water. The benchmarks for surface water are intended to be protective of aquatic biota and are not necessarily protective of mammalian and avian receptors that may be exposed to COCs through ingestion of contaminated prey or water. Refer to **3.4.2** for additional information on the kingfisher exercise.

The Texas Surface Water Quality Standards serve as the primary benchmarks for surface water or, if there is no state standard, the most current federal National Ambient Water Quality Criteria (U.S. EPA, 2013) are to be used. Current information on the standards can be obtained online

at <www.tceq.texas.gov/waterquality/standards>. Where state or federal criteria are available, they must be used. Since a comparison of concentrations in the affected property to benchmarks in required element (1) is intended to be a screening step, the chronic criteria for the protection of aquatic life are used. Acute criteria may be applied (as appropriate), given the aquatic life uses of the receiving water, at a later stage in the SLERA. The TSWQS define marine waters (or saltwater) as coastal waters with measurable elevation changes due to normal tides and further specify that marine waters are generally considered coastal waters that typically have salinities of 2 parts per thousand or greater in a significant portion of the water column. This definition should be used to select the appropriate freshwater or marine water quality benchmark. Additionally, a water body is considered tidally influenced when there is observed tidal activity, total dissolved solids are at least 2,000 mg/L, or specific conductance is at least 3,077 µS/cm (TCEQ, 2003). Freshwater and marine benchmarks should not be used interchangeably where a benchmark is unavailable for the appropriate medium. Hence, marine benchmarks are inappropriate for highly saline, yet inland, waters in West Texas. Some coastal water bodies may experience tidal elevation changes, yet support a freshwater community due to marginal salinity levels. In such cases, the person may use freshwater sediment and water benchmarks provided that information is presented to demonstrate that the water body supports freshwater organisms.

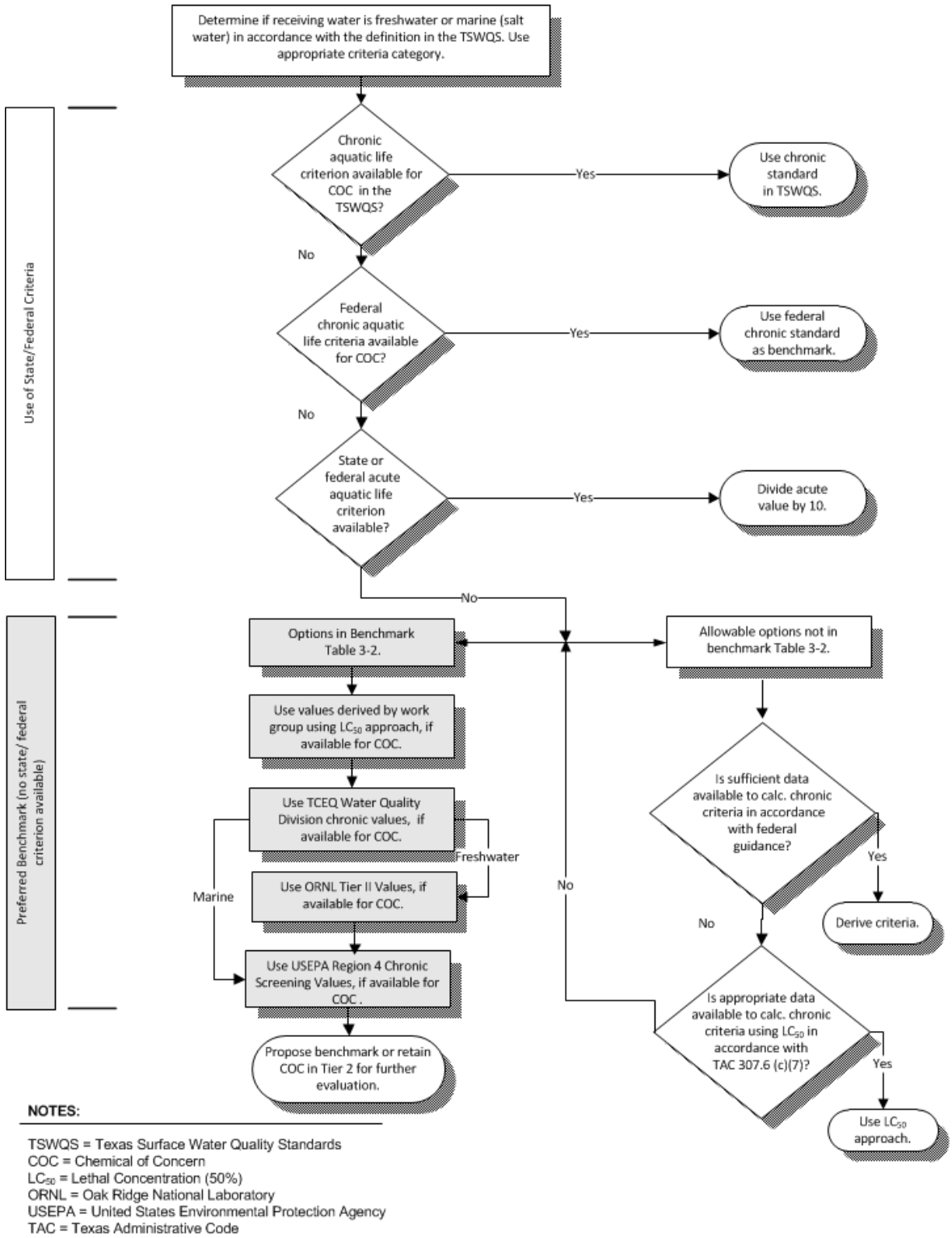


Figure 3.4. Hierachy of ecological benchmarks for surface water.

For a number of metals, the freshwater criterion is a function of hardness. The use of site-specific or default hardness values is discussed in Appendix A. Specific numerical aquatic-life criteria for metals and metalloids apply to dissolved concentrations where noted. Dissolved

concentrations can be estimated by filtration of samples prior to analysis, or by converting from total recoverable measurements in accordance with procedures approved by the commission in the latest revision of the TCEQ's Implementation Procedures (2011). This methodology is discussed in Appendix A. The TCEQ prefers dissolved metals data for surface waters, where appropriate, rather than the use of the mathematical conversion.

The person should also be aware that the TSWQS define site-specific criteria for aquatic-life protection for selected water bodies [Appendix E—Site Specific Toxic Criteria in 30 TAC 307]. As these values are higher (less conservative) than those in Table 3.2, the person should determine if there is a site-specific standard for the surface water (and COC) in question.

Since COCs for which state or federal water quality criteria are defined are few, the ecological work group sought out sources for other benchmarks for COCs that lack specified criteria. Four primary sources were used for the derivation of these other water quality benchmarks. Based on the work group's preferred order, Table 3.1 reflects the one number (when available) that should be used for comparison to the surface water data. As indicated, the preferred hierarchy for surface water benchmarks is presented in Figure 3.4. The work group first derived chronic default values for aquatic life for selected COCs using the LC_{50} approach discussed in **3.5.1.1**. An example derivation of this process appears in Appendix A. A full discussion of each value derived through the LC_{50} method is available upon request. Secondly, the work group selected the freshwater and marine water quality chronic values derived by the TCEQ for wastewater permits and requests from the Office of Waste, based on the methodology defined in the TSWQS at 307.6(c)(7). As a third choice, the work group selected the Tier II secondary chronic values developed by Suter and Tsao (1996) of the Oak Ridge National Laboratory. The Oak Ridge Tier II SCVs were derived for freshwater biota only. The last set of water quality benchmarks used—both marine and freshwater—is the U.S. EPA Region 4 Water Management Division chronic screening values (U.S. EPA Region 4, 2001).

The person may develop alternate benchmarks with appropriate justification (except where a state or federal criterion is indicated) as discussed previously and depicted in Figure 3.4. The values in Table 3.2 are preferred over alternate benchmarks.

3.5.4 Sediment Benchmarks

The benchmarks for sediment are intended to be protective of benthic biota, and are not necessarily protective of mammalian and avian receptors that may be exposed to COCs through the food chain or via the incidental ingestion of sediment. The preferred benchmarks appear in Table 3.3. The following discussion briefly overviews the benchmark approaches that were selected by the multi-stakeholder ecological work group. More detail regarding these benchmarks, and others that the group did not prefer, appears in Appendix A. Alternate sediment benchmarks (see **3.5.1**),

including those discussed in Appendix A, may be used with appropriate justification.

For freshwater sediment benchmarks, the primary benchmarks selected were the threshold-effect concentrations (TECs) from MacDonald et al. (2000). Other databases were accessed to expand the benchmark list. For antimony and silver, the effects range—low (ER-L) values from Long and Morgan (1990) were used. The freshwater benchmarks for iron, manganese, several pesticides, and individual polychlorinated biphenyl aroclors are lowest-effects levels (LELs) as discussed in Persaud et al. (1993). For marine benchmarks, the primary benchmarks selected were the ER-L values in Long et al. (1995), whose ER-L values for metals, PAHs, total DDTs, and total PCBs appear in Table 3.3. The marine TELs from Smith et al. (1996b) for chlordane, lindane, dieldrin, and phthalates were also used. The *Sum DDT, Sum DDE,* and *Sum DDE* marine sediment benchmarks originate from Canadian Council of Ministers of the Environment (1999a). Also, as described in **3.5.1.2**, an EqP process was used to derive freshwater and marine sediment benchmarks for VOCs.

Regarding PAHs, although benchmarks are provided for individual PAH compounds, low and high molecular weight (MW) PAHs (marine only), and total PAHs, the benchmarks for total PAHs are the most relevant for evaluating risk in an ERA. This is because PAHs almost always occur in the environment as mixtures. Values for individual and low- and high-MW PAHs are given as guidelines to aid in the determination of disproportionate concentrations within the mixture that may be masked by the total. Therefore, PAHs may be screened out using the total PAH benchmark even if individual, low-MW, or high-MW PAH benchmarks are exceeded. However, any exceedances of individual, low-MW, or high-MW PAH second effect levels (see Table A.2, Appendix A) may indicate adverse effects and therefore should be further discussed (e.g., in the uncertainty analysis). If appropriate, individual PAHs that exceed the second effect levels should be retained beyond the benchmark screening step (required element 1). The total PAH benchmarks are based on the 13 parent PAH compounds listed in Table 3.3 (Don MacDonald, personal communication, Oct. 2003), meaning that a person who wishes to use a total PAH benchmark for screening should include a value for all 13 PAHs in the sum by using proxy values for the analyzed but undetected PAHs, and adding them to the concentrations of the detected PAHs for comparison to the total benchmark, assuming that the individual PAH has not been eliminated in accordance with the criteria at 350.71(k) of the TRRP rule. [A general discussion of proxy values appears at 350.51(n).] However, when significantly fewer than the 13 parent PAHs are determined to be COCs, or if not all of them are included in the analyte list, screening must be based on individual PAH benchmarks.

At present, the impact of COCs associated with sediments is under widespread technical review. There is an ongoing technical debate about screening values and interpretation of sediment data. The sediment benchmarks presented in this guide are based on current sources; alternative sediment benchmarks may be justified. The existing derivation

methods are based on correlations between the presence of COCs and observed sediment conditions. Developments are likely to help identify causal relationships between sediment-associated COCs and impacts to biota. Consequently, improvements to recommended methods to derive sediment benchmarks may be discussed in future revisions of this guide.

3.5.5 Soil Benchmarks

For the most part, the recommended soil benchmarks are terrestrial plant and earthworm benchmark values from Efroymson et al. (1997) and Efroymson, Will and Suter (1997). The soil benchmark values and their respective test-organism sources (plant, earthworm, or soil invertebrate) appear in Table 3.4, along with the median Texas background concentrations for most metals. If the concentration of a COC at an affected property is at or below the median Texas background or an approved site-specific background, the benchmark value may be ignored, even if the COC is bioaccumulative in soil [see 30 TAC 350.51(l–m) for a discussion of background concentrations].

The EPA has released a set of risk-based ecological soil screening levels (Eco-SSLs) for plant or soil invertebrate endpoints for 15 contaminants that are frequently of ecological concern (U.S. EPA, 2003a,b; 2005a–g; 2006; 2007a–g). Four additional Eco-SSL documents have been issued that do not contain plant or soil invertebrate endpoints (for chromium, vanadium, dieldrin, and DDT and metabolites). When an Eco-SSL is available for either plants or soil invertebrates, they have been incorporated into the soil benchmarks recommended in Table 3.4 (antimony, arsenic, barium, beryllium, cadmium, cobalt, copper, lead, manganese, nickel, PAHs, pentachlorophenol, selenium, silver, and zinc; discussions are included for aluminum and iron that do not use numeric criteria).

Eco-SSLs were derived by a work group consisting of federal, state, consulting, industry, and academic stakeholders led by the U.S. Environmental Protection Agency Office of Superfund Remediation and Technology Innovation. The EPA emphasizes that these Eco-SSLs are appropriate to all sites where certain soil parameters fall within a range of chemical and physical limits. The Eco-SSLs for plants and soil invertebrates were derived to apply to soils for which $8.5 \geq pH \geq 4.0$, and organic matter ≤ 10 percent. Based on those parameters, the EPA emphasizes that the Eco-SSLs do not apply in some situations, such as wetland soils that are regularly flooded (i.e., sediments), sewage-sludge-amended soils where the organic matter content > 10 percent, and waste types where the pH < 4.0.

The EPA is expected to continue developing Eco-SSLs issuing related documents. Risk assessors may propose to use new Eco-SSL numbers as soil benchmarks upon their release. The Eco-SSL guidance documents, standard operating procedures, and background material are available online at <www.epa.gov/ecotox/ecossl/>.

Table 3.2. Ecological benchmarks for water.

Metals, Inorganics

CAS No.	COC (mg/L)	Freshwater	Marine
7429-90-5	aluminum	0.087[a]	
7440-36-0	antimony	0.160[g]	
7440-38-2	arsenic[d]	0.150[c]	0.078[c]
7440-39-3	barium	16[b]	25[b]
7440-41-7	beryllium	0.0053[b]	
7440-42-8	boron	1.1[e,s]	
7440-43-9	cadmium[d]	0.00015[c,f]	0.00875[c]
16887-00-6	chloride	230[a]	
7782-50-5	chlorine	0.011[a]	0.0075[a]
18540-29-9	chromium (hexavalent)[d]	0.0106[c]	0.0496[c]
16065-83-1	chromium (trivalent)[d]	0.0420[c,f]	0.103[g]
7440-48-4	cobalt	1.5[b]	
7440-50-8	copper[d]	0.00524[c,f,h]	0.0036[c,h]
57-12-5	cyanide (free)	0.0107[c,i]	0.0056[c,i]
7439-89-6	iron	1[a]	
7439-92-1	lead[d]	0.00117[c,f]	0.0053[c]
7439-93-2	lithium	0.014[e]	
7439-95-4	magnesium	3.235[b]	
7439-96-5	manganese	0.120[e]	
7439-97-6	mercury	0.0013[c]	0.0011[c]
7439-98-7	molybdenum	2[b]	
7440-02-0	nickel[d]	0.0289[c,f]	0.0131[c]
7723-14-0	phosphorus (elemental)		0.0001[a]
7782-49-2	selenium	0.005[c]	0.136[c]
7440-22-4	silver[d] (as free ion) see conversion[j]	0.00008[c,k]	0.0002[c,k]
7440-22-4	silver[d]	0.0001[a,f,k]	0.00019[a,k]
7440-24-6	strontium	1.5[e]	
7783-06-4	sulfide (hydrogen sulfide)	0.002[a]	0.002[a]

CAS No.	COC (mg/L)	Freshwater	Marine
7440-28-0	thallium	0.004[g]	0.0213[g]
7440-31-5	tin	0.073[e]	
688-73-3	tributyltin (TBT)	0.000024[c]	0.0000074[c]
7440-61-1	uranium[d]	0.700[f,l]	
7440-62-2	vanadium	0.020[e]	
1314-62-1	vanadium pentoxide	0.015[b]	
7440-66-6	zinc[d]	0.0657[c,f]	0.0842[c]
7440-67-7	zirconium	0.017[e]	

Organochloride Pesticides

CAS No.	COC (mg/L)	Freshwater	Marine
309-00-2	aldrin	0.0003[c,k]	0.00013[c,k]
319-84-6	BHC (alpha)	0.074b	0.025b
319-85-7	BHC (beta)	0.083[b]	
57-74-9	chlordane	0.000004[c]	0.000004[c]
72-54-8	4,4'-DDD	0.000011[e]	0.000025[g]
72-55-9	4,4'-DDE	0.0105[g]	0.00014[g]
50-29-3	4,4'-DDT	0.000001[c]	0.000001[c]
60-57-1	dieldrin	0.000002[c]	0.000002[c]
959-98-8	endosulfan I (alpha)	0.000056[c]	0.000009[c]
33213-65-9	endosulfan II (beta)	0.000056[c]	0.000009[c]
1031-07-8	endosulfan sulfate	0.000056[c]	0.000009[c]
115-29-7	endosulfan (all isomers)	0.000051[e,m]	
72-20-8	endrin	0.000002[c]	0.000002[c]
7421-93-4	endrin aldehyde	1.21[b]	
76-44-8	heptachlor	0.000004[c]	0.000004[c]
1024-57-3	heptachlor epoxide	0.0000038[a]	0.0000036[a]
319-86-8	delta hexachlorocyclohexane (HCH)	0.141[b]	
58-89-9	gamma hexachlorocyclohexane (Lindane)	0.00008[c]	0.000016[c,k]
77-47-4	hexachlorocyclopentadiene	0.00007[b]	0.00007[g]

CAS No.	COC (mg/L)	Freshwater	Marine
72-43-5	methoxychlor	0.00003[c]	0.00003[c]
8001-35-2	toxaphene	0.0000002[c]	0.0000002[c]

Other Pesticides, Herbicides, and PCBs

CAS No.	COC (mg/L)	Freshwater	Marine
63-25-2	carbaryl (Sevin)	0.0002[c,k]	0.0613[c,k]
2921-88-2	chloropyrifos (Dursban)	0.000041[c]	0.000006[c]
8065-48-3	demeton	0.0001[c]	0.0001[c]
333-41-5	diazinon (Spectracide)	0.00017[c]	0.000819[c]
115-32-2	dicofol (Kelthane)	0.0198[c]	
330-54-1	diuron	0.070[c]	
86-50-0	guthion (azinphos-methyl)	0.00001[c]	0.00001[c]
2691-41-0	HMX (octogen)	0.150[b]	
121-75-5	malathion	0.00001[c]	0.00001[c]
2385-85-5	mirex (dechlorane)	0.000001[c]	0.000001[c]
56-38-2	parathion (ethyl)	0.000013[c]	0.000178[g]
1336-36-3	polychlorinated biphenyls (PCBs)	0.000014[c,r]	0.00003[c,r]

Polycyclic Aromatic Hydrocarbons (PAHs)

CAS No.	COC (mg/L)	Freshwater	Marine
83-32-9	acenaphthene	0.023[o]	0.0404[o]
120-12-7	anthracene	0.0003[b]	0.00018[b]
56-55-3	benz[*a*]anthracene	0.0346[b]	
50-32-8	benzo[*a*]pyrene	0.000014[e]	
218-01-9	chrysene	0.007[b]	
53-70-3	dibenz[*a*,*h*]anthracene	0.005[b]	
206-44-0	fluoranthene	0.00616[o]	0.00296[o]
86-73-7	fluorene	0.011[b]	0.05[b]
90-12-0	1-methylnaphthalene	0.0021[e]	
91-57-6	2-methylnaphthalene	0.063[b]	0.03[b]
91-20-3	naphthalene	0.25[b]	0.125[b]

CAS No.	COC (mg/L)	Freshwater	Marine
85-01-8	phenanthrene	0.030[c]	0.0046[c]
129-00-0	pyrene	0.007[b]	0.00024[b]

Other Semi-Volatiles

CAS No.	COC (mg/L)	Freshwater	Marine
35572-78-2	2-amino-4,6-dinitrotoluene	0.74[b]	
92-87-5	benzidine	1.92[b]	
65-85-0	benzoic acid	9[b]	
100-51-6	benzyl alcohol	0.0086[e]	
92-52-4	biphenyl (diphenyl)	0.014[e,m]	
111-44-4	bis (2-chloroethyl) ether	12[b]	
108-60-1	bis (chloroisopropyl) ether	6.3[b]	
117-81-7	bis (2-ethyl-hexyl) phthalate	0.3[a,r]	
117-84-0	bis (*n*-octyl) phthalate	0.022[b]	
101-55-3	4-bromophenyl phenyl ether	0.0015[e,m]	
85-68-7	butylbenzyl phthalate	0.093[b]	0.147[b]
91-58-7	2-chloronapthalene	0.054[b]	
95-57-8	2-chlorophenol	0.13[b]	0.265[b]
124-18-5	decane	0.049[e]	
84-74-2	di-*n*-butyl phthalate	0.007[b]	0.005[b]
132-64-9	dibenzofuran	0.094[b]	0.065[b]
91-94-1	3,3-dichlorobenzidine	0.053[b]	0.037[b]
120-83-2	2,4-dichlorophenol	0.085[b]	
84-66-2	diethyl phthalate	1.043[b]	0.442[b]
131-11-3	dimethyl phthalate	0.330[g]	0.580[g]
105-67-9	2,4-dimethylphenol	0.105[b]	
99-65-0	1,3-dinitrobenzene	0.072[b]	
51-28-5	2,4-dinitrophenol	0.031[b]	0.67[b]
121-14-2	2,4-dinitrotoluene	1.22[b]	
122-66-7	1,2-diphenylhydrazine	0.012[b]	

CAS No.	COC (mg/L)	Freshwater	Marine
27176-87-0	dodecyl benzenesulfonic acid		0.058[b]
78-59-1	isophorone	6[b]	0.65[b]
1634-04-4	methyl tert-butyl ether (MTBE)	51[t]	18[t]
99-08-1	1-methyl-3-nitrobenzene (3-nitrotoluene)	0.375[b]	
59-50-7	3-methyl-4-chlorophenol	0.0003[g]	
534-52-1	2-methyl-4,6-dinitrophenol (dinitro-o-cresol)	0.012[b]	
106-44-5	4-methylphenol (p-cresol)	0.272[b]	
95-48-7	2-methylphenol (o-cresol)	0.56[b]	0.51[b]
62-75-9	*n*-nitrosodimethylamine	47[b]	165[b]
86-30-6	*n*-nitrosodiphenylamine	0.29[b]	165[b]
55-63-0	nitroglycerin	0.069[b]	
88-75-5	2-nitrophenol	0.959[b]	1.47[b]
100-02-7	4-nitrophenol	0.532[b]	0.359[b]
88-72-7	2-nitrotoluene	0.44[b]	
99-99-0	4-nitrotoluene (1-methyl,4-nitrobenzene)	0.95[b]	
621-64-7	nitrosodipropylamine	0.02[b]	0.12[b]
84852-15-3 and 25154-52-3	nonylphenol	0.0066[c]	0.0017[c]
111-13-7	2-octanone (methyl hexyl ketone)	0.0083[e]	
608-93-5	pentachlorobenzene	0.00047[e,m]	0.129[g]
87-86-5	pentachlorophenol	0.00245[c,p]	0.0096[c]
78-11-5	PETN (pentaerythrite-tetranitrate)	425[b]	
108-95-2	phenol	0.110[n]	2.75[b]
121-82-4	RDX (Cyclonite) (cyclo-trimethylenetrinitramine)	0.18[b]	
95-94-3	1,2,4,5-tetrachlorobenzene	0.05[g]	0.129[g]
95-95-4	2,4,5-trichlorophenol	0.064[c]	0.012[c]
88-06-2	2,4,6-trichlorophenol	0.0135[b]	0.061[b]
118-96-7	2,4,6-trinitrotoluene (TNT)	0.050[b]	0.050[b]

Volatiles

CAS No.	COC (mg/L)	Freshwater	Marine
67-64-1	acetone	101.2[b]	282[b]
107-02-8	acrolein (acrylic aldehyde)	0.0003[a]	0.005[b]
107-13-1	acrylonitrile	0.076[g]	0.291[b]
71-43-2	benzene	0.130[e]	0.109[g]
104-51-8	*n*-butylbenzene	0.036[b]	
103-65-1	propyl benzene	0.064[b]	
135-98-8	sec-butylbenzene	0.041[b]	
98-06-6	tert-butylbenzene	0.048[b]	
75-27-4	bromodichloromethane	2.16[b]	
78-93-3	2-butanone (mek)	42.4[b]	
75-15-0	carbon disulfide	0.105[b]	
56-23-5	carbon tetrachloride (tetrachloromethane)	0.0098[e]	1.5[g]
108-90-7	chlorobenzene	0.064[e]	0.105[g]
124-48-1	chlorodibromomethane (dibromochloromethane)	0.129[b]	
67-66-3	chloroform	0.89[q]	4.1[q]
74-87-3	chloromethane	28[b]	13.5[b]
98-82-8	cumene (isopropylbenzene)	0.255[b]	
99-87-6	cymene (4-isopropyltoluene)	0.042[b]	
95-50-1	1,2-dichlorobenzene	0.11[b]	0.099[b]
541-73-1	1,3-dichlorobenzene	0.085[b]	0.142[b]
106-46-7	1,4-dichlorobenzene	0.11[b]	0.099[b]
75-71-8	dichlorodifluoromethane (Freon-12)	1.96[b]	
75-34-3	1,1-dichloroethane	2.57[q]	
107-06-2	1,2-dichloroethane	6.3[q]	5.65[q]
75-35-4	1,1-dichloroethene	1.5[q]	12.5[q]
540-59-0	1,2-dichloroethene (mixed cis and trans isomers)	14[q]	0.68[q]
156-60-5	1,2-dichloroethene (trans)	22[q]	
78-87-5	1,2-dichloropropane	1.87[b]	2.4[g]
542-75-6	1,3-dichloropropene	0.205[b]	0.040[b]

CAS No.	COC (mg/L)	Freshwater	Marine
100-41-4	ethyl benzene	1.09[b]	0.249[b]
87-68-3	HCBD (hexachlorobutadiene)	0.00093[g]	0.00032[g]
67-72-1	hexachloroethane	0.012[e,m]	0.0094[g]
110-54-3	hexane	0.00058[e]	
591-78-6	2-hexanone (methyl butyl ketone; MBK)	6.13[b]	
108-10-1	4-methyl-2-pentanone (MIBK)	26.4[b]	61.5[b]
74-83-9	methyl bromide (bromomethane)	0.11[b]	0.6[b]
593-74-8	methylmercury (dimethylmercury)	0.0000028[e]	
80-62-6	methyl methacrylate	11.6[b]	
75-09-2	methylene chloride (dichloromethane)	11[q]	5.42[q]
98-95-3	nitrobenzene	0.27[g]	0.0668[g]
71-41-0	1-pentanol	0.110[e]	
67-63-0	2-propanol	0.0075[e]	
100-42-5	styrene (vinyl benzene)	1.25[b]	0.455[b]
79-34-5	1,1,2,2-tetrachloroethane	0.465[b]	0.451[b]
127-18-4	tetrachloroethene	0.79[q]	1.45[q]
108-88-3	toluene	1.45[q]	0.48[q]
75-25-2	tribromomethane (bromoform)	0.149[b]	1.22[b]
120-82-1	1,2,4-trichlorobenzene	0.051[b]	0.022[b]
71-55-6	1,1,1-trichloroethane	2.45[q]	1.56[q]
79-00-5	1,1,2-trichloroethane	0.90[b]	0.275[b]
79-01-6	trichloroethene	0.55[b]	0.97[q]
75-69-4	trichlorofluormethane (Freon-11)	0.871[b]	
76-13-1	1,1,2-trichlorotrifluorethane (Freon-113)	0.207[b]	
95-63-6	1,2,4-trimethylbenzene (Pseudocumene)	0.077[b]	0.217[b]
108-67-8	1,3,5-trimethylbenzene (Mesitylene)	0.071[b]	
108-05-4	vinyl acetate	0.016[e]	
75-01-4	vinyl chloride (chloroethylene)	2.82[b]	
108-38-3	*m*-xylene	0.0018[e,m]	

CAS No.	COC (mg/L)	Freshwater	Marine
1330-20-7	xylenes	1.34[q]	0.85[q]

[a] U.S. EPA, 2013.

[b] In-house water quality chronic values derived for wastewater permits and requests from the Office of Waste based on LC_{50} values in accordance with methodology defined in the TSWQS (Water Quality Division). Benchmark for bis(2-ethylhexyl) phthalate set at solubility limit given at 30 TAC 350.73(f).

[c] Texas Surface Water Quality Standards chronic (unless otherwise noted) criteria (30 TAC 307.6, Table 1).

[d] Indicates that the criteria for a specific parameter are for the dissolved portion in water.

[e] Tier II Secondary Chronic Values from Suter and Tsao (1996).

[f] Criteria calculated using a hardness value of 50 mg/L. See formula following footnotes. Resulting values using this formula will be in µg/L and will need to be divided by 1000 to correspond with the table values.

[g] U.S. EPA Region 4. 2001. Value derived from Region 4 Water Quality Management Division screening worksheet.

[h] In designated oyster waters an acute saltwater copper criterion of 3.6 micrograms per liter applies outside of the mixing zone of permitted discharges, and specified mixing zones for copper will not encompass oyster reefs containing live oysters.

[i] Compliance will be determined using the analytical method for available cyanide.

[j] Based on the procedure defined in TCEQ (2011), the percent dissolved silver that is in the free ionic form is estimated from the following regression equation:

$$Y = e^{e^{\left(\frac{1}{0.6659+0.0044Cl}\right)}}$$

where Y = percent of dissolved silver that is in free ionic form, and Cl = dissolved chloride concentration (mg/L). Persons should use the 50th percentile chloride value (from TCEQ, 2011) for the nearest downstream segment unless site-specific data is available. Because there is no readily available means to predict the percent free ion in marine waters, silver should be evaluated as dissolved silver alone.

[k] There is only an acute criterion (no chronic criterion). The indicated value is the acute criterion divided by 10.

[l] State of Colorado hardness-based water quality standard (Colorado Department of Public Health and Environment, 2013).

[m] Values calculated for OSWER 1996 as in Suter and Tsao (1996).

[n] Value calculated using Great Lakes Water Quality Initiative Tier I methodology (U.S. EPA, 1993a) as in Suter and Tsao (1996).

[o] These numbers are FCVs calculated by the EPA for use in the derivation of the sediment quality criteria (U.S. EPA, 1993a,h).

[p] Criteria calculated using a pH of 6.0. See formula following footnotes.

[q] Value derived by work group using the LC50 approach discussed in 3.5.1.1. Contact the TCEQ Technical Program Support Team (Remediation Division) for a full discussion of each value.

[r] Criteria apply to the sum of the congeners, isomers, homologs or Aroclor analysis.

[s] Value recalculated due to units in error in cited document.

[t] Mancini et al., 2002.

Cadmium: $1.101672 - [\ln(\text{hardness})](0.041838))*(we^{(0.7409[(\ln(\text{hardness})] - 4.719)})$

Chromium (Tri): $0.860we^{(0.8190(\ln(\text{hardness})) + 0.6848)}$

Copper: $0.960me^{(0.8545(\ln(\text{hardness})) - 1.6463)}$

Lead: $(1.46203 - [\ln(\text{hardness})](0.145712))*(we^{(1.273(\ln(\text{hardness})) - 4.705)})$

Nickel: $0.997we^{(0.8460(\ln(\text{hardness})) + 0.0584)}$

Uranium: $e^{(1.1021(\ln(\text{hardness})) + 2.2382)}$

Zinc: $0.986we^{(0.8473(\ln(\text{hardness})) + 0.884)}$

Pentachlorophenol: $e^{(1.005(\text{pH}) - 5.134)}$

The “m” in the preceding formulas indicates that a criterion may be multiplied by a WER or a biotic ligand model result in order to incorporate effects of local water chemistry on toxicity. The multiplier is equal to 1 except where sufficient data is available to establish a site-specific multiplier. The number preceding the m in the freshwater equation is an EPA conversion factor.

The “w” in the preceding formulas indicates that the criterion is multiplied by a water-effects ratio in order to incorporate the effects of local water chemistry on toxicity. The water-effects ratio is equal to 1 except where sufficient data is available to establish a site-specific, water-effects ratio. Water-effects ratios for individual water bodies are listed in Appendix E of the TSWQS. The number preceding the “w” in the freshwater criterion equation is a U.S. EPA conversion factor.

Table 3.3. Ecological benchmarks for sediment.

Inorganics

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
7440-36-0	antimony	2[a]	
7440-38-2	arsenic	9.79	8.2
7440-43-9	cadmium	0.99	1.2
7440-47-3	chromium	43.4	81
7440-48-4	cobalt	50b	
7440-50-8	copper	31.6	34
7439-89-6	iron	20,000[b]	
7439-92-1	lead	35.8	46.7
7439-96-5	manganese	460[b]	
7439-97-6	mercury	0.18	0.15
7440-02-0	nickel	22.7	20.9
7440-22-4	silver	1[a]	1
7440-66-6	zinc	121	150

Polycyclic Aromatic Hydrocarbons[h]

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
83-32-9	acenaphthene	0.0067	0.016
208-96-8	acenaphthylene	0.0059[i]	0.044
120-12-7	anthracene	0.0572	0.0853
56-55-3	benz[*a*]anthracene	0.108	0.261
50-32-8	benzo[*a*]pyrene	0.150	0.43
218-01-9	chrysene	0.166	0.384
53-70-3	dibenz[*a,h*]anthracene	0.033	0.0634
206-44-0	fluoranthene	0.423	0.6
86-73-7	fluorene	0.0774	0.019
91-57-6	2-methyl naphthalene		0.070
91-20-3	naphthalene	0.176	0.160
85-01-8	phenanthrene	0.204	0.24

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
129-00-0	pyrene	0.195	0.665
	low-molecular-weight PAHs		0.552[d,e]
	high-molecular-weight PAHs		1.7[d,f]
	total PAHs	1.61[d,g,h]	4.022[d,g,h]

Chlorinated Pesticides, PCBs, Benzenes

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
309-00-2	Aldrin	0.002[b]	
27323-18-8	Aroclor 1254	0.060[b]	
12674-11-2	Aroclor 1016	0.007[b]	
11096-82-5	Aroclor 1260	0.005[b]	
12672-29-6	Aroclor 1248	0.030[b]	
319-84-6	alpha-BHC	0.006[b]	
319-85-7	beta-BHC	0.005[b]	
58-89-9	gamma-BHC (Lindane)	0.00237	0.00032[c]
608-73-1	BHC	0.003[b,d]	
57-74-9	Chlordane (Total)	0.00324	0.00226[c]
60-57-1	Dieldrin	0.0019	0.000715[c]
72-20-8	Endrin	0.00222	
118-74-1	HCB (Hexachlorobenzene)	0.020[b]	
1024-57-3	heptachlor epoxide	0.00247	
2385-5	Mirex	0.007[b]	
72-55-9	sum DDE	0.00316[d]	0.00207[c,d]
72-54-8	sum DDD	0.00488[d]	0.00122[c,d]
50-29-3	sum DDT	0.00416[d]	0.00119[c,d]
	total DDT	0.00528[d]	0.00158[c,d]
1336-36-3	total PCBs	0.0598[d]	0.0227[d]

Organophosphate Pesticides

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
2642-71-9	Azinphos—ethyl	0.000018[k]	

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
86-50-0	Azinphos—methyl (Guthion)	0.000062[k]	
331-41-5	Diazinon	0.00038[k]	
121-75-5	Malathion	0.00067[l]	
	Phoxim	0.000060[k]	
	Pyrazophos	0.000015[k]	

Other Pesticides

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
8001-35-2	Toxaphene	0.00010[j]	
1912-24-9	Atrazine	0.00030[k]	
122-34-9	Simazine	0.00034[k]	

Phthalates

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
117-81-7	bis(2-ethyl-hexyl)phthalate		0.18[c]
84-66-2	diethyl phthalate	0.630[l]	

Volatiles [Footnote (m) applies to all listed volatiles, unless otherwise noted.]

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
67-64-1	acetone	60.03	167.23
107-13-1	acrylonitrile	0.05	0.17
71-43-2	benzene	0.16	0.14
104-51-8	*n*-butylbenzene	1.09	
103-65-1	propyl benzene	0.72	
135-98-8	sec-butylbenzene	0.88	
98-06-6	tert-butylbenzene	1.21	
75-27-4	bromodichloromethane	2.46	
78-93-3	2-butanone	25.71	
75-15-0	carbon disulfide	0.12	
56-23-5	carbon tetrachloride	0.02	3.67
108-90-7	chlorobenzene	0.17	0.29

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
124-48-1	chlorodibromomethane	0.16	
67-66-3	chloroform (trichloromethane)	0.94	4.30
74-87-3	chloromethane	17.80	8.74
98-82-8	cumene	8.99	
99-87-6	*p*-cymene	1.00	
95-50-1	1,2-dichlorobenzene	0.83	0.74
541-73-1	1,3-dichlorobenzene	0.19	0.32
106-46-7	1,4-dichlorobenzene	0.77	0.70
75-71-8	dichlorodifluoromethane	3.68	
75-34-3	1,1-dicloroethane	2.32	
107-06-2	1,2-dichloroethane	4.79	4.30
75-35-4	1,1-dichloroethene	1.87	15.41
156-60-5	1,2-dichloroethene (trans)	23.95	
78-87-5	1,2-dichloropropane	2.20	2.82
542-75-6	1,3-dichloropropene	0.23	0.04
100-41-4	ethylbenzene	2.86	0.65
87-68-3	HCBD (hexachlorobutadiene)	0.055[j]	0.02
67-72-1	hexachloroethane	0.23	0.18
100-54-3	hexane, *n*-	0.0031	
591-78-6	2-hexanone	4.70	
108-10-1	4-methyl-2-pentanone (MIBK)	19.43	45.34
74-83-9	methyl bromide	0.08	0.42
22967-92-6	methylmercury	0.0000017	
80-62-6	methyl methacrylate	9.50	
75-09-2	methylene chloride	7.75	3.82
98-95-3	nitrobenzene	0.51	0.13
71-41-0	1-pentanol	0.07	
67-63-0	2-propanol	0.0047	
100-42-5	styrene	10.24	3.72

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
79-34-5	1,1,2,2-tetrachloroethane	0.63	0.61
127-18-4	tetrachloroethane	1.69	3.10
108-88-3	toluene	2.88	0.94
75-25-2	bromoform	0.22	1.78
120-82-1	1,2,4-trichlorobenzene	0.88	0.39
71-55-6	1,1,1-trichloroethane	4.13	2.63
79-00-5	1,1,2-trichloroethane	0.98	0.30
79-01-6	trichloroethene	0.84	1.47
75-69-4	trichlorofloromethane	1.69	
76-13-1	1,1,2-trichlorotrifluoroethane	2.78	
95-63-6	1,2,4-trimethylbenzene	0.76	2.16
108-67-8	1,3,5-trimethylbenzene	0.77	
108-05-4	vinyl acetate	0.010	
75-01-4	vinyl chloride	1.96	
108-38-3	*m*-xylene	0.0046	
1330-20-7	xylenes	4.00	2.54

Freshwater—Unless otherwise noted, benchmarks are threshold effect concentrations (TECs) from MacDonald, Ingersoll, and Berger (2000): 20–31.

Marine—Unless otherwise noted, benchmarks are effects range low (ERL) from E.R. Long, MacDonald, Smith, and Calder (1995): 81–97.

[a] ERL from Long and Morgan (1990).

[b] Lowest effects level (LEL) from D. Persaud, Jaagumagi, and Hayton (1993).

[c] Threshold effects level (TEL) from Smith et al. (1996b).

[d] When benchmarks represent the sum of individual compounds, isomers, or groups of congeners, and the chemical analysis indicates an undetected value, the proxy value specified at 350.51(n) shall be used for calculating the sum of the respective compounds, isomers, or congeners. This assumes that the particular COC has not been eliminated in accordance with the criteria at 350.71(k).

[e] The low-molecular-weight PAH benchmark is to be compared to the sum of the concentrations of the following compounds: naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, and 2-methyl napthalene. The PAH benchmark is not the sum of the corresponding benchmarks listed for the individual compounds.

[f] The high-molecular-weight PAH benchmark is to be compared to the sum of the concentrations of the following compounds: fluoranthene, pyrene, benz[**a**]anthracene, chrysene, benzo[**a**]pyrene, and dibenzo[**a,h**]anthracene. The PAH benchmark is not the sum of the corresponding benchmarks listed for the individual compounds.

[g] Total PAH refers to the sum of the concentrations of each of low- and high-molecular-weight PAHs listed above and any other PAH compounds that are not eliminated in accordance with 350.71(k).

[h] Applies to all PAHs listed. The benchmarks for total PAHs are the most relevant in evaluating risk in an ERA as PAHs almost always occur as mixtures. Values for individual and low- and high-molecular-weight PAHs are provided as guidelines to aid in the determination of disproportionate concentrations within the mixture that may be masked by the total. See discussion in **3.5.4**.

[i] CCME (2002).

[j] NYSDEC (1999).

[k] Storteldder, Vandergaag, and van der Kooij (1989).

[l] U.S. EPA (1997b).

[m] Benchmarks derived using formula in P.C. Fuchsman (2003): 1532–34. TCEQ Surface water values from Table 3.2 were used for water quality values. **Determining PCLs for Suface Water and Sediment** (TCEQ publication RG366/TRRP-24) default values of 1 percent fraction organic carbon and 0.37 porosity were used. The person should adjust these values if sufficient site-specific data indicate they are not representative.

Table 3.4. Ecological benchmarks for soil.

Metals, Inorganics

CAS No.	Constituent (mg/kg dry wt.)	Earthworms[a]	Plants[b]	Median Background[c]
7429-90-5	aluminum[d]			30,000
7440-36-0	antimony	78[e,f]	5	1
7440-38-2	arsenic	60	18[m]	5.9
7440-39-3	barium	330[f,h]	500	300
7440-41-7	beryllium	40[f,i]	10	1.5
7440-42-8	boron		0.5	30
7726-95-6	bromine		10	
7440-43-9	cadmium	140[f,j]	32[j]	
7440-47-3	chromium	0.4	1	30
7440-48-4	cobalt		13[k]	7
7440-50-8	copper	80[f, g]	70g	15
7782-41-4	fluorine		200	190
7553-56-2	iodine		4	
7439-89-6	iron[l]			15,000
7439-92-1	lead	1700[f,n]	120[n]	15
7439-93-2	lithium		2	
7439-96-5	manganese	450[f,p]	220[p]	300
7439-97-6	mercury	0.1	0.3	0.04
7439-98-7	molybdenum		2	
7440-02-0	nickel	280[f,q]	38[q]	10
7782-49-2	selenium	4.1[f,r]	0.52[r]	0.3
7440-22-4	silver		560[s]	
7440-24-6	strontium			100
7440-26-8	technetium		0.2	
7440-28-0	thallium		1	
7440-29-1	thorium			9.3
7440-31-5	tin		50	0.9

CAS No.	Constituent (mg/kg dry wt.)	Earthworms[a]	Plants[b]	Median Background[c]
7440-32-6	titanium			2,000
7440-61-1	uranium		5	
7440-62-2	vanadium		2	50
7440-66-6	zinc	120[f, t]	160[t]	30

Polycyclic Aromatic Hydrocarbons (PAHs)

CAS No.	Constituent (mg/kg dry wt.)	Earthworms[a]	Plants[b]	Median Background[c]
83-32-9	acenaphthene		20	
86-73-7	fluorene	30		
	low-molecular-weight PAHs	29[f,u]		
	high-molecular-weight PAHs	18[f,u]		

Semi-Volatiles

CAS No.	Constituent (mg/kg dry wt.)	Earthworms[a]	Plants[b]	Median Background[c]
92-52-4	biphenyl (diphenyl)		60	
108-43-0	3-chlorophenol	10	7	
106-46-7	1,4-dichlorobenzene	20		
95-77-2	3,4-dichlorophenol	20	20	
84-74-2	di-*n*-butyl phthalate		200	
84-66-2	diethyl phthalate		100	
131-11-3	dimethylphthalate	200		
51-28-5	2,4-dinitrophenol		20	
100-02-7	4-nitrophenol	7		
86-30-6	*n*-nitrosodiphenylamine	20		
527-20-8	pentachloroaniline	100		
608-93-5	pentachlorobenzene	20		
87-86-5	pentachlorophenol	31[f,o]	5[o]	
108-95-2	phenol	30	70	
634-66-2	1,2,3,4-tetrachlorobenzene	10		
4901-51-3	2,3,4,5-tetrachlorophenol	20		
95-95-4	2,4,5-trichlorophenol	9	4	
88-06-2	2,4,6-trichlorophenol	10		

Volatiles

CAS No.	Constituent (mg/kg dry wt.)	Earthworms[a]	Plants[b]	Median Background[c]
108-90-7	chlorobenzene	40		
78-87-5	1,2-dichloropropane	700		
98-95-3	nitrobenzene	40		
100-42-5	styrene		300	
108-88-3	toluene		200	
87-61-6	1,2,3-trichlorobenzene	20		
120-82-1	1,2,4-trichlorobenzene	20		

Other Organics, PCBs, Pesticides

CAS No.	Constituent (mg/kg dry wt.)	Earthworms[a]	Plants[b]	Median Background[c]
1336-36-3	polychlorinated biphenyls (PCBs)		40	
108-42-9	3-chloroaniline	30	20	
554-00-7	2,4-dichloroaniline	100		
95-76-1	3,4-dichloroaniline	20		
77-47-4	hexachlorocyclopentadiene		10	
636-30-6	2,4,5-trichloroaniline	20	20	
3481--20-7	2,3,5,6-tetrachloroaniline	20	20	
79-07-2	chloroacetamide	2		
110-00-9	furan		600	

[a] Efroymson, Will, and Suter (1997).

[b] Efroymson et al. (1997)

[c] Texas-specific median background concentration. 30 TAC 350.51(m).

[d] Potential ecological risks associated with aluminum in soils are based on the measured soil pH. Where aluminum is a COC, it should only be retained for those soils with a pH less than 5.5. U.S. EPA. (2003a).

[e] U.S. EPA (2005a).

[f] Screening values for soil invertebrates.

[g] U.S. EPA (2007a).

[h] U.S. EPA (2005b).

[i] U.S. EPA (2005c).

[j] U.S. EPA (2005e).

[k] U.S. EPA (2005f).

[l] Iron is not expected to be toxic to plants in well-aerated soils between pH 5 and 8. Iron's relative importance is based less on its direct chemical toxicity than on its effect as a mediator in the

geochemistry of other potentially toxic metals and the potential hazard of depositing flocculent. U.S. EPA (2003b).

[m]U.S. EPA (2005d).

[n] U.S. EPA (2005g).

[o] U.S. EPA (2007d).

[p] U.S. EPA (2007c).

[q] U.S. EPA (2007b).

[r] U.S. EPA (2007g).

[s] U.S. EPA (2006).

[t] U.S. EPA (2007f).

[u] U.S. EPA (2007e).

3.6 Communities, Feeding Guilds, and Representative Species

Under the TRRP rule [350.77(c)(2)], the second required element of the Tier 2 SLERA is the identification of communities, feeding guilds, and representative species that might be supported by habitats on the affected property. Each of these ecological groupings is discussed in the following sections, much of which was adapted from U.S. EPA (1999a). In addition, the roles that these groups play in the selection of assessment endpoints and measures of effect (measurement endpoints) are discussed, as well as what ecological receptors the commission is trying to protect under the definition of *ecological PCL* [see 350.4(a)(27)].

3.6.1 Communities

Ecological communities are collections of plant and animal populations occupying the same habitat in which the various species interact with one another. However, in this publication, *communities* usually refers to those groups whose exposure to COCs can be evaluated in terms of the media in which they reside. These communities consist of soil invertebrates, terrestrial vegetation, benthic invertebrates, water-column invertebrates, algae, and rooted aquatic vegetation. COCs that exceed ecological (community-level) benchmarks but that do not subsequently prove to be a risk to higher-trophic-level receptors may still harm these community-level receptors. Depending on the site-specific circumstances, the person may be required to demonstrate that impacts to these communities will not result in unacceptable consequences for the mobile or wide-ranging receptors. Generally, it is not anticipated that exceedances of benchmarks will result in the development of COC-specific PCLs for communities, with the exception of benthic invertebrates, as discussed below.

By definition, ecological PCLs are primarily intended to be protective for more mobile or wide-ranging ecological receptors and, where appropriate, benthic invertebrate communities within Texas waters. Although benthic invertebrates play a critical role at the bottom of the aquatic food chain, as a critical pathway for the transfer of energy and nutrients to higher trophic level organisms, the TCEQ recognizes that the benthic community in some water bodies may be diminished for reasons unrelated to releases of COCs from property subject to the TRRP regulation. The bullet points that follow indicate where the agency believes it unnecessary to determine an ecological PCL for sediment that is protective of the benthic invertebrate community. This does not preclude an evaluation of risks to higher-trophic-level organisms that may forage in these types of water bodies or nearby bodies (that could see impacts from sediment COC transport). Nor does it preclude the TCEQ from requiring additional evaluations at these types of locations, case by case, where significant exposure conditions warrant (e.g., acutely toxic concentrations, presence of free product).

- Routinely dredged water bodies. This applies to the portion of the channel that is actually dredged every three years or more often. Risks to benthic communities potentially exposed to COCs in sediments not routinely dredged (such as significant areas of shallow waters near the banks that are not used for shipping) should be evaluated where the exposure pathway is complete (see discussion below).
- Intermittent streams (that dry up completely at least one week a year) without perennial pools. [See the definition at 30 TAC 307.3(a).]
- Water bodies with concrete-lined channels (bottom and sides).
- Segments 1006 and 1007 of the Houston Ship Channel (see 30 TAC 307.10, Appendix C), excluding their tidal tributaries.
- Classified and unclassified water bodies with no aquatic-life use, as indicated in 30 TAC 307.10, Appendixes A and D.

Benthic recovery in dredged areas is rapid, requiring only a few months to a year (Swartz et al., 1980 and Van Dolah et al., 1984). Hence, three years was selected as a conservative estimate of benthic recovery. For federal projects, the district offices of the U.S. Army Corps of Engineers can give the frequency of dredging for a given reach of a water body. Where appropriate, the person should evaluate the possibility of COC transport and potential impact on benthic communities downstream or downgradient of the types of water bodies indicated above. For water bodies not discussed above, or those documented to be scheduled for dredging within three years of APAR submission, the person may evaluate the suitability of a sediment PCL protective of benthic invertebrates as part of the uncertainty analysis discussed in **3.12**. This would only be required for sediment COCs that have not been eliminated from a Tier 2 SLERA.

3.6.2 Feeding Guilds and Food Webs

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

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

Habitat-specific food webs are developed for use in the ecological risk assessment to:

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

Subsection **3.6.5** presents example food webs for the major habitats found in Texas, identifying the communities and major feeding guilds within them. Users may rely upon these food webs to help them identify the appropriate communities and feeding guilds associated with a particular habitat, or they may develop their own, as discussed in **3.6.4**.

3.6.3 Representative Species

The TRRP rule defines "selected ecological receptors" as species that are to be carried through the ERA as representatives of the different feeding guilds and communities that are being evaluated. These representative species, often called "measurement receptors," may not actually be present at the affected property, but may be used to represent those within the feeding guild or community that may feed on the affected property. Representative ecological receptors should be chosen to ensure that the potentially complete exposure pathways to the associated ecological feeding guilds are included in the conceptual model. As discussed in greater detail in **3.6.7.2**, the selection of representative species or measurement receptors should be based on several factors, including: (1) ecological relevance, (2) potential for COC exposure, (3) sensitivity to the COCs, (4) social or economic importance, (5) species known or expected to be present, and (6) availability of natural-history information.

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

Government Organizations—Texas Biological and Conservation Data Systems (Texas Parks and Wildlife Department) and the U.S. Fish and Wildlife Service (National Wetland Inventory Maps at <www.fws.gov/wetlands>) give maps or lists of species based on geographic location and helpful identify threatened or endangered species or areas of special concern.

General Literature (field guides)—Examples of information describing the flora and fauna of Texas that are useful in the development of habitat-specific food webs (see **3.6.5**) include McMahan et al., 1984; Garrett and Barker, 1987; Peterson, 1988; Davis and Schmidly, 1994; and Tennant, 1998.

Private or Local Organizations—the National Audubon Society, the National Geographic Society, local wildlife clubs, state and national park systems, and universities.

3.6.4 Food-Web Development

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

The first step in developing a habitat-specific food web is to identify—based on the dietary habits and feeding strategies of receptors compiled in **3.6.3**—the major feeding guilds for birds, mammals, reptiles, amphibians, and fish. Invertebrates and plants are not assigned to guilds, but rather grouped into their respective community by the environmental media they inhabit, because the risk to these groups is characterized differently (see **3.2.2, 3.9.2**).

Once the major feeding guilds are identified (e.g., herbivore, omnivore, carnivore, insectivore), receptors should be grouped by class (e.g., mammals, birds, amphibians, and reptiles. As noted, invertebrates and plants are grouped into their respective community by the media they inhabit (i.e, soil invertebrates, terrestrial vegetation, benthic invertebrates, water-column invertebrates, algae, and rooted aquatic vegetation).

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

After species have been grouped into the appropriate feeding guilds and communities and organized by trophic level, dietary relationships between guilds and communities should be defined, based on dietary relationships, by evaluating the diet of the receptors for each guild and community.

Although most organisms have a complex diet, most of it is usually composed of a limited number of prey items and, therefore, a limited number of feeding-guild interactions occurs. For that reason, the EPA's Office of Solid Waste recommends that generally only those interactions that contribute more than 5 percent of the total diet should be considered for development of a food web (U.S. EPA, 1999a), based on the assumption that the food web can be simplified without underestimating exposure. The 5 percent cutoff only applies to prey items, not incidental ingestion of media.

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

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

3.6.5 Example Habitat-Specific Food Webs

To better illustrate food-web development as discussed in the previous sections, seven habitat-specific example food webs are presented as Figures 3.5–11. The habitats represented include:

- upland forest
- tallgrass prairie
- shortgrass prairie
- shrub-scrub
- desert-arid
- freshwater-wetland
- estuarine-wetland

These example food webs are based on information describing the flora and fauna of Texas (see references under "General Literature" in **3.6.3**). Supplemental information was also collected from the EPA's Wildlife Exposure Factors Handbook (1993i).

Although these food webs look complex, users need not evaluate risk to all feeding guilds within a web if there is a reasonable justification. For example, PCBs tend to bioaccumulate and biomagnify in food chains and are not taken up by most plants, but are accumulated by soil invertebrates. Thus, mammalian herbivores would not be exposed to PCBs in their diet, but may intake PCBs through incidental ingestion of soil attached to plants. However, this intake would be far less than that of mammalian insectivores (provided that the herbivore was not fossorial), which would be exposed both through their diet and through incidental soil ingestion. This rationale

emphasizes the importance of evaluating the mammalian-insectivore guild for PCB effects and serves as a reasonable justification for not evaluating the herbivorous-mammal guild (U.S. EPA, 1997a).

3.6.6 Selecting and Evaluating Assessment Endpoints
(modified from U.S. EPA, 1999a)

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

Generally, assessment endpoints (and measurement endpoints and measurement receptors) should be developed and selected considering the definition of ecological PCL (i.e., limited to more mobile or wide-ranging species and benthic invertebrates, where appropriate). Chemical toxicity resulting from direct exposure to a COC through media contact, or from indirect exposure through ingestion of plants or prey which have accumulated the COC, is the first and most common basis for developing an ecological PCL. In a Tier 2 SLERA, the potential for chemical toxicity is evaluated by estimating the total direct and indirect exposure of each measurement receptor to the COC and deriving a hazard quotient (based on an NOAEL or a LOAEL). The goal is to determine if the ecological risk requires further evaluation (i.e., a hazard quotient > 1) and to develop ecological PCLs for the affected medium that will protect against chemical toxicity.

However, depending on the circumstances, the person may be required to consider impacts to receptors with limited mobility or range (e.g., plants, soil invertebrates, and small rodents) by evaluating whether loss of habitat or reduced energy transfer might affect the more mobile or wide-ranging receptors that may depend on the lower trophic levels for habitat and food. In these cases, protection of the viability and presence of the more mobile or wide-ranging receptor remains the assessment endpoint, but these lower-trophic-level species would be the measurement receptors, and chronic NOAELs or LOAELs for the populations of the lower-trophic-level species would be used as the measurement endpoints (see **3.6.7**). For example, a brine spill is unlikely to cause chemical toxicity in a hawk population. The residual brine could, however, harm vegetation and rodents and so reduce the habitat or food available to the local hawks. Whether this could lead to any significant adverse impact on the hawk population depends on several

factors, particularly (1) the size of the affected property, (2) the overall area used by the hawk population (e.g., its home range), (3) the response of the habitat and food organisms to the chemical at a population or community level, and (4) the duration and availability of toxic residuals. This conceptual model is complicated, however, by the influence of factors not dependent on the affected property or any COCs present.[6] Factors completely unrelated to chemical contamination at a site may have more significant impacts on habitat use or energy transfer at any particular site (e.g., the loss of habitat due to redevelopment or the intrusion of an invasive species that alters the make up of the biological community and its function). Thus, impacts to habitat or energy transfer should not be a routine consideration in developing ecological PCLs.

Examples of potential ecological attributes are provided below to illustrate the interdependent nature of various components of the ecosystem. Numerous components of a habitat-specific food web provide critical food sources and habitats for the wildlife that constitute the assessment endpoints used to determine ecological PCLs. If terrestrial plants, soil invertebrates, or small mammals are significantly harmed, then there may be a resultant impact on the wildlife community. Consequently, elevated COC concentrations over substantial areas, particularly in regions that are undeveloped or particularly attractive to wildlife, could result in a disruption of this ecosystem or unacceptable consequences to wider ranging receptors. Under these circumstances, development of PCLs derived to protect terrestrial plants, soil invertebrates, or small rodents could be warranted. Unless threatened or endangered species are involved, PCLs would not normally be developed for these attributes concerning habitat or energy transfer, however.

Examples of critical ecological attributes for feeding guilds include:

- seed dispersal (by birds and mammals)[7]
- major source of food (for predators)
- natural selection (removal of the weak and abnormal from the gene pool)
- pollination (cross-fertilization of plants by animals)
- regulation of prey populations (e.g., small rodents)

[6] For example, the availability of suitable habitat and food outside the affected media, or other factors—such as a built environment, access corridors, or general suitability of the general habitat—may determine whether members of a particular feeding guild use a location being evaluated, especially those receptors that are mobile and typically require specific types or sizes of habitat.

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

Critical ecological attributes for communities include:

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

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

- *Herbivore productivity*—because herbivores incorporate energy and nutrients from plants and transfer it to higher trophic levels, such as first- and second-order carnivores (e.g., snakes and owls, respectively). Herbivores also are integral to the success of terrestrial plants, through such attributes as seed dispersal.
- *Omnivore productivity*—because omnivores incorporate energy and nutrients from lower trophic levels and transfer it to higher levels, such as first- and second-order carnivores.
- *First-order-carnivore productivity*—because these carnivores are food for other carnivores (both first- and second-order), omnivores, scavengers, and microbial decomposers. They also affect the abundance, reproduction, and recruitment of receptors at lower trophic levels, such as vertebrate herbivores and omnivores, through predation.
- *Second-order-carnivore productivity*—because carnivores affect the abundance, reproduction, and recruitment of species in lower trophic levels in the food web.
- *The viability and presence of upper-trophic-level receptors,* including places where their presence and viability are significantly affected through the impairment or loss of habitats, food sources, and energy transfers associated with lower trophic level receptors. Herbaceous plants provide an important pathway for energy and nutrient transfer from soil to herbivorous (e.g., rabbit) and omnivorous (e.g., mouse) receptors and also provide critically important habitat for small animals. Woody plants provide an important pathway for energy and nutrient transfer from soil to herbivorous and omnivorous vertebrates (e.g., white-tailed deer, red-cockaded woodpecker) and also provide critically important habitat for terrestrial wildlife. Terrestrial invertebrates provide a mechanism for the physical breakdown of detritus for microbial decomposition and are also a major food source for omnivores.

While there is agreement that limitations in habitat and energy transfer can conceptually affect the higher-trophic-level receptors, little ecotoxicological research is available to quantify what level of change constitutes a

significant difference, given the large areas used and the variable diets common in these species.

Evaluation of possible habitat or energy transfer effects involve a three-step process: (1) characterize habitat needs and principal food sources of measurement receptor, (2) estimate threshold loss levels at which habitat or food source depletion could result in effects on assessment receptor, and (3) define COC levels in affected media that are protective of habitat and food source species.

In practical terms, step (2) is the most significant (and likely the most difficult): the magnitude of chemical toxicity effect required to harm an upper-tropic-level receptor via habitat or energy loss will depend primarily on the size of the receptor's home range relative to the size of the affected property. For example, complete absence of a habitat or food source over a small area may have no population-level impact on an upper-tropic-level receptor with a large home range relative to the affected property. However, for an affected property much larger than the home range of the upper-tropic-level receptor, protection of the food-source population may be required to ensure protection of the assessment endpoint. Home range information for various species is provided in EPA's *Wildlife Exposure Factors Handbook* (1993i)*;* however, the person is encouraged to check the latest conservation biology and ecotoxicology literature or consult an expert.

If habitat or food-source effects are determined to be relevant, then step (3) above requires the development of ecological PCLs that are the lesser of (a) PCLs protective of the assessment endpoint species from chemical toxicity and (b) PCLs which are protective of the habitat specific to the assessment endpoints.

Table 3.5 is a compilation of some example critical ecological attributes and assessment endpoints for feeding guilds and communities in the five terrestrial and two example aquatic Texas habitat-specific food webs. As previously discussed, depending on the site-specific conditions and the selected measurement receptors, not all of these assessment endpoints will be applicable or will be associated with the development of COC-specific HQs and PCLs.

Table 3.5. Example critical ecological attributes and assessment endpoints for feeding guilds and communities in example Texas food webs. (Modified from U.S. EPA, 1999a.)

Aquatic Receptors

Community or Feeding Guild	Representive Receptors	Example Critical Ecological Attributes and Assessment Endpoints
Algae, Aquatic Vegetation	phytoplankton, epiphytes, periphytes, pond lilies, reeds, sedges, cattails, alligator weed, duckweed, grasses, rushes	Primary producers convert light energy into biomass, and are the first link in aquatic food chains supporting higher trophic level aquatic consumers and wildlife. Rooted vegetation also provides habitat and bottom stability. *Possible assessment endpoint = preservation of the viability of higher trophic level receptors using this community as habitat, food source, and/or energy transfer.*
Water-Column Invertebrates	zooplankton (rotifers, cladocerans, copepods), insects, mysid shrimp, other crustaceans	Aquatic invertebrates are an important food source for many higher trophic level consumers. Zooplankton regulate phytoplankton populations, and are a critical link in energy transfer to higher trophic levels in aquatic ecosystems. *Possible assessment endpoint = preservation of the viability of upper trophic level receptors using this community as a food source and/or energy transfer.*
Herbivorous and Planktivorous Fishes	fathead minnow, mosquito fish, golden shiner, threadfin shad, gizzard shad, paddlefish, emerald shiner, ironcolor shiner, blacktail shiner, central stoneroller, brook silverside, dusky darter, killifish, sheepshead minnow, silverside, menhaden, mullet, gulf pipefish	Herbivorous and planktivorous fish are important prey for higher-trophic-level predators in the aquatic and terrestrial ecosystems, and form a critical link for energy transfer from primary producers to higher-trophic-level consumers. They generally comprise the majority of tissue biomass in aquatic ecosystems, and play an important role in the ecosystem through regulating algae and plankton biomass. *Possible assessment endpoint = preservation of the productivity of this guild.*
Omnivorous Fishes	bluegill, green sunfish, redear sunfish, channel catfish, blue catfish, smallmouth buffalo, river carpsucker, spot, pinfish, croaker, sea (hardhead) catfish, gafftopsail catfish, naked goby, feather blenny, freckled blenny	Omnivorous fish are an important prey item for higher-trophic-level predators. Through predation, they may also regulate population levels in lower-trophic-level fish and invertebrates. *Possible assessment endpoint = preservation of the productivity of this guild.*

Community or Feeding Guild	**Representive Receptors**	**Example Critical Ecological Attributes and Assessment Endpoints**
Omnivorous Crustaceans	blue crab, stone crab	Omnivorous crustaceans are an important prey item for higher-trophic-level predators. Through predation, they may also regulate population levels in lower trophic level fish and invertebrates. They also play an important role as scavengers in nutrient cycling. *Possible assessment endpoint = preservation of the function and productivity of this guild.*
Carnivorous Fishes	largemouth bass, striped bass, alligator gar, chain pickerel, Guadalupe bass, white crappie, flounder, bull shark, red drum, spotted seatrout, bonnethead	Carnivorous fish provide an important function for the aquatic environment by regulating lower trophic level populations through predation. They are also an important prey item for many top mammal and bird carnivores. *Possible assessment endpoint = preservation of the productivity of this guild.*

Sediment Receptors

Community or Feeding Guild	Represent Receptors	Example Critical Ecological Attributes and Assessment Endpoints
Benthic Invertbrates	crayfish, snails, clams, annelids, insects, amphipods, shrimp, hermit crabs, fiddler crabs, polychaetes, sea urchins, sand dollars	Benthic invertebrates are an important food source for many higher-trophic-level predators. They also provide an important role as decomposers and detritivores in nutrient cycling. *Possible assessment endpoint = preservation of the function and productivity of this guild, where appropriate.*

Soil Receptors

Community or Feeding Guild	Represent Receptors	Example Critical Ecological Attributes and Assessment Endpoints
Terrestrial Plants	pine trees, hardwoods, brush trees, vascular plants, grasses, forbs, yucca, pricklypear, crops, sunflower, thistle, sage	Primary producers provide a critical food source and are the first link in the terrestrial food chain for higher–trophic-level consumers. In addition, vegetation provides critical habitat for wildlife. *Possible assessment endpoint = preservation of the viability of upper-trophic-level receptors using this community as habitat or food, or for energy transfer.*
Terrestrial Invertebrates	insects, arachnids, gastropods, oligochaetes, arthropods, nematodes	Terrestrial invertebrates are an important food source for many higher-trophic-level species. As decomposers and detritivores they play a critical role in nutrient cycling. They also aid in soil aeration and infiltration by increasing macro and micro porosity. *Possible assessment endpoint = preservation of the viability of upper-trophic-level receptors using this community as food or for energy transfer.*

Upper-Trophic-Level Avian and Mammalian Wildlife

Community or Feeding Guild	Represent Receptors	Example Critical Ecological Attributes and Assessment Endpoints
Herbivorous Mammals	eastern cottontail, deer mouse, hispid cotton rat, white-tailed deer, plains harvest mouse, prairie vole, gopher, desert cottontail, kangaroo rat, pocket mouse, black-tailed jackrabbit, black-tailed prairie dog, white-throated woodrat, porcupine, pronghorn antelope, mule deer, livestock, muskrat, swamp rabbit, eastern gray squirrel, American beaver, nutria, West Indian manatee	Herbivorous mammals are an important prey item for many higher-trophic-level predators. They form an important link for energy transfer between primary producers and higher–trophic-level consumers. In addition, these organisms generally comprise the majority of the terrestrial tissue biomass, and are important in seed dispersal and pollination for many plant species. *Possible assessment endpoint = preservation of the productivity of this guild.*

Community or Feeding Guild	Represent Receptors	Example Critical Ecological Attributes and Assessment Endpoints
Herbivorous Birds	white-winged dove, mourning dove, cedar waxwing, house finch, sparrow, Canada goose, lark bunting, pyrrhuloxia, snow goose, brant, whistling swan, canvasback	Herbivorous birds are an important prey item for many higher-trophic-level predators. They are important in seed dispersal for many plants in both terrestrial and aquatic ecosystems. Aquatic herbivorous birds may also play an important role in egg dispersion for fish and invertebrate species. *Possible assessment endpoint = preservation of the productivity of this guild.*
Omnivorous Mammals	Virginia opossum, nine-banded armadillo, southern short-tailed shrew, raccoon, eastern wood rat, least shrew, desert shrew, eastern mole, thirteen-lined ground squirrel, northern grasshopper mouse, mexican ground squirrel, white-footed mouse, various bat species, ringtail, livestock, marsh rice rat, cotton mouse	Omnivorous mammals are an important prey item for higher-trophic-level predators, and influence lower-trophic-level populations through predation. They play an important role in seed dispersal for many types of terrestrial vegetation and aquatic plants. *Possible assessment endpoint = preservation of the productivity of this guild.*
Omnivorous Birds	American woodcock, red-cockaded woodpecker, American robin, northern cardinal, tufted titmouse, Carolina chickadee, western meadowlark, sandhill crane, greater prairie chicken, northern bobwhite, scaled quail, lesser prairie chicken, lesser golden plover, American pipet, horned lark, dickcissel, white-necked raven, cactus wren, flycatcher, mallard, lesser scaup, herring gull, marsh wren, common snipe, red-winged blackbird, northern shoveler, swamp sparrow, roseate spoonbill	Omnivorous birds are an important prey item for higher-trophic-level predators. They play an important role in seed dispersal and pollination for many types of terrestrial vegetation and aquatic plants. In addition, aquatic species disperse eggs of some fishes and invertebrates. *Possible assessment endpoint = preservation of the productivity of this guild.*

Community or Feeding Guild	**Represent Receptors**	**Example Critical Ecological Attributes and Assessment Endpoints**
Omnivorous Amphibians and Reptiles	marbled salamander, Hurter's spadefoot toad, Eastern box turtle, slender glass lizard, central newt, rough earth snake, ornate box turtle, Texas toad, plains hognose snake, plains blind snake, Texas spotted whiptail, six-lined racerunner, Bufo toads, northern earless lizard, trans-Pecos blind snake, Eastern hognose snake, short-lined skink, Eastern green toad, snapping turtle, painted turtle, Texas river cooter, pond slider, eastern newt, green frog, leopard frog, lesser siren, loggerhead sea turtle, Atlantic ridley sea turtle, terrapins	Omnivorous amphibians and reptiles are an important food source for predators. They also disperse seeds of many plants and regulate lower-trophic-level populations through predation. *Possible assessment endpoint = preservation of the productivity of this guild.*
Insectivorous Reptiles	horned lizard, desert side-blotched lizard, southern prairie lizard	Insectivorous reptiles play an important functional role to the environment by regulating lower–trophic-level prey and are an important prey item for other upper-trophic-level predators. *Possible assessment endpoint = preservation of the productivity of this guild.*
Carnivorous Mammals	red fox, bobcat, spotted skunk, long-tailed weasel, coyote, swift fox, badger, gray fox, mountain lion, striped skunk, mink, river otter, bottlenose dolphin	Carnivorous mammals have an important environmental function: regulating lower-trophic-level prey populations. *Possible assessment endpoint = preservation of the productivity of this guild.*
Carnivorous Birds	turkey vulture, red-tailed hawk, broad-winged hawk, barn owl, great horned owl, American kestrel, Cooper's hawk, ferruginous hawk, Swainson's hawk, burrowing owl, white-tailed hawk, rough-legged hawk, Mississippi kite, crested caracara, prairie falcon, greater roadrunner, golden eagle, osprey, bald eagle, northern harrier, barred owl, pelicans, terns	Carnivorous birds have an important environmental function: regulating lower-trophic-level prey populations. *Possible assessment endpoint = preservation of the productivity of this guild.*

Community or Feeding Guild	**Represent Receptors**	**Example Critical Ecological Attributes and Assessment Endpoints**
Carnivorous Shore Birds	spotted sandpiper, belted kingfisher, great blue heron, greater yellowlegs, willet, snowy egret, American bittern, black-crowned night heron	Carnivorous shore birds have an important environmental functions: regulating lower-trophic-level prey populations, and influencing species composition in terrestrial and aquatic ecosystems. They also disperse eggs of some fishes and aquatic invertebrates. *Possible assessment endpoint = preservation of the productivity of this guild.*
Carnivorous Reptiles and Amphibians	Texas rat snake, canebrake rattlesnake, southern copperhead, eastern yellowbelly racer, great plains rat snake, Texas garter snake, bullsnake, western Diamondback rattlesnake, Western coachwhip, Great Plains skink, longnose leopard lizard, desert kingsnake, bullfrog, Harter's water snake, pig frog, American alligator, cottonmouth water moccasin, spiny softshell turtle, king snake, diamondback water snake	Carnivorous reptiles and amphibians have an important environmental functions: regulating lower-trophic-level prey and serving as an important prey item for other upper-trophic-level predators. *Possible assessment endpoint = preservation of the productivity of this guild.*

3.6.7 Selecting Measures of Effect (Measurement Endpoints)
(modified from U.S. EPA, 1999a)

Measures of effect are measures used to evaluate "the response of the assessment endpoint when exposed to a stressor (formerly measurement endpoints)" (U.S. EPA, 1997a). *Measures of exposure* are measures of how exposure may be occurring, including how a stressor may co-occur with the assessment endpoint (ibid.). Measures of effect, in conjunction with measures of exposure, are used to make inferences about potential changes in the assessment endpoint (ibid.).

Measures of effect are selected as: (1) toxicity values developed or adopted by state or federal agencies (e.g., TCEQ surface water quality criteria, NOAA ER-L values) for protection of media-specific communities, or (2) receptor-specific chronic NOAELs or their equivalent for ecologically relevant endoints for this screening assessment. Measures of exposure are selected as the COC concentrations in media or dose (e.g., ingestion of impacted media or tissue) to which exposure occurs, and determined as discussed in **3.9.2**.

The evaluation of the measure of effect to the assessment endpoint requires identification of a measurement receptor representative of the assessment endpoint. A measurement receptor, specific to each feeding guild, may be selected as a species, population, community, or assemblage of communities. Measurement receptors for communities and for feeding guilds are discussed in **3.6.7.1** and **3.6.7.2**, respectively. Appendix B discusses selection of some potential measurement receptors for the example food webs from **3.6.5**.

3.6.7.1 Measurement Receptors for Communities
(modified from U.S. EPA, 1999a)

A community that occupies surface water, sediment, or soil or an assemblage of such communities is selected as the measurement receptor, and no specific species are selected. Therefore, it may be inferred that critical ecological attributes of these communities are not adversely affected if a COC concentration in the respective medium does not exceed the toxicity benchmark specific for that community (see Tables 3.2–3.4). Remember that this exercise in comparative concentrations was conducted as required element (1) (**3.5**). COCs that exceed these benchmarks but do not subsequently prove a risk to higher-trophic-level receptors may still have an impact on these community-level receptors. As indicated during the discussion of the definition of *ecological PCL*, depending on site-specific circumstances, the person may be required to demonstrate that impacts on these communities will not result in unacceptable consequences for the more mobile or wide-ranging receptors. Generally, exceedances of benchmarks should not result in the development of COC-specific PCLs for communities, with the exception of benthic invertebrates (see **3.6.1**).

Representative measurement receptors for communities include:

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

3.6.7.2 Measurement Receptors for Feeding Guilds (modified from U.S. EPA, 1999a)

A measurement receptor should be selected for each **evaluated** class-specific feeding guild to model the COC dose ingested, and the whole-body COC concentration in prey eaten by predators. The selected measurement receptor should be representative of other species in the guild with respect to its feeding niche in the ecosystem. The risk assessment should demonstrate that using the measurement receptor ensures that risk to other species in the guild is not underestimated, particularly if threatened or endangered species are an issue. The following factors should be evaluated when selecting a representative measurement receptor:

- **Ecological relevance.** Highly relevant receptors fulfill an important function or form an important structure in the ecosystem. Attributes of highly relevant receptors typically fall under the categories of food, habitat, production, seed dispersal, pollination, and decomposition. Critical attributes include those that affect or determine the function or survival of another population in the same community. For example, a population of forage fish might be critical to the sustainability of a population of carnivorous game fish.
- **Exposure potential.** Receptors with high exposure potentials are those that tend to have higher potentials for exposure than other receptors, due to their metabolism, feeding habits, location, or reproductive strategy. For example, the metabolic rates of small receptors are generally higher than those for large animals. This results in a higher ingestion per body weight—i.e., increased exposure potential.
- **Sensitivity.** Highly susceptible receptors include those with low tolerances to a COC as well as receptors with enhanced COC susceptibility due to other concomitant stressors that may not be related to a COC, such as reduced habitat availability. For example, raptors are highly sensitive to the effects of chlorinated pesticides that bioaccumulate through the food chain.
- **Social or economic importance.** An assessment endpoint may also be based on socially or economically important receptors, including species valued for social importance, such as songbirds, and economic importance, such as game fish. For these receptors, critical attributes include those that affect survival, production, and fecundity characteristics.
- **Presence of known, expected, or protected species.** The receptor need not always be present or even occur at all at the affected property to be selected as a measurement receptor. A migratory species may be the most appropriate measurement receptor for a particular feeding guild.

Also, a species that is expected to occur, based upon its range and the availability of suitable habit, but does not occur because of its susceptibility to COCs from the affected property, may be a good measurement receptor. Threatened and endangered species do not usually make good measurement receptors for obvious reasons. However, if a protected species is potentially present at the affected property, a measurement receptor should be selected that will not underestimate risk to the protected species.

- **Availability of information on natural history or toxicology.** Information on natural history and toxicology is essential to the quantitative evaluation of risk to measurement receptors. If this information (e.g., body weight, food and media ingestion rates, and COC-specific TRVs) is unavailable for the desired measurement receptor, information for one or more of these exposure variables should be obtained from a closely related species. Uncertainty associated with using such surrogate information should be discussed (**3.12**).

Note that more than one measurement receptor can be selected per assessment endpoint. Also, although each of these factors should be evaluated when selecting the measurement receptor, one of the measurement receptors selected to represent a class-specific feeding guild should have the highest exposure potential (i.e., highest ingestion rate on a per unit body weight) and the greatest sensitivity (i.e., lowest NOAEL or LOAEL). Identification of a receptor that is both the most exposed and the most sensitive to a particular COC and its use as the measurement receptor ensures that risk to other species in the guild is not underestimated. As stated in **3.6.5**, users will not have to evaluate all feeding guilds for a particular food web provided there is a reasonable justification.

The U.S. EPA's *Wildlife Exposure Factors Handbook* (1993i) is an example of an excellent source of information on diet and other natural history. However, receptor information obtained from it or any source should be verified and documented.

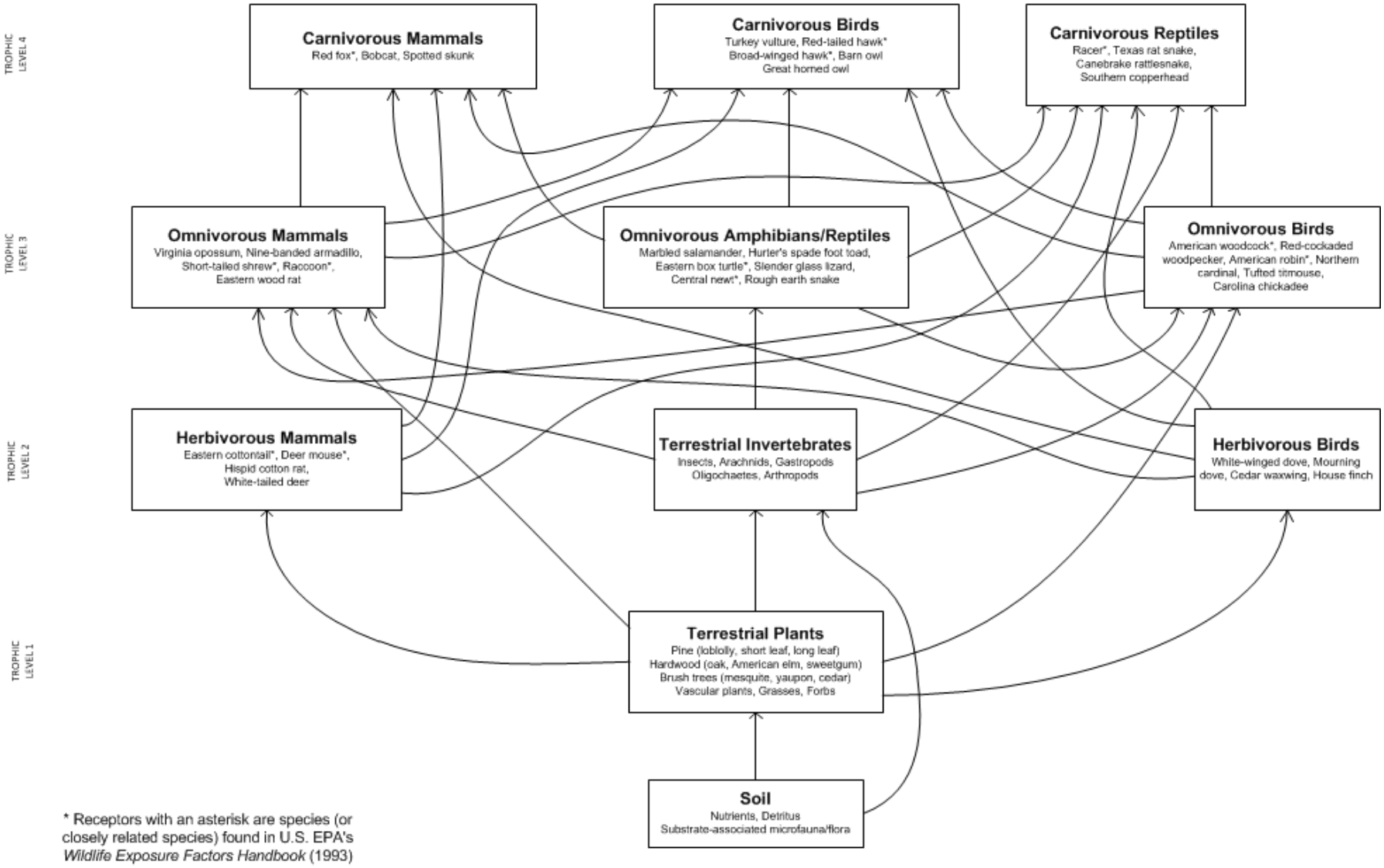


Figure 3.5. Example Texas upland-forest food web.

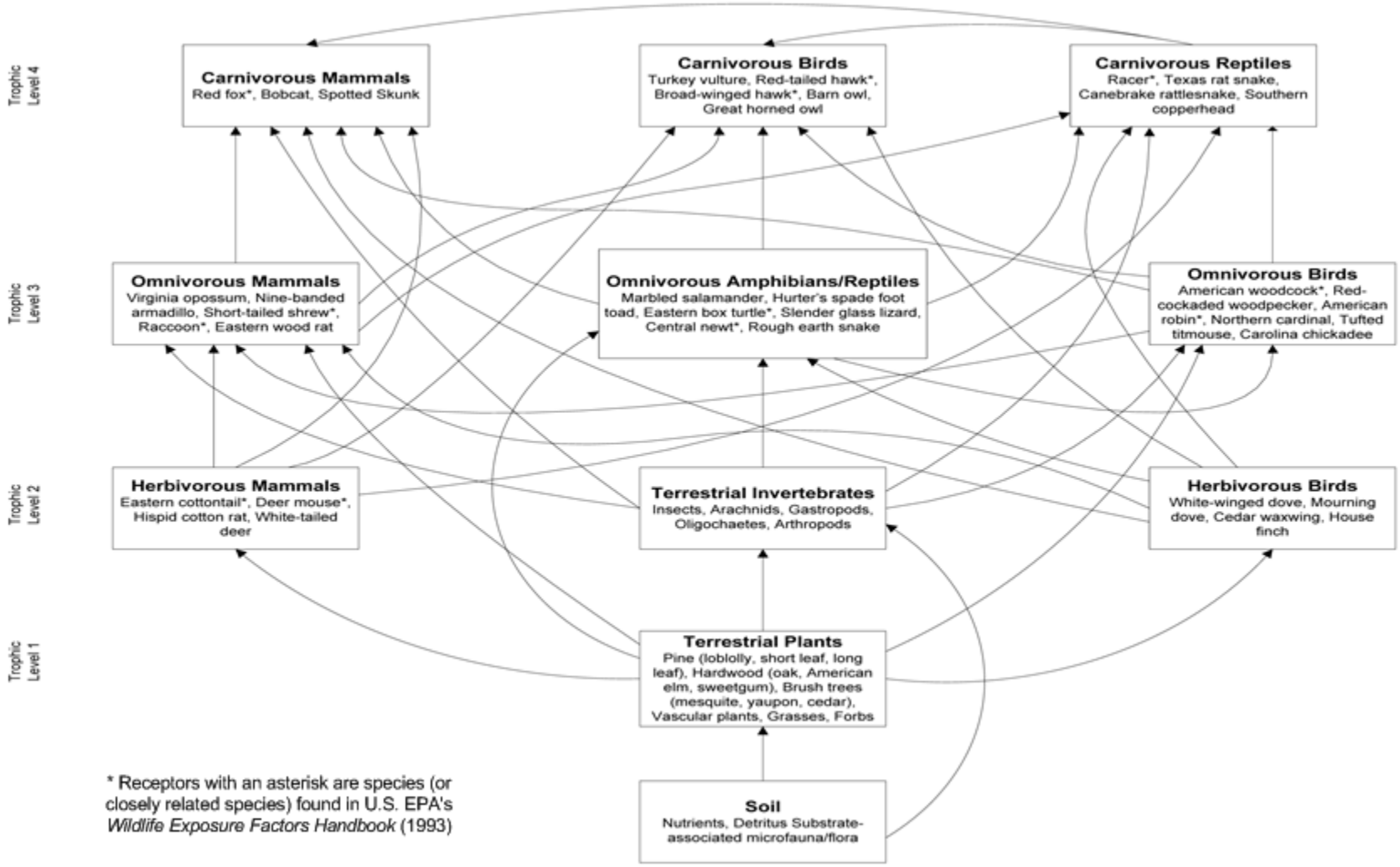


Figure 3.6. Example Texas tallgrass-prairie food web.

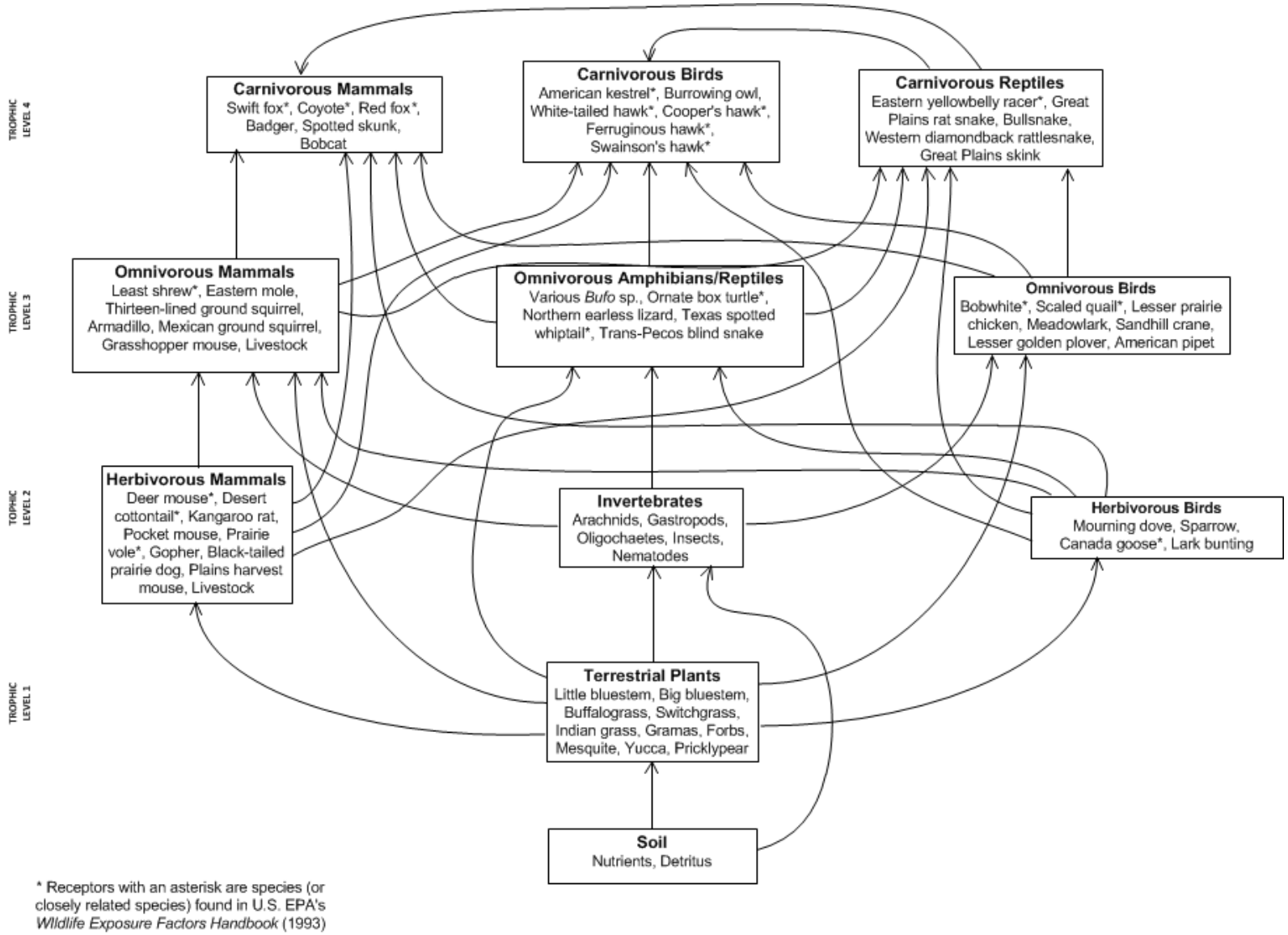


Figure 3.7. Example Texas shortgrass-prairie food web.

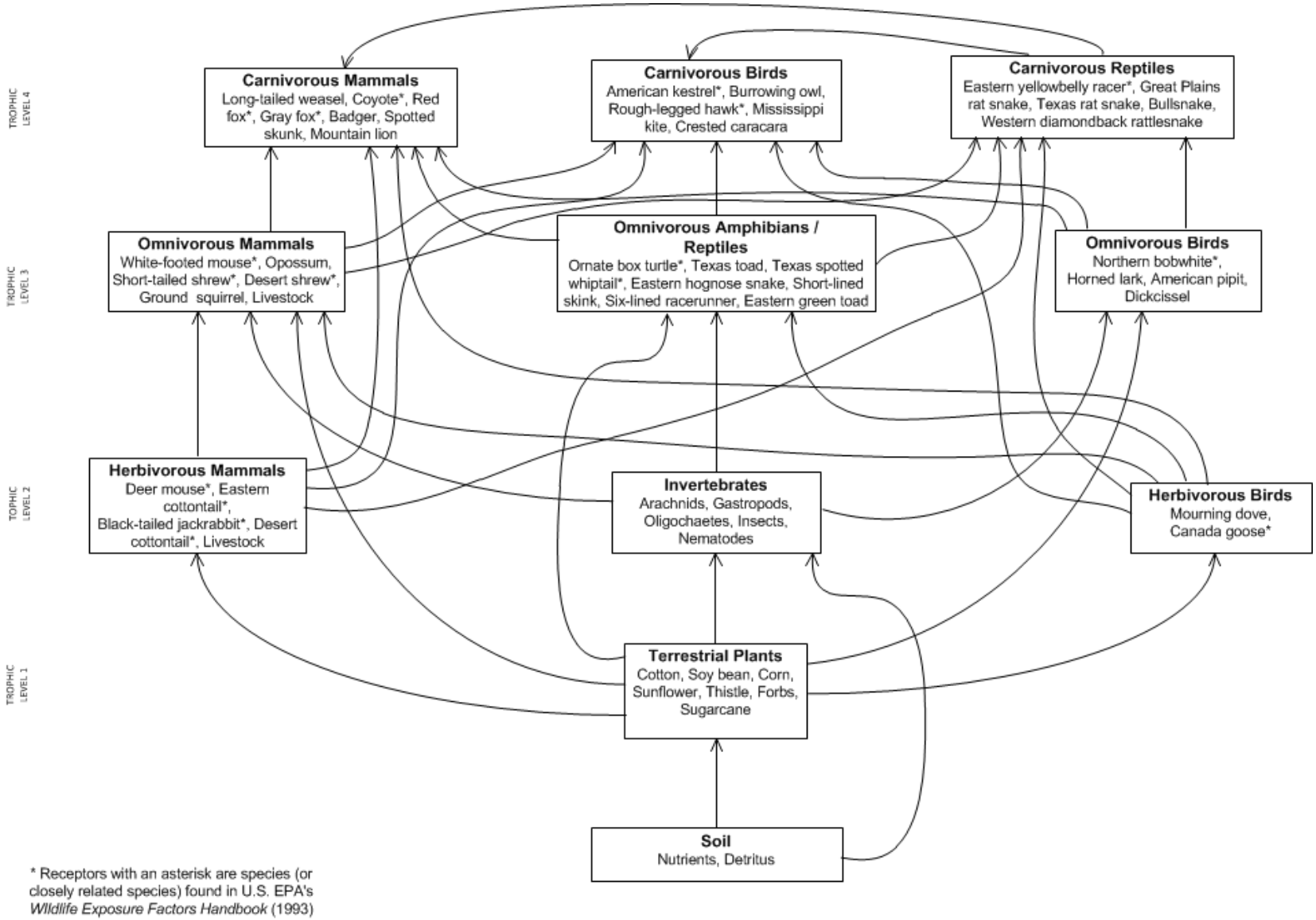


Figure 3.8. Example Texas shrub-scrub food web.

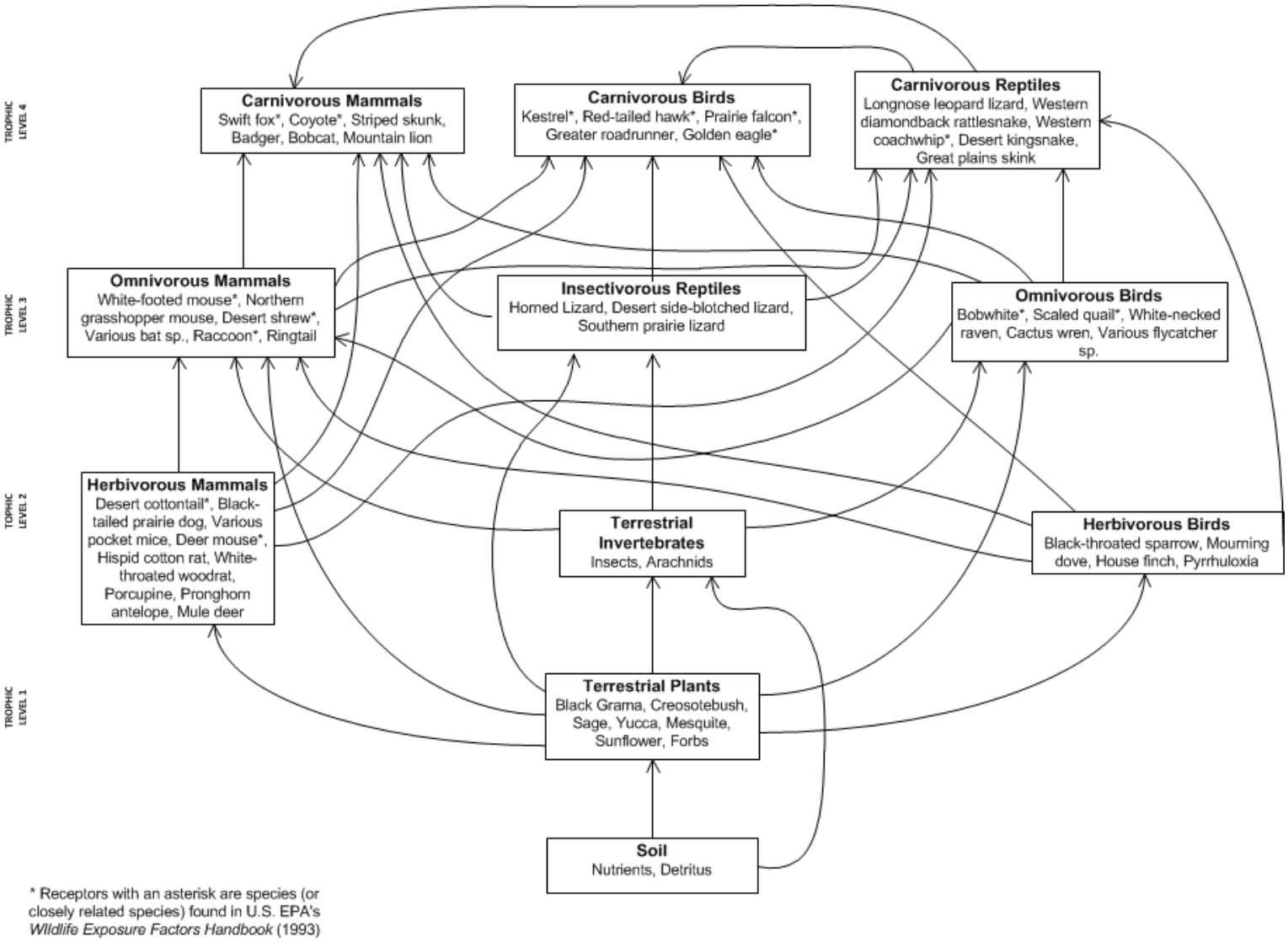


Figure 3.9. Example Texas desert-arid food web.

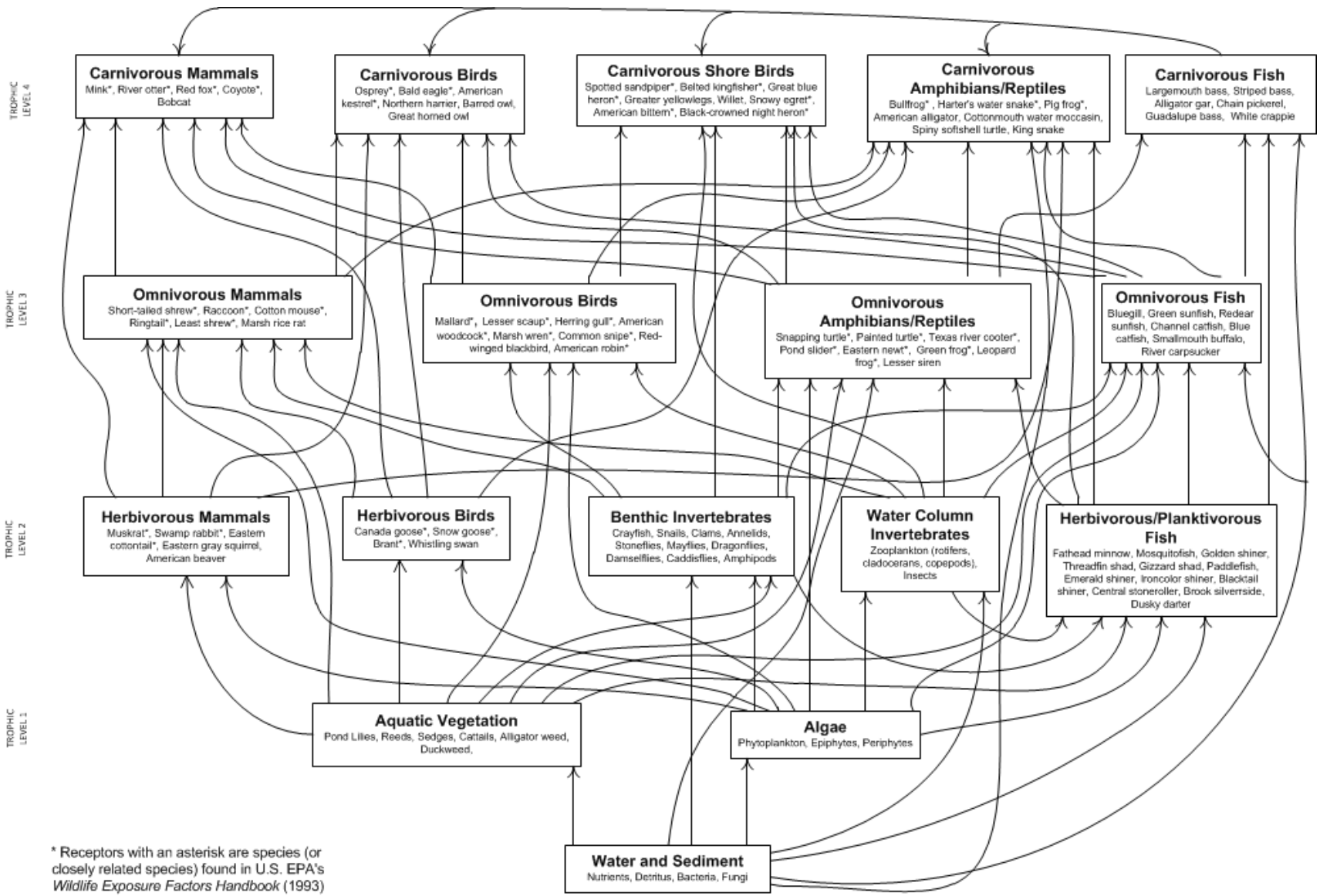


Figure 3.10. Example Texas freshwater-wetland food web.

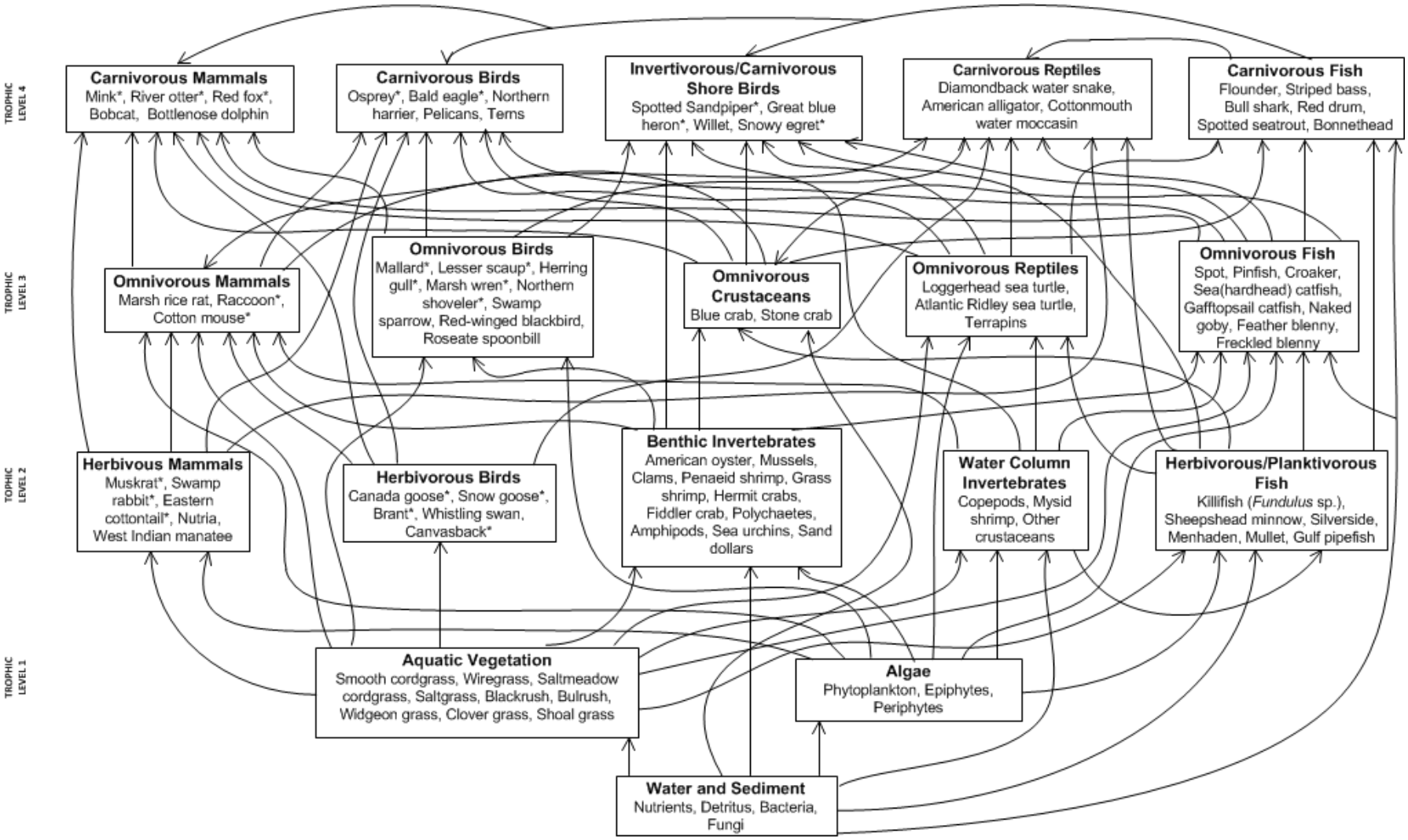


Figure 3.11. Example Texas estuarine-wetland food web.

3.7 Conceptual Model

The purpose of the conceptual model is to illustrate the exposure pathways that are complete or reasonably anticipated to be completed ecological exposure pathways and are to be evaluated in the ERA. The steps to building the conceptual model include:

1. Determine the types of habitat supported by or on the affected property.
2. Relate habitats on the affected property to one or more of the seven example habitats and their associated food webs (see **3.6.5** and Figures 3.5–3.11).
3. Add in the direct (media) exposure pathways and define the routes of exposure to each feeding guild or the representative species of a guild.
4. Test and refine the pathways or routes of exposure for applicability of COCs to any feeding guild.
5. Build the site conceptual model for the food web and COCs.

The conceptual model graphically depicts the movement of COCs from sources through media to the feeding guilds or to the selected ecological receptors of those guilds (i.e., measurement receptors). An overview of conceptual model development for ecological exposure pathways appears in Figure 3.12. Although a graphic presentation is preferred, this may be replaced or supplemented with a narrative or table to present the source-to-receptor exposure relationships.

3.8 Fate and Transport, Toxicological Profiles

A determination should be made as to whether the COCs at the affected property are likely to persist, be degraded, or move beyond the extent of contamination determined in the affected property assessment. COCs to be included in an affected property assessment, remedial investigation, or the equivalent are identified based on the specific requirements of the triggering program, and may include the levels present at the affected property, the media where the COCs are located, and the potential for adverse ecological effects associated with a particular COC. During the assessment, the person characterizes the nature, extent, and potential fate and transport of COCs.

This characterization includes physical, chemical, and biological processes and their influence on the movement, persistence, form, toxicity, and availability of COCs to the degree necessary to understand and characterize risk.

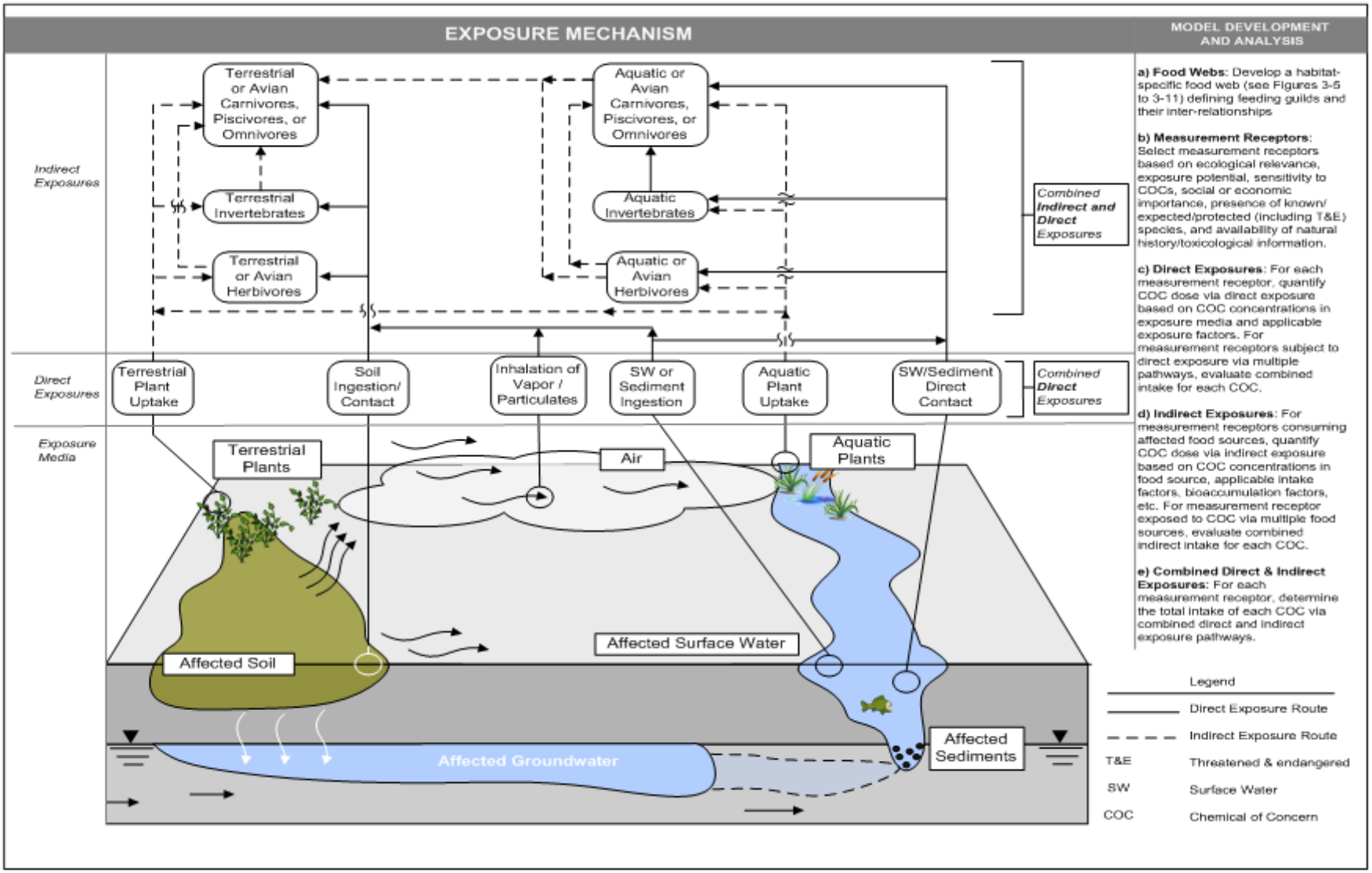


Figure 3.12. Overview of conceptual model development for ecological exposure pathways.

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

- degradation
- complexation
- ionization
- precipitation
- adsorption

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

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

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

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

Additional information should be gathered on past as well as current mechanisms of COC release from source areas at the affected property. The mechanisms of release along with the chemical and physical form of a COC can affect its fate, transport, and potential for reaching ecological receptors. Although information on the fate and transport of COCs can be found throughout the literature, the TRRP rule gives a variety of simple screening-level fate and transport equations that apply to COC migration in soil, air, groundwater, and surface water, and that should be used in ecological exposure evaluations. In addition, certain chemical and physical parameters (e.g., vapor pressure, solubility, log K_{ow}) must be used where appropriate. Consistent with the TRRP rule, pH-dependent factors may be adjusted to account for site-specific pH conditions [see 30 TAC 350.73(f)(1)].

Toxicological profiles available in various databases (e.g., HSDB, Medline, Toxline, ECOTOX) are used to quantify toxicity (e.g., dose-response) and to evaluate the likelihood of toxic effects in different groups of organisms. Effects levels may be expressed either as dose-response values (e.g., NOAEL, LOAEL) or as medium-specific exposure limits (e.g., surface water quality criteria for aquatic life). The toxicity profile should describe the toxic mechanisms of action, to the degree known or available, for the exposure routes being evaluated and the dose or environmental concentration that causes a specified adverse effect. Understanding the toxic mechanism of a COC helps to evaluate the importance of potential exposure pathways and focuses the selection of assessment endpoints. An example fate and transport-toxicological profile for zinc appears as Attachment 2. Toxicological profiles should focus on the information needs relevant to the site-specific situation and need not follow the exact format or content of Attachment 2, depending on the site. Professional judgment should be used to determine which COCs (or groups of COCs) should be described, but relevant considerations should include COC toxicity, bioaccumulation, and site concentrations.

3.9 Input Data and Exposure Calculation

As presented in Figure 3.1, input data and exposure calculations for Tier 2 SLERAs are developed in accordance with required element (5) of 30 TAC 350.77(c). Figure 3.3 depicts how exposure estimates for wildlife are conducted for COCs detected in environmental media that bioaccumulate in wildlife food (i.e., plant and animal tissue), exceed ecological benchmarks, or do not have benchmarks. Conversely, COCs that are not bioaccumulative for an environmental medium and that do not exceed benchmark levels are not carried forward into an exposure evaluation of wildlife receptors. For example, the estimated total dose to a piscivorous wildlife receptor would not include the component attributable to water ingestion if that COC does not bioaccumulate in water and does not exceed surface water benchmark levels.

3.9.1 Initial and Refined Exposure Assessments

Required element (5) of 350.77(c) provides for an initial exposure assessment incorporating reasonably conservative assumptions to minimize the potential for overlooking ecological risks. These assumptions may incorporate site-specific data when available. Required element (7) allows for a refined exposure assessment when less conservative exposure can be justified based on conditions at the affected property (see **3.11**). Initial and refined exposure estimates typically incorporate a range of values for the following exposure variables:

- bioavailability (see **3.9.2.4, 3.9.4**)
- area-use factors and exposure frequency (see **3.9.2.4**)
- diet (see **3.9.2.2**)

Both initial and refined exposure estimates may also be developed to evaluate risk associated with other parameters that can vary widely. For example, when available bioconcentration factors for a particular COC are not within an order of magnitude of one another, both an initial and refined exposure estimate may be warranted.

However, other data such as NOAEL-based toxicity reference values will be carried through the Tier 2 SLERA without modification.

3.9.2 Exposure Characterization

Exposure for fish, benthic macroinvertebrates, soil invertebrates, and plants is generally expressed in terms of media concentration (mg/L or mg/kg), while exposure for terrestrial wildlife such as birds and mammals is generally expressed in terms of dose (mg / kg body weight / day). In both the initial and refined exposure assessments, appropriate statistical methods (e.g., 95 percent upper confidence limit of the arithmetic mean) may be used to compute exposure point concentrations from COC concentrations in the exposure medium in accordance with TRRP guidance.

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

Dermal and inhalation exposure routes for wildlife are typically not addressed in ERAs due to limited toxicity information and, generally, the lesser significance of these exposure routes relative to oral exposure at many affected properties. Note that circumstances exist where dermal and inhalation exposure may be significant, such as for amphibians, burrowing wildlife, and those species that inhabit the burrows of others. Carlsen (1996) compares inhalation exposures to burrowing and nonburrowing mammalian vertebrates exposed to trichloroethylene and cadmium at a hazardous-waste site. Gallegos et al. (2007) develop inhalation screening levels for VOCs for small burrowing receptors based on the pocket gopher. Archbold et al. (2007) provide inhalation TRVs for small mammal exposure to VOCs and select metals. If evaluation of dermal or inhalation routes for wildlife is warranted, available uptake and response data should be obtained. Hope (1995) and U.S. EPA (1993i) give exposure-estimation methods for dermal and inhalation pathways. Maughan (1993) includes a methodology for determining ecological effects levels for muskrat and beaver from inhalation and dermal exposures to benzene, toluene, ethylbenzene, xylenes and PAHs.

The following general equation can be used to estimate oral exposure for wildlife receptors:

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

where:

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

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

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

EMF_{food} = exposure modifying factor for food (unitless)

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

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

EMF_{water}= exposure modifying factor for water (unitless)

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

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

EMF_{soil} = exposure modifying factor for soil (unitless)

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

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

EMF_{sed} = exposure modifying factor for sediment (unitless)

BW = body weight of receptor (kg)

These variables are discussed in **3.9.2.1–5**. Literature sources used for intake and exposure variables should be clearly indicated and justified in the Tier 2 SLERA. Where literature information is modified for use in a Tier 2 SLERA, the modifications should be clearly indicated in the discussion. For example, as indicated, a literature-derived food ingestion rate may be adjusted to reflect wet weight or dry weight as appropriate. Another example is the use of U.S. EPA (1993i) to determine the relative food types in a receptor's diet when the person elects not to use every referenced type. Where a variety of choices are available for a receptor (e.g., body weight, dietary composition), the person should indicate how any one reference was selected from those available, particularly where he or she has selected a less conservative exposure factor. Also, the current scientific literature should be reviewed to determine if new information is available for previously less-studied wildlife taxa, such as reptiles and amphibians. A database called the Reptile and Amphibian Toxicological Literature (RATL) from the Canadian Wildlife Service (Pauli et al., 2000) should be reviewed for applicable toxicological data; it is available online at < publications.gc.ca/collections/Collection/CW69-5-357E.pdf>. Ecotoxicological information on amphibians and reptiles is also presented in Sparling et al. (2000) and Gardner and Oberdörster (2006).

3.9.2.1 COC Uptake in Food Items

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

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

Bioaccumulation is a general term describing a process by which COCs are taken up by an organism either directly from exposure to a medium or by consumption of food containing the COC [350.4(a)(8)]. A bioaccumulation factor is the ratio of the concentration of a COC in an organism to the concentration in the ambient environment at steady state, where the organism can take in the COC through both ingestion of food and direct contact. A biota-sediment accumulation factor (BSAF) is a specific type and form of bioaccumulation factor: the ratio of lipid-normalized tissue chemical residue to the chemical concentration in organic carbon–normalized sediment (Rand, 1995).

Bioconcentration is a net accumulation of a COC directly from an exposure medium (e.g., water) into an organism. A bioconcentration factor (BCF) is the ratio of the concentration of a COC in an organism to the concentration in the ambient environment at steady state.

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

As presented below and discussed in **3.9.5.5** and **3.13.2**, available methods for predicting tissue residues in a Tier 2 SLERA include the use of laboratory- and field-derived BCF and BAFs obtained from the scientific literature, as well as regression models based on literature-derived data, kinetic models, and thermodynamic (equilibrium partitioning) models. Laboratory- or field-derived BCFs and BAFs are available for a number of compounds and receptors. These values should be closely scrutinized in order to assess differences between laboratory and field conditions and differences in the biology and ecology of affected properties. The person should consult the original literature document whenever possible to determine applicability to the affected property conditions. Particular attention should be given to ensuring that values are based on similar underlying factors (e.g., organic carbon content, expression as wet versus dry weight, and normalization to a specific lipid content).

Several methods are available to develop BAFs using existing information and to further describe the methodology used in the belted-kingfisher exercise (**3.4.2**). In order to determine total dose for persistent and bioaccumulative COCs, exposure from environmental media, food intake, and magnification between successive trophic levels should be considered. A BAF provides a simplified way to use environmental concentrations (e.g., water or sediment analytical values) to determine a total dose for a wildlife receptor. In general, the estimated dietary concentration is the product of the environmental concentration and the BAF. The dietary exposure is then the product of this concentration and the ingestion rate for the receptor.

BAFs are highly dependent on site-specific characteristics, so any BAF based on generalized characteristics will have limited precision. To improve precision, the following preferences are usually accepted:

(a) data relevant to Texas and Gulf of Mexico ecosystems and species rather than other generic site data

(b) field data rather than laboratory data

(c) biotic data (e.g., fish bioconcentration studies) rather than physical- or chemical-based models (e.g., bioconcentration estimated from octanol-water partitioning studies)

The approach selected should reflect the availability of field data, the relevance of tested species, the food chain or trophic structure of the community and the level of modeling deemed acceptable for a Tier 2 SLERA. The method used for COCs evaluated in the kingfisher exercise is identified below.

Ideally, BAFs are available for the specific trophic level of the prey of the wildlife receptor. In other cases, a food-chain multiplier (FCM) is sometimes used to predict bioaccumulation not considered in available data (e.g., a bioconcentration factor). Octanol/water partitioning (log K_{ow}) is used as a predictor for trophic level transfer. Observations suggest that nonpolar organic COCs with log $K_{ow} < 5$ (Mackay, 2001: 219; Gobas et al., 1999) or log $K_{ow} < 5.5$ (Russell et al., 1999) have minimal differences via trophic-level transfers, so no adjustment is needed (i.e., FCM = 1). For other nonpolar organic COCs, FCM values have been developed based on log K_{ow} (U.S. EPA, 1999a) derived from models of a few selected COCs at a limited number of sites.

BAFs can be developed by a range of data types. Three are commonly used.

Field-derived BAFs: Stephan (1993) reviewed a number of field studies for aquatic bioaccumulation factors. Where several field studies were available, the geometric mean was taken. BAF values were normalized for percent lipid, and a final BAF was presented based on a 5 percent lipid value in the fish. The data used were from higher-trophic-level fish (e.g., salmonids). Because the field data represent exposure from environmental media, food intake, and magnification between successive trophic levels, no food chain multiplier is needed. In the kingfisher exercise (see **3.4.2**), field-derived BAFs were used for mercury, methylmercury, chlordane, toxaphene, mirex, photomirex, PCBs, and hexachlorocyclohexanes.

Fish Bioconcentration Studies and log K_{ow}: If field BAF data are not available, an alternative approach is to use a laboratory study of fish bioconcentration to derive a BCF, and to use log K_{ow} to derive a FCM. If several BCF studies are available, the geometric mean is taken. BCF values are not usually normalized for percent lipid. The log K_{ow} is used to derive an FCM, e.g., FCM = 1 if log $K_{ow} < 5$, or FCM = Table 5.2 value in U.S. EPA 1999a. In order to express the COC concentration on a dry-weight basis (to accommodate diet models for wildlife receptor consumption), the assumption is made that a fish is 80 percent water (Holcombe et al., 1976). In the kingfisher exercise, fish BCF-derived BAFs were used for tributyltin, DDT, DDE, dieldrin, endrin, heptachlor, and heptachlor epoxide.

log K_{ow}: Using the observed correction between fish BCF values and log K_{ow} values, an estimate of BCF can be derived from the log K_{ow}. The log K_{ow} is also used

to derive an FCM, as for the previous situation. The result is converted to a dry weight basis using the same assumptions as above. In the kingfisher exercise, log K_{ow}–derived BAFs were used for Aldrin and DDD.

When multiple literature values for a COC are deemed equally acceptable by the risk assessor (based on supporting documentation), an aggregate value may be obtained as the geometric mean of the available values, provided the values are for similar taxa (such as the same genus). BCF and BAF data for surrogate compounds may also be used in the risk assessment. For organic COCs, uptake factors can frequently be estimated from regression equations based on K_{ow}. When an appropriate BCF cannot be found or derived, a default value of 1 may be the best alternative; such a proxy assignment should be discussed when addressing the uncertainty analysis of required element (8). In general, BCF data for terrestrial biota, plants excluded, are much scarcer than for aquatic receptors.

Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities (U.S. EPA, 1999a) contains useful approaches to calculating COC uptake in representatives of the various feeding guilds present in Texas food webs. This peer-review draft contains an extensive treatment of COC uptake in ecological receptors at trophic level 1 as well as herbivorous, omnivorous, and carnivorous aquatic and terrestrial receptors (including birds) within the three higher trophic levels. However, the person should verify relevant equations and data from the original references.

Because no single document contains all acceptable approaches to quantifying COC uptake by way of diet, the ecological risk assessor is encouraged to review a variety of documents and publications. The person should clearly indicate the source of any uptake factor used in the ERA and justify its use relative to the COC and pathway in question. Additionally, the person should ensure that the uptake factor is expressed as wet weight or dry weight, but consistent with the corresponding wildlife ingestion rate. If different uptake factors are used in the initial and refined exposure evaluation, the person should clearly justify the use of any uptake factor more relevant to the exposure scenario at the affected property. Additional documents that may be considered for affected properties under TRRP include those listed in **3.9.2.1.2**. The person should also see Issue No. 27 in TCEQ (2005).

3.9.2.1.1 Uptake in Aquatic and Terrestrial Plants (Trophic Level 1)

In general, COCs may be present in plants largely as a result of transfer from the soil to soil pore water and, ultimately, into the roots. Rooted aquatic plants exhibit the same general mechanism: uptake of COCs from surface water into the roots. Where COC exposure at an affected property occurs primarily from air emissions, transfer through the leaves or cell membrane as well as direct deposition onto the outer surfaces of the plants can be relevant pathways for uptake. Plant uptake from air emissions is typically not addressed at sites subject to TRRP since this exposure pathway is generally of lesser significance than root uptake. If a particular affected property warrants evaluation of plant uptake from air emissions, appropriate uptake equations should be used (e.g., U.S. EPA, 1999a).

Modeling Uptake in Non-Rooted Aquatic Plants. Ecological risk assessors may obtain bioconcentration factors for uptake of COCs in non-rooted aquatic plants

such as algae from any of a variety of sources, including the EPA's 1980 criteria for ambient water quality and more recent published scientific literature, reports issued by the EPA or other reliable sources, unpublished data, and electronic databases such as ECOTOX. General reference texts such as Verschueren (1983) also contain published bioconcentration factors for algae. Users of secondary sources are cautioned that transcription errors may have occurred. Thus, they are advised to consult the primary source to verify that the bioconcentration factors contained in secondary references are accurate. Organic COC concentrations may be estimated by regression-based equations that account strictly for bioconcentration from surface water into tissues of non-rooted plants (U.S. EPA, 1999a). As in the rest of this guide, when data on chemical properties (e.g., K_{ow} values) are needed to perform the calculation, they must be obtained from the figure at 30 TAC 350.73(f).

Modeling Uptake in Rooted Aquatic Plants and Terrestrial Plants. For rooted aquatic plants, the equation used to calculate uptake of COCs through the roots of terrestrial plants is acceptable because the terrestrial model is based on uptake of COCs from the soil water. Note, however, that there is no accounting for any COCs penetrating the cuticle of the chronically submerged portions of rooted aquatic plants (i.e., direct bioconcentration). If a suitable model is identified to account for both root uptake and penetration through the cuticle, it will be referenced in future editions of this document.

For sites where the source of the COCs was a historical release of solid or liquid COCs to soil, the primary transfer mechanism into terrestrial plants is likely root uptake of soil pore water. In addition, rooted plants may take up COCs from other sources. COCs released to groundwater may be available for direct uptake by rooted plants. Uptake by rooted (and non-rooted) aquatic plants may also occur following discharge of groundwater to surface water. Finally, uptake may occur from irrigation using groundwater or surface water that contains COCs.

In a Tier 2 SLERA, concentrations of organic COCs in terrestrial vegetation may be quantified using published plant uptake factors. For inorganic COCs, soil-to-plant biotransfer factors (BTFs), expressed in units of g soil / g dry plant tissue, are available in TRRP. Figure 30, TAC 350.73(f), contains BTFs for calculation of concentrations of inorganic COCs in aboveground and belowground portions of plant matter. Note, however, that the aboveground values were based on an assumed vegetable diet for humans (50 percent protected vegetables, 25 percent exposed non-leafy vegetables, and 25 percent exposed leafy vegetables) and applicable uptake factors from U.S. EPA (1992d) and Baes et al. (1984). Therefore, those values should only be used in ecological evaluations when site-specific ecological receptors are believed to have a similar dietary composition. Nevertheless, the underlying uptake factors in Baes et al. (1984) and U.S. EPA (1992d) are applicable to ecological risk assessments when combined with appropriate dietary information. Segregation of inorganic constituents according to exposed and unexposed portions of plants lends enhanced specificity to risk assessments when receptors that ingest considerable quantities of root matter are evaluated.

Uptake of organic COCs in plant matter may be quantified using the following equation:

$$RCF = 10^{((0.77 \times \log K_{ow}) - 1.52)} + 0.82$$

where:

RCF = the ratio of COC concentration in roots to the ratio of COC concentration in pore water (unitless).

K_{ow} = the octanol-water partition coefficient, expressed as a log, obtained from the figure at 30 TAC 350.73(f).

The above equation, unlike the version in 30 TAC 350.75(b)(1), does not include a dry weight–to–wet weight conversion factor of 0.222 (such an adjustment is not appropriate if the corresponding wildlife ingestion rates are based on wet weight).

Note that the regression-derived portion of the equation, from Briggs et al. (1982), applies to uptake of organic constituents only in belowground portions of plants. Because organic constituents generally are poorly translocated in plants, the TCEQ is limiting consideration of uptake of organic constituents to below ground portions of plants only. Further, the TCEQ recognizes that highly volatile constituents, defined here as constituents having a Henry's Law constant of 0.03 atm-m^3/mol or greater, will not remain in soil within the root zone long enough to be taken into plant roots. Therefore, ecological risk assessments involving highly volatile constituents will not require evaluation of those constituents via ingestion of underground portions of plants.

3.9.2.1.2 Uptake of COCs in Herbivorous, Omnivorous, and Carnivorous Animals (Trophic Levels 2, 3, and 4)

As stated earlier, exposure estimates for wildlife are conducted for COCs detected in environmental media that bioaccumulate in wildlife food (i.e., plant or animal tissue), or exceed or lack ecological benchmarks. Uptake of COCs in diet at the higher trophic levels (2, 3, and 4) is a function of the types of food ingested daily and the concentration of COC in each food item (see **3.9.2.1**).

The EPA's Great Lakes Water Quality Initiative has made serious advances in quantifying COC uptake in aquatic organisms. A technical support document, *Great Lakes Water Quality Initiative Technical Support Document for the Procedure to Determine Bioaccumulation Factors* (U.S. EPA, 1995) contains a large number of BAFs for estimating COC uptake at various levels in the aquatic food chain. With due caution in applying these BAFs to aquatic sites in Texas (as the BAFs were derived for cold-water fish species, while the vast majority of Texas fishes are warm-water species), there is merit in reviewing those uptake approaches.

Attachment 4-1 to the EPA's *Guidance for Developing Ecological Soil Screening Levels (Eco-SSLs)*, Attachment 4-1, available at <www.epa.gov/ecotox/ecossl/pdf/ecossl_attachment_4-1.pdf>, contains basic equations for calculating COC uptake within the various trophic levels in terrestrial ecosystems.

U.S. EPA Region 10 Supplemental Ecological Risk Assessment Guidance for Superfund (U.S. EPA, 1997c) contains general uptake equations. More extensive treatment of uptake may be found in U.S. EPA, 1994a.

The Oak Ridge National Laboratory's has extensive documents on uptake of COCs in ecological receptors are available online at <www.esd.ornl.gov/programs/ecorisk/tools.html>. Example documents include Bechtel Jacobs Company 1998a, b. *Estimating Exposure of Terrestrial Wildlife to Contaminants* (Sample and Suter, 1994) contains equations for estimating uptake in a variety of terrestrial species ranging from trophic-level-2 herbivores (e.g., the eastern cottontail) to level-4 carnivores (e.g., the red-tailed hawk). Other available ORNL guidance for calculation of uptake in receptors includes *Development and Validation of Bioaccumulation Models for Small Mammals* (Sample et al., 1998a) and *Development and Validation of Bioaccumulation Models for Earthworms* (Sample et al., 1998b).

The EPA has produced a considerable body of information on uptake, relating to equilibrium partitioning of selected COCs from sediment and water into aquatic species tissues. Various reports and fact sheets available at <www.epa.gov/waterscience/pc/> can be used to estimate COC concentrations in applicable food items using equilibrium partitioning, also discussed in **3.5.1.2, 3.9.5.5**, and **3.13.2**.

3.9.2.2 Rates of Food and Water Ingestion

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

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

IR_{food} (wet weight) = IR_{food} (dry weight) / (1-percent water)

C_{food} (wet weight) = C_{food} (dry weight) × (1-percent water)

In the initial risk estimate, the most contaminated food item is sometimes assumed to comprise 100 percent of a receptor's diet (barring site-specific data to the contrary). A refined exposure assessment may consider data that more accurately reflect the receptor's diet. The person should also see Issue Nos. 8 and 28 in TCEQ (2005).

3.9.2.3 Soil and Sediment Ingestion Rates

Wildlife may ingest soil or sediment intentionally to obtain nutrients or incidentally during feeding, grazing, preening, cleaning, or burrowing. Information on soil or sediment ingestion rates is limited, and unlike food and water consumption,

generalized models do not exist for estimating soil ingestion by wildlife receptors. The *Handbook* estimates of soil and sediment ingestion for a variety of wildlife species (U.S. EPA, 1993i); however, current literature should be reviewed.

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

When information on a measurement receptor's soil or sediment ingestion rate is stated as a percentage of dry matter in the gut (as is common), the converted fractional value should be multiplied by the food-ingestion rate to obtain the soil-sediment ingestion rate. Percentages of food items in the diet of the measurement receptor should sum to 100 and should not be normalized to include the soil-ingestion percentage. The person should also see Issue No. 8 in TCEQ (2005).

3.9.2.4 Exposure Modifying Factors

In this document, exposure modifying factors (EMF, unitless) include bioavailability, home range, mobility, and life-cycle attributes. Bioavailability, the ratio of COC that reaches a site of toxic action in an organism to the total load of that COC in the environment, is discussed in **3.9.4**. Other examples of exposure modifying factors include the area-use factor (AUF) and exposure frequency (EF). AUF is defined as the ratio of home range, or feeding-foraging range, to the area of the affected property under investigation. EF accounts for migration or other seasonal activity patterns (U.S. ACE, 2010). Both AUF and EF are unitless values $> 0, \leq 1$. A variety of factors may influence home range or territory size, including habitat quality, prey abundance, and population density. The AUF for soil is the fraction of the receptor's total soil exposure that occurs in the area of affected soil. Similarly, the AUF for water is the fraction of the receptor's total water exposure that occurs in the area of affected water.

Both bioavailability and area use factor will vary for each exposure medium (soil, sediment, water, and food), though sufficient data may not be available at all affected properties to justify use of medium-specific values for these parameters. At such properties, a single value may be used.

A default 100 percent area use factor (i.e., 1) is often used in the initial exposure assessment of required element (6) of the Tier 2 SLERA if data to suggest otherwise are lacking. If an area use factor of 100 percent is assumed, then no further justification is required, but a rationale for the assumption is required. Since required element (6) is intended to be a conservative evaluation, the specific factor used should be near the conservative end of an appropriate range of values that match site conditions. Further justification an even less conservative value can be made in the evaluation under required element (7).

Caution is necessary when estimating a receptor's area use at affected properties that comprise a fraction of the receptor's home range. There may not be a direct relationship between the probability of exposure and the size of the affected

property relative to home range due to limited open space and variability in habitat throughout the receptor's home range. Also, the receptors may forage in impacted habitats from the affected property as well as in unimpacted habitats adjacent to the affected property. When applying an AUF to receptors in these situations, the person should avoid additional exposure adjustments to the food intake that would duplicate the adjustment already factored into the AUF. The *Handbook* provides species-specific home range values, etc. for selected wildlife receptors, and Sample et al. (1997) provides regression models relating home range to body weight for various receptor types. See also Issue Nos. 20, 21, and 30 in TCEQ (2005).

3.9.2.5 Body Weight

Body weights are generally reported in the literature as fresh weight, as would be obtained by weighing a live animal in the field. In addition to the *Handbook*, literature sources of wildlife body weights include Davis and Schmidly (1994), Dunning (1984, 1993), Burt and Grossenheider (1980), and Silva and Downing (1995). A representative (average) body weight for the ecological receptor should be selected and remain constant through both the initial and refined assessments.

3.9.2.6 Points of Exposure

The TRRP defines the point of exposure (POE) as 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, a plane, or an area within or beyond some location. In all cases, sampling and analytical methods should be documented in the affected property assessment report, including a discussion of the number of samples and where and when they were collected.

Soil

When estimating exposure for wildlife receptors, the appropriate medium depth should be reflected in the dose equation. More often than not, the ecological exposure-point concentration for surface soil (ground surface to 0.5 ft deep) will be used. However, the ecological exposure-point concentration for subsurface soil (0.5–5.0 ft deep) may be appropriate for evaluating burrowing wildlife and those that share their burrows. The person should also consider the appropriate soil depth when soil concentrations are used to estimate COC concentrations in wildlife food or prey. For example, earthworms may be present at depths greater than 0.5 ft during portions of the year, and some plants may take up COCs through the roots at depths greater than 5.0 ft.

Soil and Sediment

Should the bottom of a dry watercourse be evaluated as soil or sediment? The TRRP rule defines the material lying below surface waters, including intermittent streams, as sediment. As appropriate, the person should evaluate exposure both from the dry stream bottom and from sediment associated with intermittent streams. Hence, the person should evaluate exposure to land-based ecological receptors when the stream bottom is dry, and should perform normal sediment evaluations when the stream bottom is wet. The exposure duration for a particular

receptor can be adjusted to reflect the usual dry-wet cycles for the water body in question. The person may choose to evaluate one scenario or the other based on site-specific considerations. In that case, the person should discuss the rationale for not quantitatively evaluating the remaining scenario in the uncertainty analysis. The person should also see Issue No. 18 in TCEQ (2005).

Sediment

The TRRP also provides for a site-specific determination of the POE for ecological receptors exposed to sediment. The depth of the sample should target the aerobic layer which represents more recent deposition and is where most benthic infauna will occur. Although this biologically active layer is frequently considered to be the upper 4 inches (10 cm) of sediment, it may be considerably shallower in some environments. In any case, the POE specific to the site should be evaluated, since some receptors are known to burrow deeper.

The person will need to justify the sample depth used in the ERA based on sediment characteristics observed during sampling. Observations of differing color intervals, texture and consistency, and biological inclusions (worm tubes, evidence of movement) may help distinguish between the biologically active layer and deeper layers. These types of observations should be noted in the ERA to justify the depth interval sampled. Typically the sediment will exhibit a light brown silt layer on top, followed by a gray aerobic zone overlying a typically black anaerobic layer (TCEQ, 2012). The key is to sample a depth interval that will best represent expected exposure, including exposure to predators that feed on burrowing or tube-dwelling biota. For example, freshwater oligochaetes may burrow 4–15 cm into sediment (U.S. EPA, 1994b). Alternatively, Whitlatch (1982) as cited in Armstrong (1987) found that most polychaetes, amphipods, and bivalves were in the top 2–4 cm in estuarine mudflats. However polychaetes have been observed as deep as 20 cm in some waters after the sediments were colonized by deep-burrowing species (enteropneusts) (Armstrong, 1987). Similarly, angel-wing clams burrow 30 cm or more beneath the surface, depending on their size (Fotheringham, 1980). Deeper sediments should also be evaluated on a site-specific basis when COCs present in sediment below the biologically active layer are likely to be exposed at some future date from either natural causes (e.g., hurricanes) or anthropogenic events (e.g., dredging).[8]

Other sampling depths may be relevant depending on the circumstances in question. For example, samples for analyses of acid volatile sulfide (AVS) and simultaneously extracted metal (SEM) should be collected specifically in the oxic interval. Thus, where sediment samples are collected at deeper depths to account for organisms that may have burrows in the anaerobic, black, sulfide-rich sediment, separate samples should be collected for AVS and SEM analyses in the more shallow oxic zone where the density of benthic organisms is greatest. Alternatively, if the purpose of the sampling effort is to evaluate COC uptake through

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

methylation, it is reasonable to limit sampling to the upper few centimeters down to the anoxic-oxic interface. The person should also see Issue No. 23 in TCEQ (2005).

Surface Water

For toxic COCs such as metals and organics, TSWQS 307.9(c)(4) specifies that numerical 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 ft 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 ft below the water surface are acceptable.

Groundwater Releases to Surface Water and Sediment

It is not appropriate to sample surface water for groundwater releases to surface water except where sampling is needed to confirm a property-specific dilution factor or where the PCLE exceedance zone discharges across significant areas of a surface water body. Normally the groundwater–to–surface water pathway is evaluated using sample data from one or more groundwater-monitoring wells immediately upgradient of the zone of discharge to surface water in accordance with the TRRP rule [30 TAC 350.51(f) and 350.37(i)]. More detailed discussion of the groundwater–to–surface water pathway appears in *Determining PCLs for Surface Water and Sediment* (RG-366/TRRP-24). Where allowed by the TRRP rule [at 350.75(i)(4)], a dilution factor can be applied to the groundwater concentrations used in the intake calculations for a wildlife receptor or for benchmark comparisons. This approach is further described below. Consult RG-366/TRRP-24 for guidance on determining the dilution factor (default or property-specific).

The TRRP rule gives direction [350.74(i)(4)] on using a dilution factor in determining the groundwater PCL protective of surface water. Under this approach, the dilution factor is applied to the surface water RBEL protective of surface water. The rule also states that compliance with the groundwater PCL protective of surface water will be determined via a groundwater monitoring well placed immediately upgradient of the zone of groundwater discharge to surface water.

1. How is the groundwater PCL determined for a receptor or a water-column receptor?

In this case, the ecological PCL should be used with the application of the appropriate dilution factor (DF) to derive the groundwater PCL protective of surface water:

$$Groundwater\ PCL = \frac{Surface\ Water\ PCL\ (for\ wildlife\ receptor\ \ or\ TSWQS)}{DF}$$

2. When is a dilution factor **not** appropriate? A dilution factor is not appropriate if:

- COC concentrations measured at the groundwater monitoring wells do not exceed the appropriate surface water PCL, or
- a specific COC is listed on the current 303(d) List as impairing the nearest classified segment at or downstream of the affected property. The current 303(d) List is available online at <www.tceq.texas.gov/goto/tirswq>.

3. What surface water data should be used for the ERA calculations for an ecological receptor?

If the source of COCs in surface water is groundwater, ambient surface water data should not be used for this pathway. Rather, groundwater-monitoring-well data should be used with the application of a DF (default or property specific). For example, assume that selenium is a COC in groundwater being discharged to surface water.

(a) Determine the groundwater exposure-point concentration for selenium. For example, assume this value is 30 µg/L.

(b) Modify this concentration with the appropriate DF (groundwater concentration × DF). For example: 30 µg/L × 0.15 = 4.5 µg/L.

(c) Enter the value from b into the ERA as the **surface water** exposure-point concentration for the ecological receptor.

(d) Determine the surface water ecological PCL, where appropriate. If the value in b is less than the PCL, no further action is necessary. Assume a PCL of 2.1 µg/L is calculated as the ecological surface water PCL.

(e) Once a surface water ecological PCL is established, determine the groundwater PCL protective of surface water using the formula indicated above. Continuing the example:

$$Groundwater\ PCL = \frac{Surface\ Water\ PCL(for\ wildlife\ receptor\ or\ TSWQS)}{DF} = \frac{2.1}{0.15}\ 14\ \mu/L$$

Compare the groundwater PCL protective of ecological receptors with the selenium groundwater concentration and perform a response action as necessary.

4. When should ambient surface water samples be collected and evaluated in an ERA?

Certainly, ambient surface water data can be and should be evaluated in the ERA wherever surface water impacts result from multiple pathways (spills, runoff) or historical contamination is present. The use of the well data and the application of a dilution factor only applies for the groundwater–to–surface water scenario.

Where groundwater is released to surface water, releases to sediment should also be evaluated. Generally, sediments should be sampled in accordance with the preceding text. However, sampling should be modified to account for a site's unique fate and transport considerations, e.g.:

- Sample collection and analysis should be planned to minimize loss of volatiles where volatile COCs are present in the groundwater release.
- Pore-water sampling should be considered where groundwater COCs are principally expected to occur in the dissolved phase.
- Sediment samples should be collected in the bottom or side walls of the channel where the plume interface is known or expected to occur based on site hydrogeology.
- The person should consider if the overall release to surface water and sediment is patchy. The geologic layer that intercepts the surface water may be larger than the actual seepage areas or discrete discharge zones due to the nature of the sediment materials and geologic features. See also Issue No. 29 in TCEQ (2005).

3.9.2.7 Hot Spots

A *hot spot* is a discrete area of substantially elevated COC concentrations relative to the surrounding area. No standard approach has been developed for defining such areas. 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.

A variety of statistical and geostatistical techniques can be used to evaluate the possible presence of hot spots. Gilbert (1987) presents a method for locating a hot spot by sampling on a square, rectangular, or triangular systematic sampling grid. Geostatistical methods such as factorial kriging analysis (Goovaerts, 1992) have also been used to identify hot spots. The analytical tools included in the Spatial Analysis and Decision Assistance software include integrated modules for visualization, geospatial analysis, and statistical analysis. This software is available free of charge at <www.sadaproject.net/>.

In addition to statistical techniques, cost-benefit approaches have been used to determine the action level below which the incremental costs of corrective action are disproportionate to the incremental benefits (Graves, 1997). Determinations of what constitutes a hot spot and the appropriate response are ultimately risk-management decisions specific to the site.

3.9.3 Livestock and Crops

Ecological risk assessment can extend from evaluation of COC occurrence in native plants and animals to crops and domestic livestock. Exposure scenarios may involve area wildlife feeding on crop plants that have taken up COCs. Examples include mallard ducks or mourning doves feeding in fields of rice, milo, or similar seed-bearing cultivated plants that may be produced on soil that contains COCs or irrigated with water that contains COCs. Besides a potential risk to wildlife feeding directly on affected crops, ecological risk can extend into higher trophic levels as a result of exposure of wildlife to crops that contain COCs. A mourning dove feeding on crop plant products that contain COCs can in turn be eaten by a raptor.

Livestock—including swine, horses, cattle, sheep, goats, and poultry—may be vulnerable to exposure to COCs in environmental media or the diet. The implications of exposure are likely to be of lesser importance to wildlife than to humans who consume these animals. For example, cattle are especially sensitive to molybdenum poisoning when grazing on pastures where copper and inorganic sulfate are deficient, whereas Eisler (1989) showed wildlife tested to be at least 10 times as resistant. Opportunistic predation of livestock by wildlife occurs, but is not typically expected to result in a transfer of COCs into the predators at concentrations sufficient to cause ecological risk. More likely consequences of uptake of COCs by livestock are livestock-related or human health related concerns. If, for example, a COC affects reproductive capability in laboratory mammals, reproductive capability in mammalian livestock might be affected. Impaired reproductive capability in economically valuable domestic livestock such as bulls can substantially affect their market value and dramatically reduce the birth of calves. Thus, the owners of affected livestock can suffer financial loss.

Water and soil quality standards and criteria for protection of livestock and crops are in existence or are being developed by regulatory agencies. For example, the Canadian Council of Ministers of the Environment has developed an integrated compendium of guidelines <ceqg-rcqe.ccme.ca> that includes criteria appropriate for agricultural uses of water and soil (e.g., watering of livestock and use of soils as cropland). Although existing standards for livestock and crop protection likely focus on protection of humans as the ultimate receptor, valuable information in such documents may be extrapolated to ecological receptors.

The same uptake equations used for predicting COC concentrations in aboveground and belowground vegetables may be used to estimate COC concentrations in crops. Similarly, uptake equations for herbivorous mammals may be used for virtually all mammalian livestock. Uptake in poultry may be estimated using the same equations appropriate for modeling uptake in herbivorous and omnivorous birds. Food and water intakes for a variety of livestock (beef cattle, dairy cattle, sheep, goats, swine, chickens, and laying hens) are available from *Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities* (U.S. EPA, 2005h). This document is available online at <www.epa.gov/epawaste/hazard/tsd/td/combust/risk.htm#hhrad>. Because human health–oriented risk assessments for hazardous waste combustion facilities routinely include evaluation of risk to humans due to ingestion of COCs taken up by livestock and crop plants, information specific to uptake in crops and livestock is available in this document. In addition, uptake is addressed to varying degrees in the various guidance documents referenced in **3.9.2.1.1–2**.

3.9.4 Bioavailability

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

and biological processes evident in air, water, soil, and sediment, as well as biological factors present in the bodies, organs, tissues, or cells of exposed organisms that act to change that organism's rate of COC exposure. For example, the form of a COC can affect the degree of stomach absorption—e.g., soluble (barium sulfate) versus insoluble (barium carbonate) metal compounds. For aquatic receptors, the bioavailable fraction of COCs is closely related to the concentration dissolved in water. Thus, exposures in ERAs are likely to be over-predicted when bioavailability is assumed to be 100 percent. It is recommended that if appropriate data are available for site conditions, modifications to media concentrations or the dose equation (or both) can be incorporated to reflect site-specific bioavailability—typically, in required element (7) of the Tier 2 SLERA, but some obvious considerations may be factored in during required element (6), if they can be justified as accommodating assumptions that would otherwise be too conservative, as discussed below.

Although regarded as a key factor in generating realistic and quantitative exposure estimates for ecological resources, there are few universally accepted ways to quantify bioavailability. Hence a default of 100 percent is often used in required element (6) of the Tier 2 SLERA if contrary data specific to a chemical and a site are lacking. If a bioavailability of 100 percent is assumed, then no further justification is required. However, the rationale for the use of bioavailability factor less than 100 percent would need to be provided in the ecological risk assessment. Since required element (6) is intended to be a conservative evaluation, the specific factor used should tend toward the conservative end of an appropriate range of values for the chemical that match site conditions. Justification for the use of a less conservative value can be made in the required element (7) evaluation.

Common adjustments that may be considered even in required element (6) might be surface water calculations that recognize the dynamic nature of physiochemical properties on the amount of soluble metals (e.g., water hardness and pH). Similar efforts are under way to account for bioavailability from sediments using AVS and SEM for metals and carbon partitioning for organics. Such adjustments may be appropriate for exposure calculations in required element (6). For soils, bioavailability research has been limited to plant nutrient uptake and food chain bioaccumulation. The result is few rules of thumb, and few approaches that may be considered for required element (6).

Three layers of analysis can be distinguished when evaluating biological availability in its totality: environmental availability, environmental bioavailability and toxicological-pharmacological bioavailability (Figure 3.13). These mechanisms can support site-specific evaluations, and serve as the basis for exposure adjustments conducted under required element (7) in the Tier 2 SLERA and the Tier 3 SSERA.

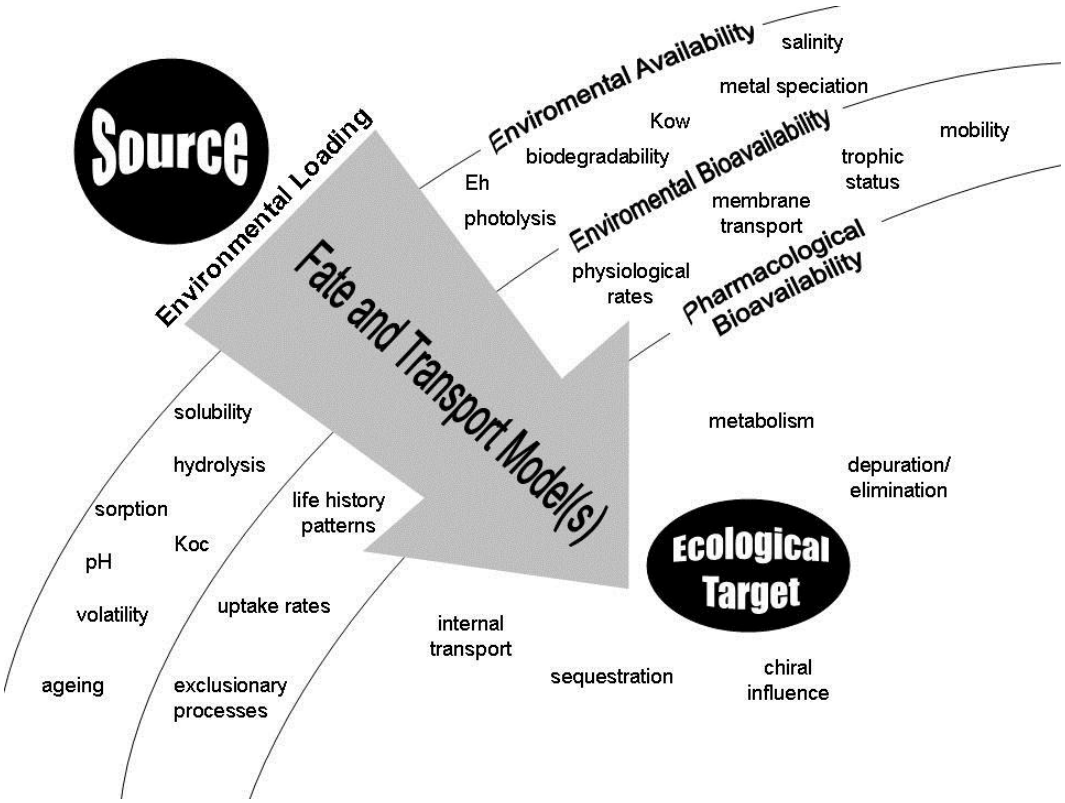


Figure 3.13. Types of biological availability.

Environmental availability depends on the chemical and physical properties of the compound and the environment to which it is released. This aspect of bioavailability reflects a number of environmental-fate factors operating in the media (air, water, soil and sediments, and food) in which a COC occurs. The physical and chemical properties of the COC interact with physical and chemical properties of the environment, affecting the disposition of the COC in terms of compartmentalization, chemical form, concentration (activity), and persistence.

Environmental bioavailability depends on environmental availability and the interactions of an organism with its environment throughout its life history. Plants and animals may be exposed singly, intermittently, or continually during their life cycle. Each life state may differ exposure route and intensity, uptake rates, and transfer efficiency. Exposure occurs at the interface between the media and the biological membranes contacting the contaminated media.

Toxicological and pharmacological bioavailability depends on environmental bioavailability and the internal functions of the receptor. It is the portion of the dose that reaches cells, tissues, or organs, and initiates a biological response. This is a function of specific metabolic and biochemical processes and capabilities at various life stages that influence effective intracellular concentrations.

If appropriate data are available for site conditions, modifications to media concentrations or the dose equation can be incorporated to reflect site-specific bioavailability based on these mechanisms. Factors that should be considered in making site-specific determinations include:

- *Environmental factors such as:*
  - factors affecting the physical or chemical state, solubility, or sorption of a COC in the environment and the environmental media with which it becomes associated;
  - factors such as temperature, salinity, pH, and eH which influence the chemical form (speciation) in the environmental media where the COC resides; and
  - physical and chemical interactions between the COC and the environmental media that affect sorption and organic partitioning.
- *Environmental bioavailability factors such as:*
  - dispersal patterns and behavioral activity, physiological rates (e.g., rates of respiration, water intake), and trophic status (e.g., uptake or consumption of nutrients or food); and
  - chemical exchange rates, active or passive uptake processes, and membrane transport.
- *Toxicological and pharmacological bioavailability factors such as:*
  - an organism's metabolic capabilities and functions which reduce COC exposure at vulnerable organs, tissues, or cellular structures via specific degradation pathways;
  - processes that lead to COC sequestration in nonsensitive organs, tissues, or cellular structures before critical body burdens are attained; and
  - depuration or elimination rates.

All of these considerations would require full documentation from literature sources and site data as to their applicability to the affected property under consideration.

3.9.5 Characterization of Ecological Effects

For each COC with a complete exposure pathway that is not excluded by a comparison to ecological benchmarks, a toxicity reference value (TRV) should be developed from published studies or developed for potential receptor species. The TRV is used in the ecological hazard quotient methodology described in required elements (6–9) of 30 TAC 350.77(c) and discussed in **3.10–13**. Two quotients are described in the rule, one based on "no unacceptable risk" derived from using NOAEL endpoints [required element (6)], and a second based on "less conservative assumptions" derived from LOAEL endpoints [required element (7)]. An uncertainty analysis of the quotients [required element (8)] is required to justify whether a medium-specific PCL [required element (9)] should be developed. Consequently, two TRV values will need to be derived, one based on NOAEL or NOAEL-like endpoints reflecting no unacceptable risk, and one based on LOAEL or LOAEL-like endpoints reflecting less conservative exposure assumptions. For clarity, the resulting values will be distinguished as TRV_{NOAEL} and TRV_{LOAEL}.

Ecological effects of concern are those that can impact populations, such as development, reproduction, and survivorship. NOAEL endpoints reflect the highest

exposure level that causes no statistically significant difference in effect compared with the controls. LOAEL endpoints reflect the lowest exposure level shown to produce similar adverse effects in a potential receptor species. Some alternative endpoints that may serve as reliable estimates of a NOAEL or LOAEL are discussed below and in Appendix C. The TRV_{NOAEL} and TRV_{LOAEL} are derived from a review of NOAEL and LOAEL endpoints or estimates of such endpoints, respectively.

Significant tasks in the Tier 2 SLERA effects characterization are: (1) collecting data from the literature, (2) critical evaluation of the data, (3) selection of appropriate endpoint values, and (4) extrapolation between tested systems and the major feeding guilds that are the focus of the screening risk assessment. See also Issue No. 6 in TCEQ (2005).

3.9.5.1 Collecting Data from the Literature

In the ideal case, data from the literature will be available for a COC with the species selected to be representative of the feeding guild supported by habitats on the affected property for each exposure pathway that is complete or reasonably anticipated to be complete. In such a case, the overall quality of the study should be evaluated and, if the study is acceptable, the NOAEL and LOAEL endpoints should be determined from the report and used as TRVs in the subsequent risk characterization.

If multiple literature values are available for the same species and same endpoint, an aggregate TRV_{NOAEL} may be obtained as the geometric mean of the NOAEL values and, similarly, an aggregate TRV_{LOAEL} derived as the geometric mean of the LOAEL values. This procedure, used by the EPA in deriving ambient water quality criteria (Stephan et al., 1985), improves the reliability of the values by increasing the database used to derive them.

The most readily accessible information sources for ecotoxicity data are computerized databases such as ECOTOX, which integrates single-COC toxicity data from the following former databases: AQUIRE (aquatic organisms), TERRATOX (terrestrial wildlife), and PHYTOTOX (terrestrial plants). The ECOTOX system can be accessed free of charge at <cfpub.epa.gov/ecotox/>. Other databases, such as the Oil and Hazardous Materials / Technical Assistance Data System (OHMTADS), Chemical Hazards Response Information System (CHRIS), and the TOXNET Toxicology Data Network at <toxnet.nlm.nih.gov/> summarize available information. In addition, Suter et al. (1993) identify a number of other compendia of test results. Also, the RATL is available at <publications.gc.ca/collections/Collection/CW69-5-357E.pdf>.

Searching by COC and test species is the more traditional approach. Compilations or synoptic reviews of specific COCs may be available, such as the series by the U.S. Fish and Wildlife Service or monographs prepared for trade groups (see for example, Eisler, 1985; API, 1998; API, 1999; and Markarian et al., 1993). Another accessible source of ecotoxicity data is the Oak Ridge National Laboratory's documents presenting ecological screening benchmarks for aquatic species (Suter and Tsao, 1996), sediment-associated biota (Jones, et al., 1997), wildlife (Sample et al., 1996), and terrestrial plants (Efroymson, et al., 1997), available at the ORNL website <www.esd.ornl.gov/programs/ecorisk/ecorisk.html>.

One concern with use of documents summarizing data or presenting reference values is the nature of the data collection and screening used to derive the values. Compendia of values may in fact be tertiary collections of values derived by others. The original papers summarized in these compendia should be evaluated whenever possible to ensure acceptable data quality (see discussion that follows) and relevance to the ecological PCLs defined in the rule (e.g., impairment of microbial processes would not be useful information). For example, Friday (1998) observed that numerous soil and sediment screening values reflected microbial processes rather than lethality, growth, and reproduction endpoints.

Reviews of original literature should be performed, particularly for references from which critical endpoint values are derived. Absence of one of the following items may not be sufficient to reject the endpoint, but absence of many should lead to serious concerns about the overall quality assurance. High-quality publications and references should include:

- Information about the study protocol used and whether it conforms to a standard test method, such as those published by the EPA, the Organization for Economic Cooperation and Development (see <www.oecd.org/home/0,3305,en_2649_201185_1_1_1_1_1,00.html>), the ASTM <www.astm.org>, or others.
- Statements about what concentrations were tested and the exposure regime.
- Clarification about whether concentrations were measured or nominal.
- Sufficient data to permit a reader to arrive independently at the stated conclusions.
- Summaries of the statistical analyses performed, the resulting summary statistics, and their significance levels. If no statistically significant results were observed, extreme caution applies to any conclusions based on “trends.”
- Quality assurance information about test organisms, such as species verification, age and condition, and reference toxicant performance.

3.9.5.2 Critical Evaluation of the Data: Data Quality Evaluation

Not all available data may be of acceptable quality or relevance. Simple compilations of test endpoints usually do not permit the reader to evaluate the quality of the study behind the reported value. Consequently, an independent evaluation of data behind the critical value (the test result used to set the TRV) may be necessary (Durda and Preziosi, 2000; Clark et al., 1999). If formal data-quality objectives have been developed, they may be used to exclude studies of insufficient quality.

If procedural concerns are evident in the descriptions of test methods, then closer scrutiny is appropriate because the test results will probably be very dependent on the procedural details. Classic examples of situations where procedures can significantly affect endpoints include aquatic toxicity tests of poorly water-soluble materials or terrestrial tests of highly volatile materials.

Peer review is another check on data quality. Most journals publish technical articles only after expert review. However, such review may not evaluate the factors

that could affect interpretation of the study results. Consequently, even peer-reviewed published articles may be suited to the development of ecological TRVs in this context.

How a test chemical was administered or measured may matter in selecting the most relevant studies. Different salts, chelators, solvents, or carrier systems may have important effects on the apparent sensitivity of a species. For example, metals are generally less toxic if not dissolved, or if chelated. Similarly, organic COCs that have been in contact with organic soils are typically less toxic than freshly added COCs. Different salts of a metal may have characteristically different toxicities (e.g., a chloride salt is less toxic than a sulfate salt) (API, 1998).

Poorly water-soluble or highly volatile COCs or mixtures pose specific challenges to study protocols and interpretation. Studies of petroleum-related materials expressed as the amount of test material added per volume water gave significantly different toxicity endpoints than studies of actual concentrations (Markarian et al., 1993). Similarly, studies of styrene with close attention to maintaining constant exposure by flow-through or sealed systems led to different endpoint values than studies allowing natural volatilization or polymerization (Cushman et al., 1997). Inattention to how the test material was introduced and maintained in the test system can lead to significant errors in applying those endpoints to a different situation. Consequently, it may be appropriate to exclude certain studies in favor of others because of how the test COC was introduced or measured in a test system.

The use of a standard test protocol should also be considered in deciding whether to include a study. Test methods such as presented by the EPA, ASTM, or OECD have been developed to improve the reliability and comparability of test results. Nonstandard protocols may have been appropriate from the researcher's perspective, but such studies require careful evaluation to ascertain whether the modified protocol has affected the reliability, credibility, or comparability of the endpoints. In some cases, the modifications are so severe that the study should be excluded.

3.9.5.3 Selection of Appropriate Endpoint Values: Ecological Relevance

Not all literature values will present useful information, so one task is to exclude those results with measured endpoints that do not help evaluate the relevant ecological endpoint. Behavioral endpoints, such as an aversion to a COC, might suggest altered patterns of habitat use, but may not readily correspond to the ecological endpoints of reproduction, development, and survival.

Similarly, biomarkers have been widely used as evidence of organism exposure to a stressor, but have not been accepted as good endpoints of ecological effects. Biomarkers are a type of endpoint obtained using molecular, genetic, biochemical, histological, or physiological techniques. While such endpoints suggest that the organism has been exposed to a stressor, the response or biomarker cannot be easily extrapolated to endpoints reflecting impacts at a population level, such as survivorship. Consequently, scientific studies would be needed to demonstrate that a particular biomarker quantitatively reflects exposure and predicts significant ecological effects. At this time, it seems unlikely that biomarker endpoints can be generally useful in Tier 2 ecological effects characterizations.

Two different approaches to choosing among relevant data have been taken in cases where several studies of a particular COC are available. One approach is to survey all the studies in the literature and include all studies of acceptable quality, then to derive a value from the more stringent (lower values) part of this distribution. In many cases, the most stringent endpoint (lowest value) is used, while in others, several or all relevant data are used to statistically define a criterion.

The second approach—the one advocated by the TCEQ—is to select the most relevant study available, based on professional judgment or other specific criteria, and use the results of that study in preference to others. This approach is commonly used in assessment of studies related to specific human-health concerns, such as carcinogenicity. The non-preferred studies may be used to corroborate conclusions or to contribute to a weight-of-evidence case, but the endpoint is derived from the preferred study. The study selected (species and endpoint) should reflect its relation to the Texas feeding guilds supported by the habitat at the affected property, and the endpoints most likely to affect populations (e.g., development, reproduction, and survivorship). Primary references are preferred and should be included with the test organism, effect, and dose. TRV selection is not an issue where regulatory criteria (e.g., state or federal) exist (see **3.5.1**). In such cases, the TRV may not exceed the criteria and the TRV may equal the ecological benchmark value used previously.

The advantage of this approach is that a single TRV is derived as most relevant and so need not be replaced by a less conservative TRV during the refined exposure assessment (see **3.9.1**). A similar discussion on representative TRVs appears in **3.10**.

3.9.5.4 Extrapolation to Texas-Specific Ecological Receptors

Though, ideally, the literature would report endpoints for a member of the feeding guild supported by the habitat at the affected property (i.e., if there were a complete match between test species and ecological receptor, and between route of administration in the test and environmental exposure, and between measured endpoint and the development, reproductive, and survival endpoints emphasized herein). However, few available test studies will exactly match the representative species using the pathways relevant to an affected property.

Models of several types are used to address the mismatches. Use of models requires making assumptions and simplifications that always result in questions about the estimates produced by the models. Use of estimates should be discussed in the uncertainty analysis of the ERA (see **3.12**).

Data selected should preferably be for test species that resemble species in the Texas feeding guilds. For example, taxonomic similarity is a widely used criterion. Data from a species more closely related to the species occurring on the affected property should be preferred to data from a more distant species. The mode of exposure (e.g., dietary or media) should reflect the pathway analysis for the affected property. The measured effect should correspond to the development, reproductive, or survival endpoints.

The feeding guilds used in the site conceptual model (see **3.7**) can form the basis for screening for the most relevant taxonomic or scaling extrapolations, with emphasis on data that permits extrapolating between members of the same guild. For example, when considering data to describe effects to a raptor, more emphasis should be given to studies of other raptors than to studies of granivorous birds.

Where data are insufficient for data selection based on feeding guilds, other approaches have been investigated. Three such approaches—allometric estimates of NOAEL, uncertainty factors, and species sensitivity distributions—are discussed below. Regardless of the approach, toxicity data should not be extrapolated across taxonomic classes of receptors (e.g., birds to mammals), with the possible exception of extrapolating from birds to reptiles and amphibians when information is completely lacking in the scientific literature and when threatened or endangered species are exposed.

Allometric Estimation of NOAEL typically incorporates the use of body-scaling parameters (i.e., body mass) to estimate a toxic concentration for a class of organisms (e.g., the toxic dose to mammals). Suter and others have evaluated data sets from multiple taxonomic groups to derive regressions or distributions describing how the endpoints for different taxa might vary (Suter, 1993). According to Sample and Arenal (1999), the endpoint for an avian or mammalian guild member can be estimated using an equation of the form:

$$NOAEL_w = NOAEL_t \left[\frac{bw_t}{bw_w} \right]^{(1-b)}$$

Where:

$NOAEL_w$ = NOAEL for avian or mammalian wildlife species

$NOAEL_t$ = NOAEL for avian or mammalian test species

bw_t = body weight of avian or mammalian test species

bw_w = body weight of avian or mammalian wildlife species

b = allometric scaling factor that is specific to either birds or mammals.

The typical scaling factor b had been previously estimated at about 0.75 (Sample et al., 1996). Sample and Arenal (1999) evaluated body-scaling parameters for COCs from avian and mammalian data to determine whether the conventional scaling factor 0.75 fit the available toxicity data. While body weight accounted for much variability in the acute toxicity of COCs to birds (r^2 = 0.76) and mammals (r^2 = 0.89), they found little basis for scaling factors of 0.66, 0.75 or 1.0. They recommend use of COC-specific allometric scaling, and present parameters for about 180 COCs, though the data includes mostly pesticides. In the absence of COC-specific parameters, they suggest a scaling factor of 1.2 for birds and 0.94 for mammals. The TCEQ accepts these recommendations. When a COC-specific scaling factor is available from Sample and Arenal (1999) it should be used; otherwise, the default scaling factors (b) of 1.2 for birds and 0.94 for mammals should be used.

Uncertainty factors (modified from U.S. EPA, 1999a). The second approach often used to derive an estimated NOAEL involves uncertainty factors (UFs). When knowledge of the toxicity of a COC is incomplete, UFs can reduce the likelihood of underestimated risk. Historically, UFs have been used for various extrapolations, and their applications reflect policy to provide conservative estimates of risk (Chapman et al., 1998). As discussed below, UFs may reduce the probability of underestimating ecological risk from exposures to releases of COCs. A toxicity value is multiplied by a UF to produce a TRV reflecting a NOAEL for a chronic-exposure duration.

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

Specifically, UFs should be used to account for extrapolation uncertainty due to differences in test endpoint and exposure duration:

- Test endpoint uncertainty—extrapolation from a non-NOAEL endpoint (e.g., LOAEL, LD_{50}) to an NOAEL endpoint
- Duration uncertainty—extrapolation from a single dose, acute, or subchronic duration to a chronic duration
- Except as noted above for irregular toxicity data, the following UFs (Calabrese and Baldwin, 1993) may be used to convert a toxicity test endpoint to a TRV that is equivalent to a chronic NOAEL:
    - A chronic LOAEL should be multiplied by a UF of 0.1 to convert it to a chronic NOAEL.
    - A subchronic NOAEL should be multiplied by a UF of 0.1 to convert it to a chronic NOAEL.
    - An acute lethal value (such as an LC_{50} or LD_{50} should be multiplied by a UF of 0.01 to convert it to a chronic NOAEL.

Alternatively, UFs may also be obtained from U.S. ACE (2010) and Wentsel et al. (1996).

The allometric estimate and the UF approaches are often combined when deriving an estimated NOAEL or LOAEL endpoint. For example, if a chronic NOAEL is needed for a raccoon for a particular COC and all that can be found in the literature is a chronic LOAEL for a rat, then the extrapolation would be to convert the LOAEL into a NOAEL by multiplying it by 0.1 and then to use the body weights of the rat and raccoon to adjust the NOAEL.

Species Sensitivity Distributions. The third approach, although not as common as the other two, is still viable and centers on the use of sensitivity distributions. Where a number of species have been tested with a common toxicant (e.g., many

metals), a distribution of endpoints can be prepared showing the range of sensitivities observed. This pattern represents a cumulative frequency distribution. The distribution may then be fitted to a mathematical expression, such as a logistic curve using regression techniques, and the resulting curve used to estimate the endpoint for the more sensitive species. The endpoint for the fifth percentile could be estimated, and taken as a lower bound of the endpoint for the receptor of concern. Hence, an estimated TRV for a Texas feeding-guild species would be derived that is expected to be conservative (the true TRV for that species is not less than the estimate) 95 percent of the time, such that no more than 5 percent of species NOAEL values occur at lesser concentrations. This approach has been described as the *hazardous concentration—5 percent* (HC5) or the *potentially affected fraction* and has been proposed for use in the Netherlands (Van den Brink, Posthuma, and Brock, 1999; Van de Meent and van Straalen, 1999; and Ownby et al., 1999). The resulting estimated NOAEL would then be used as the TRV_{NOAEL}.

3.9.5.5 Tissue Residues

As further discussed in **3.13.2**, equilibrium partitioning methods may be used to predict tissue burdens, which can subsequently be used to characterize ecological effects in a Tier 2 SLERA. At present, two fundamental approaches form the basis for use of tissue residue concentrations. While these approaches are evolving and lack universally accepted definitions, herein they will be referred to as:

1. the empirical ("diagnostic" approach)
   A. the experimental or bioconcentration-based ("critical body residue" approach)

The differences between the two approaches lie in their origins.

3.9.5.5.1 Empirical Approach

The empirical approach is based on monitoring data where concentrations of COCs were measured in field-collected or laboratory-exposed organisms or organs. The emphasis of this approach has historically been on *monitoring* for the evaluation of the appropriateness of open-water disposal of dredged material, and *diagnosis* of wildlife deaths and abnormalities. However, in practice, it is difficult to prove that residues of field-collected organisms are linked to adverse effects. Nevertheless, Beyer et al. (1996) have recommended certain residues associated with adverse effects in birds and mammals, and others (Jarvinen and Ankley 1999; U.S. ACE, 2007) have compiled residue databases based on bioassays of fish and aquatic invertebrates that, in some cases, may be used to link residues to adverse effects. In addition, the California Water Resources Control Board (CWRCB, 1990) has compiled residue data for live coastal and freshwater fishes and categorized the data by reference to the median value found in all fish for each chemical. However, since these fish were collected alive, no adverse effects are linked to these residues. Although no specific database of tissue residues for terrestrial invertebrates, vertebrates, or vascular plants is presently available, several sources of residue-effect data along with some preliminary tissue guidelines are accessible (e.g., CCME 1999b; U.S. EPA 2001; Kabata-Pendias and Pendias, 1992; Beyer et al., 1996).

3.9.5.5.2 Critical Body Residue Approach

The emphasis in this area of work is whether a constant body burden (on a molar basis) can be associated with lethality[9] (i.e., critical body residue, CBR). Most of this work has been conducted in aquatic animals with nonpolar organics that exert toxicity through a common physiological pathway, narcosis (e.g., McCarty et al., 1992; Connell and Markwell, 1992; van Wezel et al., 1995a), although some polar organics are also amenable to this approach (Smith et al., 1990; van den Heuvel, 1991; van Wezel et al., 1995b).

The critical body residue postulate assumes that a given residue is always associated with a given toxicological response, e.g., lethality. In fact, for nonpolar organics, a nearly constant CBR (e.g., 3 to 6 mmol/kg wet weight) has demonstrably caused death by narcosis, and sublethal effects have also been linked to lower residues (Mortimer and Connell, 1995). The emphasis for this approach has been on **prediction** rather than diagnosis or monitoring.

From these studies, quantitative structure-activity relationships (QSARs) have been constructed that allow prediction of residues and adverse effects based upon the octanol-water partition coefficient (K_{ow}) and the lipid content of the organism. The QSAR is a refined version of a bioaccumulation factor with the additional advantages of (a) allowing prediction of residues for any nonpolar organic chemical and (b) linking those residues to an adverse effect. In turn, the predicted residue can be used in conjunction with empirical or derived biota-sediment or water accumulation factors (e.g., Wong et al., 2001; U.S. EPA, 1997b) to estimate whether field-collected water or sediments are likely have adverse effects. Trophic modeling approaches (e.g., Thomann, 1989; Thomann et al., 1992; Gobas, 1993) can also predict the tissue burdens of chemicals of concern in higher-trophic-level receptors and estimate risk by comparing the results to the available residue-effects databases or critical body residues developed as discussed above.

Virtually all investigations of the effect of narcotic-behaving chemicals are based on acute and water only exposures. Although the CBR should theoretically be constant regardless of the exposure pathway, there could be different CBRs for water and diet. The other significant issue for the CBR approach is the assumption that the acute-to-chronic toxicity ratios (ACR) observed for water exposures used as the basis for this approach are indeed functional for the CBR—that is, whether the acute and chronic toxicological effects for narcosis vary as the ACR would predict. These have been used in attempts to establish chronic CBRs (Shepard, 1998; Di Toro et al., 2000; Di Toro and McGrath, 2000) but such a relationship has not been empirically validated.

The person should also consider the measure of effect for an ERA during any data selection for tissues thresholds. It has been shown that the CBR approach can work well in a context of mixtures and individual nonpolar organics using lethality or growth as the endpoint (e.g., McCarty et al., 1992; Mortimer and Connell, 1995; Di Toro and McGrath, 2000). However, this assumes additivity or at least equivalency, which for some endpoints (e.g., cancer or tumor formation) is an invalid assumption (e.g., for PAHs—see Reeves et al., 2001). The relationship

[9] As this field progresses, other, less extreme, endpoints may prove useful.

between the additivity of mixtures for lethality may or may not be the same for endpoints other than narcosis. In fact, CBR may decrease over time, perhaps as the result of the metabolism of nonpolar organics to polar metabolites.

CBR is unlikely to be useful for predicting the effects of metals on aquatic organisms for at least five reasons: (1) different organisms have different residues of naturally-occurring micronutrients (e.g., copper, zinc, chromium) and these concentrations fluctuate over time and with reproductive and nutritional status; (2) metals, such as cadmium and mercury, bioaccumulate naturally over time, so that older organisms will have higher concentrations than younger organisms; (3) if exposure is low enough, the metal residue will be compensated for without measurable adverse effect (e.g., mercury in swordfish); (4) metals such as copper cause toxicity to aquatic organisms, by reacting at the gill surface and, at high concentrations, there may not be time for a measurable change in residues to occur before death occurs; and (5) exposure to sublethal levels of metals results in the production of metal-binding proteins (i.e., metallothionines) that can alter toxicity.

There is a general bias towards using sediment and soil concentrations over CBR or measured residues in biota, primarily due to simplicity of measurement: it is much easier (and generally cheaper) to collect soil, sediment, and water than to collect biological tissue samples. But use of the tissue residue approach can provide another useful line of evidence in screening and monitoring programs because of its applicability to: alternate receptors associated with sediment or soil contamination (e.g., fish, herbivorous insects); integration of multiple exposure pathways and evaluation of irregular exposure concentrations and duration of exposures; and site-specific field validation of ERA assumptions.

In summary, equilibrium partitioning methods may be used to predict tissue burdens, which can subsequently be used to characterize ecological effects in a Tier 2 SLERA. The two fundamental approaches discussed here have their limitations and may have substantial uncertainty. Before even considering the use of either the diagnostic approach or CBR approach, it should be carefully established whether this approach is consistent with the identified assessment endpoints and measures of effect for the ERA. The CBR approaches can be effective tools in a weight-of-evidence evaluation.

3.9.6 Special COC Classes

Several classes of COCs commonly detected during affected property assessments require unique analytical techniques, interpretation, analysis, or receptor identification. The following discussion, while not comprehensive, highlights some of these issues and provide discusses their incorporation into the Tier 2 requirements.

3.9.6.1 Metals

The chemical form (or *species*) of a metal is important in determining its toxicity. For example, chromium III is much less toxic than chromium VI. Additionally, certain metals can exist in organic forms (e.g., methylmercury and alkyllead) which are generally more toxic than their elemental forms. Determination of a metal's species, however, is often prohibitively expensive and in some situations a chemical

speciation model may be appropriate. In the absence of compelling information concerning the speciation of a particular metal, assume that the most toxic form is present.

The quantity of metals in an abiotic medium is not always indicative of the toxicity of that medium to biota. The fraction of the total metal concentration which is bioavailable is usually a better indicator of toxicity. Factors that can influence the bioavailability of a metal include organic carbon, cation exchange capacity, pH, sulfides, and water hardness. For aquatic organisms, the dissolved, or soluble fraction of a metal may be a better measure of potential short-term toxicity in both water and pore-water exposures. However, accurate determination of the dissolved fraction can be difficult and expensive and underestimates exposure to receptors with certain feeding behaviors (Lee et al. 2000). Recommended Texas Surface Water Quality Standards have been converted from total recoverable metals to dissolved metals, and are subsequently reported as dissolved metals. Water quality characteristics, which influence the toxicity of some metals, can be accounted for using empirical relationships (TCEQ, 2012). A more complete discussion of bioavailability appears in **3.9.5**.

Some inorganic metals such as selenium can bioaccumulate to high levels in forage species, and certain organometallic compounds (e.g., methylmercury) can adversely influence higher trophic levels as a result of biomagnification. Additionally, numerous potentially toxic metals are also essential micronutrients (e.g., copper, zinc).

3.9.6.2 Dioxins, Furans, and Polychlorinated Biphenyls

Polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) are by-products of the manufacturing of certain herbicides, chlorophenols, and polychlorinated biphenyls (PCBs). They are also generated during the incineration of chlorinated compounds and are associated with the bleach-kraft pulp-mill process. There are 75 possible PCDD congeners and 135 possible PCDF congeners, and—as with PCBs—their toxicity depends on the number and location of chlorine atoms. Additionally, these COCs have large K_{ow} values that dictate their mobility and partitioning in environmental and biological media (Eisler, 1986a).

PCDDs and PCDFs and dioxin-like PCBs have a wide range of relative potencies and are usually found in complex mixtures in the environment, making determination of risk expensive and difficult. To simplify this process, internationally recognized toxic-equivalency factors (TEFs) are used (Van den Berg et al., 1998, 2006). A TEF is an order-of-magnitude estimate of the toxicity of an individual congener relative to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD—see Table 3.6 for a sample calculation). The literature assumes and generally supports the principle that effects from different congeners are concentration additive (Van den Berg et al., 1998, 2006; Tillitt, 1999). The congener concentration multiplied by the TEF equals the toxic equivalent (TEQ) concentration. The sum of toxic equivalent concentrations is then incorporated into the calculation of a hazard index (see **3.10**).

Table 3.6. Example derivation of 2,3,7,8 TCDD equivalents (TEQs).

Dioxins

COC	TEFa	Concentration (pg/g)	TEQ (pg/g)
2,3,7,8-tetraCDD	1	3.7	3.7
1,2,3,7,8-pentaCDD	1	6.4	6.4
1,2,3,4,7,8-hexaCDD	0.1	3.9	0.39
1,2,3,6,7,8-hexaCDD	0.1	34	3.4

Furans

COC	TEFa	Concentration (pg/g)	TEQ (pg/g)
2,3,7,8-tetraCDF	0.1	3.1	0.31
1,2,3,7,8-pentaCDF[b]	0.03	0.5	0.015
2,3,4,7,8-pentaCDF[b]	0.3	11	3.3

Non-ortho PCBs

COC	TEFa	Concentration (pg/g)	TEQ (pg/g)
3,3',4,4'-tetraCB	0.0001	350	0.035
3,3',4,4'-pentaCB	0.1	330	33

Mono-ortho PCBs[b]

COC	TEFa	Concentration (pg/g)	TEQ (pg/g)
2,3,3',4,4',5-pentaCB	0.00003	210	0.0063
2,3,4,4'-pentaCB	0.00003	162	0.0049

Total TEQs	**50.561**

[a] Adapted from Giesy and Kannan, 1998.

[b] From Van den Berg et al., 2006.

The rationale for the use of TEQs or TEFs is based on a common mechanism of action described for planar halogenated hydrocarbons (PHHs), including dioxins, furans. and some PCBs. The intracellular target of PHHs is the aryl hydrocarbon receptor (AhR), which—while bound to the ligand—mediates the transactivation and inhibition of a variety of target genes, with a wide array of deleterious effects. The structure of a ligand is critical to its affinity for the AhR, and this affinity correlates well with the resulting toxicity and biochemical responses (Safe, 1997). Due to its high potency, the AhR mediated toxicity of 2,3,7,8-TCDD is the standard to which all other PHH potencies are normalized. The relative AhR-mediated toxicity of PHHs has been determined using a combination of *in vitro* and *in vivo* studies, and TEFs are available for mammalian, avian and fish species [Van den Berg et al., 1998, 2006 (mammals only)].

PCBs are highly lipophilic and resistant to degradation in the environment. This class of COCs is generally strongly sorbed to particulates and ultimately resides in sediments or soils. PCBs readily cross most biological membranes and partition into fatty tissues. Concentrations of these COCs are expected to be higher in upper-trophic-level receptors as a result of food-chain transfer.

Increasing chlorination results in higher octanol-water partition coefficients (K_{ow}), increased sorption, and biomagnification (Fisk et al., 1998).

There are 209 possible PCB congeners and, until recently, PCBs have been quantified by comparison of chromatograph patterns to Aroclor standards. This approach has significant associated uncertainties, particularly when samples contain Aroclor mixtures or are highly degraded. Current technology allows for more specific quantitation of homologues or individual congeners which definitively measure the type and quantity of PCBs (ATSDR, 2000). Congener-specific analysis may be more expensive but allows for a better estimation of risk associated with environmental PCBs.

The toxicity of PCBs depends on the congener as confirmed by the number and position of chlorine substitutions. The mechanism of toxicity of non- and mono-ortho-substituted PCB congeners (planar) is initiated through activation of the AhR in the same manner as TCDD. AhR-mediated toxicity results in a broad spectrum of maladies and has been studied extensively in many species (e.g. Safe, 1994; Poland and Knutson, 1982) but is still an active area of research.

Nonplanar PCBs are neurotoxic and carcinogenic, and also elicit a wide range of responses including behavioral and endocrinal changes (Van den Berg et al., 1998, 2006; Safe, 1994). These responses, which may resemble responses to dioxin in some cases, are, however, not mediated by the AhR and therefore cannot be normalized to the TCDD mechanism of toxicity using TEFs. Planar PCBs generally constitute a small percentage of the total PCBs in technical PCB mixtures but often pose the greatest risk of toxicity in the environment (Tillitt, 1999; Giesy and Kannan, 1998). Dioxin-like activity is generally the more potent in eliciting toxic responses.

Additionally, biotic and abiotic weathering of technical PCB mixtures dramatically alters the congener profile (Tillitt, 1999). Generally the literature indicates greater toxicity is associated with weathered PCB mixtures and the proportion of TEQs

relative to total PCB mass increases with trophic level (Zimmermann et al., 1997; Giesy and Kannan, 1998). This indicates that the use of reference doses derived from technical mixtures may be under protective of ecological receptors (Giesy and Kannan, 1998) and that the TEQ-TEF approach may more accurately indicate risk (Tillitt, 1999). However, there is not sufficient information available to determine conclusively that the risk associated with dioxin-like PCB toxicity is greater than that associated with the non-dioxin-like PCBs. Significant uncertainty could exist if the TEQ-TEF approach was used as a surrogate for total PCBs exposure. Therefore, both dioxin-like PCBs (using TEQ-TEF) and non-dioxin-like PCBs need to be evaluated separately in a Tier 2 SLERA. Additionally, congener-specific data needs to be collected in subsequent site-specific investigations to allow for accurate assessment of risk associated with both dioxin-like and non-dioxin-like PCBs.

Extensive information is available on the effects of PCDDs, PCDFs, and PCBs on fish and wildlife and consultation of the literature is encouraged to determine receptors and effects most appropriate for the affected property. Literature sources for reptiles and amphibians, such as Sparling et al. (2000) and Pauli et al. (2000), should be reviewed for information regarding the effects from exposure to these COCs. Toxic effects on fish-eating birds include developmental abnormalities, malformations, embryo mortality, and edema (Grasman et al., 1998; Henshel et al., 1997; White and Hoffman, 1995). Effects in mammals include cognitive disabilities, wasting syndrome, impaired immune response, decreased reproduction, reduced offspring survival, and mortality (Seo, et al., 1999; Leonards et al., 1995; De Swart et al., 1995). Effects on fish include cranial deformities, yolk-sac edema, vascular hemorrhage, fin necrosis, hyperpigmentation, weight loss, and death (Black et al., 1998; Johnson et al., 1998; and Cantrell et al., 1996). Mink and some predatory fish and piscivorous birds are very sensitive to the toxicity of these COCs (Eisler, 1986b).

3.9.6.3 Polycyclic Aromatic Hydrocarbons

PAHs are multiple-ring structures of carbon and hydrogen, both natural and anthropogenic, and ubiquitous in the environment. There are hundreds of individual PAHs, but the compounds with molecular weights ranging from 128.17 (naphthalene, $C_{10}H_8$) to 300.36 (coronene, $C_{24}H_{12}$) are of particular environmental concern. The higher the molecular weight of the PAH, the more lipophilic, less volatile, and less soluble the COC will be (Table 3.7). When released to the environment, PAHs rapidly adhere to particulate matter where they are subject to photolysis, volatilization, metabolic degradation, deposition, and sedimentation (Eisler, 1987).

In aquatic systems PAHs are largely associated with particulate matter, however the solubility of lighter compounds does not preclude exposure via water. Despite the lipophilic nature of PAHs, biomagnification is considered unlikely because they tend to be rapidly transformed or eliminated in many fish and mammalian species (Yuan et al., 1999; Nezda et al., 1997).

Table 3.7. Classification of PAHs for a Tier 2 SLERA.

Low-Molecular-Weight PAHs

PAHs	Molecular Weight[a]	Number of Rings[a]	Solubility (mg/L)[b]	TRRP log K_{ow}
Naphthalene	128.2	2	30	3.17
2-Methyl naphthalene	142.2	2	—	3.72
Acenaphthylene	152.2	3	3.93	3.94
Acenaphthene	154.21	3	1.93	4.15
Fluorene	166.2	3	1.68–1.98	4.02
Anthracene	178.2	3	0.076	4.35
Phenanthrene	178.2	3	1.2	4.35

High-Molecular-Weight PAHs

PAHs	Molecular Weight[a]	Number of Rings[a]	Solubility (mg/L)[b]	TRRP log K_{ow}
Fluoranthene	202.3	3	0.2–0.26	4.93
Pyrene	202.3	4	0.077	4.93
Benzo[a]anthracene	228.3	4	0.01	5.52
Chrysene	228.3	4	2.8×10^{-3}	5.52
Benzo[a]pyrene	252.3	5	2.3×10^{-3}	6.11
Benzo[b]fluoranthene	252.3	5	0.0012	6.11
Benzo[k]fluoranthene	252.3	5	7.6×10^{-4}	6.11
Indeno[1,2,3-c,d]pyrene	276.3	6	0.062	6.70
Benzo[g,h,i]perylene	276.34	6	2.6×10^{-4}	6.70
Dibenzo[a,h]anthracene	278.35	5	5.4×10^{-4}	6.70

[a] Adapted from Parametrix, 1997.

[b] ATSDR, 1995.

However, large bioconcentration factors have been reported for aquatic invertebrates which are believed incapable of the metabolic transformations necessary for efficient excretion of PAHs (Eisler, 1987; Sheedy et al., 1998). The route of exposure, environmental conditions, and feeding behavior are important factors to consider during the risk assessment of PAHs. Low-molecular-weight PAHs (LPAHs) are expected to be more available to benthic invertebrates because of their higher solubility in water. Deposit feeders that ingest sediment particles, however, accumulate high levels of high-molecular-weight PAHs (HPAHs) relative to suspension feeders exposed under similar conditions (Kaag, 1998). Additionally, evidence suggests that environmental factors such as turbidity may influence the profile of PAHs in the tissues of filter feeders. Mussels sampled from calm waters primarily accumulated more water-soluble LPAHs, while mussels from turbid waters had PAH profiles that resembled the profile of the sediment particles (Baumard et al., 1999). Organisms that selectively feed on certain aquatic invertebrates may be exposed to substantial amounts of LPAHs and HPAHs.

Little information is available on accumulation of PAHs in terrestrial biota. PAHs have been detected in plants, but result from aerial deposition rather than active transport (O'Conner et al., 1991). Much of the literature indicates minimal bioconcentration of PAHs in terrestrial invertebrates; however, Gile et al. (1982) reported elevated phenanthrene concentrations in earthworms.

Generally, PAH toxicity involves the disruption of the normal function of enzyme systems or DNA damage by reactive metabolic intermediates. LPAHs (generally, 2–3 rings) are acutely toxic to many invertebrates, possibly as a result of their higher solubility, but are generally considered noncarcinogenic. HPAHs (generally, 4–7 rings) are less acutely toxic, but many are mutagenic, teratogenic, or carcinogenic. Additionally, HPAHs are more recalcitrant in abiotic and some biotic media. Genetic injury resulting in tumor formation depends on the metabolic transformation of the parent compound to one or more reactive intermediates.

Individual species have widely differing abilities to metabolize PAHs to carcinogenic intermediate compounds (Yuan et al., 1999; Livingston, 1998; Eisler, 1987). Invertebrate species are susceptible to acute toxicity (Thompson et al., 1999) and metabolic disruption (Saint-Denis et al., 1999) upon exposure to PAHs; however, stimulatory effects on invertebrate communities have been reported (Erstfeld and Snow-Ashbrook, 1999). The effects of PAHs on mammals—particularly carcinogenesis—have been extensively studied (Eisler, 1987). Many fish species (especially bottom-dwelling fishes) are also very sensitive to the carcinogenic effects of PAHs (Anulacion et al., 1998; Baumann, 1998). Additionally, decreased circulating hormones, disruption of vitellogenesis and oocyte maturation, decreased reproductive success, and altered immune function have been reported in fish exposed to PAHs (Nicolas, 1998; Karrow, 1999). Very little information exists on their effects on avian species (Eisler, 1987). Literature sources for reptiles and amphibians, such as Sparling et al. (2000) and (Pauli et al., 2000), should be reviewed for information regarding effects of PAH exposure.

Generally, the characteristics of the individual PAH will determine its fate in the environment and toxicity. For a Tier 2 SLERA is the TCEQ recommends classification of PAHs into high- and low-molecular-weight categories as illustrated

in Table 3.7 to simplify interpretation of the data and facilitate comparison with ecological benchmarks listed in Tables 3.2–3.4. Individual PAHs are considered to exert their toxic effects through the same mechanism, so derived hazard quotients for PAHs must be summed.

3.9.6.4 Total Petroleum Hydrocarbons

Total petroleum hydrocarbons (TPHs) are the measurable amount of petroleum-based hydrocarbons in an environmental medium. TPHs comprise several hundred COCs and can be broadly categorized as *aliphatic hydrocarbons* (straight-chained alkanes, alkenes, alkynes, and cyclic compounds) and *aromatic hydrocarbons* (BTEX and PAHs) (ATSDR, 1999). Additionally, petroleum products may contain metals; nitrogen-, sulfur-, and oxygen-containing organics and additives such as oxidants; scavengers; and organolead compounds (Suter, 1997).

The origin of a TPH cannot always be determined due to differences in crude oil, refinery processes, and chemical, physical, and biological weathering. Due to the complexity and variety of these hydrocarbon mixtures, TPH analysis generally offers little necessary information for an ecological risk assessment because it does not elucidate the properties that determine potential fate and toxicity of the material. Therefore characterization of the individual components of TPH is necessary to facilitate a defensible assessment of ecological risk. TPH analysis may however, be useful for determining the nature and extent of contamination (Suter, 1997).

Only limited toxicological information is available for TPHs (Dorn et al., 1998; Barron et al., 1999; Saterback et al., 1999), so the use of surrogate COCs for various TPH fractions may be necessary. Recommended surrogate COCs for human health are given in 30 TAC 350.76(g) and environmental concentrations may be compared to appropriate ecological benchmarks for a Tier 2 SLERA. *Canada-Wide Standards for Petroleum Hydrocarbons in Soil* (CCME, 2001, 2008) have been derived to protect key ecological receptors on four types of land use. Most values were based on toxicity studies for soil contact (plants and invertebrates) rather than soil ingestion. It is anticipated that advancements in the assessment of the ecological risk posed by TPHs will continue and readers are encouraged to search for the latest information.

3.10 Hazard Quotient

Hazard quotients (and hazard indices, as appropriate) must be calculated for each COC paired with each selected ecological receptor, as identified in required element (6) of the Tier 2 SLERA [350.77(c)(6)]. A hazard quotient (HQ) states the ratio of the predicted exposure to an acceptable exposure, for a specific COC and a specific representative measurement receptor. An HQ (unitless) is calculated as follows:

HQ = Exposure / TRV

where:

Exposure = measured or estimated exposure point concentration (e.g., mg/L, mg/kg) or dose (e.g., mg/kg body weight/day);

TRV = Toxicity Reference Value (e.g., based on a NOAEL or LOAEL) in units matching the exposure point concentration or dose.

In the risk estimate generated in required element (6), an HQ is based on reasonably conservative exposure assumptions and a representative NOAEL-based TRV (TRV_{NOAEL}). COCs with a hazard quotient (and any associated hazard index) ≤ 1 are dropped from further evaluation for that particular measurement receptor for that medium. If all COCs associated with a receptor are eliminated from the Tier 2 SLERA, no further evaluation of that receptor is required. HQ approaches are advantageous in screening-level risk assessments, as a relatively simple and transparent means of deciding which COCs might be carried forward in more detailed evaluations. Generally, the assessments are purposefully conservative in their measure of exposure and selection of TRVs. Often, HQs can be calculated with minimal effort using existing site-characterization data and literature values on toxic effects. They expedite the process of focusing subsequent risk assessment on COCs likely to pose the greatest risks. (See also **3.9.5** and Appendix C on the selection and development of TRVs.)

To expedite such a process, the TCEQ supports derivation of HQs in required element (6) from the most representative TRV_{NOAEL} that can be technically defended. Though arguably a less conservative derivation of such HQs (see the discussion of less conservative approaches below), this may form a more reasonable basis for screening ecological risks. In particular, selection of the TRV should reflect efforts discussed in **3.9.5** and Appendix C regarding evaluation of results from several studies of differing design, quality, and scope. Therefore, conservative decisions and assumptions may be incorporated into selection or derivation of the value to be put forth as the TRV_{NOAEL} for screening-level HQ calculations. Often this involves assessments of tests done with surrogate species or differing forms of the COC, tests with differing durations or exposure-dose-response regimes, or tests with different or conflicting results. These considerations should be fully documented in derivation of the TRV values in required element (5) of the Tier 2 SLERA. Once the TRV_{NOAEL} is selected, it should be carried through the Tier 2 SLERA.

Several limitations and cautions apply to using HQs in risk characterization (Ferenc et al., 1999). The quotient usually is calculated from point estimates of exposure and toxicity. Thus, no quantitative assessment of risk should be inferred. For example, an HQ based on a point estimate of toxicity such as TRV_{NOAEL} may indicate the presence of an adverse ecological effect but not its magnitude, because the slope of the dose-response curve is not used. Note that the hazards posed by exposure to a COC do not increase linearly as the HQ increases linearly. Also, an HQ should not be viewed as a statistical value. For example, an HQ of 0.01 indicates that an exposure is 100 times less than the associated TRV, not a 1-in-100 probability of an adverse ecological effect occurring (U.S. ACE, 2010).

Because HQs focus only on individual COCs, they do not represent the potential for toxic effects on ecological receptors from COCs acting in concert. Subsection **3.9.6** gives example classes of COCs where such concerns may arise; to address them, risk assessors have summed the risks expressed in individual HQs to form a

cumulative expression of risk in a hazard index. HI is the sum of two or more hazard quotients for different COCs:

$$HI = \sum HQ_i$$

where:

i = COCs with a common toxic mechanism

These hazard indices are calculated as a measure of the potential for impacts of multiple COCs, assuming that the effects are additive. Therefore, this computation is limited to COCs with the same toxic mechanism (i.e., the same mode and site of action). For example, it is not appropriate to add the HQs of two COCs that are both reproductive toxicants if one affects fertilization potential and the other reduces egg production; the risks are independent, and therefore not additive. Note that ecological benchmarks may have been derived from situations where multiple COCs were present (e.g., sediment ER-L values) and from additional conservative generalizations. Consequently, comparisons against ecological benchmarks do not routinely sum different COC concentrations unless the benchmark was explicitly derived for a class of COCs (e.g., a total-PAH benchmark).

When sufficient information is available, an HI should be calculated for any suite or class of COCs with the same toxicological mechanisms. However, this information is not often available outside of a few well-studied groups of COCs, such as those discussed in **3.9.6**. Thus, hazard indices commonly are considered only for a few groups of COCs known to act through a common toxic mechanism for common test species. For example, HIs may be appropriate for PCBs, chlorinated benzenes, dioxins and furans, LPAHs, and HPAHs. Alternatively, if toxicity equivalency factors are used to combine exposures within a class of COCs (e.g., dioxins and furans), then the HQ of the surrogate already represents the cumulative exposure for the whole class adjusted for constituent-specific toxicity potency, and calculation of an HI is not appropriate. Note that, where evidence exists that these referenced groups of COCs do not act through a common toxic mechanism, the assumption of additivity may be adjusted if the appropriate technical merit can be defended and the rationale discussed in the uncertainty analysis (see **3.1.2**).

HQs and subsequent HIs should be calculated using reasonable and representative exposure estimates for the appropriate media and habitats considered for measurement receptors. In most cases, the representative value will be an estimate of the mean exposure concentration. *Determining Representative Concentrations of Chemicals of Concern for Ecological Receptors* (RG-366/TRRP-15eco) should be consulted for the specific types of statistical considerations and analytical procedures for estimating of representative exposure values for ecological receptors.

For any COCs with the same toxic mechanism, the corresponding HI must also equal 1 for their elimination from further consideration in the Tier 2 SLERA. See also Issue No. 5 in TCEQ (2005).

3.11 Less Conservative Assumptions and HQs

Required element (7) of the Tier 2 SLERA [30 TAC 350.77(c)(7)] allows for calculation of hazard quotients using less conservative exposure assumptions and TRVs based on both NOAEL and LOAEL data. Applicable exposure variables for the refined risk estimate generally consist of bioavailability, area use factors or exposure frequency, and diet. These variables should be less conservative in their totality, and the person must justify the use of such data on based on site-specific information or some other clear rationalee. Additional discussion appears in **3.9**, and related TRRP guidance may be forthcoming.

If comparison of a less conservative exposure estimate with a NOAEL-based TRV results in an HQ (and any HI) ≤ 1, the COC may be dropped from further evaluation in the Tier 2 SLERA. Dropping COCs with HQs or HIs derived from technically defensible, LOAEL-based TRVs that are < 1 may be proposed, and supporting information included as part of the discussion of uncertainty (**3.12**). HQs > 1 based on less conservative exposure assumptions and LOAEL-based TRVs form a reasonable basis for beginning remedial planning, as ecological impacts may be expected. In lieu of initiating response actions based on Tier 2 evaluations, COCs with HQs > 1 (and any associated HI) may be carried forward to Tier 3 for more site-specific risk evaluations. Consultations with the TCEQ are recommended at this stage to determine a course of action based on the identified risk drivers.

Adjusting estimates of exposure based on differing assumptions or new data is considered a less conservative approach [30 TAC 350.77(c)(7)]. What constitutes conservatism has evolved, but this approach been generally accepted as less conservative since reevaluating the numerical values used in calculating the HQ commonly decreases exposure estimates and lowers an HQ. However, the level of risk (or acceptable risk) has not been altered in adjusting such calculations. That is, the decision point is still whether the HQ exceeds 1 for the specified measurement receptor for the same endpoints of concern at the same level of effect. Therefore, these adjustments are neither more nor less conservative in the level of protection used in deciding whether a resource will be adversely affected. These adjustments may enhance the focus of the SLERA without incurring the more substantial effort often associated with the higher-tier risk assessments. As stated in the TRRP rule, less conservative HQs present a more representative or certain estimate of exposure or hazard based on site-specific considerations. Exclusion of a COC from further consideration or a decision to take no further action at a site following recalculation of the HQ should not be considered a decision to be less conservative in the level of protection offered to the resources under consideration. See also Issue No. 5 in TCEQ (2005).

3.12 Uncertainty Analysis

After calculating the HQs in required element (7) and analyzing the results of the risk assessment, the person will need to evaluate the uncertainty associated with the ERA in required element (8). The nature of the uncertainties should be clearly summarized in the SLERA (or SSERA, as appropriate). Uncertainty analysis can be used to justify the need for calculating or not calculating a PCL for a given COC (required element 9), considering indications of potential ecological risk in context

with the likelihood of that risk. Factors to be evaluated include the location and extent of the COCs, the degree to which the TRV is exceeded, and the expected half-life of the COCs in the particular environment. If, after completing the HQ exercises in the preceding section, it is determined that for a particular COC the NOAEL HQ/HI > 1 but that the LOAEL HQ/HI < 1, the uncertainty analysis may state that no PCL is necessary for that COC. This is justified because, ideally, any potential remediation of a medium would be to a PCL that is bounded by those two effects levels. However, justification is required when the LOAEL HQ or HI approaches unity and there are indications that risk may have been underestimated in other areas. PCL calculations for a given COC can be justified qualitatively or quantitatively, based on strengths and weaknesses in the data. In most cases, the uncertainty analysis will be qualitative.

More traditional uncertainties include: (1) uncertainty in the conceptual model; (2) uncertainty in the parameters used to evaluate risk; (3) uncertainty in the models used to interpret risk; and (4) stochasticity (natural variability). Uncertainty in the development of the conceptual model may be one of the most important uncertainties in the entire ERA process. If relationships between sources and receptors are missing or incorrectly identified in the conceptual model, then risks could be under- or overestimated. Proper conceptual-model development can help reduce this source of uncertainty.

Uncertainties in the parameters used to evaluate risk that can affect ERA results include variability, uncertainty about a quantity's true value, and data gaps. For example, the exposure of ecological receptors to volatile COCs in soil via inhalation or dermal absorption is rarely evaluated because of the lack of toxicological data; however, this pathway can be significant for burrowing mammals, amphibians, and animals that use the burrows of others.

Consequently, the potential for exposure via inhalation or dermal absorption should be considered when determining the need to develop a soil PCL for a volatile COC. The approaches used to bridge data gaps during the ERA should be described as much as practicable. Sources of uncertainty can often be reduced by collecting additional data, when practicable. Quality assurance–quality control evaluations of COC analyses and effects measurements can help. Discussions on these and other types of uncertainty can be found in Warren-Hicks and Moore (1998). The inherent limitations and typical uncertainties associated with the SLERA are discussed in **3.12.1–2**.

Uncertainty analysis in ecological risk assessment is an evolving field, covering development of more comprehensive qualitative considerations to more sophisticated modeling and statistical analysis. When completing risk assessments where uncertainty analysis is a major component of the conclusions or rationale for actions or decisions, the person should track new technical advances and evolving guidelines in application of uncertainty analysis. See also Issue No. 26 in TCEQ (2005).

3.12.1 Limitations Typical of a SLERA
(modified from U.S. EPA, 1999a)

The approach used to select the measurement receptors is based, in part, on the premise that, if key components of the ecosystem are protected, protection will be conferred to populations and, by extension, communities and the ecosystem. Though this approach is reasonable given the nature of the analysis and the availability of the data, protecting measurement receptors may not always adequately protect all ecologically significant assessment endpoints. Similarly, the selection of ecological receptors relies on feeding guilds. As a result, representative species may not be the most sensitive to particular compounds, but may have been chosen as a function of their ecological significance and the availability of information on natural history and toxicology.

The toxicity of COCs varies with the measurement receptors and with the availability and form of a given COC. If a COC is more bioavailable to an organism for absorption or uptake (such as through increased solubility in the surface soil, surface water, or sediment), then its toxic potential increases. Availability and chemical form are affected by factors such as pH, temperature, alkalinity, seasonal variation, microbial activity, organic carbon content, and complexation with other COCs. In the risk assessment, bioavailability of COCs is assumed to be similar to that observed in the toxicity studies reported in the literature. Thus, toxicity may be over- or underestimated, depending in part on the extent to which site-specific COC bioavailability differs from those in studies reported in the literature.

Attempts to quantify and correct for uncertainty resulting from the use of surrogate species are common, but controversial. Calabrese and Baldwin (1993) discuss the use of uncertainty factors to adjust for extrapolations among taxa, between laboratory and field responses, and between acute and chronic responses. Other sources that describe the use of uncertainty factors are U.S. ACE (2010) and Wentsel et al. (1996). These multipliers are expected to adjust for differences in responses among taxa resulting from differences in physiology and metabolism. When extrapolating from laboratory to field settings, important considerations are differences in physical environment, organism behavior, and interactions with other ecological components. Extrapolation between responses will be necessary in some cases, particularly when data on relevant endpoints are not available (most commonly when extrapolating from an LOAEL to an NOAEL). The net effect of uncertainty factors on the accuracy of the risk assessment depends on the accuracy of assumptions.

3.12.2 Uncertainties Typical of a SLERA
(modified from U.S. EPA, 1999a)

The SLERA typically uses some default parameter values in place of site-specific measured data, incorporating assumptions as a result of data gaps. The absence of site-specific information and the need for these assumptions may result in uncertainty in calculating HQs. An understanding of the uncertainties is necessary for understanding the significance of the HQs. After identifying the major uncertainties associated with the risk-assessment results, their significance to

the computed HQs should be evaluated. Uncertainties that generally should be evaluated in a SLERA for a remediation site are:

- Evaluation of non-detects used in the risk assessment.
- The site-specific representativeness of food webs used in the risk assessment.
- The exposure potential of the measurement receptors.
- The representativeness of diet assumptions for measurement receptors.
- The effect of COC physicochemical properties on estimates of fate and bioavailability.
- The effect of site-specific environmental conditions affecting the fate, transport, and bioavailability of the COCs.
- The assumption that, once exposed, a measurement receptor does not metabolize or eliminate a COC.
- The potential risk to measurement receptors from COCs with no TRVs.

3.13 PCL Calculation

Ecological PCLs must be calculated for each COC which has not been eliminated from consideration under required elements (1), (6), (7), or (8) of the Tier 2 SLERA [see 30 TAC 350.77(c)]. The ecological PCL must be protective of more wide-ranging ecological receptors that may frequent the affected property and use less mobile receptors as a food source and, where appropriate, benthic invertebrate communities within the waters in the state. The ecological PCL is not directly intended to be protective of on-site receptors with limited mobility or range (e.g., plants, soil invertebrates, and small rodents) unless these receptors are threatened or endangered species or unless impacts on these receptors disrupt the ecosystem or have other unacceptable consequences for the more mobile or wide-ranging receptors (e.g., impacts to a grassland habitat eliminate rodents, which causes a desirable owl population to leave the area). The ERA should be conducted in a manner that results in the protection of ecological receptors subject to management by other federal and state agencies and consistent with those agencies' statutory authority. For each COC not eliminated from consideration under required elements (1), (6), (7), or (8), a medium-specific PCL bounded by the NOAEL and LOAEL is calculated for each relevant measurement receptor. Guidelines for deriving ecological PCLs bounded by the NOAEL–LOAEL range appear in **3.13.2**. Since exposure for community-level receptors such as fish and benthic macroinvertebrates is generally expressed in terms of media concentration, any PCLs related to such receptors are based on a simple comparison of representative media concentrations to applicable TRVs. For wildlife receptors (where exposure may be due to ingestion of impacted food or media), there are a variety of techniques for deriving media-specific PCLs. Some of these techniques are discussed below. See also Issue No. 5 in TCEQ (2005).

3.13.1 PCL Calculation Methods

Frequently at affected properties, only a single medium contributes to the ecological exposure. See **3.13.1.1** for a simple method for calculating a PCL range when the total dose to a wildlife measurement receptor arises from one medium; with multimedia exposure, there is no single set of valid PCLs, since their derivation requires solving one equation (the general dose equation discussed in **3.9.2**) for multiple unknowns (media-specific PCLs). The text in **3.13.1.2–4** suggests three example methods for deriving valid yet non-unique sets of multimedia PCLs for a particular receptor.

3.13.1.1 One-Medium Method

If the COC exposure to a measurement receptor at an affected property is from only one medium (usually soil), a simple equation using the NOAEL- and the LOAEL-based HQs can give the PCL range. Using soil as an example, the PCL range is derived by dividing the representative soil concentration by the appropriate HQ:

NOAEL PCL = Representative soil COC concentration / NOAEL HQ

LOAEL PCL = Representative soil COC concentration / LOAEL HQ

This PCL range would be appropriate for addressing both direct (soil ingestion) and indirect (food consumption) soil exposure from the COC to the measurement receptor. This method cannot be reliably applied to multimedia exposure. For instance, if COC exposure from both soil and sediment were contributing significant risk to a measurement receptor, remediating the soil—even to a zero concentration level—would not affect the significant risk remaining from the sediment. However, application of this method may be considered when the relative risk contributions from the other media are insignificant.

3.13.1.2 Percentage Method

The first, and perhaps the simplest, technique for deriving media-specific PCLs for wildlife measurement receptors is to calculate the percentage that each medium contributes to the HQ and then adjust each media concentration to the same relative proportions so that the adjusted HQ equals 1.0. The wildlife dose equation presented in **3.9.2** becomes the starting point, with the applicable media-specific PCLs represented by the media concentration terms in the numerator of that equation. For example, if 30 percent of a NOAEL-based HQ of 5.5 results from **either direct or indirect** soil exposure, the soil NOAEL PCL (PCL_{soil}) is set at a concentration that results in an overall contribution of 0.3 to the adjusted NOAEL HQ of 1.0. Similar calculations are performed for the LOAELs and the other applicable media.

3.13.1.3 Dose-Equation Method

A second method for deriving media-specific PCLs for wildlife measurement receptors also starts with the wildlife dose equation presented in **3.9.2**, with the equation rearranged to solve for a single medium-specific PCL while COC concentrations in other media are held constant. This may be accomplished by using a background value, an ecological benchmark, or the lowest value in the data set.)

$$(1)\ Dose_{oral} = \frac{\left[\left(IR_{food} \times C_{food} \times EMF_{food}\right) + \left(IR_{water} \times C_{water} \times EMF_{water}\right) + \left(IR_{soil} \times C_{soil} \times EMF_{soil}\right) + \left(IR_{sed} \times C_{sed} \times EMF_{sed}\right)\right]}{BW}$$

As described in Section 3.10, an HQ compares exposure (e.g., dose) to a TRV:

$$(2)\ HQ = \frac{Exposure}{TRV} = \frac{Dose}{TRV}$$

The derivation of a PCL requires the HQ to equal 1.0, resulting in the following relationship:

$$(3)\ Dose_{oral} = TRV$$

TRV is then substituted for Doseoral in Equation 1:

$$(4)\ TRV = \frac{\left[\left(IR_{food} \times C_{food} \times EMF_{food}\right) + \left(IR_{water} \times C_{water} \times EMF_{water}\right) + \left(IR_{soil} \times C_{soil}\ EMF_{soil}\right) + \left(IR_{sed} \times C_{sed} \times EMF_{sed}\right)\right]}{BW}$$

If the COC concentration in the primary food or prey of the wildlife receptor has not been measured (as would typically be the case for Tier 2 SLERAs), then the COC concentration in food (C_{food}) can be predicted from the applicable media concentration and uptake factor (UF) as discussed in **3.9.2.1**. For the purposes of this example, C_{food} is represented solely in terms of C_{soil} (i.e., PCL_{soil}) and an applicable uptake factor.

$$(5)\ TRV = \frac{\left[\left(IR_{food} \times PCL_{soil} \times UF \times EMF_{food}\right) + \left(IR_{water} \times C_{water} \times EMF_{water}\right) + \left(IR_{soil} \times PCL_{soil} \times EMF_{soil}\right) + \left(IR_{sed} \times C_{sed} \times EMF_{sed}\right)\right]}{BW}$$

Both sides of the equation are multiplied by BW to move that variable to the left side.

$$(6)\ TRV \times BW = \left[\left(IR_{food} \times PCL_{soil} \times UF \times EMF_{food}\right) + \left(IR_{water} \times C_{water} \times EMF_{water}\right) + \left(IR_{soil} \times PCL_{soil} \times EMF_{soil}\right) + \left(IR_{sed} \times C_{sed} \times EMF_{sed}\right)\right]$$

Dose terms relating to water and sediment ingestion are subtracted from both sides of the equation to move those variables to the left side:

$$(7)\ TRV \times BW - \left[\left(IR_{water} \times C_{water} \times EMF_{water}\right) + \left(IR_{sed} \times C_{sed} \times EMF_{sed}\right)\right] = \left[\left(IR_{food} \times PCL_{soil} \times UF \times EMF_{food}\right) + \left(IR_{soil} \times PCL_{soil} \times EMF_{soil}\right)\right]$$

PCL_{soil} is factored in the right side of the equation:

$$(8)\ TRV \times BW - \left[\left(IR_{water} \times C_{water} \times EMF_{water}\right) + \left(IR_{sed} \times C_{sed} \times EMF_{sed}\right)\right] = PCL_{soil}\left[\left(IR_{food} \times UF \times EMF_{food}\right) + \left(IR_{soil} \times EMF_{soil}\right)\right]$$

The equation is rearranged to solve for PCL_{soil}:

$$(9)\ PCL_{soil} = \frac{TRV \times BW - \left[\left(IR_{water} \times C_{water} \times EMF_{water}\right) + \left(IR_{sed} \times C_{sed} \times EMF_{sed}\right)\right]}{\left[\left(IR_{food} \times UF \times EMF_{food}\right) + \left(IR_{soil} \times EMF_{soil}\right)\right]}$$

The PCL_{soil} term will reflect the NOAEL PCL if the NOAEL was used as the TRV in Equation (4) or the LOAEL PCL if the LOAEL was used as the TRV; however, the TRRP rule requires that both PCLs must be calculated.

3.13.1.4 Algorithmic Method

A third method of deriving multimedia PCLs for wildlife measurement receptors uses an automated algorithm such as the Solver add-in for Microsoft Excel to simultaneously solve for the various PCLs in Equation 1. Briefly, the dose equation is written into a spreadsheet format using Excel or a similar program that includes appropriate algorithm codes. The values in the "adjustable cells" (i.e., PCLs) are manipulated to produce the result specified from the "target cell" (i.e., *Dose = TRV*). Appropriate constraints are set on the derived PCLs such that the results make sense for remedial decision-making (e.g., disallow negative PCL values; disallow PCL values greater than the representative concentrations). Note that the terms in the dose equation (Equation 1) and, by extension, the PCL equation (Equation 9) are not independent variables. The concentrations in the various environmental media could be dependent on each other due to fate and transport and equilibrium principles.

Incorporation of a NOAEL as the TRV in Equations (1) or (9) results in the lower bound to the PCL; incorporation of an LOAEL, the upper bound to the PCL. Note that the preceding exercise must be repeated for all selected receptors for which a PCL derivation is warranted. As discussed in **3.13.2**, the final medium-specific PCL for a particular COC should be based on the selected receptor predicted to be at the greatest potential risk and should lie between the upper and lower bounds.

Equations (1) and (9) illustrate the general concepts for calculating a PCL for a wildlife measurement receptor. Of course, these precise equations are not suited to all circumstances. For example, ingestion rates found in the literature may be normalized to body weight, in which case the body-weight term would be eliminated from the combined oral dose and PCL equations. At other times, ingestion rates may be reported for each type of food or prey or for total food (including media), and "fraction ingested" terms may need to be incorporated into the PCL equation.

3.13.2 Selection of Comparative and Final Ecological PCLs

Under 30 TAC 350.77(c)(9), the person must develop medium-specific PCLs bounded by the NOAEL and the LOAEL before choosing a final PCL. These *comparative PCL*s are developed for each remaining COC associated with each relevant measurement receptor for a medium and, where appropriate, for the medium itself in the case of benthic invertebrates. The final ecological PCL[10] for a COC in a particular medium should be the lowest of the comparative PCLs and, except as discussed below, should lie between the NOAEL and the LOAEL for the most susceptible measurement receptor or community because the ERA process allows every realistic exposure

[10]The final ecological PCL should not be confused with the critical PCL. The critical PCL is the lower of the human-health PCL and the final ecological PCL for a particular COC within a specific medium. See additional discussion of the critical PCL in **5.2** and in *Critical PCLs* (TCEQ publication RG-366/TRRP-25).

assumption (and any site-specific data) before PCLs are developed. When these realistic exposure assumptions are incorporated into the analysis, the TCEQ can be reasonably confident that any COC with a LOAEL HQ ≥ 1 resulting from the exercise in required element (7) has the potential to pose unacceptable ecological risk. This also means that remediation to a LOAEL-based PCL derived from realistic exposure assumptions may result in unacceptable ecological risk still remaining within that specific medium. As discussed below (see item 5), the foundation for selecting a comparative PCL that is skewed toward either the NOAEL-based or LOAEL-based PCL should be made in the uncertainty analysis.

The TRRP rule is intentionally silent on how to select a comparative ecological PCL that is bounded by the NOAEL and LOAEL to allow flexibility in making this determination. However, the TCEQ has developed guidelines to assist in this determination, although these may be adapted to site-specific circumstances. Generally, when trying to establish comparative ecological PCLs for the relevant measurement receptors, the following guidelines should be followed.

1. Except for the benthic PCL (see item 6), as a default, the average between the NOAEL- and LOAEL-based PCLs for a COC in a specific medium can be used as the comparative PCL, provided that the NOAEL and LOAEL do not differ more than a factor of 10. For example, if the NOAEL-based PCL for a COC for a measurement receptor was determined to be 12 mg/kg and the LOAEL-based PCL was 60 mg/kg, then the average (i.e., comparative PCL) would be (12 + 60) ÷ 2 = 36 mg/kg. Selection of this average value does not require any further justification, merely a statement identifying the comparative PCL as the average of the NOAEL- and LOAEL-based PCLs.
2. One exception to having the comparative PCL lie between the upper and lower bounds occurs whenever a threatened or endangered species is at risk. In this case, the NOAEL-based PCL should be chosen as the comparative PCL because the TCEQ will not advocate remediation to a concentration level (i.e., > the NOAEL-based PCL) likely to have any adverse effect on individualss of a threatened or endangered species.
3. Another exception to having the comparative PCL lie between the upper and lower bounds may occur in situations where only conservative exposure assumptions are used. As this is somewhat contrary to required element (7) in the TRRP rule—mandating justification for less conservative exposure assumptions—this situation should not occur often. However, if costs, technical impracticability, or other reasons prevent either the collection or the use of data to support the use of less conservative exposure assumptions, the LOAEL-based PCL could be used as the comparative PCL. Additional scenarios may warrant the consideration of using the LOAEL-based PCL, but would need to be examined case by case. See also Issue No. 26 in TCEQ (2005).
4. If a mixture of conservative and justified less conservative assumptions is used, the person could either select the average value as discussed in item 1 or a value biased toward the LOAEL bound. The degree of bias

should be commensurate with the measure of conservatism remaining in the exposure assumption. For example, if at an affected property consisting of 2 acres, less conservative exposure assumptions were justified for a measurement believed to be wide-ranging but whose exact range is unknown and therefore causing the choice of an area use factor of 1, then a bias toward the LOAEL bound (e.g., 75 percent of the LOAEL-based PCL) could be appropriate. As usual, the selected value would depend on site-specific circumstances.

5. Although the uncertainty analysis precedes the development of PCLs, that is where the person should discuss any remaining conservative or non-conservative practices within the refined exposure analysis or any other circumstances that would ultimately influence the selection of a comparative PCL biased toward the LOAEL or NOAEL bound. Presumably, the person has carefully considered and has already chosen the most relevant NOAELs and LOAELs (see **3.10**). There are, however, times when a discussion of this TRV uncertainty can aid the determination of the comparative PCL. For example, if the most relevant (or only) TRV for a COC is derived from a particularly toxic form of that COC and there is no reason to believe that the more toxic form is present or can develop naturally (e.g., via methylation) at the affected property, then the bias could be toward the LOAEL bound (e.g., 90 percent of the LOAEL-based PCL), as to do otherwise would be overly conservative.[11] Bias toward the NOAEL bound (e.g., 75 percent of the average) could be justifiable whenever a complete ecological exposure pathway was not accounted for in the exposure analysis because of a lack of toxicological data (see the example of the volatile COC and inhalation pathway in **3.12**).
6. A benthic invertebrate community potentially harmed by COC releases from an affected property may require the development of a PCL, which could apply to the COC-laden groundwater discharging to the sediments or to the sediment itself. Again, the TCEQ does not advocate the benchmark values in Tables 3.2–4 as cleanup goals because of their conservatism. However, as presented in Appendix A, for the sediment benchmarks (Table 3.3) there is usually a corresponding second level of effects value from the same database (see Table A.2). For example, among the primary benchmarks used for freshwater sediment were the threshold effect concentrations from MacDonald et al. (2000). The second effect levels from this same database are called *probable effect concentrations* (PECs). The TEC was intended to estimate the concentration for a given COC below which adverse biological affects only rarely occurred. The PEC was chosen to represent the concentration for a given COC above which adverse biological affects frequently occur.

   For the development of benthic PCLs, the TCEQ recommends the primary sediment benchmarks in Table 3.3 as NOAELs, and the second

[11]The use of this guideline requires—in the uncertainty analysis—a demonstration, ideally based on sampling and analysis—that the more toxic form of the COC is not present and likely could not develop at the affected property.

effect levels in Table A-2 as LOAELs. In developing the benthic PCL, the same logic presented in the preceding guidelines should be applied, with one notable exception. The person may continue to propose the midpoint as the comparative PCL without further justification, but the midpoint will not be considered the default when the weight of evidence suggests the value is not protective of the benthic community (also see the discussion of EqP-derived benchmarks in **3.5.1.2** and Appendix A). For instance, the TEC for copper in freshwater is 31.6 mg/kg; the PEC is 149 mg/kg. The average to be proposed as the comparative PCL would be (31.6 + 149) ÷ 2 = 90.3 mg/kg. However, if reliable or more relevant (e.g., local) studies from the literature show that values above 80 mg/kg cause unacceptable impact on the benthic community, then the midpoint value of 90.3 mg/kg may be questionable.

Alternatively, the person may evaluate the individual studies that comprise the effects-level databases, find the most applicable study, and recommend a PCL he or she considers more suited to the actual circumstances (e.g., affected species, sediment composition) of the affected property. Of course, such a recommendation will need to be adequately justification and documentation.

Finally, the person may also choose an EqP method for developing a benthic PCL based on the inherent uncertainties (see **3.5.1.2**); it is recommended as an alternative method only where data gaps necessitate such measures. The person proposing an EqP-based sediment PCL approach should clearly discuss the uncertainties, including the gaps in available sediment data that prevent alternate means of deriving a benthic PCL.

7. The derivation of PCLs for aquatic life (water-column receptors) does not parallel the derivation of PCLs for other media (e.g., benthic receptors in sediment) where a range is determined. Surface water PCLs are point values representing the TSWQS, values derived in accordance with the TSWQS, or federal criteria. Here, it is not appropriate to use the midpoint between the acute and chronic values.

   B. The final media-specific ecological PCL for a COC should be the lowest concentration among the comparative PCLs determined for each relevant measurement receptor and the benthic community, where appropriate. Accordingly, the measurement receptor or benthic community requiring the lowest comparative PCL is considered the most susceptible for that medium. Cost and remediation technology should never be factored into the determination of the final ecological PCL. These are risk-management considerations.

As stated previously, the guidelines listed above may vary between affected properties. This list is **not** comprehensive. The person should rely on site-specific circumstances and the availability of toxicological data for selection of comparative and final ecological PCLs.

3.14 Risk-Management Recommendation

The Tier 2 SLERA concludes with required element (10), a recommendation on how to manage the ecological risk at the affected property. If all COCs and pathways have been eliminated by this point, it suffices to state that the ERA guidance has been followed and there is no apparent unacceptable ecological risk at the affected property. However, if ecological PCLs were calculated in Tier 2, the person must do one or more of the following:

- Proceed to additional risk assessment under Tier 3 to develop final ecological PCLs or to determine that there is no apparent unacceptable ecological risk at the affected property.
- Compare the PCL values generated in Tier 2 to relevant levels protective of human health (e.g., RRR Standard 2 numbers, values generated from a BLRA, or TRRP human health–groundwater PCLs generated at any tier) in order to determine the critical PCL (see **5.2**) and remediate to those levels; or
- Evaluate and state whether the human health remedy would eliminate all ecological exposure pathways.
- Request permission to conduct an ecological services analysis (see **5.3**).

Other management strategies may be possible, but the ecological risk management recommendation must describe an action that will address any exceedances of ecological PCLs. A more detailed discussion of ecological risk management options appears in Chapter 5.

4 TIER 3: SITE-SPECIFIC ECOLOGICAL RISK ASSESSMENT

4.1 Purpose and Use

In accordance with the TRRP rule at 30 TAC 350.77(d), the purpose of the optional SSERA is to incorporate additional information obtained through the performance of site-specific studies designed to encourage an empirical evaluation of ecological risk at the affected property. An SSERA can be conducted when the person believes that any of the Tier 2 PCLs are inappropriate or do not reflect existing conditions at the affected property, or when otherwise elected. A person who elects to begin the ecological evaluation of the affected property by proceeding directly to an SSERA must incorporate applicable components of a Tier 2 SLERA, including required elements (2), (3), (4), (8), (10), and other requirements as determined appropriate by the TCEQ. Completion of a Tier 1 exclusion criteria checklist may also be useful to focus the SSERA with regard to the applicable exposure pathways. If a Tier 2 SLERA has been conducted, particular aspects of the SLERA may need to be refined to reflect the focused specificity of the Tier 3 SSERA. The result of the SSERA will be the development of site-specific Tier 3 PCLs, a determination that there is no ecological risk, or a conclusion that ecological risk is not apparent based on site-specific information. The Tier 3 SSERA can consist of any site-specific study approved by the commission and can include, but is not limited to:

- Development of site-specific bioaccumulation factors through the collection and analysis of tissue samples from appropriate ecological receptors.
- Performance of toxicological testing of the impacted media via exposure to an appropriate test species.
- Comparison of site data (e.g., macroinvertebrate-diversity surveys) to like data from a reference area.
- Other studies designed to obtain a preponderance of evidence for conclusions about ecological risk.

Examples of studies that may qualify as Tier 3 assessments are discussed in the sections that follow. As indicated, the Tier 3 SSERA is not required and is only conducted at the discretion of the person. Because Tier 3 involves the collection of site-specific information, it can be costly and time consuming; therefore, persons are strongly encouraged to communicate with the TCEQ ERA staff regarding the study objectives, conceptual model, study methodology, and additional sampling and site investigations before proceeding.

4.2 Types of Studies

This is a general listing of the types of studies commonly used in Tier 3 SSERAs. The person is encouraged to seek out additional scientific literature.

4.2.1 COC-Residue Studies

Tissue residue analysis and bioaccumulation studies can be used to evaluate the degree of transference of COCs through the food chain, to measure the bioavailability of COCs and their concentrations in foods consumed by receptors of concern, to generate site-specific estimates of exposure to higher trophic level organisms, and to relate the tissue concentrations to concentrations in environmental media (U.S. EPA, 1997a). Residue studies are appropriate if COCs in question will bioaccumulate.

4.2.2 Toxicity Tests

Toxicity tests, when combined with COC analyses, can be an important tool in a Tier 3 SSERA to determine if COC present in an affected property are toxic (and bioavailable) to test organisms. To measure toxicity, a specific biological endpoint (e.g., mortality, reductions in growth or reproduction, relevant changes in behavior) is used to assess the response of the test organisms to COCs in the affected media. Toxicity tests can be used to demonstrate whether COCs are bioavailable, can evaluate the aggregate toxic effects of all COCs in a medium and the toxicity of substances whose toxicity is not well characterized or known, can characterize the nature of the toxic effect (lethal or sublethal), can characterize the distribution of toxicity at an affected property, and can be used to develop PCLs and facilitate remediation decisions. Toxicity tests may be more sensitive to low levels of contamination than other monitoring methods (e.g., COC analyses of media). Most standard toxicity tests are performed at laboratories on media transported from the affected property. This allows for constant conditions, standardized test protocols, and readily available equipment.

4.2.3 Field Studies

Ecological field studies take place in the actual property under scrutiny, focusing on its habitats and biota. Field studies are especially valuable for terrestrial habitats, for which there are fewer available standardized toxicity tests and toxicological reference values than for aquatic habitats. Field studies generally focus on populations and communities and the associated habitats rather than individual organisms. Results are usually analyzed by comparing the observations in the affected property (e.g. biomass, abundance, diversity, richness of species and communities, age structure, and trophic structure) to corresponding measurements from a reference or control area.

Population metrics include measurements of density patterns, abundance, biomass, rates of recruitment, size and age distribution, spatial distribution, growth, and survival. Community metrics include measurements of species

composition, richness, diversity, dominance, abundance, community structure, trophic dynamics, seasonal patterns, and age classes. Behavioral and physiological measurements such as respiration, photosynthesis, reproduction, burrowing, predation, and courtship may also be evaluated. These measurements are typically compared to those of a reference area or are evaluated for changes along a concentration gradient.

4.2.4 Reference Area

A *reference site* or *area* is defined as an area that is outside the COC influence of the affected property, but possesses similar characteristics such as habitat and substrate type, allowing for comparison of data between areas with and without impacts. This applies whether the reference area is used for a community or population study or to collect reference samples for toxicity tests on media from the affected property. Mortality, vegetation stress, tissue data (histopathologic and COC concentrations), habitat degradation, presence or absence of key species, population assessment of key species, community indices, and ecosystem function determined at the affected property can be compared with those of the reference site. Reference areas give valuable information about naturally occurring compounds or ubiquitous COCs. The area selected must be of similar habitat type and species composition to those of the affected property, and should lie outside the area of influence of the affected property, preferably in an area of minimal impact or disturbance (Appendix B of U.S. EPA, 1997a). Sampling and surveying of reference areas should use the same techniques and the same level of detail employed at the affected property to ensure a valid comparison.

4.2.5 Other Studies, Weight of Evidence
(Adapted from U.S. EPA, 1997a)

For Tier 3 SSERAs that entail more than one type of study (or line of evidence), a weight-of-evidence approach is used to integrate multiple types of data to support a conclusion. Generally, confidence in the risk assessment conclusions will be increased using several lines of evidence. The data might include toxicity test results, assessments of existing impacts at an affected property, or risk calculations comparing exposures estimated for the affected property with toxicity values from the literature. Balancing and interpreting the different types of data can be major tasks requiring professional judgment, as not all data are of equal value. The weight of evidence from different types of tests and the precedence of one type of study over another should be established in planning for the SSERA. This approach ensures that data interpretation is objective, not biased to support a preconceived answer. Additional considerations at this stage include the degree to which data-quality objectives were met, and whether confounding factors became evident in the analysis phase.

4.3 Tier 3 Reporting and Conclusions

As similarly required for the end of a Tier 2 SLERA, the person must submit the SSERA, including recommendations for managing ecological risk, to the TCEQ as part of the APAR, ensuring that the SSERA is adequate and that its information supports the recommended decision for risk management. This determination may require technical advice from the TCEQ ERA staff. The possible recommendations for risk management at this point are:

- Information is adequate to conclude that there is negligible ecological risk and, therefore, no need for remediation on the basis of the Tier 3 SSERA.
- The information indicates there may be risk, and therefore, a specified control mechanism (possibly to protect human health also) will be implemented to manage ecological risk.
- The information indicates there may be risk, and therefore, Tier 3 PCLs will be evaluted.
- The information indicates that an ecological services analysis is needed.

Ecological services analyses and other forms of ecological risk management are discussed in Chapter 5.

5 ECOLOGICAL RISK MANAGEMENT

Decisions about ecological risk management can be made using either the Risk Reduction Rule or the Texas Risk Reduction Program (but not both). The RRR standards (Risk Reduction Standards 1, 2 or 3) or the TRRP Remedy Standards (Remedy Standard A or B) must include protection of ecological receptors as a remedy goal. Below is a brief discussion of risk management under the two rules, followed by discussion of the Ecological Services Analysis (ESA) remedy option, remedy implementation, compliance sampling and confirmation, and reporting.

5.1 Risk Management under the Risk Reduction Rule

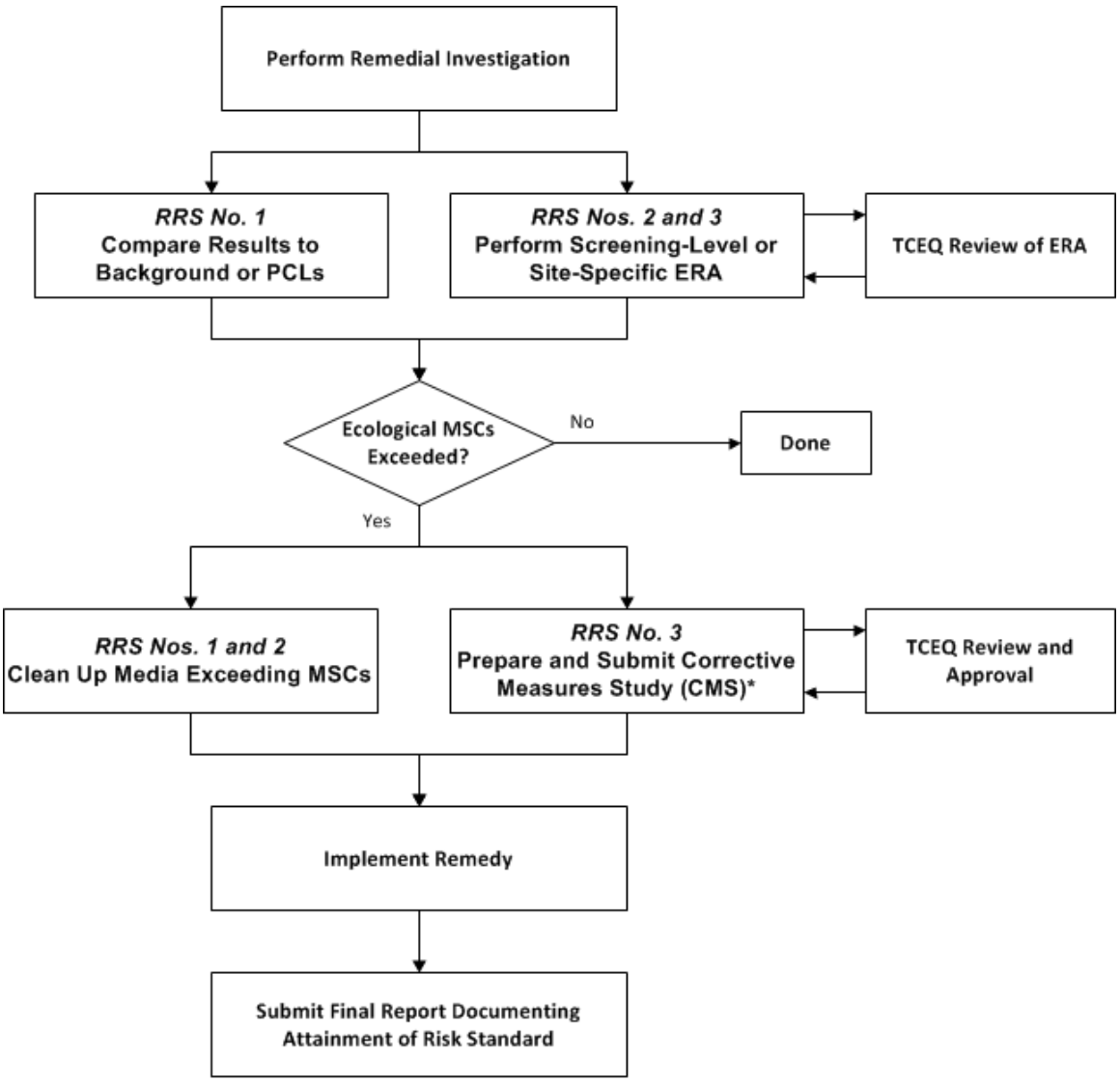
The Risk Reduction Rule allows the person to decide how to attain the risk-reduction standards (1, 2, or 3). Under the RRR, the following risk management actions would preclude the need for Tier 2 or 3 ERA.

- Risk Reduction Standard 1 (background or PQL, whichever is higher) is attained.
- Application of the Tier 1 exclusion criteria checklist excludes the affected property.
- Implementation of a risk-management decision to protect human health by controlling exposure pathways also will eliminate exposure pathways for ecological receptors.

If a Tier 2 or 3 ERA is needed, the following are potential outcomes of risk assessment:

- A finding of “no unacceptable ecological risk,” so no further action to address ecological receptors is required.
- Human-health standards (RRR Standard 2 MSCs or values generated from a BLRA under RRR Standard 3) are lower than ecologically protective levels, and actions taken to address human-health risk will protect ecological receptors.
- If at Tier 2 level, further risk assessment is slated under Tier 3 for all or some of the COCs or media, to better characterize site-specific conditions and risk.
- An action is required to address risk to ecological receptors (e.g., removal, remediation, or control of ecological exposures).

The person should report the results of the ERA and recommend risk-management decisions to the TCEQ [in a format such as a RRR Standard 2 report per 30 TAC 335.553(a) or a baseline risk-assessment or corrective measures study per 30 TAC 335.553(b)] for review and approval. If the TCEQ concurs with the ecological risk assessment and management decision, the process is complete and the remedy should be implemented. Figure 5.1 presents a flowchart for deciding upon remedies under Standards 1, 2, or 3.



Notes:

PQLs = Practical Quantitation Limits
RRS = Risk Reduction Standards
ERA = Ecological Risk Assessment
MSCs = Media-specific Concentrations
TCEQ = Texas Commission on Environmental Quality

*The person may elect to use the Ecological Services Analysis option available under the TRRP rule. If pursued, the person must follow the process described in Section 5.3 and Figure 5-4 of this guidance.

Figure 5.1. Choosing among risk-reduction standards for ecological exposures.

Under the RRR, a person may propose to use the ESA described under the TRRP rule; however, doing so, entails compliance with all ESA requirements in the TRRP rule [see 30 TAC 350.33(a)(3)(B)] and as described herein, including seeking approval from and coordinating with the Natural Resource Trustee agencies. (Note that it is the absence of

specificity regarding ecological analysis in the RRR and the frequent use of "may" that allows the TRRP procedure to be imported into the RRR in this instance. Ordinarily the rules cannot be mixed and matched.)

5.2 Risk Management under the TRRP Rule

As a component of required element (10) of the Tier 2 SLERA, the person must provide risk management recommendations for the affected property if the affected media contains COCs in excess of applicable PCLs. Such risk-management recommendations are confined to the response options available under Remedy Standard A or B of the TRRP rule. The remedy must address both human health and ecological exposure. For this purpose, human health–based and ecological PCLs are compared to identify *critical PCLs* (i.e., the lowest concentration levels) for each COC and affected medium, and the remedy is directed toward addressing critical PCLE zones. The remedy is complete when either Standard A or Standard B response objectives have been achieved, the TCEQ has approved all requisite reports, and any necessary post-response care has been performed and financial assurance maintained (see 30 TAC 350.34 and 30 TAC 350.91–96). It is the responsibility of the person to select the appropriate remedy, and, if selecting Remedy Standard B, to submit a response-action plan for review and approval by the TCEQ.

For each COC for which the ecological PCLs are determined to be the critical PCL and the corresponding media concentration of that COC exceeds the critical PCL, the person must consider the need for further assessment (e.g., a Tier 3 ERA) or select one or a combination of the available remediess available under the TRRP rule.

The person may propose a remedy for managing the PCLE zone. The specific remedy options may be summarized as follows (see the full presentation in 30 TAC 350, Subchapter B):

- **Removal** (Remedy Standards A or B) of concentrations of COCs that exceed the critical PCLs. Examples include excavation or dredging and the subsequent placement or disposal in a manner protective of ecological risks. Waste or affected environmental media must be removed and taken to another location.
- **Decontamination** (Remedy Standards A or B)—meaning a permanent and irreversible treatment that eliminates concentrations of COCs that exceed their respective critical PCLs. Examples include biological degradation (natural and enhanced), chemical fixation or sequestration, detoxification, or natural attenuation. (Other remedies are possible.)
- **Control** (Remedy Standard B only) contemplates physical or institutional controls that prevent the exposure of ecological receptors to concentrations of COCs that exceed their respective ecological PCLs. Examples include capping an affected area with an impermeable or semi-impermeable cap or layer (e.g., concrete, clay, geotextiles), physical containment of potential sources of COCs, or institutional controls that

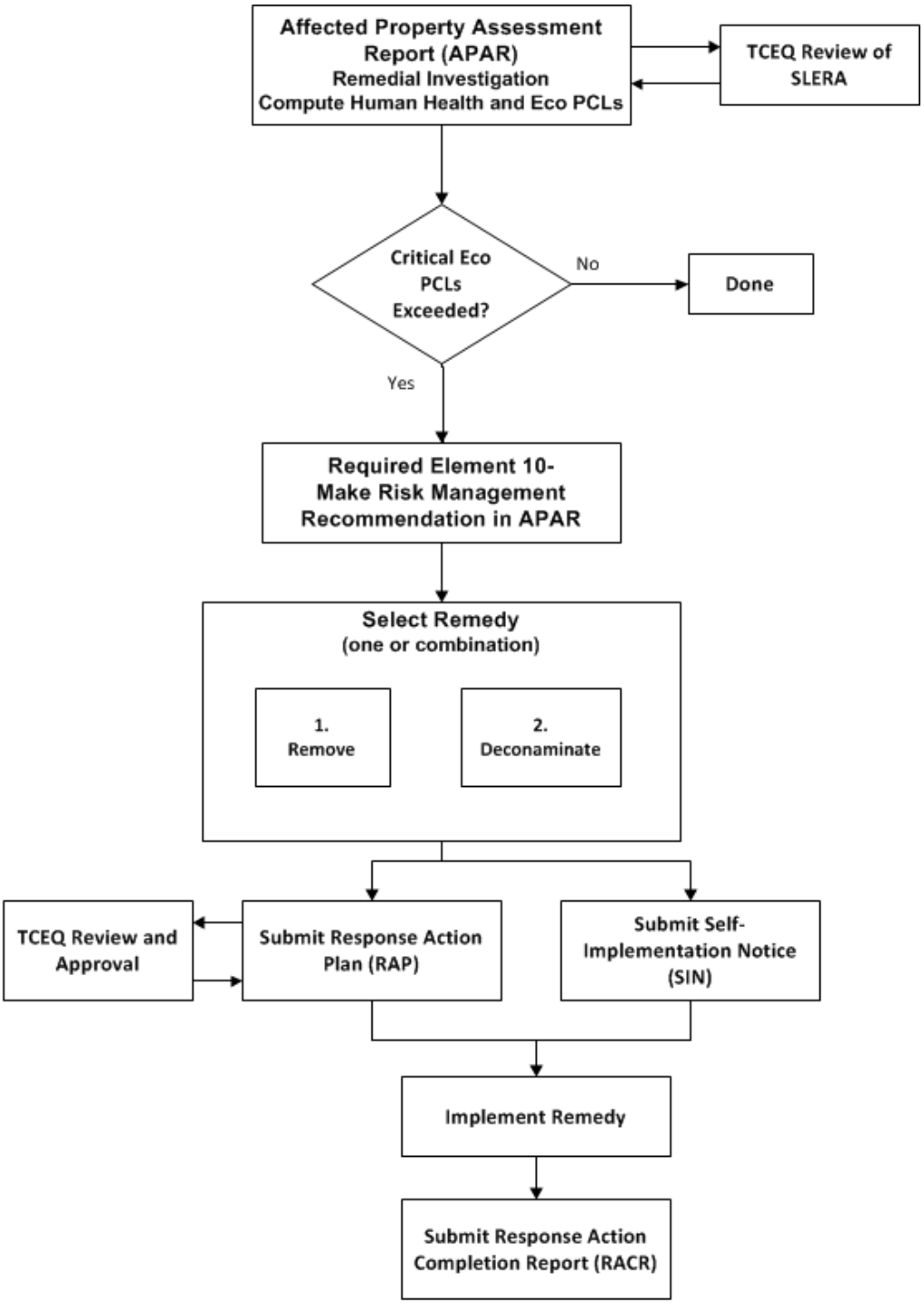
establish maintenance and housekeeping requirements for the property (e.g., remedy-related requirements, habitat maintenance in commercial or industrial areas if appropriate ecological protection is achieved).

- **Ecological Services Analysis** (Remedy Standard B only) involves the completion of an ESA and, where appropriate, the completion of an on- or off-site compensatory ecological-restoration project. This option allows consideration of the potential impacts of the remediation as well as risks associated with exposure to COCs that exceed their respective PCLs. The ESA will involve an analysis of the ecological service flows associated with options that include, but are not limited to, natural attenuation and partial or full-scale remedial actions. In certain cases, ecological restoration may be used to compensate for potential ecological services losses associated with a selected remedy.

Figures 5.2 and 5.3 give an overview of deciding on Remedy Standards A and B, respectively. For practical purposes, there is little technical difference in removal or decontamination under Standards A and B. The key technical differences deal with assumptions based on calculations of human-health risk calculations (e.g., "assume direct contact, no lateral transport"). In application, though, there are many differences between Standards A and B. First, Remedy Standard A is self-implementing. The person need only submit a self-implementation notice (SIN) to the TCEQ before initiating remediation. Second, post-closure care, institutional controls, and possibly financial assurance[12] are required for a Standard B response action.

Neither of those requirements affects Remedy A. However, when submitting a SIN for undertaking a remedy that addresses ecological PCLs, it is recommended that the person consult with the TCEQ and gain approval of the Tier 2 or 3 ERA before initiating the remedy. Otherwise, there is the risk of being required to perform additional response work if the TCEQ disagrees or disapproves the assumptions or calculations made in the Tier 2 or 3 ERA.

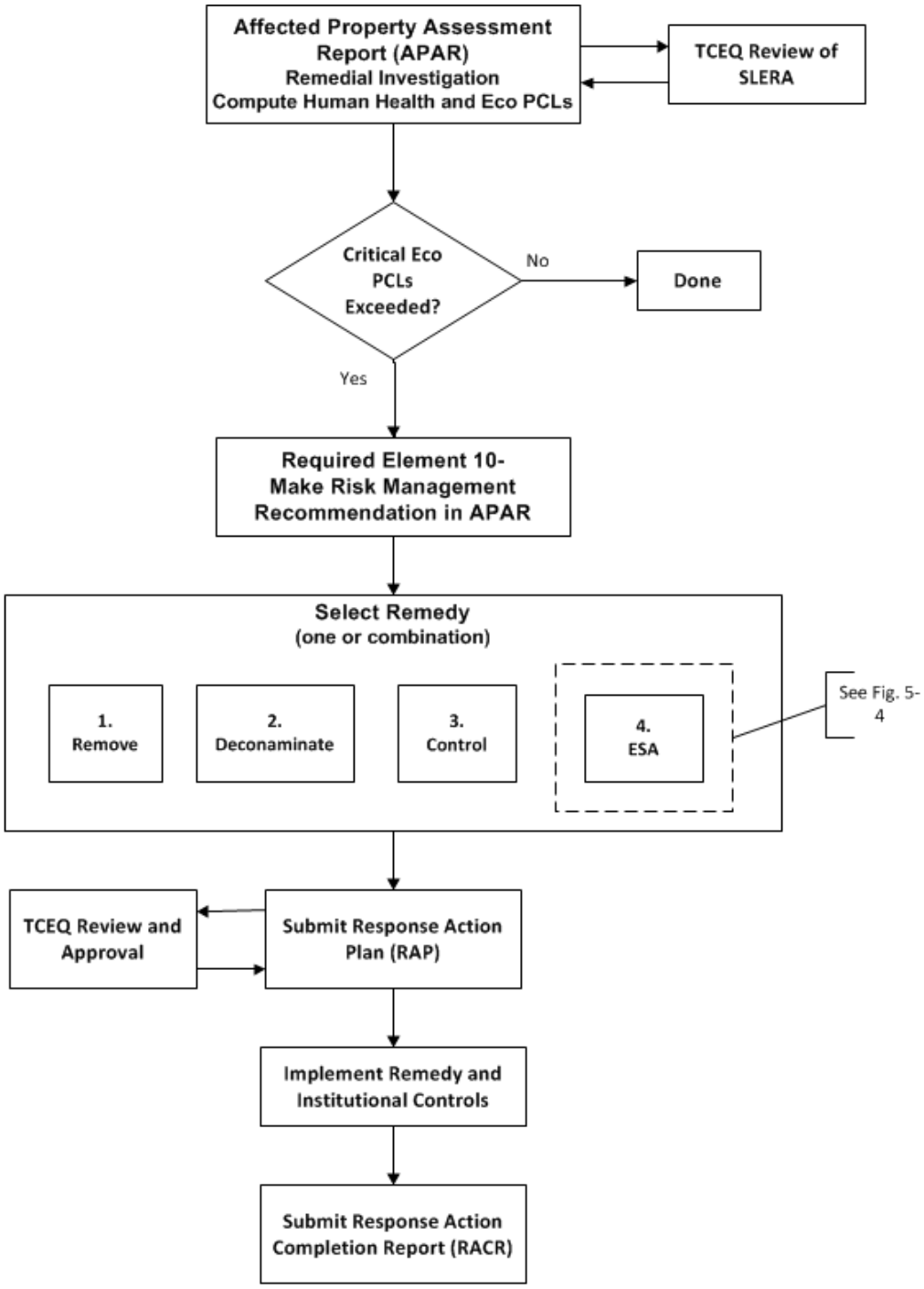
[12]Financial assurance is only required if a physical control is used.



Notes:

PCLs = Protective Concentration Levels
SLERA = Screening Level Ecological Risk Assessment
TCEQ = Texas Commission on Environmental Quality

Figure 5.2. Remedy Standard A for ecological exposures.



Notes:

ESA = Ecological Services Analysis
PCLs = Protective Concentration Levels
SLERA = Screening Level Ecological Risk Assessment
TCEQ = Texas Commission on Environmental Quality

Figure 5.3. Remedy Standard B for ecological exposures.

In recommending a specific remedy [per required element (10); see **3.14**], the person should consider existing background levels (if not previously considered); current and likely future land uses; current and likely future resource uses in the area; the local, regional, and national ecological significance of the affected property; and the potential impacts of available responses, including the potential impacts associated with leaving COCs in place.

When human-health PCLs are exceeded, if human health risks demonstrably are minimal and a human health–based response action would have a "significant and highly disproportionate effect" on ecological receptors, the person may propose that the response not be performed [see 30 TAC 350.33(a)(3)]. In addition to those options, the TCEQ may require some form of post-response sampling or monitoring (see **5.5**).

5.3 Ecological Services Analysis

The ecological-risk-management options available represent an area of significant change from the 1996 edition of this manual. For example, as stated in the TRRP rule, after the ecological risk has been quantified, PCLs established, and the ecological PCL determined to be the critical PCL (i.e., the risk driver) or the only PCL, the person may act to remove, decontaminate, or control contaminated media and COCs. However, to afford additional flexibility where concentrations of COCs do not exceed human health-based levels[13] (either before or after a response action) but do exceed ecological PCLs, the TCEQ allows an ESA to be conducted, as described below and at 30 TAC 350.33(a)(3)(B). The performance of the ESA and any required compensatory ecological restoration must be done in cooperation with and with approval from the Natural Resource Trustees for Texas, including the TCEQ, the Texas Parks and Wildlife Department (TPWD), the Texas General Land Office (TGLO), the National Oceanic and Atmospheric Administration (NOAA), and the U.S. Department of the Interior (DOI), hereinafter collectively referred to as the "Trustees." Additional information on the various Trustee programs, including their legal authority, can be reviewed at <www.tceq.state.tx.us/goto/ntrp>

The ESA considers the present and predicted ecological services of the affected property, as well as beneficial and detrimental effects on services associated with potential responses to residual ecological risks. Furthermore, where appropriate and based upon the results of the ESA, a plan for compensatory ecological restoration may also be combined with some type of response action (e.g., hot-spot removal, monitored natural attenuation) for the affected property. Compensatory ecological restoration provides or restores alternative services when a response at the affected property may cause additional, unwarranted risks to ecological receptors.

The key tenets of the ESA are: (1) even impaired habitats can provide valuable ecological services (e.g., food source, breeding, rearing, shelter),

[13]Except as allowable under 30 TAC 350.33(a)(3).

(2) the "environment" is an ecosystem that extends beyond the perimeter of an affected property, and (3) reduction in habitat services in one location can be addressed by increased services elsewhere in the same ecosystem. The advantage of this option is a net environmental gain (in the form of restoration and conservation of unaffected habitat) with potentially lower associated costs than full-scale remediation. According to the TRRP rule [350.33(a)(3)(B)], the ecological services produced by the restoration must exceed the future ecological service decreases potentially associated with the continued exposure to COCs or any selected response action at the affected property (or both)—i.e., the person is required to compensate beyond actual impacts. Figure 5.4 gives an overview of the ESA as outlined herein, consistent with Remedy Standard B.

5.3.1 Coordination with the TCEQ and the Natural Resource Trustees

If the ESA process is pursued, the person is required to consult with and obtain approval from the TCEQ or other Trustees at two points in the process. First, when the person requests to perform an ESA, he or she must obtain the approval of the TCEQ after it consults with the Trustees. Second, if compensatory ecological restoration is proposed by the person or required by the Trustees as part of the remedy under the ESA option, the person must obtain approval to proceed with an ESA from both the TCEQ and the Trustees [see 30 TAC 350.33(a)(3)(B)]. At any time, the person may elect to withdraw from the ESA process and revert to another appropriate TRRP or RRR response action.

5.3.1.1 TCEQ Approval to Use ESA

Regarding the first requirement [30 TAC 350.77(c)(10) and (f)(2)], the preferred method for requesting approval to pursue an ESA is making the request to the TCEQ as part of the risk-management recommendation under required element (10). Alternatively, the person may submit a separate written request as part of the APAR. In either case, the request should include at least: (1) the location of the area in the affected property where it is proposed that COCs be left in place above ecological PCLs, (2) a description of the habitats within the area, (3) a list of receptors at risk (as defined in the ERA), and (4) a list of COCs within the area. To the extent possible, any other relevant information which may assist the TCEQ in evaluating the request should be included or referenced from the ERA.

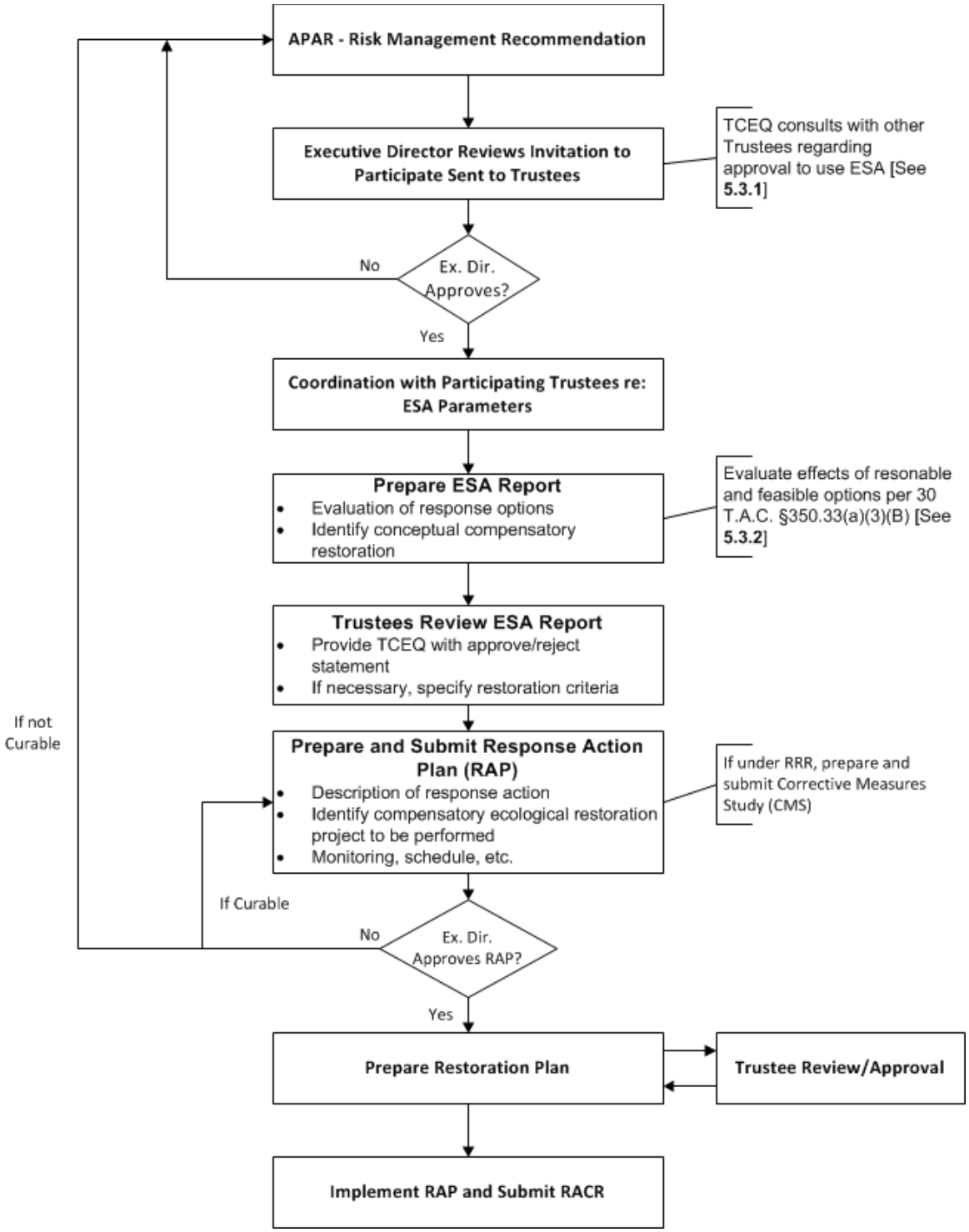


Figure 5.4. Ecological services analysis response action.

The TRRP rule requires that the TCEQ consult with the Trustees prior to approval of a request to conduct the ESA [30 TAC 350.33(a)(3)(B) and 30 TAC 350.77(f)(2)]. After it receives the request, the TCEQ will give timely notification to predesignated contacts within each Trustee agency. Each agency must then respond within a designated time and state whether it needs to be consulted on the request. If all of the other Trustees opt not to

participate or fail to give a timely response, the person has the choice of continuing with the ESA through collaboration with the TCEQ Trustee Program or discontinuing the approach and reverting to a standard TRRP or RRR response.

Upon receipt of a Trustee's intent to be consulted, the TCEQ will coordinate with the Trustee agencies to determine whether the request is appropriate. Any Trustee that recommends against TCEQ approval of the request must supply a reasoned explanation. The TCEQ ecological risk assessor will evaluate the Trustee comments on the request to perform an ESA and the TCEQ project manager will incorporate them into the TCEQ's response to the person, as appropriate. In the event that the TCEQ disapproves the request to perform an ESA, the requester must recommend an alternate response.

5.3.1.2 Performance of ESA Evaluation and Reporting

A memorandum of understanding (MOU) between the TCEQ and Trustees has been developed to ensure the timely and efficient coordination of the consultation with the Trustees on a person's request to perform an ESA. This MOU has been adopted as 30 TAC 7.124. The MOU describes procedures for the distribution of relevant documents and coordination of meetings, sets deadlines for the submission of Trustee comments, and outlines a process for the resolution of conflicting comments. The MOU also institutes a mechanism for the reentry of any Trustee agency into the process even if it elects not to participate at the outset or exits early.

Upon TCEQ approval of the request, the person must coordinate the development of the ESA directly with the Trustees, which have the responsibility of informing the TCEQ remedial-project manager of all ESA activities, copying the manager on all comments, and inviting the manager to all meetings with the person concerning performance of the ESA. To enhance coordination and ensure efficient development of the ESA, the person should initiate a dialogue with the Trustees as soon as the ESA option is considered and maintain open communications with them throughout the process.

The MOU also addresses timely and efficient coordination between the Trustees and the person in the development and implementation of the ESA, prescribing procedures review and approval of the ESA, including designation of a lead Trustee representative as liaison with the person and a mechanism for ensuring a unified Trustee position on all written comments and statements to the person. If the Trustees cannot reach agreement with the person or he or she fails to perform the ESA as proposed, the Trustees will refer the affected property to the TCEQ for further decisions on remedial action.

The culmination of the ESA is the preparation of a report recommending a final remedy for the affected property: e.g., removal, decontamination, control, natural recovery, or compensatory ecological restoration. If the ESA demonstrates that compensatory ecological restoration is required or the

person proposes it, the person must submit a restoration project that produces ecological services greater than the ecological service decreases potentially associated with the continued exposure to COCs or any selected response action at the affected property [see 30 TAC 350.33(a)(3)(B)]. The Trustees will review the ESA report and will forward a written statement to the person and the TCEQ project manager approving or rejecting the conclusions in the report. If approved, the Trustees will specify necessary restoration criteria for the project. The person will then prepare a response action plan giving details of any actions selected, including a conceptual plan of any necessary restoration work and other RAP requirements as specified by 30 TAC 350.94.

5.3.1.3 Compensatory Ecological Restoration as a Response

When any necessary compensatory ecological restoration is completed consistent with Trustee-approved criteria, the Trustees will also send a written statement to both the person and the TCEQ project manager confirming satisfactory completion of the compensatory ecological restoration (under the performance criteria established in the RAP). If the compensatory ecological restoration cannot be completed to the Trustees' satisfaction, they will refer the affected property to the TCEQ for further decisions on remedial action. Frequent coordination with Trustees on the items discussed in this section will facilitate timely and successful completion of the ESA.

Unless otherwise negotiated with the Trustees, the ESA framework addresses only potential prospective ecological service losses (e.g., commencing on the date the affected property assessment data were collected). The TCEQ will not construe a decision to undertake an ESA as an admission that natural resource injuries have occurred or are associated with the affected property.

5.3.2 Evaluation of Response Actions under the ESA

To evaluate which response actions will be selected, the person should compare realistically feasible active and passive alternatives available under Remedy Standard B. Realistic estimates of the positive and negative effects of implementing an option must be demonstrated as part of the evaluation. Ultimately, any responses chosen should allow environmental conditions to return to full function in a reasonable period of time. Refer to *Application of Remedy Standards A and B* (TCEQ publication RG-366/TRRP-28) for a discussion of reasonable time periods.

Responses to consider should include, for example, removal (e.g., excavation), isolation (e.g., capping), and recovery via natural attenuation (e.g., burial by sedimentation or COC degradation). The decision to select natural attenuation should consider the magnitude and spatial scale. As set forth in 30 TAC 350.33(a)(3)(B), combinations of active and passive remedy alternatives, with or without compensatory restoration, may be appropriate. For example, highly contaminated sediments could be removed, while the

remainder of the affected property may best be addressed through a combination of natural attenuation and compensatory ecological restoration. The TRRP rule at 350.33(a)(3)(B) states:

> The ecological services analysis must, at a minimum, include an evaluation of the effects of reasonable and feasible remediation alternatives, including complete removal/decontamination to PCLs and a control measure to prevent ecological exposure to COCs in excess of ecological PCLs, with respect to present and predicted losses of ecological services; and clear justification for leaving COCs in place above ecological PCLs.

Equivalency-analysis tools should be used to compare the negative and positive effects of implementation of these various remedial options and natural process recovery. One of the most readily accessible tools is the habitat-equivalency analysis. HEA is an economic model originally developed by NOAA and the DOI for use in scaling restoration projects to compensate for potential ecological injuries in natural resource damage (NRD) actions. HEA can be used to determine the net present value of ecological services provided by 1 acre of habitat over a specified time measured in discounted service-acre-years (dSAYs). The HEA sees frequent use, including at several sites in Texas by the TCEQ, TPWD, and TGLO.

The implicit assumption behind HEA is that the public is willing to accept a trade-off between lost ecological services and restoration project services. Out-of-kind services can often be normalized for comparison. HEA is applicable when productivity services are considered comparable. A thorough explanation of HEA and associated inputs can be found at <www.darrp.noaa.gov/library/pdf/heaoverv.pdf>. For an example of HEA use, see **5.7**. Other equivalency-analysis tools and methods for scaling compensatory ecological restoration may be considered, but should be presented to the Trustees for consideration prior to the analysis.

Inputs to the HEA and other equivalency models typically include: (1) the date the assessment begins, (2) the area of the impact (as defined by exceedances of ecological PCLs), (3) the severity of the impact (based upon the results of the Tier 2 or 3 ERA), (4) the duration of the impact and recovery time from it (in years), and (5) a discounting factor.

For the ESA, information generated as part of the assessment is critical (e.g., delineation of the extent of the affected area as defined by exceedances of the ecological PCLs in the various environmental media including surficial sediment, surface water, or shallow soil). The quantification of risk in terms of potential ecological-service losses is based on site-specific factors and the resources involved. The process may differ for each site, each case, and each negotiated resolution. While ecological risk and ecological-service losses may not be equivalent, for expediency and cost-effectiveness, the ESA process intends that risk estimation and remedial effects be used to determine potential losses; therefore, areas of excess ecological risk may be useful in initial consideration of the ESA process. As appropriate and after consultation with the Trustees, other factors (e.g.,

biological effects) may also be used to determine potential ecological-services losses and their associated potential responses.

Generally, mortality, reproductive effects (e.g., fecundity reductions, sterility), and growth effects that are used in the ERA should be interpreted as resulting in greater ecological-service losses. Behavioral effects, such as avoidance of contaminated media, are also interpreted as service losses, but to a lesser degree than those involving mortality and reproductive and growth effects.

The ecological "debit," for this discussion, is the amount of restoration necessary to offset the ecological-service losses due to continued exceedances of the ecological PCLs and any impacts from the response action. One key feature of the ESA should be a comparison among the ecological debits for each of the alternatives. Generally, the alternative selected should balance the severity of remaining ecological risk, the length of time necessary for the affected property to recover to pre-release conditions, appropriate compensation for the public, and cost. At any time during the ESA, a person may elect to withdraw from the process and initiate another response under Remedy Standards A or B.

In developing the inputs for the debit calculation, geographic information systems (GIS) can help identify, describe, and measure the spatial extent of potential ecological-service losses, using information from the SLERA. In many cases, literature cited in the SLERA will be useful to develop conservative service-loss values, from the perspective of public trust. Some properties may have developed site-specific interpretive risk data on sediment-quality triads or community structure, etc. Estimates of recovery time may come from literature, site-specific information, or other investigations at the affected property. Regardless, all estimates should be completely justified, thoroughly documented, and reasonably conservative.

5.3.3 Restoration Planning

Three steps in the ESA process involve restoration planning. The first is the development of the report. If the ESA remedy evaluation shows that compensatory ecological restoration is appropriate as a response, the person must first indicate in the ESA report that such restoration will be carried out to compensate for losses of ecological service associated with the continued exceedances of COCs in environmental media. The discussion of restoration at this stage will be conceptual and will identify the habitat types to be addressed as part of the restoration planning (e.g., intertidal marsh, upland forest, shrub and scrub) and the ecological credits necessary to offset the debit associated with the PCL exceedances. Second, if the compensatory ecological restoration proposed in the ESA report is approved by the Trustees, the person will then incorporate more details on the project, in conjunction with the Trustees, for inclusion in the RAP. The compensatory ecological restoration is the response and may be coupled with others as the ESA remedy evaluation shows to be necessary. The RAP should include the following general information regarding the restoration

project: (1) ecological credits required based upon remedy evaluation, (2) discussion of the candidate restoration project (e.g., location, habitat types, proposed restoration actions, acreage, duration, maintenance), and (3) the ecological credits generated by the project. It is strongly recommended that the person coordinate with the Trustees before submitting this information in the RAP so as to avoid delay in its approval. Finally, once the RAP is approved, the person will then develop a detailed restoration plan (detailing the tasks to be conducted and the performance criteria, similar to construction details for other responses) for review and approval by the Trustees. Once approved, the person will then implement the restoration project pursuant to the plan.

To facilitate the identification and selection of a restoration project that compensates for the future ecological risks and lost ecological services associated with the affected property, it is necessary to screen candidate restoration projects against certain criteria. Factors to be considered—to ensure maximum long- and short-term benefits to the ecosystem—include (but are not limited to) proximity of the restoration site to the affected property, hydrology, current uses of the restoration site, and topography. Examples of specific criteria for the selection include:

- The preferred option is for the restoration site to be within the same watershed or ecosystem as the affected property.
- The restoration site should benefit from the enhancement, acquisition, or preservation of the same or similar types of habitat (e.g., vegetation and soil types) as at the affected property.
- The project should be designed to produce the same type of ecological services evaluated as part of the remedy evaluation for the affected property.
- The project and site must have the capacity for long-term success.

These criteria were developed specifically to identify a project site with the potential for habitat restoration, ensure the project has more than enough acreage to compensate for the future potential ecological risks and associated lost services being offset, and allow for a timely and cost-effective project. The ecological credits to be generated by candidate restoration projects should be determined using the same equivalency process used as part of the remedy evaluation.

5.4 Implementing the Response

Under Remedy Standard A, implementation begins 10 days after submission of the SIN to the TCEQ. Under Standard B, implementation begins upon approval of the RAP by the TCEQ. If the person is conducting the remedy under the RRR, implementation of the remedy will only commence upon approval of the ERA by the TCEQ.

5.5 Confirmation Sampling and Monitoring

Final details on the need for confirmation sampling and monitoring should be discussed with the TCEQ project manager and the TCEQ risk assessor, and agreement should be reached before sampling and monitoring begin. Under Remedy Standard B, the sampling and monitoring plan should be included within the RAP for approval [30 TAC 350.94(d) and (f)]. To ensure consistency with previous work for the affected property, all sampling and analytical methods should comply with 30 TAC 350.31 and 350.54.

Confirmation sampling of soil, sediment, groundwater, and surface water may be needed to determine whether concentration-based remedial goals have been achieved. Generally speaking, the confirmation sampling is likely to be less robust than is needed to determine the PCLE zone. In designing an approach to confirmation sampling, it is important to consider an appropriate level of statistical significance that would support any conclusions drawn from the sampling. For example, confirmatory samples may be necessary only at 10 percent of the locations sampled previously, provided that satisfies the statistical criteria used. In some situations, particularly where a natural-attenuation remedy is implemented, confirmatory sampling will not be needed immediately after the remedy is implemented, since there may be no immediate measurable change. Where more active remediation is implemented, confirmatory sampling may need to begin once a discrete area is cleaned, but before the remedial equipment is removed from the affected property. This may prove a cost-effective approach at large properties where equipment is being moved from one area to another over time. In other situations, it may not be feasible, due to the lag time between taking of samples and obtaining the results. In those situations, the person and the TCEQ may agree on more rapid analytical methods that lack the precision and accuracy needed for determining the remedial footprint.

In addition, it is unlikely that biological tissues will need to be sampled immediately after remedial actions are completed. More importantly, where no tissue-specific remedial goal is involved, tissue sampling may be unnecessary. Whether natural attenuation or more active remediation is undertaken, the decline in tissue-specific concentrations may require months or longer to appear, and may be highly variable when sampling animals whose home range is greater than the remedial area. This type of sampling approach requires great caution.

Often, confirmation sampling and monitoring are linked, but they are not the same. Confirmation sampling may be implemented to determine whether a concentration-based remedial goal has been met, whereas post-remedial monitoring generally involves an examination of the chemical and biological characteristics of the affected property over some time period to confirm that continuing conditions conform with the requirements of an approved RAP. Monitoring may involve COC analysis of remediated media such as soil, sediment, groundwater, and surface water. In addition, some monitoring plans may include biological assessments to determine whether

the remedy has achieved improvements in particular biological parameters. Some biological assessments may involve simple surveys of the number and types of flora and fauna on the affected property once remediation is completed (greater abundance and diversity for example), or—in more complex situations—may involve tissue-specific monitoring, as noted previously. An important consideration, however, is that the results of biological assessments may vary widely depending on the type of parameter monitored. The number and type of organisms on an affected property will vary with season and local conditions (food, competition, weather, etc.). Given that animals can forage over large areas, tissue-specific monitoring programs may obtain highly variable and perhaps uninterpretable results if the life history and home range of an animal are not well known. Plants and other sessile organisms will likely serve as more reliable surrogates for tissue-specific concentration monitoring.

The duration of a monitoring program depends on its remedial goals. Some goals can be reached quickly and the need for extensive, long-term monitoring may be limited. Where natural attenuation is applied, monitoring may be needed over several years, at some frequency, to determine whether or not the remedial goals have been achieved. In general, monitoring programs should be consistent with the predicted recovery periods to ensure that remedial goals have been met.

5.6 Reporting and Documentation

There are numerous reporting requirements under the RRR and TRRP rules and, in many cases, under the controlling program for the affected program (e.g., the Superfund or RCRA program). Details of the various TRRP reports are described in 30 TAC 350, Subchapter E. RRR reports (the Remedial Investigation Report, Baseline Risk Assessment, and Corrective Measures Study) are only required when remedial actions under Risk Reduction Standard 3 are pursued.

As to reporting under the TRRP rule, the person must present the PCLs and associated backup information in the APAR. This report essentially contains the results of the investigation of the affected property, as well as the human-health and ecological PCL calculations (if the ERA is subsumed within the APAR). In the APAR, the person is required to make a risk-management recommendation per required element (10) for Tier 2 SLERAs. While the APAR may not specifically identify all aspects of the proposed remedy, it should include enough detail so that the TCEQ can make an informed evaluation of the risk assessment and associated calculations.

A person electing a remedy under Standard A must submit a self-implementation notice <www.tceq.state.tx.us/assets/public/remediation/trrp/forms/10323sin.pdf>. A person submitting a SIN for undertaking a remedy that addresses ecological PCLs is strongly counseled to consult with the TCEQ and gain its approval of the Tier 2 ERA before initiating the remedy. Otherwise, the person runs the risk of being required

to perform additional response work if the TCEQ disagrees with or disapproves of the assumptions or calculations made in the Tier 2 ERA.

A person that elects a remedy under Standard B must submit a RAP to the TCEQ for review and approval. A person pursuing an ESA must prepare and submit an ESA report to the Trustees for review and approval. Once approved and where restoration is required, the person must include in—or submit concurrently with the RAP—a compensatory ecological restoration plan. The response may not commence until the person receives approval or approval with modification from the TCEQ, but that does not preclude interim measures. Additionally, the Trustees must approve any restoration under the ESA option.

Upon approval of the RAP, the person must submit response action effectiveness reports every three years until the response action is complete and, once complete, must then submit a response action completion report (RACR). After the response action is completed, if post-response action care continues, post response action care reports will be required.

5.7 Hypothetical Example Using ESA

5.7.1 Assumptions

Assume the following: (1) COCs were systematically released over an extended period of time from a facility to an adjacent marsh; (2) the release from the facility was recently terminated through source control at the site; (3) a Tier 2 SLERA was conducted using analytical results from sediment samples collected from the marsh; (4) the marsh provides food and habitat for a variety of small and large birds, mammals, fish, and invertebrates; (5) the findings of the risk assessment conclude that there are potential risks to the invertebrates (because of direct contact with the sediment) and the small birds that eat the invertebrates (referred to as small invertivorous birds); and (6) as a result of biodegradation and natural attenuation, these risks are projected to dissipate to acceptable levels within 10 years (as demonstrated through modeling).

5.7.2 Removal, Decontamination, Control

Given the above assumptions (and absent or minimal risks to human health), the site risk manager must consider all appropriate risk-management options. Options to remove or control (e.g., cap) all of the sediment exhibiting concentration levels that are contributing to the risks will eliminate risks to the invertebrates and small invertivorous birds by breaking the exposure pathway. However, the marsh is also inhabited by many other organisms not at risk (e.g., large birds, mammals). These options would result in the destruction of a viable habitat currently providing many valuable direct and indirect ecological services. Moreover, these options might create a condition worse than that posed by the COCs. Last, both of these options can be extremely costly. Generally, decontamination options include natural attenuation, degradation, or

a variety of *in situ* methods. Costs associated with natural attenuation (or degradation) would be small compared to removal costs, but could result in the continuing exposure of receptors to COCs above ecological PCLs for years. *In situ* methods such as phytoremediation would likely not be appropriate in a marsh.

5.7.3 Ecological Services Analysis

The ESA process addresses potential losses of ecological service associated with COCs in a habitat or remedial activities. As stated in **5.3.2**, the use of HEA or other suitable models is appropriate when an ESA is undertaken. A HEA can be used to determine the net present value of ecological services provided by 1 acre of habitat over a specified time period and can estimate potential reduction in ecological-service debits while putting into context habitat structure and quality and the relative importance of the ecological receptors. Through a HEA, the person can evaluate and quantitatively rank the most environmentally protective response options (in dSAY debits) in comparison to the ecological-service reductions (or theoretical risks) currently posed by the COCs. Concurrently, the person can evaluate and rank potential restoration projects that create ecological credits that seek to replace or offset debits at the affected property.

In the hypothetical affected marsh, a HEA analysis of the available remedy options might generate the following results: (1) natural attenuation would achieve a debit of 100 dSAYs that is attributable to the continuing presence of COCs in the sediment above ecological PCLs, including hot spots; (2) the small-scale removal of COCs would achieve a net debit of 120 dSAYs, and (3) large-scale removal would achieve a net debit of 150 dSAYs. Under both the removal and the control options, the benefit is offset by services lost as a result of the destruction or alteration of the habitat. Assuming these are the only reasonable and feasible options, there is no ecological benefit; moreover, the associated costs are likely to far outweigh the monitoring costs associated with the natural-attenuation option, which thus appears to be the most appropriate. However, depending on the magnitude of the risk, the types of ecological receptors and habitats exposed, and the area affected by the COCs, natural attenuation alone may not suffice.

It may be appropriate to combine the natural attenuation option with a commitment to compensatory ecological restoration and routine COC monitoring at the property. The restoration activities would focus on repairing or rehabilitating the types of resources (invertebrates and small invertivorous birds) potentially affected by the COCs present in the marsh. The final result is an exchange of the ecological service debits associated with the theoretical ecological risks and uncertainties with ecological service credits generated by a restoration project located within the same ecosystem. For example, it was assumed above that the natural attenuation option would result in a debit of 100 dSAYs. Conceivably, restoration of that marsh or enhancement of a similar marsh could generate 200 dSAYs credits. Combining the two results in an overall service credit of 100 dSAYs, which directly and indirectly benefits the environment, agency

stakeholders, public, and the property owner. Depending on the magnitude of the ecological risks and other factors, another combination of remedies might include limited or small-scale removal and compensatory ecological restoration.

6 THE FUTURE OF ERAs

Several topics associated with ERAs are not presented in detail in this revision. Potential candidates for future updates include discussions of probabilistic risk assessment, endocrine disruptors, equilibrium partitioning, and fugacity. A brief discussion follows.

6.1 Probabilistic Risk Assessment

The TCEQ is not currently accepting the use of probabilistic risk assessments. In the future, once a process is in place for standardization and review of PRAs, the agency may allow their use. A change in the TRRP rule may be needed for use of PRAs in human health risk assessments because of existing limitations on exposure parameters used for human health. However, for ecological assessments, no specific limitations on exposure assessments exist. As currently understood, a PRA would require extensive data sets that are seldom available for ecological assessments. Consequently, the issue is of low priority within the TCEQ. However, a TRRP work group is considering PRA issues for human-health assessments and its future recommendations may be applicable to ERAs.

6.2 Endocrine Disruptors

At present, information about endocrine disruptors is insufficient to create a suitable validated procedure for identifying or quantifying risk. The EPA's recent report to Congress observed that there are no validated assays for endocrine disruption (see <www.epa.gov/scipoly/oscpendo/>). The report has begun to identify the parameters necessary to begin screening for substances with potential endocrine effects, and to confirm and characterize those effects. It concluded that a proposed screening tool (the High Throughput Pre-Screening Process) needs further development. The EPA is continuing research in cooperation with the Committee on Environment and Natural Resources of the National Science and Technology Council. Endpoints under consideration include various biochemical markers, often reflecting endocrine status or maturation, and future work may link those to reproductive endpoints within the scope of the TRRP. More information can be found at the Web page linked above, or in various review volumes (e.g., DeFur et al., 1999; DiGiulio and Tillitt, 1999; Henshel et al., 1999). Future revisions of this manual may discuss endocrine disruption.

6.3 Equilibrium Partitioning

As the use of EqP to develop sediment quality benchmarks and PCLs evolves in practice (e.g., U.S. EPA, 1999b–d; Di Toro et al., 2000; Di Toro and McGrath, 2000; Fuchsman, 2003) the TCEQ will continue to evaluate the concept and may increase its reliance on that approach.

6.4 Fugacity

Fugacity-based modeling of bioaccumulation in food webs has been advocated as a consistent framework (Sharpe and Mackay, 2000). The framework, based on the partitioning tendencies (*fugacities*) of chemicals, uses physical properties such as log K_{ow}, water solubility, and the Henry's Law constant. However, application of the approach requires estimation of the mass balance of COCs in the system under evaluation in order to estimate the fugacities. The Gobas (1993) model used in U.S. EPA (1999a) to derive the FCM tables is actually fugacity based. In practice, projects to create fugacity-based models have only been conducted infrequently and using large data sets. The number of site-specific parameters is consistently rather large, although it may be reduced by making assumptions about depth, water-to-sediment ratios, levels of suspended solids, and organic carbon content. Fugacity modeling may be discussed in future revisions of this publication.

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8 DEFINITIONS

95 percent upper confidence limit (of a mean)—A value that, when calculated repeatedly for randomly drawn subsets of site data, equals or exceeds the true mean 95 % of the time.

affected property[R]—The entire area (i.e., on-site and off-site; including all environmental media) which contains releases of chemicals of concern at concentrations equal to or greater than the assessment level applicable for residential land use and groundwater classification.

area-use factor—The ratio of an organism's home range, breeding range, or feeding and foraging range to the area of contamination of the site under investigation.

assessment endpoint—An explicit expression of an environmental value to be protected.

assessment level[R]—A critical protective concentration level for a chemical of concern used for affected property assessments where the human health protective concentration level is established under a Tier 1 evaluation as described in §350.75(b) of [Title 30, TAC] (relating to Tiered Human Health Protective Concentration Level Evaluation), except for the protective concentration level for the soil-to-groundwater exposure pathway which may be established under Tier 1, 2, or 3 as described in §350.75(i)(7) of [Title 30], and ecological protective concentration levels which are developed, when necessary, under Tier 2 and/or 3 in accordance with §350.77(c) and/or (d), respectively, of [Title 30] (relating to Ecological Risk Assessment and Development of Ecological Protective Concentration Levels).

background[R]—A population of concentrations characterized from samples in an environmental medium containing a chemical of concern that is naturally occurring (i.e., the concentration is not due to a release of chemicals of concern from human activities) or anthropogenic (i.e., the presence of a chemical of concern in the environment which is due to human activities, but is not the result of site-specific use or release of waste or products, or industrial activity). Examples of anthropogenic sources include non-site specific sources such as lead from automobile emissions, arsenic from use of defoliants, and polynuclear aromatic hydrocarbons resulting from combustion of hydrocarbons. There are some commonalities regardless of the activity; specifically, the chemicals of concern have resulted from the use of a product in its intended manner and may be present at generally low levels over large areas (tens of square miles up to hundreds of square miles). Background is required for use in a statistical model appropriate for testing the hypothesis that the background area characterized by these kinds of models has the same concentrations of the chemical of concern as the affected property. The background area characterized is as "close" as possible to the affected property, in either space or time, as required.

bedrock[R]—The solid rock (i.e., consolidated, coherent, and relatively hard naturally formed material than cannot normally be excavated by manual methods alone) that underlies gravel, soil or other surficial material.

benthic community—The community of organisms dwelling at the bottom of a pond, river, lake, or ocean.

bioaccumulation—General term describing a process by which chemicals are taken up by an organism from both water and food containing the chemicals.

bioccumulation factor—The ratio of the concentration of a chemical of concern in an organism to the concentration in the ambient environment at steady state.

bioaccumulative chemical of concern[R]—A chemical of concern which has the tendency to accumulate in the tissues of an organism as a result of food consumption or dietary exposure and/or direct exposure (e.g., gills and epithelial tissue) to an environmental medium.

bioavailability—The degree to which a material in environmental media can be assimilated by an organism.

bioconcentration—Net accumulation of a chemical directly from an exposure medium into an organism.

biomagnification—An increase in tissue concentrations of chemicals in organisms as those materials pass up the food chain.

chemical of concern[R]—Any chemical that has the potential to adversely affect ecological or human receptors due to its concentration, distribution, and mode of toxicity. Depending on the program area, chemicals of concern may include the following: solid waste, industrial solid waste, municipal solid waste, and hazardous waste as defined in the Texas Health and Safety Code, §361.003, as amended; hazardous constituents as listed in 40 Code of Federal Regulations Part 261, Appendix VIII, as amended; constituents on the groundwater monitoring list in 40 Code of Federal Regulations Part 264, Appendix IX, as amended; constituents as listed in 40 Code of Federal Regulations Part 258 Appendices I and II, as amended; pollutant as defined in Texas Water Code, §26.001, as amended; hazardous substance as defined in the Texas Health and Safety Code, §361.003, as amended, and Texas Water Code, §26.263, as amended; other substances as defined in Texas Water Code §26.039(a), as amended; and daughter products of the aforementioned constituents.

closure[R]—The act of permanently taking a waste management unit or facility out of service.

commercial/industrial land use[R]—Any real property or portions of a property not used for human habitation or for other purposes with a similar potential for human exposure as defined for residential land. Examples of commercial/industrial land use include manufacturing; industrial research and development; utilities; commercial warehouse operations; lumber

yards; retail gas stations; auto service stations; auto dealerships; equipment repair and service stations; professional offices (lawyers, architects, engineers, real estate, insurance, etc.); medical/dental offices and clinics (not including hospitals); financial institutions; office buildings; any retail business whose principal activity is the sale of food or merchandise; personal service establishments (health clubs, barber/beauty salons, mortuaries, photographic studios, etc.); churches (not including churches providing day care or school services other than during normal worship services); motels/hotels (not including those which allow residence); agricultural lands; and portions of government-owned land (local, state, or federal) that have commercial/industrial activities occurring. Land use activities consistent with this classification have the North American Industrial Classification System code numbers 11–21 inclusive; 22 except 22131; 23–56 inclusive; 61 except 61111, 61121, and 61131; 62 except 62211, 62221, 62231, 62311, 62322, 623311, 623312, 62399, and 62441; 71 except 71219; 72 except 721211 and 72131; 81 inclusive; and 92 excluding 92214.

community[R]—An assemblage of plant and animal populations occupying the same habitat in which the various species interact via spatial and trophic relationships (e.g., a desert community or a pond community).

compensatory ecological restoration[R]—The creation of ecological services by or through restoration or the setting aside of, preferably, a comparable type of habitat as that which is impacted to offset residual ecological risk at an affected property. A net environmental benefits analysis or similar evaluation of ecological services may be used in the determination of the appropriate level of compensation.

complete exposure pathway[R]—An exposure pathway where a human or ecological receptor is exposed to a chemical of concern via an exposure route (e.g., incidental soil ingestion, inhalation of volatiles and particulates, consumption of prey, etc.)

conceptual model—A series of working hypotheses of how a stressor might affect ecological components. Describes ecosystem or ecosystem components potentially at risk and the relationships between measurement and assessment endpoints and exposure scenarios.

control[R]—To apply physical or institutional controls to prevent exposure to chemicals of concern. Control measures must be combined with appropriate maintenance, monitoring, and any necessary further response action to be protective of human health and the environment.

critical protective concentration level[R]—The lowest protective concentration level for a chemical of concern within a source medium determined from all of the applicable human health exposure pathways as described in 350.71 of [Title 30, TAC] (relating to General Requirements), and when necessary, protective concentration levels for applicable ecological exposure pathways as required in 350.77 of [Title 30] (relating to Ecological Risk Assessment and Development of Ecological Protective Concentration Levels).

decontaminate[R]—Application or occurrence of a permanent and irreversible treatment process to a waste or environmental medium so that the threat of release of chemicals of concern at concentrations above the critical protective concentration levels is eliminated.

deed notice[R]—An instrument filed in the real property records of the county where the affected property is located that is intended to provide to owners, prospective buyers and others notice and information regarding, but which does not, by itself, restrict use of the affected property.

de minimis[R]—The description of an area of affected property comprised of one acre or less where the ecological risk is considered to be insignificant because of the small extent of contamination, the absence of protected species, the availability of similar unimpacted habitat nearby, and the lack of adjacent sensitive environmental areas.

disturbed ground (also **disturbed area** or **setting**)—A location where any ecological habitat that may have once existed has been altered, changed, or reduced to a degree that it is no longer conducive to use by ecological receptors (e.g., pavement, process areas, buildings). These locations are predominantly urban or commercial/industrial and are often characterized by human presence and activities.

dose—A measure of exposure. Examples include the amount of a chemical ingested, the amount of a chemical absorbed, and the product of ambient exposure concentration and the duration of exposure.

dose-response curve—The relationship between a change in effect on an organism caused by differing levels of exposure (or doses) to a stressor (usually a chemical) after a certain exposure time.

ecological benchmark[R]—A state standard, federal guideline, or other exposure level for a chemical of concern in water, sediment, or soil that represents a protective threshold from adverse ecological effects. An ecological benchmark may also be a toxicity reference value that is established by the person based on scientific studies in the literature.

ecological hazard index[R]—The sum of individual ecological hazard quotients of COCs within a class of compounds that exert ecological effects which have the same toxicological mechanism or endpoint (e.g., PAHs, PCBs).

ecological hazard quotient[R]—The ratio of an exposure level to a chemical of concern to a toxicity value selected for the risk assessment for that chemical of concern (e.g., a no observed adverse effects level).

ecological protective concentration level[R]—The concentration of a chemical of concern at the point of exposure within an exposure medium (e.g., soil, sediment, groundwater, or surface water) which is determined in accordance with §350.77(c) or (d) of [Title 30, TAC] (relating to Ecological Risk Assessment and Development of Ecological Protective Concentration Levels) to be protective for ecological receptors. These concentration levels

are primarily intended to be protective for more mobile or wide-ranging ecological receptors and, where appropriate, benthic invertebrate communities within the waters in the state. These concentration levels are not intended to be directly protective of receptors with limited mobility or range (e.g., plants, soil invertebrates, and small rodents), particularly those residing within active areas of a facility, unless these receptors are threatened/endangered species or unless impacts to these receptors result in disruption of the ecosystem or other unacceptable consequences for the more mobile or wide-ranging receptors (e.g., impacts to an off-site grassland habitat eliminate rodents which causes a desirable owl population to leave the area).

ecological risk assessment[R]—The process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors; however, as used in this context, only chemical stressors (i.e., COCs) are evaluated.

ecological services[R]—The physical, chemical, or biological functions of natural resources that one natural resource provides for another or to the public. Examples include provision of food, protection from predation, and nesting habitat, among others.

ecological services analysis[R]—A measurement of the potential change in ecological services based on considerations which may include but are not limited to: the percent change in ecological services at the affected property that are attributable to COCs and/or potential response actions; the spatial extent of the affected property; and the recovery period.

ecosystem—The biotic community and abiotic environment at a specified location and time.

ecotoxicity—Toxic effects on nonhuman organisms, populations, or communities.

environmental medium[R]—A material found in the natural environment such as soil (including non-waste fill materials), groundwater, air, surface water, and sediments, or a mixture of such materials with liquids, sludges, gases, or solids, including hazardous waste which is inseparable by simple mechanical removal processes, and is made up primarily of natural environmental material.

exclusion criteria[R]—Those conditions at an affected property which preclude the need to establish a protective concentration level for an ecological exposure pathway because the exposure pathway between the chemical of concern and the ecological receptors is not complete or is insignificant.

exposure—Co-occurrence of or contact between a stressor and an ecological component. The contact reaction between a chemical and a biological system, or organism.

exposure area[R]—The smallest property surface area within which it is believed that exposure to chemicals of concern in soil or air by a receptor would be limited under reasonably anticipated current or future use scenarios.

exposure assessment—The determination or estimation (qualitative or quantitative) of the magnitude, frequency, duration, and route of exposure.

exposure medium[R]—The environmental medium or biologic tissue in which or by which exposure to chemicals of concern by ecological or human receptors occurs.

exposure pathway[R]—The course that a chemical of concern takes from a source area to ecological or human receptors and includes a source area, a point of exposure, and an exposure route (e.g., ingestion), as well as a transport mechanism if the point of exposure is different from the source area.

facility[R]—The installation associated with the affected property where the release of chemicals of concern occurred.

feeding guilds[R]—Groups of ecological receptors used to represent the variety of species that may be exposed to chemicals of concern at the affected property. The feeding guilds are generally based on function within an ecosystem, potential for exposure, and physiological and taxonomic similarity. Examples include carnivorous mammals, carnivorous birds, and piscivorous birds.

food-chain transfer—A process by which substances in the tissues of lower-trophic-level organisms are transferred to the higher-trophic-level organisms that feed on them.

fossorial—Describes an animal that is adapted for burrowing.

functioning cap[R]—A low permeability layer or other approved cover meeting its design specifications to minimize water infiltration and chemical of concern migration, and prevent ecological or human receptor exposure to chemicals of concern, and whose design requirements are routinely maintained.

groundwater-bearing unit[R]—A saturated geologic formation, group of formations, or part of a formation which has a hydraulic conductivity equal to or greater than 1×10^{-5} centimeters/second.

habitat—Place where a plant or animal lives, often characterized by a dominant plant form and physical characteristics.

home range—The area to which an animal confines its activities including foraging and nesting.

Implementation Procedures[R]—The most current version of *Procedures to Implement the Texas Surface Water Quality Standards,* as amended.

institutional control[R]—A legal instrument placed in the property records in the form of a deed notice, Voluntary Cleanup Program Certificate of Completion (VCP Certificate of Completion), or restrictive covenant which indicates the limitations on or the conditions governing use of the property which ensures protection of human health and the environment or equivalent zoning and governmental ordinances.

landscaped area[R]—An area of ornamental, introduced, commercially installed, or manicured vegetation which is routinely maintained.

LC_{50}—The concentration of a toxicant that is lethal (fatal) to 50 percent of the organisms tested in a specified time period.

LD_{50}—The dose of a toxicant that is lethal (fatal) to 50 percent of the organisms tested in a specified time period.

long-term effectiveness[R]—The ability of a remedy to maintain the required level of protection of human health and the environment over time.

lowest observed adverse-effect level (LOAEL)—The lowest level of a stressor evaluated in a toxicity test or biological field survey that has a statistically significant adverse effect on the exposed organisms compared with unexposed organisms in a control or reference site.

measurement endpoint—A measurable ecological characteristic that is related to the valued characteristic chosen as the assessment endpoint. Measurement endpoints often are expressed as the statistical or arithmetic summations of the observations that make up the measurement. Can include measures of effect and measures of exposure.

method detection limit[R]—The minimum concentration of a substance that can be measured and reported with 99% confidence that the analyte concentration is greater than zero and is determined for each COC from the analysis of a sample of a given matrix type containing the COC.

method quantitation limit[R]—The lowest non-zero concentration standard in the laboratory's initial calibration curve and is based on the final volume of extract (or sample) used by the laboratory.

monitored natural attenuation[R]—The use of natural attenuation within the context of a carefully controlled and monitored response action to achieve protective concentration levels at the point of exposure.

natural attenuation[R]—The reduction in mass or concentration of a chemical of concern over time or distance from the source of a chemical of concern due to naturally occurring physical, chemical, and biological processes, such as: biodegradation, dispersion, dilution, adsorption, and volatilization.

natural attenuation factor[R]—The numerical value which represents the natural attenuation (i.e., reduction) in chemical of concern concentrations during transport from the source area to the point of exposure. The natural

attenuation factor is the concentration at the source area divided by the concentration at the point of exposure. The natural attenuation factor is always greater than or equal to one for the purposes of this rule.

Natural Resource Trustees[R]—The federal agencies as designated by the President and the state agencies as designated by the Governor pursuant to the National Contingency Plan, Oil Pollution Act, and CERCLA §107(f)(2)(A) and (B) to act on behalf of the public as trustees of natural resources (e.g., water, air, land, wildlife). The Trustees include TCEQ, Texas Parks and Wildlife Department, Texas General Land Office, National Oceanic and Atmospheric Administration, and the Department of the Interior.

"no observed adverse effect" level (NOAEL)—The highest level of a stressor evaluated in a toxicity test or biological field survey that causes no statistically significant difference in effect compared with the controls or a reference site.

off-site property (off-site) [R]—All environmental media which is outside of the legal boundaries of the on-site property.

on-site property (on-site) [R]—All environmental media within the legal boundaries of a property owned or leased by a person who has filed a self-implementation notice or a response action plan for that property or who has become subject to such action through one of the agency's program areas for that property.

person[R]—An individual, corporation, organization, government or governmental subdivision or agency, business trust, partnership, association, or any other legal entity.

[Specifically, "person," throughout this document, is meant to denote the regulated entity or environmental consultant that is performing the ERA or ecological services analysis.]

physical barrier[R]—Any structure or system, natural or manmade, that prevents exposure or prevents migration of chemicals of concern to the points of exposure.

physical control[R]—A structure or hydraulic containment action which prevents exposure to and/or migration of chemicals of concern when combined with appropriate post-response action care to protect human health and the environment. Examples of physical controls are caps, slurry walls, sheet piling, hydraulic containment wells, and interceptor trenches, but typically not fences.

point of exposure[R]—The location within an environmental medium where a receptor will be assumed to have a reasonable potential to come into contact with chemicals of concern. The point of exposure may be a discrete point, plane, or an area within or beyond some location.

population—An aggregate of individuals of a species within a specified location in space and time.

practical quantitation limit—The lowest concentration of an analyte which can be reliably quantified within specified limits of precision and accuracy during routine laboratory operating conditions. The PQL minimizes to the extent possible the effects of instrument and operator variability and the influences of the sample matrix and other COCs or substances upon the quantitation of the analyte. "Specified limits of precision and accuracy" are the criteria which have been included in applicable regulations or which are listed in the quality control sections of the analytical method. The PQL may be directly obtained or derived from the following sources with preference given to the most recent, scientifically valid method: federal regulations; EPA guidance documents; calculation from interlaboratory studies; and experimentally determined analytical methods not available from other existing sources.

prescribed points of exposure[R]—The prescribed on-site and off-site locations within an environmental medium where an individual human or population will be assumed to come into contact with chemicals of concern from an affected property.

protective concentration level[R]—The concentration of a chemical of concern which can remain within the source medium and not result in levels which exceed the applicable human health risk-based exposure limit or ecological protective concentration level at the point of exposure for that exposure pathway.

protective concentration level exceedance zone[R]—The lateral and vertical extent of all wastes and environmental media which contain chemicals of concern at concentrations greater than the critical protective concentration level determined for that medium, as well as, hazardous waste. A protective concentration level exceedance zone can be thought of as the volume of waste and environmental media which must be removed, decontaminated, and/or controlled in some fashion to adequately protect human health and the environment.

reasonably anticipated to be completed exposure pathway[R]—A situation with a credible chance of occurrence in which an ecological or human receptor may become exposed to a chemical of concern (i.e., complete exposure pathway) without consideration of circumstances which are extreme or improbable based on property characteristics.

reference site—A relatively uncontaminated site used for comparison to contaminated sites in environmental monitoring studies, often incorrectly referred to as a “control.”

release[R]—Any spilling, leaking, pumping, pouring, emitting, emptying, discharging, injecting, escaping, leaching, dumping, or disposing into the environment, with the exception of:

A. A release that results in an exposure to a person solely within a workplace, concerning a claim that the person may assert against the person’s employer;

B. An emission from the engine exhaust of a motor vehicle, rolling stock, aircraft, vessel, or pipeline pumping station engine;

C. A release of source, by-product, or special nuclear material from a nuclear incident, as those terms are defined by the Atomic Energy Act of 1954, as amended (42 U.S.C. §2011 et seq.), if the release is subject to requirements concerning financial protection established by the Nuclear Regulatory Commission under §170 of that Act;

D. For the purposes of the environmental response law §104, as amended, or other response action, a release of source, by-product, or special nuclear material from a processing site designated under §102(a)(1) or §302(a) of the Uranium Mill Tailings Radiation Control Act of 1978 (42 U.S.C. §7912 and §7942), as amended; and

E. The normal application of fertilizer.

remediation[R]—The act of eliminating or reducing the concentration of chemicals of concern in environmental media.

remove[R]—To take waste or environmental media away from the affected property to another location for storage, processing or disposal in accordance with all applicable requirements. Removal is an irreversible process that results in permanent risk reduction at an affected property.

residential land use[R]—Property used for dwellings such as single family houses and multi-family apartments, children's homes, nursing homes, and residential portions of government-owned lands (local, state or federal). Because of the similarity of exposure potential and the sensitive nature of the potentially exposed population, day care facilities, educational facilities, hospitals, and parks (local, state or federal) shall also be considered residential.

response action[R]—Any activity taken to comply with these regulations to remove, decontaminate and/or control (i.e., physical controls and institutional controls) chemicals of concern in excess of critical PCLs in environmental media, including actions taken in response to releases to environmental media from a waste management unit before, during, or after closure.

risk—The expected frequency or probability of undesirable effects resulting from exposure to known or expected stressors.

risk-based exposure limit[R]—The concentration of a chemical of concern at the point of exposure within an exposure medium (e.g., soil, sediment, vegetables, groundwater, surface water, or air) which is protective for human health. Risk-based exposure limits are the fundamental risk-based values which are initially determined and used in the development of protective concentration levels. Risk-based exposure limits do not account for cumulative effects from exposure to multiple chemicals of concern, combined exposure pathways, and cross-media or lateral transport of chemicals of concern within environmental media.

sample detection limit[R]—The method detection limit, as defined in this section, adjusted to reflect sample-specific actions, such as dilution or use of smaller aliquot sizes than prescribed in the analytical method, and to take into account sample characteristics, sample preparation, and analytical adjustments. The term, as used in [30 TAC 350], is analogous to the sample-specific detection limit.

sediment[R]—Non-suspended particulate material lying below surface waters such as bays, the ocean, rivers, streams, lakes, ponds, or other similar surface water body (including intermittent streams). Dredged sediments which have been removed from below surface water bodies and placed on land shall be considered soils.

selected ecological receptors[R]—Species that are to be carried through the ecological risk assessment as representatives of the different feeding guilds and communities that are being evaluated. These species may not actually occur at the affected property, but may be used to represent those within the feeding guild or community that may feed on the affected property.

sensitive environmental areas[R]—Areas that provide unique and often protected habitat for wildlife species. These areas are typically used during critical life stages such as breeding, hatching, rearing of young, and overwintering. Examples include critical habitat for threatened and endangered species, wilderness areas, parks, and wildlife refuges.

soil protective concentration level exceedance zone[R]—A protective concentration level exceedance zone within the surface soil or subsurface soil which may extend down to a groundwater-bearing unit(s). These protective concentration level exceedance zones may also be present below or between groundwater-bearing units.

source area[R]—The volume of a chemical of concern in environmental media (e.g., soil or groundwater) which is leaching, dissolving or emitting chemicals of concern. Of primary regulatory concern are the source areas that are leaching, dissolving or emitting chemicals of concern at unprotective concentrations under natural conditions, and not in consideration of any physical controls (e.g., slurry walls, caps), that will result in protective concentrations being exceeded at the point of exposure. The source area need not be the horizontal and vertical extent of the protective concentration level exceedance zone when cross-media or lateral chemical of concern transport is required for a point of exposure to be reached. Generally, a source area is located in the vicinity of or below primary release sources (e.g., tanks, pipelines, drums, lagoons, landfills, etc.).

source medium[R]—An environmental medium containing chemicals of concern which must be removed, decontaminated and/or controlled in order to protect human health and the environment. The source medium may be the exposure medium for some exposure pathways.

stressor[R]—Any physical, chemical, or biological entity that can induce an adverse response; however, as used in this context, only chemical entities apply.

subsurface soil[R]—For human health exposure pathways, the portion of the soil zone between the base of surface soil and the top of the groundwater-bearing unit(s). For ecological exposure pathways, the portion of the soil zone between 0.5 feet and 5 feet in depth.

surface cover[R]—A layer of artificially placed utility material (e.g., shell, gravel).

surface soil[R]—For human health exposure pathways, the soil zone extending from ground surface to 15 feet in depth for residential land use and from ground surface to 5 feet in depth for commercial/industrial land use; or to the top of the uppermost groundwater-bearing unit or bedrock, whichever is less in depth. For ecological exposure pathways, the soil zone extending from ground surface to 0.5 feet in depth.

surface water[R]—Any water meeting the definition of surface water in the state as defined in §307.3 of [Title 30, TAC] (relating to Definitions and Abbreviations), as amended.

toxicity reference value[R]—An exposure level from a valid scientific study that represents a conservative threshold for adverse ecological effects.

trophic level—A functional classification of taxa within a community based on feeding relationships (e.g., aquatic and terrestrial plants make up the first trophic level, and herbivores make up the second).

[R] Definition as it appears in the Texas Risk Reduction Program rule 30 TAC 350. Current at time of publication.

APPENDIX A: Derivation of the Ecological-Screening Benchmarks

Water Benchmarks

The benchmarks for surface water are intended to protect aquatic biota—not necessarily mammals and birds that may be exposed to COCs through ingestion of contaminated prey or water. The following discussion details the derivation of the water benchmarks, discusses the use of site-specific data, and sets forth a preferred hierarchy for selecting a water benchmark. (See also Figure 3.4).

State and Federal Criteria

The Texas Surface Water Quality Standards (30 TAC 307, as amended), serve as the primary benchmarks for surface water. When there is no state standard, the federal National Ambient Water Quality Criteria are be used (U.S. EPA, 2013). Benchmark values appear in Table 3.2. The federal and state standards are generally protective of sensitive aquatic species, have undergone rigorous review and comment, and are generally accepted by the regulated community. Since a comparison of concentrations to benchmarks is intended to be a screening step, the chronic criteria for the protection of aquatic life are used. According to the TSWQS, marine waters (= salt water) are coastal waters with measurable elevation changes due to normal tides, typically having salinities of 2 parts per thousand or greater in a significant portion of the water column.

For a number of metals, the freshwater criterion is a function of hardness. The benchmark values indicated in Table 3.2 are based on a hardness value of 50 mg/L. The person has several options for using an alternate hardness value to calculate the benchmark value. The person may use the segment-specific 15th percentile hardness value (for the nearest downstream segment) or property-specific hardness data using site-sample results in accordance with the *Implementation Procedures* (TCEQ, 2011). Specific numerical aquatic-life criteria for metals and metalloids apply to dissolved concentrations where noted. Dissolved concentrations can be estimated by filtration of samples prior to analysis, or by converting from total recoverable measurements in accordance with the latest revision of the *Implementation Procedures*. The TCEQ usually prefers dissolved-metals data for surface waters rather than the mathematical conversion. If the conversion method is used, the person must use either the concentration of total suspended solids (TSS) for the nearest classified downstream or downgradient segment (as listed in the *Implementation Procedures*), or property-specific TSS data from site sample results (in accordance with the *Implementation Procedures*). An example calculation appears in Box A.1.

Box A.1 Example conversion of dissolved copper to total conversion for copper.

Segment 0806: West Fork of the Trinity River
Hardness = 140 mg/L (from *Implementation Procedures*)
TSS = 10 mg/L (from *Implementation Procedures*)
Freshwater chronic standard for dissolved copper:

$0.960e^{(0.8545(\ln(\text{hardness}))-1.386)} = 16.4\ \mu g/L$

Determination of total copper:
Partition Coefficient, $K_d = 10^b \times TSS^m$

$K_d = 10^{(6.02)} \times (10)^{-0.74}$
$K_d = 1.9055 \times 10^5$

Fraction of Metal Dissolved = C/C_T

$$\frac{C}{C_T} = \frac{1}{1+(K_d \times TSS \times 10^{-6})}$$

$$= \frac{1}{1+(1.9055\times10^5)(10)(10^{-6})}$$

= 0.344

Total Chronic Standard = 16.4 µg/L ÷ 0. 344 = 48 µg/L.
See *Implementation Procedures* for formulas and variables.

Since the Texas silver criterion is for the free-ion form, persons should convert dissolved to total silver, in addition to the chloride-dependent estimation of the percentage of silver in the free-ion form (see the footnotes to Table 3.2). An example calculation appears in Box A.2.

Box A.2 Example dissolved free ion to total conversion for silver.

Segment 0604: Neches Below Lake Palestine
Hardness = 36 mg/L
TSS = 10 mg/L
Chlorides = 24 mg/L
Freshwater Standard for the Free Ion form of Silver = 0.08 µg/L. (Acute standard ÷10)
Determination of percent dissolved silver (Y):

$$Y = e^{e^{\left(\frac{1}{0.6659 + 0.0044\ Cl}\right)}}$$

= 41.18 percent dissolved silver.

Determination of total silver:*
Partition Coefficient, $K_d = 10^b \times TSS^m$

$K_d = 10^{(6.38)} \times (10)^{-1.03}$
$K_d = 2.239 \times 10^5$

Fraction of Metal Dissolved = C/C_T

$$\frac{C}{C_T} = \frac{1}{1 + (K_d \times TSS \times 10^{-6})}$$

$$= \frac{1}{1 + (2.239 \times 10^5)(10)(10^{-6})}$$

= 0.31

Total Standard = 0.08 µg/L÷ ((0.31)(0.41)) = 0.63 µg/L
See *Implementaion Procedures* for formulas and variables.

Alternatively, the person may use the freshwater federal silver benchmark from Table 3.2.

Few COCs have defined state or federal water quality criteria. In developing the benchmarks specified in this guidance, the TCEQ and the ecological guidance work group sought out other benchmarks for COCs that lack specified criteria. Four primary sources were used.

A preferred hierarchy for surface water benchmarks is depicted in Figure 3.4. Table 3.2 reflects one number (if available) that should be used for comparison of site surface water data to the screening benchmarks, since a preferred order of use accompanied the benchmark derivation. As a first tactic, the work group derived aquatic-life numbers for a small number of COCs (marked with a superscript *q* in Table 3.2) using the LC_{50} approach

discussed in **3.5.1.1**. A full discussion of each value is available upon request from the TCEQ Technical Support Team. An example of the derivation process (for chloroform) appears below. The work group next selected the freshwater and marine water quality chronic values from TCEQ (2003a) for wastewater permits and requests from the Office of Waste, based primarily on the methodology defined in the TSWQS at 30 TAC 307.6(c)(7). Since the TCEQ water quality program routinely receives requests to develop "criteria" for COCs that do not have standards, the list of chronic values will be revised and expanded periodically.

The Tier II secondary chronic values (SCVs) developed by Suter and Tsao (1996) of the Oak Ridge National Laboratory (ORNL) were the third source for benchmark values selected by the work group. The SCVs rely on fewer data than the national criteria, and generally use the Great Lakes guidance method described in U.S. EPA (1993a; see 40 CFR 132, Appendix A). The Tier II values are concentrations expected to be higher than the national water quality criteria in no more than 20 percent of the cases if ample test data were available to calculate a chronic criterion. The Tier II values developed by ORNL differ from those in the Great Lakes guidance in that the Great Lakes secondary acute values (SAVs) require an LC_{50} for a daphnid. ORNL used adjustment factors (see Appendix B of Suter and Tsao, 1996 and references therein) for calculating SAVs when the daphnid LC_{50} data are not available. The SAV is used to calculate the SCV. The Oak Ridge Tier II SCVs were derived for freshwater biota only.

The last set of water quality benchmarks used is the U.S. EPA Region 4 (2001) chronic screening values for both marine and freshwater for compounds without federal criteria, the lowest chronic effect level reported in the applicable EPA ambient water quality criteria document was used with the application of a safety factor of 10. When chronic values were not reported, the lowest acute effect level was used, applying a safety factor of 100. Finally, the State of Colorado's hardness-based freshwater standard for uranium was used (CDPHE, 2005), rather than the ORNL Tier II approach. In derivation of the criterion, Colorado used data from seven toxicity tests involving three fish species over a range of hardness.

Example Aquatic Life Calculation Using the LC_{50} Approach for Chloroform (CAS: 67-66-3)

Fate and Transport in Water

When released into water, chloroform will be primarily lost by evaporation into the atmosphere due to its high Henry's Law constant of 3.68×10^{-3} atm-m^3/mole (TRRP rule). Modeling studies suggest that the volatilization half-life is 36 hours in a river, 40 hours in a pond, and 9–10 days in a lake (U.S. EPA, 1984: 21–22). Field monitoring data indicate the half-life of chloroform to be 1.2 days in the Rhine River and 31 days in a lake in the Rhine Basin (Zoeteman et al., 1980). Mackay et al. (1999) suggests a semi-quantitative classification of half-lives using nine different classes based on average environmental conditions. These half-life classes (referred to as

reactivity classes) integrate reaction rates of the chemical for all relevant processes, including biodegradation, volatility, and photolysis. Chloroform is classified as Class 6 with a mean half-life of 1700 hours and a range of 1,000–3,000 hours. Howard et al. (1991) indicates a half-life of 672–4,320 hours in surface water, based on aqueous aerobic biodegradation. The reported fate and transport data for chloroform suggest that it is likely to persist in water environments for more than 4 days. Based on these data, chloroform is considered persistent for deriving the water quality value.

Bioaccumulation Potential

Chloroform has little or low potential to bioconcentrate. The log bioconcentration factor is less than 1 for four species of fish (Anderson and Lusty, 1980). A BCF for chloroform was estimated using a log K_{ow} value of 1.52 (TRRP rule) and the BCF equation referenced in U.S. EPA (1999b) for water–to–aquatic invertebrates BCFs.

$$\log BCF = 0.819 \times \log K_{ow} - 1.146$$

$$BCF = 1.26$$

The estimated BCF value for chloroform suggests that it has a very low potential to accumulate in the food chain.

Toxicity to Aquatic Organisms

Freshwater: Chloroform is moderately toxic to aquatic organisms; several toxicity values are in the range of 13 mg/L to 353 mg/L (see Table A1). The 96-hr LC_{50} values for freshwater species are 70.7–171 mg/L for *Pimephales promelas* (fathead minnow), 13.3–115 mg/L for *Lepomis marochirus* (bluegill), 15.1–66.8 mg/L for *Oncorhynchus mykiss* (rainbow trout), and 45.4–55.8 mg/L for *Micropterus salmoides* (largemouth bass). For freshwater invertebrates, the 48-hr LC_{50} values for water fleas (*Daphnia magna* and *Ceriodaphnia dubia*) ranged from 29 to 353 mg/L.

Marine: Only one marine-species study was referenced in ECOTOX: the 48- and 72-hr LC_{50} value for *Penaecus duorarum* (northern pink shrimp) is 81.5 mg/L. See Table A.1 for a summary of the results of aquatic toxicity studies.

Water Quality Criteria

Freshwater: As indicated above, several acute toxicity studies have been conducted in a number of freshwater species for chloroform. Geiger et al. (1990; referenced in ECOTOX) evaluated the acute toxicity of chloroform in the fathead minnow and reported a 96 hr LC_{50} of 70.7 mg/L. Anderson and Lusty (1980) evaluated the acute toxicity of chloroform in four freshwater species: channel fish, largemouth bass, bluegill, and rainbow trout). Statistical analyses of the toxicity test results were analyzed using program that computes LC_{50}s by probit, moving average, and binomial computation. The 96 hr LC_{50} values reported ranged from 13.3 to 75 mg/L. The channel catfish and the largemouth bass were not very sensitive to the toxicity of chloroform. The reported 96-hr LC_{50} for the channel catfish is 75 mg/L.

Three 96 hr LC_{50} values were reported for largemouth bass, 45.4, 52.5, and 55.8 mg/L, with a geometric mean of 51 mg/L.

The LC_{50} results for the rainbow trout and bluegill indicated that these species were more sensitive to chloroform. Acute toxicity tests with juvenile rainbow trout gave 96 hr LC_{50} values between 15 and 22 mg/L, with a geometric mean of 18 mg/L. Acute toxicity tests in the bluegill gave 96 hr LC_{50} values ranging from 13.3 through 22.3 mg/L, with a geometric mean of 17.8 mg/L.

The LC_{50} results for the channel catfish, the largemouth bass, and the fathead minnow were eliminated from further consideration because chloroform was moderately toxic to those species in comparison to the bluegill and rainbow trout. The studies for the bluegill and rainbow trout indicated similar 96 hr LC_{50} values (a geometric mean of 17.8 and 18 mg/L, respectively). Since the rainbow trout is not a species native to Texas, the LC_{50} data for the trout were dropped from further consideration.

The geometric mean of the flow-through exposure 96 hr LC_{50} values (17.8 mg/L) for the bluegill was used in the derivation of freshwater acute and chronic criteria for chloroform. The following calculation assumes persistence and non-bioaccumulative potential for the chronic criterion:

freshwater acute criterion (mg/L) $= 0.3 \times 17.8$
$= 5.34$ mg/L

freshwater chronic criterion (mg/L) $= 0.05 \times 17.8$
$= 0.89$ mg/L

Marine: Only one marine species study was referenced in ECOTOX. Bentley et al. (1979) evaluated the toxicity of chloroform in *Penaecus duorarum* (pink shrimp) using static methods, reporting a 48 hr LC_{50} value (the same as the 72 hr value) of 81.5 mg/L, used in the derivation of marine acute and chronic criteria for chloroform. The following calculation assumes persistence and non-bioaccumulative potential for the chronic criterion:

marine acute criterion (mg/L) $= 0.3 \times 81.5$
$= 24.5$ mg/L

marine chronic criterion (mg/L) $= 0.05 \times 81.5$
$= 4.1$ mg/L

Table A.1. Summary of aquatic toxicity data for chloroform.

Common Name	Latin Name	Water Type	Duration (hr)	Exposure Type	Endpoint	Effect	Conc. (µg/L)	ECOTOX Reference
Channel catfish	*Ictalurus punctatus*	FW	96	F	LC_{50}	MOR	75,000	5267
Water flea	*Ceridophnia dubia*	FW	48	S	LC_{50}	MOR	290,000	212
Water flea	*Daphnia magna*	FW	48	S	LC_{50}	MOR	29,000	5184
Water flea	*D. magna*	FW	48	S	LC_{50}	MOR	353,000	212
Water flea	*D. magna*	FW	48	S	LC_{50}	MOR	66,500	12055
Water flea	*D. magna*	FW	48	S	LC_{50}	MOR	63,800	12055
Bluegill	*Lepomis macrochirus*	FW	96	F	LC_{50}	MOR	16,200	5267
Bluegill	*L. macrochirus*	FW	96	F	LC_{50}	MOR	22,300	5267
Bluegill	*L. macrochirus*	FW	96	F	LC_{50}	MOR	13,300	5267
Bluegill	*L. macrochirus*	FW	96	F	LC_{50}	MOR	18,300	5267
Bluegill	*L. macrochirus*	FW	96	F	LC_{50}	MOR	20,800	5267
Bluegill	*L. macrochirus*	FW	96	S	LC_{50}	MOR	100,000	2644
Bluegill	*L. macrochirus*	FW	96	S	LC_{50}	MOR	115,000	2644
Largemouth bass	*Micropterus salmoides*	FW	96	F	LC_{50}	MOR	55,800	5267

Common Name	Latin Name	Water Type	Duration (hr)	Exposure Type	Endpoint	Effect	Conc. (µg/L)	ECOTOX Reference
Largemouth bass	*M. salmoides*	FW	96	F	LC_{50}	MOR	52,500	5267
Largemouth bass	*M. salmoides*	FW	96	F	LC_{50}	MOR	45,400	5267
Rainbow trout	*Oncorhynchus mykiss*	FW	96	F	LC_{50}	MOR	18,200	5267
Rainbow trout	*O. mykiss*	FW	96	F	LC_{50}	MOR	18,400	5267
Rainbow trout	*O. mykiss*	FW	96	F	LC_{50}	MOR	22,100	5267
Rainbow trout	*O. mykiss*	FW	96	F	LC_{50}	MOR	15,100	5267
Rainbow trout	*O. mykiss*	FW	96	F	LC_{50}	MOR	17,100	5267
Rainbow trout	*O. mykiss*	FW	96	F	LC_{50}	MOR	43,800	2644
Rainbow trout	*O. mykiss*	FW	96	F	LC_{50}	MOR	66,800	2644
Fathead minnow	*Pimephales promelas*	FW	96	F	LC_{50}	MOR	70,700	3217
Fathead minnow	*P. promelas*	FW	96	F	LC_{50}	MOR	129,000	10432
Fathead minnow	*P. promelas*	FW	96	S	LC_{50}	MOR	171,000	10432
Fathead minnow	*P. promelas*	FW	96	S	LC_{50}	MOR	103,000	10432
Northern pink shrimp	*Penaecus duorarum*	SW	48	S	LC_{50}	MOR	81,500	2644

Common Name	Latin Name	Water Type	Duration (hr)	Exposure Type	Endpoint	Effect	Conc. (µg/L)	ECOTOX Reference
Northern pink shrimp	*P. duorarum*	SW	72	S	LC_{50}	MOR	81,500	2644

Sediment Benchmarks

The benchmarks for sediment are intended to protect of benthic biota, and are not necessarily mammals and birds that may be exposed to COCs through the food chain or via the incidental ingestion of sediment. Although there are a variety of existing sources for sediment benchmarks, most are derived using one of two basic approaches. First is the correlative or integrative approach, which relies largely on paired field and laboratory data to relate the incidence of adverse biological effects to the dry-weight sediment concentration of a COC. The toxicity values are derived through a number of approaches including toxicity tests of spiked sediment and field sediment, equilibrium partitioning (EqP), apparent effects threshold, and benthic community surveys. Since these benchmarks rely in part on the total concentration of COCs in sediment, the resultant screening values may have no relationship with the actual toxicity of individual COCs in a mixture, or their ability to bioaccumulate in aquatic or terrestrial organisms. The second, EqP, is theoretically based, and relies on the physical and chemical properties of sediment and COCs to predict the level of contamination that would not cause an adverse effect on aquatic life. In selection of the preferred sediment benchmarks specified in this guidance, the TCEQ and the ecological work group relied primarily on references that used the correlative approach, though EqP-based benchmarks were developed for VOCs.

Preferred Sediment Screening Benchmarks

The preferred benchmarks appear in Table 3.3. The following sections discuss the sediment benchmark approaches that were evaluated, including those not chosen as preferred. This discussion does not attempt to cover every possible sediment screening benchmark or method. These and other methodologies are discussed elsewhere (e.g., MacDonald, 1994; Neff, 1986; U.S. EPA, 1992; Ingersoll et al., 1997). Alternate sediment benchmarks, including those discussed herein may be used with appropriate justification (**3.5.1**.).

Freshwater Sediment Benchmarks

The primary freshwater benchmarks selected were the threshold-effect concentrations from MacDonald et al. (2000). Other sources include the effects range—low values in Long and Morgan (1990) and the lowest-effects levels from Persaud et al. (1993). The values in Jones et al. (1997) derived using the EqP approach were evaluated, but were not included in the listing of preferred benchmarks. However, EqP benchmarks were developed for VOCs by the ecological work group using the TCEQ's LC_{50} database and the modified approach suggested by Fuchsman (2003). All of these approaches are discussed below.

Threshold Effect Concentrations

MacDonald et al. (2000) developed sediment guidelines for 29 COCs in freshwater sediments. Two values were developed—a threshold effect

concentration (TEC), and a probable effect concentration (PEC). The TEC represents a sediment concentration below which adverse effects are not expected to occur; the PEC, a concentration above which adverse effects are expected to occur more often than not. Published sediment-quality guidelines were compiled and divided into two categories depending on their original intent. Where three or more acceptable sediment guidelines were available for a COC, TECs and PECs were calculated by determining the geometric mean of the previously published sediment guidelines. The predictive ability of the TECs was evaluated using matching sediment toxicity and chemistry data from field studies. Concentrations in sediment were compared to the corresponding TEC for that COC; samples were predicted to be not toxic if the measured concentrations were lower than the corresponding TEC. The TEC was considered reliable if more than 75 percent of the sediment samples were correctly predicted to be not toxic. The authors concluded that most of the TECs (21 of 28) form an accurate basis for predicting the absence of sediment toxicity for four trace metals, eight individual PAHs, total PAHs, total PCBs, and seven organochlorine pesticides.

ER-Ls

Long and Morgan (1990) first proposed the use of ER-Ls as part of the NOAA informal guidelines to help evaluate sediment chemistry data collected in the National Status and Trends Program (NSTP). The authors assembled a large sediment database that included effects and no effects field and lab data, for freshwater, estuarine, and marine organisms. COC concentrations (dry-weight normalized) observed or predicted by these methods to be associated with biological effects were ranked using percentiles. The lower 10th percentile concentration for those sediment COC concentrations associated with biological effects was chosen as the ER-L value. Values below the ER-L were considered to represent the “no effects” range. The 50th percentile concentration for the ranked sediment COC concentrations associated with biological effects was defined as the effects-range median (ER-M). COC concentrations between the ER-L and the ER-M values were considered to represent the possible effects range, and those above the ER-M were considered to represent the probable effects range. The ER-L values for antimony and silver were the only benchmarks used from this reference, since other similar approaches (such as Smith et al. 1996a) have incorporated more recent data sets.

Lowest-Effects Levels

The Ontario Ministry of the Environment derived sediment guidelines for evaluation of sediments throughout Ontario (Persaud, et al., 1993). The ministry defined a lowest-effects level (LEL) as a level of sediment contamination tolerated by the majority of benthic organisms, and a severe-effects level (SEL) as the level at which pronounced disturbance of the sediment-swelling community can be expected. These benchmarks were derived from matching sediment-chemistry and benthic-community data from various geographic areas. The OME used the screening-level concentration (SLC) approach as developed by Neff et al. (1986). This is a two-step process where an individual species SLC is first calculated for each COC by plotting the frequency distribution of the COC concentrations over all sites (at least 10) where that particular species is present. The 90th percentile is then selected as the SLC for that species. Then 90th percentiles for all the species present are plotted and the 5th (the LEL) and 95th (SEL) percentiles are calculated. Hence the 5th-percentile SLC (LEL) is the COC concentration above which 95 percent of the species SLCs are found (the highest level of a COC that can be tolerated by 95 percent of the benthic species). The adequacy of the SLC relates directly to the size of the database and its variability. An advantage of this approach is that it is based on chronic population-level effects on indigenous biota and can be used for polar and ionic organics and metals as well as nonpolar organics. However, it does not establish a direct cause-and-effect relationship between a single COC and benthic survival, and it requires a large amount of data, including sediment analyses and benthic assessments. LELs for iron, manganese, several pesticides and individual polychlorinated biphenyl aroclors were used as freshwater benchmarks in Table 3.3.

Interim Sediment-Quality Guidelines (ISQGs)

CCME (1999) describes the derivation of the Canadian freshwater and marine sediment quality guidelines for dichlorodiphenyltrichloroethane (DDT) and its degradation products, dichlorodiphenyldichloroethylene (DDE), and dichlorodiphenyldichloroethane (DDD). This reference was used in the guidance as the source for marine sediment benchmarks for Sum DDT, Sum DDE, or Sum DDE (the sums of the concentrations of the *p*, *p*', and *o,p*' isomers). The reference includes interim sediment quality guidelines based on the three sums for the protection of marine and estuarine life. Interim sediment-quality guidelines for sum DDD and sum DDE for the protection of freshwater aquatic life and a provisional value for sum DDT for freshwater aquatic life are also given but were not used. These values were developed in accordance with the procedures established by the CCME (1995), which relies on the NSTP approach (Long and Morgan, 1990) (with modifications) and the approach using the spiked-sediment-toxicity test in combination. If insufficient information is available to derive interim guidelines, Canada uses other approaches or guidelines that may be adapted as a provisional ISQG. Modifications of the NSTP approach include the separate evaluation of information for freshwater and marine systems, an expanded data set, and use of derivation procedures that consider all

compiled information (effect and no-effect data). All values are TELs with the exception of sum DDE for freshwater sediments, since the minimum data requirements (at least 20 entries in both the "no effects" and "effects" data sets) were not met. Based on available data, Environment Canada determined that freshwater and marine crustaceans are affected at similar concentrations of sum DDT, and elected to use the marine ISQG as a provisional freshwater sediment value.

Equilibrium Partitioning

The TCEQ evaluated the EqP approach to expand the list of freshwater sediment benchmarks for organic COCs. The Oak Ridge National Laboratory (Jones et al., 1997) has used this approach to calculate benchmarks for 75 nonionic organic COCs. Although it did not use that database, the ecological work group did use a modified EqP approach (Fuchsman, 2003) to develop freshwater and marine sediment benchmarks for 57 volatile COCs. EqP remains an alternative method for developing sediment benchmarks (and PCLs) provided there is adequate justification (**3.5.1.2**, **3.13.2**, **6.3**).

Marine Sediment Benchmarks

For marine benchmarks, the primary benchmarks selected were the ER-L values in Long et al. (1995). Other sources include the TELs from Smith et al. (1996b), the ISQG from Environment Canada (1997), and the EqP-derived benchmarks for volatiles developed by the ecological work group. The Apparent Effects Threshold approach and the Florida TEL approach were evaluated, but were not used as preferred benchmarks.

ER-Ls and ER-Ms

Long et al. (1995) established effects range—low (ER-L) and effects-range median (ER-M) values from an updated version of the BEDS database developed by Long and Morgan (1990). Here, freshwater data were omitted, and new data added. As in Long and Morgan (1990), studies included chemistry data and bioassays of field samples, toxicity tests using spikes of clean sediments, benthic community analyses, and equilibrium-partitioning modeling. No-effects data were separated from data where an effect was observed; then the effects data were sorted by increasing concentrations of each COC. For each COC, the ER-Ls and ER-Ms were defined as in Long and Morgan (1990).

Using amphipod-survival bioassasys, O'Connor, et al. (1998) used the EPA Environmental Monitoring and Assessment Program—Estuaries and the NOAA Status and Trends Bioeffects Surveys to test the applicability of various sediment toxicity guidelines. Of the 481 samples without an ER-L exceedance, only 5 percent were determined to be toxic—considered a good indication that toxic effects are unlikely at concentrations below an ER-L. Similarly, Long et al. (1998) found false negatives (toxic responses) for ER-Ls and TELs of 11 percent and 9 percent respectively when synoptically collected chemistry and amphipod toxicity test data for 1,068 samples from

studies compiled by the EPA and NOAA during 1990–93 were evaluated. The ER-L values for metals, PAHs, total DDTs, and total PCBs were the marine benchmarks used from this reference. The ER-L values have been widely used as screening tools in ecological risk assessments.

TELs for Marine Waters

Smith et al. (1996b) used CCME (1995) protocols to derive environmental-quality guidelines. They used a modified version of the NSTP approach (Long and Morgan, 1990) and the NOAA BEDS database. The data for the "effects" the "no effects" data sets were sorted for each COC and arranged in ascending order of COC concentrations. A TEL was calculated to be the geometric mean of the lower 15th-percentile concentration of the "effects" data set, and the 50th-percentile concentration of the "no effects" set. The TEL was intended to estimate the concentration for a given COC below which adverse biological affects only rarely occurred. The PEL was calculated to be the geometric mean of the 50th percentile concentration of the "effects" data set, and the 85th percentile of the "no effects" set. The PEL was intended to represent the concentration for a given COC above which adverse biological affects frequently occurred. TELs are draft interim sediment guidelines in the absence of adequate spiked-sediment toxicity test data. The marine TELs for chlordane, lindane, dieldrin and phthalates were used herein.

Apparent-Effects Thresholds

The AET methodology was first developed by Barrick et al. (1988) using empirical data from Puget Sound, Washington. The AET is the sediment concentration of a COC above which statistically significant (p 0.05) biological effects are always expected. The AET values are empirically derived from paired (field and laboratory) sediment chemistry and biological effects measures such as sediment toxicity tests or benthic-community surveys. The significance of adverse effects is assessed by statistical comparisons with suitable reference or control sediments. For a given data set, the AET value for a particular COC is the sediment concentration above which a particular adverse biological effect has always been found to be statistically significant, relative to a reference condition. Use of these values for a specific location or region elsewhere may be overprotective or underprotective. The AET approach requires a large database of COC data with at least one biological indicator. Unless site-specific data is used to derive an AET, the other benchmarks discussed in this guidance are presumed to be more appropriate in a Tier 2 ERA, since the AET represents the sediment concentration above which statistically significant biological effects are always expected and therefore presents a substantial risk of underprotection.

Other Marine-Sediment Quality Guidelines for PAHs

In addition to PAH benchmarks proposed for individual and combined PAHs in this guidance, other methods have been proposed. Swartz et al. (1995) proposed the ∑PAH model, which predicts the probability of acute toxicity of PAH contaminated marine sediments using a combination of EqP, Quantitative Structure Activity Relationships (QSARs), and toxic-unit, additivity, and concentration-response models. Assuming that the toxic effects of PAHs are additive, the total number of toxic units of 13 PAH compounds were used to predict the probability of toxicity to amphipods using a concentration-response model derived from spiked sediment toxicity tests. With inputs of f_{oc} and bulk concentrations of PAHs in sediment, the model predicts the probability that a sediment sample will be acutely toxic to amphipods (mortality > 24 percent), not toxic (mortality < 13 percent), or cause uncertain toxicity (mortality 13–24 percent).

The authors concluded that the model accurately predicts toxicity of PAH-contaminated sediments when PAHs are the principal COCs and the 13 PAHs used in the model development are the dominant PAHs. In another paper (Swartz, 1999) discusses the "mixture paradox" for PAHs in that sediment quality guidelines derived from experimental determination of toxicological effects of individual PAHs (spiked-sediment tests) will greatly underestimate ecological effects in the field that are associated with the guideline but actually caused by the PAH mixture, whereas guidelines derived from the correlation of ecological effects with the concentration of an individual PAH in field-collected sediment will greatly overestimate the effects actually caused by the single compound. For this reason, the author believes that guidelines for individual PAHs are inappropriate. With PAH concentrations normalized for organic carbon, the author proposes guidelines for threshold (TEC) of 290 µg/g organic carbon (OC), median (MEC) of 1,800 µg/g OC, and extreme (EEC) of 10,000 µg/g OC effects concentrations as a mixture of total PAHs (TPAHs) in marine or estuarine sediments. The author concludes that the TEC is the most useful guideline because mixtures of PAHs are unlikely to cause adverse effects on benthic ecosystems below the TEC, and that the TPAH guidelines agree with others (EqP, ∑PAH toxicity threshold, ER-L and SLC) within a factor of 2.

Using Sediment Benchmarks to Derive PCLs for Benthic Communities

As detailed in **3.1.3.2**, one approach to developing a PCL protective of the benthic invertebrate community employs the same databases used to derive the sediment screening benchmarks. The benchmarks listed in Table 3.3 were based on conservative primary effect levels. For each of these values, there is a corresponding higher value that represents a second level of effects. These second effect levels appear in Table A-2. As indicated in **3.13.2**, the person may view the primary effect levels as NOAELs and the second effect levels as LOAELs. In developing the benthic PCL, the same logic presented in **3.13.2** pertaining to NOAELs and LOAELs should be applied, with the understanding that the midpoint value may be proposed as the PCL but it is not a default (i.e., if sufficient evidence suggests that the

midpoint is not protective, then this value will be questioned). Also, when using Tables 3.3 and A.2 to develop a sediment PCL for a volatile COC that was derived from the EqP approach, the person should remember that even though the surface water benchmarks and TCEQ's LC_{50} database were used as input, the resulting values are not effects-based for benthics, and therefore it may be prudent to be conservative (i.e., proposing a PCL that is less than the midpoint). This is particularly true when the acute surface water input number used to derive the second effects level was developed from equations in Di Toro et al. (2000) as shown in Box A.3. These values appear in Table A.2 with an *o* footnote. Alternatively, the person could use the acute-to-chronic ratio of 5.09 suggested in Di Toro et al. (2000) and input a chronic surface water number into the sediment benchmark equation. In this case, the midpoint between the resulting values and the corresponding initial effect levels in Table 3.3 would be an acceptable PCL.

Box A.3. Example calculation of acute surface water benchmarks using the Di Toro method.

Step 1: Di Toro et al (2000), Equation 37 without the acute-to-chronic ratio (ACR):

log (WQB mmol/L) = log (35.3) + C_l –0.945 × log(K_{ow})

Step 2: Units Conversion mmol/L to mg/L

WQB (mg/L) = WQB (mmol/L) × MW

Where:
WQB = acute water quality benchmark in surface water
C_l = chemical class correction (0 for aliphatic COCs, –0.244 for halogenated chemicals)
K_{ow} = octonol-water partition coefficient (unitless)
MW = molecular weight (g/mol)

Example Calculation: Benzene (log K_{ow} = 1.99, MW = 78.1 g/mol)

log (WQB) = log (35.3) + 0 – 0.945 × 1.99
log (WQB) = 1.55 – 1.88
log (WQB) = -0.33
WQB = 0.47 mmol/L × 78.1 g/mol
WQB = 36.1 mg/L for benzene

A chronic WQB can be obtained by dividing by the ACR of 5.09:
36.1 mg/L ÷ 5.09 = 7.09 mg/L

*Example calculatio*n: carbon tetrachloride (log K_{ow} = 2.44 MW = 154 g/mol)

log (WQB) = log (35.3) – 0.244 – 0.945 × 2.44
log (WQB) = 1.55 – 0.244 – 2.31
log (WQB) = – 1.00
WQB = 0.100 mmol/L × 154 g/mol
WQB = 15.2 mg/L for carbon tetrachloride

A chronic WQB can be obtained by dividing by the ACR of 5.09:
15.2 mg/L ÷ 5.09 = 2.99 mg/L.

Table A.2. Second effects levels for sediment.

Inorganics

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
7440-36-0	antimony	25[a]	
7440-38-2	arsenic	33	70
7440-43-9	cadmium	4.98	9.6
7440-47-3	chromium	111	370
7440-50-8	copper	149	270
7439-89-6	iron	40,000[b]	
7439-92-1	lead	128	218
7439-96-5	manganese	1,100[b]	
7439-97-6	mercury	1.06	0.71
7440-02-0	nickel	48.6	51.6
7440-22-4	silver	2.2[a]	3.7
7440-66-6	zinc	459	410

Polycyclic Aromatic Hydrocarbons[j]

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
83-32-9	acenaphthene	0.089[j]	0.500
208-96-8	acenaphthylene	0.128[j]	0.640
120-12-7	anthracene	0.845	1.1
56-55-3	benz[*a*]anthracene	1.05	1.6
50-32-8	benzo[*a*]pyrene	1.45	1.6
218-01-9	chrysene	1.29	2.8
53-70-3	dibenz[*a,h*]anthracene	0.140[j]	0.260
206-44-0	fluoranthene	2.23	5.1
86-73-7	fluorene	0.536	0.540
91-57-6	2-methyl naphthalene		0.670
91-20-3	naphthalene	0.561	2.1
85-01-8	phenanthrene	1.17	1.5
129-00-0	pyrene	1.52	2.6

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
	low-molecular-weight PAHs		3.16[d,k]
	high-molecular-weight PAHs		9.6 [f,i]
	total PAHs	22.8[f,h,i]	44.79[f,h,i]

Chlorinated Pesticides, PCBs, Benzenes

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
309-00-2	Aldrin	0.08[b,g]	
27323-18-8	Aroclor 1254	0.34[b,g]	
12674-11-2	Aroclor 1016	0.53[b,g]	
11096-82-5	Aroclor 1260	0.24[b,g]	
12672-29-6	Aroclor 1248	1.5[b,g]	
319-84-6	alpha-BHC	0.1[b,g]	
319-85-7	beta-BHC	0.21[b,g]	
58-89-9	gamma-BHC (Lindane)	0.00499	0.00099[c]
608-73-1	BHC	0.12[b,g,h]	
57-74-9	Chlordane (total)	0.0176	0.00479[c]
60-57-1	Dieldrin	0.0618	0.00430[c]
72-20-8	Endrin	0.207	
118-74-1	HCB (Hexachlorobenzene)	0.24[b,g]	
1024-57-3	heptachlor epoxide	0.016	
2385-85-5	Mirex	1.3[b,g]	
72-55-9	sum DDE	0.0313[h]	0.374[c,h]
72-54-8	sum DDD	0.028[h]	0.00781[c,h]
50-29-3	sum DDT	0.0629[h]	0.00477[c,h]
NA	total DDT	0.572[h]	0.0461[h]
1336-36-3	total PCBs	0.676[h]	0.180[h]

Other Pesticides

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
8001-35-2	toxaphene	0.032[l]	

Phthalates

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
117-81-7	bis(2-ethyl-hexyl)phthalate		2.647[c]
NA	di-*n*-butyl phthalate	0.043[k]	

Volatiles [Footnote (m) applies to all listed volatiles unless otherwise noted]

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
67-64-1	acetone	360.18	1003.36
107-13-1	acrylonitrile	1.36	1.04
71-43-2	benzene[n]	45.01	45.01
104-51-8	*n*-butylbenzene	6.57	
103-65-1	propyl benzene	4.35	
135-98-8	sec-butylbenzene	5.28	
98-06-6	tert-butylbenzene	7.26	
75-27-4	bromodichloromethane	14.74	
78-93-3	2-butanone	154.26	
75-15-0	carbon disulfide	0.78	
56-23-5	carbon tetrachloride[n]	37.33	37.33
108-90-7	chlorobenzene[n]	19.87	19.87
124-48-1	chlorodibromomethane	0.94	
67-66-3	chloroform (trichloromethane)	5.63	25.8
74-87-3	chloromethane	106.8	52.43
98-82-8	cumen	53.95	
99-87-6	*p*-cymene	5.98	
95-50-1	1,2-dichlorobenzene	4.95	4.44
541-73-1	1,3-dichlorobenzene	0.35	1.95
106-46-7	1,4-dichlorobenzene	4.65	4.21
75-71-8	dichlorodifluoromethane	22.09	
75-34-3	1,1-dichloroethane	13.89	
107-06-2	1,2-dichloroethane	28.69	25.80

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
75-35-4	1,1-dichloroethene	11.22	92.47
156-60-5	1,2-dichloroethene (trans)	71.84	
78-87-5	1,2-dichloropropane	13.17	
542-75-6	1,3-dichloropropene	1.37	0.26
100-41-4	ethylbenzene	17.18	3.93
87-68-3	HCBD (hexachlorobutadiene)	0.55[l]	12.76[m,n]
67-72-1	hexachloroethane[n]	13.77	13.77
110-54-3	*n*-Hexane[n]	12.77	
591-78-6	2-hexanone	28.20	
108-10-1	4-methyl-2-pentanone (MIBK)	116.59	272.06
74-83-9	methyl bromide	0.46	2.49
22967-92-6	methylmercury	N/A	
80-62-6	methyl methacrylate	56.98	
75-09-2	methylene chloride	46.52	22.91
98-95-3	nitrobenzene[n]	161.06	161.06
71-41-0	1-pentanol[n]	N/A	
67-63-0	2-propanol[n]	443.99	
100-42-5	styrene	61.42	22.31
79-34-5	1,1,2,2-tetrachloroethane	3.80	3.69
127-18-4	tetrachloroethene	10.05	18.59
108-88-3	toluene	17.29	5.66
75-25-2	bromoform	1.31	10.67
120-82-1	1,2,4-trichlorobenzene	5.31	2.32
71-55-6	1,1,1-trichloroethane	24.80	15.83
79-00-5	1,1,2-trichloroethane	5.88	1.80
79-01-6	trichloroethene	5.07	8.82
75-69-4	trichlorofloromethane	10.12	
76-13-1	1,1,2-trichlorotrifluoroethane	16.70	
95-63-6	1,2,4-trimethylbenzene	4.58	12.95

CAS No.	COC (mg/kg dry wt.)	Freshwater	Marine
108-67-8	1,3,5-trimethylbenzene	4.59	
108-05-4	vinyl acetate[n]	366.29	
75-01-4	vinyl chloride	11.78	
108-38-3	m-xylene[n]	2.08	
1330-20-7	xylenes	12.01	7.47

Freshwater: Unless otherwise noted, values are PECs from MacDonald, Ingersoll, and Berger (2000).

Marine: Unless otherwise noted, values are ERMs from Long, MacDonald, Smith, and Calder (1995).

[a] ERM from Long and (1990).

[b] SEL from Persaud, Jaagumagi, and Hayton (1993).

[c] PEL from Smith, MacDonald, Keenleyside, and Gaudet (1996).

[d] The sum of the concentrations of the following compounds: naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, and 2-methyl napthalene.

[e] The sum of the concentrations of the following compounds: fluoranthene, pyrene, benz[**a**]anthracene, chrysene, benzo[**a**]pyrene, and dibenzo[**a,h**]anthracene.

[f] The sum of the concentrations of each of low- and high-molecular-weight PAHs listed above and any other PAH compounds that are COCs.

[g] Values in the original reference were based on percent total organic carbon. These values were converted to bulk sediment values by assuming 1 percent TOC (SEL × 0.01).

[h] When benchmarks represent the sum of individual compounds, isomers, or groups of congeners, and the chemical analysis indicates an undetected value, the proxy value specified at 350.51(n) shall be used for calculating the sum of the respective compounds, isomers, or congeners. This assumes that the particular COC has not been eliminated in accordance with the criteria at 350.71(k).

[i] Applies to all listed PAHs. The benchmarks for total PAHs are the most relevant in evaluating risk in an ERA as PAHs almost always occur as mixtures. Values for individual, low-molecular-weight, and high-molecular-weight PAHs are provided as guidelines to aid in the determination of disproportionate concentrations within the mixture that may be masked by the total. See discussion in **3.5.4**.

[j] Canadian Council of Ministers of the Environment. 2002. **Canadian environmental quality guidelines.** Winnipeg, Manitoba.

[k] Cubbage, Batts, and Briedenbach (1997).

[l] NYSDEC (1999).

[m] Benchmarks derived using formula in P.C. Fuchsman (2003). The TCEQ's LC_{50} database was used for water quality values, except where noted. TRRP-24 default values of 1 percent fraction organic carbon (f_{oc}) and 0.37 porosity were used. The person should adjust these values if sufficient site-specific data indicate they are not representative.

[n] Acute water quality values were used as input for these COCs and were derived from Di Toro, McGrath, and Hansen (2000).

Soil Benchmarks

The soil benchmarks herein were chosen after a careful review of those available in the current scientific literature. The soil benchmarks provided here are considered to be the most relevant and useful of those currently available. The source documents for soil benchmarks reviewed include Efroymson (1997); Efroymson, Will and Suter (1997); and U.S. EPA Region 5 (1998). The U.S. EPA Region 5 Ecological Data Quality Limits (ibid.) were considered too conservative for ecological screening benchmarks. Their intended purpose is different from that of a screening criterion for a Tier 2 SLERA. The ORNL Preliminary Remediation Goal (PRG) values (Efroymson, Suter, Sample, and Jones, 1997) were considered inappropriate because bioaccumulation is accounted for in their derivation and is considered as a separate exercise in the screening process in Tier 2. Note that most of the PRG values are equivalent to the benchmarks cited for either terrestrial plants or earthworms in the benchmark documents produced by ORNL for terrestrial plants and soil invertebrates, respectively.

The recommended soil benchmarks are the lower of the terrestrial plant and earthworm (or soil invertebrate) benchmark value from Efroymson (1997) and Efroymson et al., (1997) and the Eco-SSLs developed by the EPA (U.S. EPA, 2003a, b; 2005a–g; 2006; and 2007a–g). The soil benchmark values and their respective test-organism source (i.e., plant, earthworm, or soil invertebrate) appear in Table 3.4. If the soil concentration is less than the associated benchmark, then that COC does not need further consideration in the ERA; otherwise, it requires further evaluation (see **3.5**). If the soil benchmark value in Table 3.4 for a particular COC is greater than the median Texas background concentration (or site-specific background concentration) cited for the same COC in this publication, then the soil benchmark value may be ignored [see 30 TAC350.51(l) for a discussion of background concentrations].

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APPENDIX B: Measurement Receptors for Example Texas Food Webs

(modified from U.S. EPA, 1999)

As stated in **3.6.6**, measurement receptors should generally be selected with consideration to the definition of *ecological PCL* (i.e., limited to more mobile or wide-ranging species and benthic invertebrates, where appropriate). However, the person may be required to demonstrate that impacts to receptors with limited mobility or range (e.g., plants, soil invertebrates, and small rodents) will not result in unacceptable consequences for the more mobile or wide-ranging receptors. With that in mind, the following list and discussion of measurement receptors includes limited-range species (mice, rats, shrews) that should be evaluated when impacts to these receptors, particularly in undeveloped or off-site areas, are likely to significantly disrupt the ecosystem (see the definition of *ecological PCL* in the glossary).

Consistent with **3.6.7**, measurement receptors were selected for the example food webs presented in **3.6.5**. Information from the *Wildlife Exposure Factors Handbook* (U.S. EPA, 1993) and other available literature was evaluated to determine suitable measurement receptors for each class-specific guild represented in the example food webs.

Ecological relevance; exposure potential; sensitivity, social or economic importance; the presence of known, expected, and protected species; and the availability of information on natural history and toxicology (see **3.6.7.2**) were evaluated to identify measurement receptors for the example food webs. Since these measurement receptors are included as examples to facilitate understanding of the selection, not every feeding guild expected in a screening-level ERA at an affected property may have been included. A discussion of each of the example measurement receptors follows.

American Kestrel

The American kestrel (*Falco sparverius*), or sparrow hawk, could be selected as the measurement receptor for the carnivorous-bird feeding guild in the example Texas tallgrass prairie, shortgrass prairie, shrub-scrub, desert-arid, and freshwater-wetland food webs based on the following:

- The kestrel is important in regulating small mammal populations through predation. Predators of the kestrel include larger raptors such as red-tailed hawks, golden eagles, and great horned owls.
- The kestrel's prey includes a variety of invertebrates such as worms, spiders, scorpions, beetles, and other large insects, as well as an assortment of small to medium-sized birds and mammals. Winter home ranges vary from a few hectares to hundreds of hectares, depending on the amount of available prey in the area.
- The availability of natural-history information (e.g., home range, ingestion rates, body weights) also supports its selection as a measurement receptor.

American Robin

The American robin (*Turdus migratorius*) could be selected as the measurement receptor for the omnivorous bird feeding guild in the example Texas upland forest and freshwater-wetland food webs based on the following:

- The robin serves an important function in seed dispersion for many fruit species, making it a valuable component of the ecosystem.
- Habitats include forests, wetlands, swamps, and habitat edge where forested areas are broken with agricultural and range land. The robin forages on earthworms, snails, and other soil invertebrates, seeds, and fruit.
- The availability of natural-history information (e.g., home range, ingestion rates, body weights) also supports its selection as a measurement receptor.

Belted Kingfisher

The belted kingfisher (*Ceryle alcyon*) could be selected as the measurement receptor for the carnivorous shore bird feeding guild in the example Texas freshwater-wetland and estuarine-wetland food webs based on the following:

- As a higher-trophic-level predator, the belted kingfisher is an important component to the ecosystem in influencing the population dynamics of its prey.
- The belted kingfisher is found statewide along rivers, streams, lakes, ponds, coasts, and estuaries. Its diet consists mostly of fish that swim

near the surface or in shallow water. It nests in burrows in earthen banks, generally near suitable fishing areas.

- Measured in shoreline length, its home range averages approximately 1.5 kilometers. Due to its dietary habits and relatively small body weight, there is a high potential for exposure.
- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weights, TRVs) also supports its selection as a measurement receptor.

Canada Goose

The Canada goose (*Branta canadensis*) could be selected as the measurement receptor for the herbivorous bird guild in both example aquatic food webs based on the following:

- The Canada goose plays a valuable functional role in aquatic habitats by dispersing seeds for aquatic vegetation.
- It varies greatly in size, from the largest goose in Texas to the smallest, depending on the subspecies. Its diet consists of aquatic vegetation, grains, and grasses.
- The availability of natural-history information (e.g., home range, ingestion rates, body weights) also supports its selection as a measurement receptor.

Deer Mouse

The deer mouse (*Peromyscus maniculatus*) could be selected as the measurement receptor for the herbivorous mammal feeding guild in the example Texas upland forest, tallgrass and shortgrass prairie, shrub-scrub, and arid-desert food webs based on the following:

- The deer mouse is preyed upon by owls, snakes, and small carnivorous mammals, making it a very important prey item. This animal also plays an important ecological role in seed and fruit dispersion for many types of vegetation. In addition, its burrowing activities influence soil composition and aeration.
- It is distributed statewide but is uncommon in the eastern, coastal, and southern parts of Texas. It lives in underground burrows, brush piles, or crevices among rocks. It is almost strictly nocturnal and feeds chiefly on seeds, fruits, bark, roots, and herbage. Due to its burrowing and dietary habits, there is a high potential for direct and indirect exposure. The home range area for a deer mouse is rarely over 0.8 ha (50 m radius), and it spends most of its day in an underground burrow.
- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weights, TRVs) also supports its selection as a measurement receptor.

Eastern Cottontail Rabbit

The eastern cottontail (*Sylvilagus floridanus*) could be selected as the measurement receptor for the herbivorous mammal feeding guild in the example Texas upland forest, tallgrass prairie, shrub-scrub, and both aquatic food webs based on the following:

- The eastern cottontail is preyed upon by hawks, barn owls, opossums, coyotes, foxes, and small weasels, making it a very important prey item. This animal also plays an important ecological role in seed dispersion for many types of vegetation.
- It is active mostly in the twilight hours and at night when it ventures to open pastures, meadows, or lawns to forage on grasses and forbs primarily, but also on twigs and bark. The home range for an eastern cottontail is approximately 1.8 ha, and it spends most of its day in beds in thickets and underground burrows. Due to its burrowing and dietary habits, there is a high potential for direct and indirect exposure.
- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weights, TRVs) also supports its selection as a measurement receptor.

Least Shrew

The least shrew (*Cryptotis parva*) could be selected as the measurement receptor for the omnivorous-mammal feeding guild in the example Texas tallgrass- and shortgrass-prairie food webs based on the following:

- Because of the shrew's abundance and high population density, it makes up a large portion of the diet of owls, hawks, and snakes.
- It is the smallest mammal in Texas, occurring in grasslands in eastern and central portions of the state, westward in the Panhandle to the New Mexico line, and to Val Verde County along the Rio Grande. It feeds on snails, insects, sow bugs, and other small invertebrates. The home range area is on average 0.39 ha (35 m in radius). Its diet of invertebrates and burrowing behavior result in a high potential of direct and indirect exposure to COCs.
- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weights, TRVs) also supports its selection as a measurement receptor.

Long-Tailed Weasel

The long-tailed weasel (*Mustela frenata*) could be selected as the measurement receptor for the carnivorous-mammal feeding guild in the example Texas tallgrass-prairie and shrub-scrub food webs based on the following:

- The long-tailed weasel is important in regulating small mammal populations through predation. Its predators include cats, foxes, snakes, and large raptors such as hawks and owls.
- It is distributed statewide, except for the extreme northern Panhandle, but scarce in western Texas. Habitats are varied and include forested, brushy, open areas including farm lands preferably near water, where it preys on ground squirrels, pocket gophers, rats, small rabbits, and insects.
- The availability of natural-history information (e.g., home range, ingestion rates, body weights) also supports its selection as a measurement receptor.

Mallard Duck

The mallard duck (*Anas platyrhynchos*) could be chosen as the measurement receptor for the omnivorous bird feeding guild for the example Texas freshwater-wetland and estuarine-wetland food webs based on the following:

- The mallard serves as a valuable component in aquatic food webs dispersing seeds for aquatic vegetation, and due to their role in the nutrient cycle of wetlands. In addition, the mallard is a major prey item for carnivorous mammals, birds, and snakes.
- It is present in a diversity of aquatic habitats in Texas. Although it is considered omnivorous, 90 percent of its diet may be plant material at some times of the year. Mallards are dabblers that will often filter through soft mud and sediment searching for food items.
- It is a very important game species, representing approximately one-third of all waterfowl harvested in the U.S.
- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weights, TRVs) also supports its selection as a measurement receptor.

Marsh Rice Rat

The marsh rice rat (*Oryzomys palustris*) could be selected as the measurement receptor for the omnivorous mammal feeding guild in the example Texas freshwater-wetland and estuarine-wetland food web based on the following:

- The marsh rice rat plays an important role in seed dispersal and is a major food item for many predators including raptors, cats, weasels and snakes. The marsh rice rat has a high potential for exposure due to its aquatic diet and direct contact with media.
- It is found in eastern Texas west to Brazos County and south to Cameron County. It inhabits marsh and wetland areas where it feeds on crabs, insects, fruits, snails, and aquatic plants.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weights, TRVs) also supports its selection as a measurement receptor.

Marsh Wren

The marsh wren (*Cistothorus palustris*) could be selected as the measurement receptor for the omnivorous bird feeding guild in the example Texas freshwater-wetland and estuarine-wetland food webs based on the following:

- The marsh wren consumes large numbers of aquatic insects thus regulating their populations, which make it a valuable component of the ecosystem. Its main predators are snakes and turtles which prey heavily upon the eggs.
- It is common throughout Texas, inhabiting freshwater, brackish, and saltwater marshes. Its diet consists mainly of aquatic invertebrates (though it may eat snails and spiders), rendering it susceptible to accumulation and toxicity of bioaccumulative COCs.
- The availability of natural-history information (e.g., home range, ingestion rates, body weights) also supports its selection as a measurement receptor.

Mink

The mink (*Mustela vison*) could be selected as the measurement receptor for the carnivorous mammal feeding guild in the example Texas freshwater-wetland and estuarine-wetland food webs based on the following:

- As a high-trophic level-predator, the mink is an important component to the ecosystem in influencing the population dynamics of its prey. Its main predators include foxes, bobcats, and great horned owls.
- It is known from the eastern half of the state westward to the northern Panhandle. It lives near rivers, creeks, lakes, and marshes. It is predominantly a nocturnal hunter, although it is sometimes active during the day. It is an opportunistic feeder and will consume whatever prey is most abundant, including small mammals, fish, birds, reptiles, amphibians, mollusks, crustaceans, insects, and carrion.
- It has been shown to be sensitive to PCBs and similar COCs, and has a high potential for exposure due to its aquatic diet and direct contact with the media.
- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weights, TRVs) also supports its selection as a measurement receptor.

Mourning Dove

The mourning dove (*Zenaida macroura*) could be selected as the measurement receptor for the herbivorous bird feeding guild in the example Texas upland forest, tallgrass prairie, shortgrass prairie, shrub-scrub, and desert-arid food webs based on the following:

- The dove plays an important functional role in seed dispersion for many grasses and forbs, and is an important prey item for many higher-trophic-level omnivores and carnivores. Predators of the mourning dove include falcons, hawks, fox, and snakes.
- It inhabits open woodlands, forests, prairies, and croplands. It feeds mostly on seeds, which comprise 99 percent of its diet. It may ingest insignificant amounts of animal matter and green forage incidently.
- It has a high potential for exposure through ingestion of inorganic COCs.
- It is an important game species, contributing significantly as a food and economic resource.
- The availability of natural-history information (e.g., home range, ingestion rates, body weights) also supports its selection as a measurement receptor.

Northern Bobwhite

The northern bobwhite (*Colinus virginianus*) could be selected as the measurement receptor for the omnivorous-bird feeding guild in the example Texas tallgrass-prairie, shortgrass-prairie, shrub-scrub, and arid-desert food webs based on the following:

- The bobwhite plays an important role in seed dispersion for many plant species, and is an important prey item for snakes and small mammals. If habitat conditions permit, its numbers will increase rapidly, providing an additional food source for many predators. It is also valuable in controlling insect populations during certain times of the year.
- Its diet is mainly seeds and invertebrates although, in the winter, green vegetation can dominate the diet. During breeding season, the bobwhite's home range may encompass several hectares, including areas for foraging, cover, and a nest site. In non-breeding season, the bobwhite's home range can be as large as 16 ha. It has a high potential for exposure through ingestion and dermal contact with soil during dust bathing.
- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weights, TRVs) also supports its selection as a measurement receptor.

Northern Harrier

The northern harrier (*Circus cyaneus*), also called the marsh hawk, could be selected as the measurement receptor for the carnivorous-bird feeding guild in the example Texas freshwater-wetland and estuarine-wetland food webs based on the following:

- The marsh hawk plays an important role in the ecosystem in regulating populations of small mammals through predation.
- Its diet consists of small mammals, birds, and occasionally snakes, frogs, and insects. Its preferred habitats include marshes and other wetlands.
- In addition, the marsh hawk has demonstrated sensitivity to pesticides, which bioaccumulate through food chains.
- The availability of natural-history information (e.g., home range, ingestion rates, body weights) also supports its selection as a measurement receptor.

Red Fox

The red fox (*Vulpes vulpes*) could be selected as the measurement receptor for the carnivorous mammal feeding guild in the example Texas upland-forest, shortgrass-prairie, shrub-scrub, freshwater-wetland, and estuarine-wetland food webs based on the following:

- The red fox has a high potential for exposure due to bioaccumulation though the food chain. It is a valuable component to ecosystem structure in regulating the abundance, reproduction, distribution, and recruitment of lower-trophic-level prey.
- Although it is omnivorous, the majority of its diet consists of small rodents, rabbits, birds, wild fruits and berries, and insects. This animal could be chosen because of its status as a top carnivore and its distribution across eastern and central Texas to the Trans-Pecos, inhabiting mixed woodlands interspersed with farms and pastures, coastal areas, and crevices in rocky outcrops.
- The availability of natural-history information (e.g., home range, ingestion rates, body weights) also supports its selection as a measurement receptor.

Red-Tailed Hawk

The red-tailed hawk (*Buteo jamaicensis*) could be selected as the measurement receptor in the carnivorous bird feeding guild in the example Texas upland forest, and desert-arid food webs based on the following:

- The red-tailed hawk's position as a high trophic level predator makes it a valuable component of terrestrial food webs through its regulation of populations of lower-trophic-level prey species.

- It is widely distributed across Texas among a diversity of habitat types ranging from woodlands to pastures. Its diet includes small mammals (such as mice, shrews, voles, rabbits, and squirrels), birds, lizards, snakes, and large insects. It is an opportunistic feeder, preying on whatever species is most abundant. Red-tailed hawks are territorial throughout the year, and have home ranges that can exceed 1,500 ha.
- Red-tailed hawks have shown sensitivity to many COCs that disrupt reproduction or egg development.
- The availability of natural-history information (e.g., home range, ingestion rates, body weights) also supports its selection as a measurement receptor.

Southern Short-Tailed Shrew

The southern short-tailed shrew (*Blarina carolenensis*) could be selected as the measurement receptor for the omnivorous mammal feeding guild in the example Texas upland forest, shrub-scrub, and freshwater-wetland food webs based on the following:

- The short-tailed shrew, valued as a prey species by many high level predators, is very important to the health of an ecosystem. It also plays an important role in soil recycling and aeration, through tunnel excavation.
- Found in the eastern one-fourth of the state, it is a small, mostly insectivorous mammal that represents secondary consumers (insectivores) present in terrestrial ecosystems. Its diet of invertebrates such as earthworms and its burrowing behavior result in a high potential of direct and indirect exposure to COCs. It has a very high metabolism rate, which requires almost constant feeding. Its most common habitats are wooded and wet areas in the drier parts of the range.
- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weights, TRVs) also supports its selection as a measurement receptor.

Spotted Sandpiper

The spotted sandpiper (*Actitis macularia*) could be selected as the measurement receptor for the invertivorous-carnivorous shore bird feeding guild in the example Texas freshwater-wetland and estuarine-wetland food webs based on the following:

- The spotted sandpiper inhabits a wide variety of habits usually associated with water or marsh.
- It has a high potential for exposure through ingestion of aquatic insects, worms, fish, crustaceans, mollusks, and carrion.
- The availability of natural-history information (e.g., home range, ingestion rates, body weights) also supports its selection as a measurement receptor.

Swift Fox

The smallish swift (or kit) fox (*Vulpes velox*) could be selected as the measurement receptor for the carnivorous mammal guild in the example Texas tallgrass- and shortgrass-prairie, and desert-arid food webs based on the following:

- The swift fox fills an important functional role by regulating the population dynamics of many prey species.
- It is known from the western third of the state east to Menard County. It is mainly nocturnal; its diet consists of small mammals, insects, birds, lizards, and amphibians. It spends most of its days in a den, emerging at night to hunt. Their home range extends up to a 3 km radius.
- The availability of natural-history information (e.g., home range, ingestion rates, body weights) also supports its selection as a measurement receptor.

Western Meadowlark

The western meadowlark (*Sturnella neglecta*) could be selected as the measurement receptor for the omnivorous-bird feeding guild in the example Texas tallgrass- and shortgrass-prairie food webs based on the following:

- The western meadowlark serves an important function in seed dispersion for many forb and grass species, making it a valuable component of the ecosystem.
- Itsabitats include grassland, savanna, pasture, and cultivated fields. It forages on spiders, sow bugs, snails, and grass and forb seeds.
- The availability of natural-history information (e.g., home range, ingestion rates, body weights) also supports its selection as a measurement receptor.

White-Footed Mouse

The white-footed mouse (*Peromyscus leucopus*) could be selected as the measurement receptor for the omnivorous mammal feeding guild in the example Texas shrub-scrub and desert-arid food webs based on the following:

- The white-footed mouse plays an important role in seed dispersal and provides an important food source for raptors, snakes and other mammals including cats, weasels and foxes.
- It is a medium-sized, short-tailed mouse distributed across the state. It feeds on nuts, seeds, fruits, beetles, caterpillars, and other insects and has a home range of 0.2 ha. Due to its burrowing and dietary habits, there is a high potential for direct and indirect exposure.

- The availability of information on natural history and toxicology (e.g., home range, ingestion rates, body weights, TRVs) also supports its selection as a measurement receptor.

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APPENDIX C: Derivation of Toxicity Reference Values

As stated in **3.9.5**, TRVs are not derived for every COC for every exposure medium for every ecological receptor. TRVs are developed if COCs are present in a particular medium above the ecological benchmark for that medium or are bioaccumulative. Appendix A describes derivation of ecological screening benchmarks for water, sediments, and soils.

TRVs are used in a hazard-quotient or hazard-index application, wherein a predicted exposure is compared with the concentration associated with no unacceptable risk, a TRV derived from NOAEL endpoints. A subsequent comparison may reflect less conservative assumptions for both the exposure and effects. For clarity, the TRV derived from NOAEL endpoints reflecting no unacceptable risk can be described as TRV_{NOAEL}, while a TRV derived from less conservative exposure assumptions (from LOAEL endpoints) can be described as TRV_{LOAEL}.

TRV derivation should reflect an increased focus on the site-specific conditions under evaluation and so may involve the review and refinement of data used initially to derive ecological benchmarks. In some cases, the TRVs may numerically equal the benchmark values shown in **3.5**. In other cases, the TRVs may be greater than benchmark values because they need not reflect the same generic conservative assumptions used in derivation of benchmarks. Where ecological benchmarks reflect Texas or federal criteria—such as ambient water quality criteria for the protection of aquatic life and its uses—toxicity reference values will generally be the same as these numerical criteria.

Estimating an NOAEL from Short-Term Test Endpoints

A typical limitation of studies is their duration. Acute studies may be available, but the ecological concern may be for long-term exposures (e.g., during the entire sensitive life stage or lifespan). In addition, use of NOAEL and LOAEL endpoints from toxicity tests has been widely criticized on technical statistical grounds. Statisticians prefer a regression procedure to estimate the EC_{50}, IC_{25}, or similar endpoints, which avoids many statistical problems associated with determining a true “no effect” concentration.

The following estimation procedures have been used in different contexts:

1. In development of water quality standards, the EPA experimentally derives an acute-chronic ratio (ACR) for a test COC, and then applies the ratio to acute studies of other species to estimate the long-term no-effect level from the most sensitive LC_{50} value (Stephan et al., 1985). Default ACRs have been proposed where data are inadequate, and may be adjusted for the amount of data available (e.g., the Tier II method for Great Lakes initiative proposed a default ACR of 18—U.S. EPA, 1995). This is roughly equal to the factor of 0.05 described in Appendix A for deriving water benchmarks for persistent COCs that do not

bioaccumulate. This procedure for estimating an NOAEL would lead to the following calculation:

$TRV_{NOAEL} = LC_{50} \times 0.056$ [U.S. EPA default ACR]

$TRV_{NOAEL} = LC_{50} \times 0.05$ [per Appendix A]

2. The EPA has recommended that an ACR of 10 be used for evaluating mixtures of COCs, i.e., complex effluents, where data are not otherwise available (U.S. EPA, 1991). This equals the factor of 0.10 described in Appendix A for non-persistent COCs. This procedure for estimating an NOAEL would lead to the following calculation:

   $TRV_{NOAEL} = LC_{50} \times 0.10$

3. Regression of chronic thresholds from acute data has been conducted for a variety of aquatic studies. Sloof et al. (1986) reported the regression for fish and daphnid data as:

   $\log (TRV_{NOAEL}) = \log (NOEC) = 0.95 \log LC_{50} - 1.28$ (values in µg/L; NOEC = no observed effects concentration)

4. Suter et al. (1983, 1986) proposed a number of regressions of similar form (see Table 7.8 in Suter, 1993). For example, for fish:

   $\log (MATC) = 0.90 \log LC_{50} - 1.16$ (values in µg/L)

   or TRV_{NOAEL} = Maximum Acceptable Toxicant Concentration (MATC)

5. Other reviews of wildlife and fish concluded that statistically significant effects on sublethal or nonlethal endpoints predominantly occur within a factor of $^1/_5$ to $^1/_6$ of the lethal endpoint (Mayer et al., 1986; Tucker and Lietzke, 1979). This would be equivalent to:

   TRV_{NOAEL} = NOAEL = 16 to 20 percent × LC_{50}

6. Other recommendations (U.S. EPA, 1999, following Calabrese and Baldwin, 1993) include estimating the chronic NOAEL to be 10 percent of a subchronic NOAEL, or as 1 percent of an acute lethal value (LC_{50}). This is equal to the 0.01 factor described in Appendix A for persistent, bioaccumulative COCs. This procedure for estimating an NOAEL would lead to the following calculation:

   $TRV_{NOAEL} = LC_{50} \times 0.01$

7. Following a different logic, the Office of Pesticide Programs of the U.S. EPA has estimated the average ratio of wildlife LD_{50} to $LD_{0.01}$ to be 5, so that protection of a wildlife species is inferred at exposures of 20 percent of the LD_{50} or less (Urban and Cook, 1986). This procedure for estimating an NOAEL would lead to the following calculation:

   $TRV_{NOAEL} = LD_{0.01} = LD_{50} \times 0.20$

8. U.S. EPA (1991), in reviewing the results of whole effluent toxicity tests, concluded that the IC_{25} endpoint served as a reasonable estimate of the

NOEC. This procedure for estimating an NOAEL would lead to the following calculation:

$TRV_{NOAEL} = IC_{25}$

In studies of a large river-reservoir system, Cook et al. (1999) determined as potentially significant a reduction of 20 percent or greater in an endpoint measured in the field, or a 20 percent reduction in survivorship, growth, or reproduction measured in a toxicity test.

This procedure for estimating an NOAEL would lead to the following calculation:

$TRV_{NOAEL} = EC_{20}$

Consequently, the use of an IC_{25} or EC_{20} would seem reasonable procedure to estimate a NOAEL and the resulting TRV_{NOAEL} for purposes of a Tier 2 SLERA for non-bioaccumulative COCs.

Extrapolations

From one endpoint to another for the same species: Few standard approaches appear broadly useful in extrapolating from one endpoint within a species to other endpoints, particularly to endpoints that the TCEQ has determined to be useful: development, reproduction, and survivorship. However, the TCEQ recommends the use of uncertainty factors, as discussed in **3.9.6.4**, for these extrapolations.

Extrapolations from short- to long-duration tests are described above.

Extrapolations between life stages or sizes of individuals of a species vary, although the following generalizations have been made (Suter, 1993):

- For fish, larval stages are generally more sensitive than either adult fish or eggs, and production of viable eggs is generally the most sensitive life stage.
- For aquatic invertebrates, immature forms are more sensitive than mature forms.
- For birds, no systematic consistent pattern of sensitivity related to age or size is evident.

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ATTACHMENT 1: Example Fate and Transport Toxicological Profile for Zinc

ZINC

Environmental Fate and Transport

Sources

Zinc is a naturally occurring trace element, constituting 0.027 percent of the earth's crust (Merck 1983: 1455–58). Natural weathering of materials is the single largest source of zinc released to the environment. Sources of anthropogenic zinc in the environment include electroplators, smelting and ore processors, mine drainage, domestic and industrial sewage, combustion of solid wastes and fossil fuels, road-surface runoff, corrosion of zinc-alloy and galvanized surfaces, and erosion of agricultural soils (Environment Canada, 1996). Limited information is available on total releases of zinc to soil. Zinc is often present in soils and grasses as a result of atmospheric deposition. The National Academy of Sciences (1977: 205–488) has reported that approximately 22,000 tons of zinc is used in the United States each year in fertilizers. Due to lack of evaluation, zinc runoff into soils has not been explored. Hazardous wastes sites are additional sources of zinc in soil. Municipal sludge applied to cropland soils can also be an important source of trace metals, including zinc (Chang et al., 1987).

Transport and Fate

The most common form of zinc found in the environment occurs in the +2 oxidation state. Zinc is highly reactive in soils and can be adsorbed to clay minerals or metallic oxides (Sachdev et al. 1992). This metal forms stable complexes with organic substances such as humic and fulvic acids. Metallic zinc is insoluble while the solubilities of different zinc compounds range from insoluble (oxides, carbonates, phosphates, silicates) to extremely soluble (sulphates and chlorides) (CCME, 1999).

The concentration of zinc in soil porewater depends on soil pH, zinc forms, contents of clays and minerals, organic matter, and other factors. Zinc becomes more soluble with decreasing soil pH and hence more mobile and bioavailable in acidic soil conditions, particularly at a pH < 5 (Duquette and Henershot, 1990). In soils with pH > 7.7, $Zn(OH)^2$ becomes the dominant form and solubility is very low. Zinc in a soluble form, such as zinc sulfate, is fairly mobile in most soils. However, relatively little land-disposed zinc is in soluble form, and mobility is, therefore, limited by a slow rate of dissolution. Consequently, movement towards groundwater is expected to be slow unless zinc is applied to the soil in soluble form (such as in agricultural applications) or accompanied by corrosive substances (such as in mine tailings) (U.S. EPA, 1980). Yet soil conditions not suitable for zinc sorption may lead to leaching. Low pH (< 7) and high ionic strength of the leaching solution favor desorption (U.S. EPA, 1987; Saeed and Fox, 1977).

Speciation and Bioavailability

Zinc is an essential trace element to both plants and animals. The active zinc species in the adsorbed state is the singly charged zinc hydroxide species $Zn(OH)^+$ (Sanders and El Kherbawy, 1987). For calcareous soils, the relationship between zinc solubility and pH is nonlinear. At a high pH, zinc in solution is precipitated as zinc carbonate, or calcium zincate (Saeed and Fox 1977). Clay and metal oxides are capable of sorbing zinc and tend to retard its mobility in soil.

The amount of bioavailable zinc will be determined by the amount of zinc present that is soluble or may be solubilized. Plant uptake, losses by leaching, input of zinc in various forms, changes in moisture content of soil, pH changes, mineralization of organic matter, and changing redox potential of the soil will influence the equilibrium. Due to the complexity of zinc interactions therein, zinc transport behavior in soil cannot be predicted accurately (Hinz and Selim, 1994).

Zinc availability decreases as pH increases (Rehm and Schmitt, 1997); usually, increased zinc levels occur on soils with pH < 5.0 (Vitosh et al., 1994). Killorn (1984) reports that highly organic soils both increase and decrease zinc availability to plants. In addition, zinc availability decreases in cool soil (Killorn, 1984; Rehm and Schmitt, 1997; Mahler and Westermann., 1981). Furthermore, copper, iron, and manganese can inhibit plant uptake of zinc (Heckman, 2010). Plant species have different tolerance levels to the availability of zinc. Grasses can tolerate high levels of available zinc while vegetables are sensitive (Vitosh et al., 1994b). For example, fruit trees and corn are very sensitive to zinc deficiency, but carrots and peas have low sensitivity to zinc deficiency (Heckman, 2010).

Toxicological Profile

Summary

Zinc is ubiquitous in nature and considered an essential trace element, particularly for plants. The nutritional roles of zinc in plants include the regulation of growth, chlorophyll synthesis, carbohydrate formation, biogenesis of growth hormones, and regulation of enzymatic reactions. Similarly, zinc is necessary for normal growth and development in birds and mammals; its presence in these organisms is mostly as metalloenzymes involved in primary and secondary metabolic pathways. Adverse effects of zinc on plants are normally at higher levels and include effects such as decreased seed yield and decreased weight of most plant parts. In invertebrates, zinc decreases the growth rate and reproduction. In microbes, zinc affects a variety of enzymatic reactions involved in respiration and secondary pathways such as nitrification, dehydrogenation, denitrification and respiratory enzymes.

Plants

Zinc is a micronutrient for plants and is required to sustain regulation of growth, chlorophyll synthesis, and carbohydrate formation, and to regulate

enzymatic reactions and hormonal functions. At higher concentrations, however, zinc could produce toxic effects in exposed organisms. The toxicity of zinc in ecosystems has been well-documented in the available literature. The studies readily available on zinc plant toxicity cover a variety of endpoints. Small amounts (3.3 mg/kg) have been shown to decrease the annual ring growth of trees (Hagemeyer et al., 1993). At relatively low levels of 25 and 50 mg/kg, zinc decreases seed yields (Sheppard et al., 1993; Aery and Sakar, 1991). At higher levels, decreased leaf and plant weights and repressed grain yields are observed. Most plant studies use zinc sulfate.

Microbes

Microbial studies for zinc focus largely on decreased enzymatic activity, which commonly has occurred at relatively high levels of over 1600 mg/kg. Along with nitrification, many studies examined zinc's effect on plant respiration, in which a decrease was observed at a range of dose levels.

Invertebrates

Invertebrate studies are available for earthworms, along with an assortment of other organisms. Almost all of the earthworm studies resulted in a decrease in cocoon production or growth rate at levels spanning from 136 to 300 mg/kg. Effects on other invertebrates included, death, decreased population size and decreased growth.

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