

Chapter 25 Analysis: Characterization of Ecological Effects

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25.1 Introduction

As noted in the previous chapter, the analysis step of ecological risk assessment includes characterization of exposures and characterization of ecological effects. Chapter 24 described the approaches and methods used for exposure characterization. This chapter describes the approaches and measures used for characterization of ecological effects. The discussion in this chapter is based largely on EPA's *Ecological Risk Assessment Guidelines*.⁽¹⁾ Readers are referred to that document for a more complete discussion of available approaches and methods.

The methodology used to characterize ecological effects is generally similar to that used for human health toxicity assessment. One of the distinctive features of ecological effects characterization relates to the more general management goal of protecting a receptor population or community rather than a single individual. This has led to the development of water, sediment, and soil quality criteria that are designed to protect the communities of organisms that inhabit surface waters and soils. It also provides the option of using a distribution or range of values to characterize chemical toxicity (an option not generally available in human health risk assessment).

Characterization of ecological effects involves describing the potential effects resulting from exposure to a stressor, linking these effect to the assessment endpoints identified during problem formulation, and evaluating the **stressor-response relationship** (i.e., how the effects will change with varying stressor levels). The characterization begins by evaluating effects information to specify the resulting effects, verifying that these effects are consistent with the assessment endpoints, and confirming that the conditions under which the effects occur are consistent with the conceptual model. Once this has been done, the effects characterization involves two additional steps: (1) performing an ecological response analysis, and (2) developing a **stressor-response profile** which also contains an analysis of uncertainty and variability. Each of these additional steps is discussed in a separate section below.

25.2 Ecological Response Analysis

Ecological response analysis examines three primary elements: identifying stressor-response relationships, establishing causality, and determining the linkages between measurable ecological effects and assessment endpoints. Each is described in a separate subsection below.

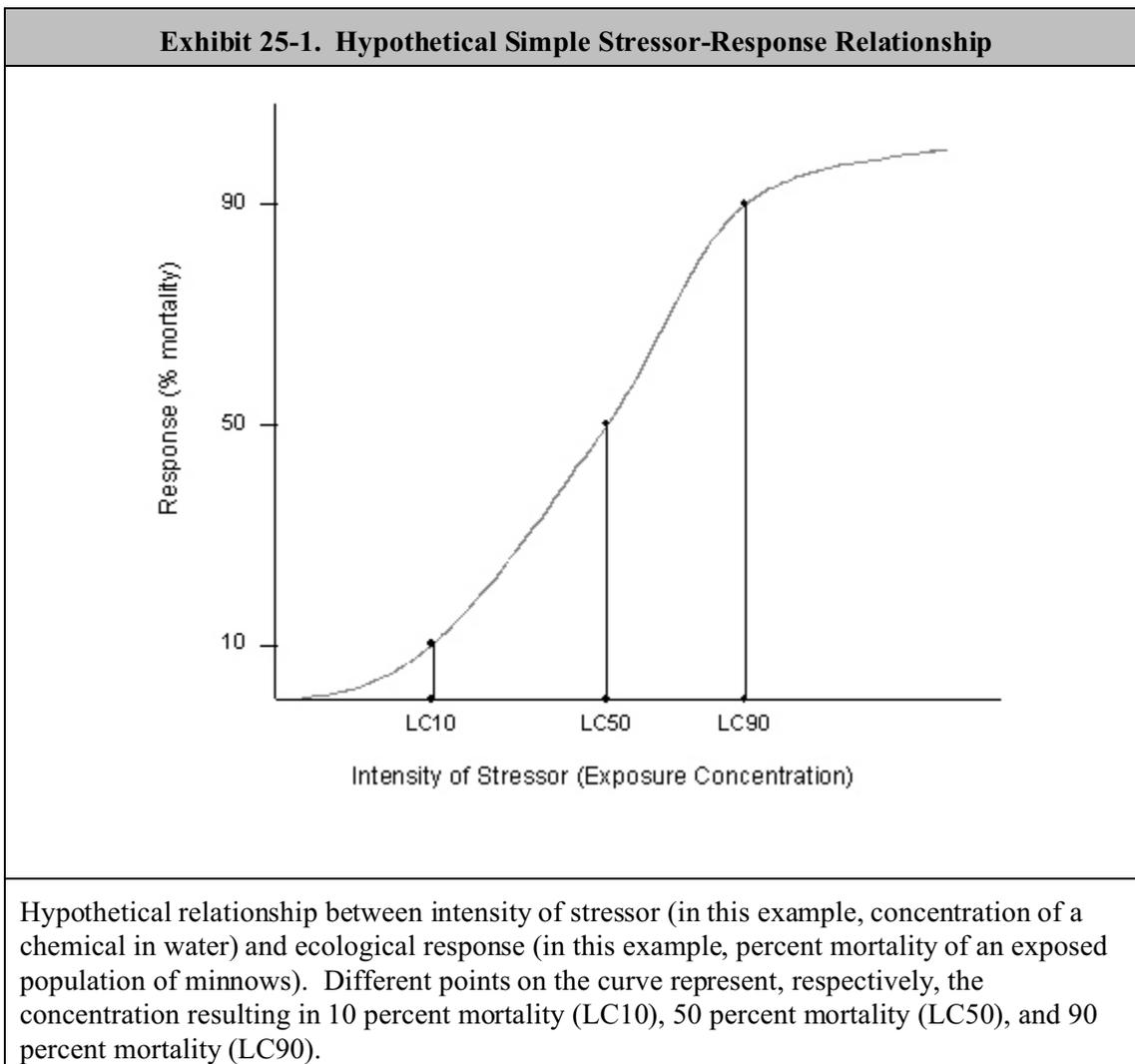
25.2.1 Stressor-Response Analysis

Stressor-response analysis for ecological effects is functionally similar to dose-response analysis for human health effects (e.g., see Chapter 12). The specific stressor-response relationship(s) used in a given risk assessment depend on the scope and nature of the assessment as defined in the problem formulation and reflected in the analysis plan. Three types of stressor-response relationships are commonly used: point estimates, stressor-response curves, and cumulative distribution functions. Each of these is discussed in a separate subsection below.

25.2.1.1 Ecological Effect Levels

Ecological effect levels are point estimates of an exposure associated with a given effect (e.g., a concentration that results in 50 percent mortality in the exposed population, or LC₅₀) used to

compare with an environmental exposure concentration. Data on the toxicity of a chemical is usually obtained from laboratory studies in which groups of organisms (e.g., invertebrates, benthic organisms, plants, earthworms, laboratory mammals, fish) are exposed to varying levels of the chemical, and one or more responses (endpoints such as survival, growth, reproduction) are measured. Various statistical methods are used to establish thresholds for adverse ecological effects associated with acute or chronic exposures. Risk assessors often choose no-effect or low-effect levels as screening values. Stressor-response relationships may be relatively simple (as illustrated in Exhibit 25-1) or may be very complex.



Several specific point estimates are commonly used to characterize ecological effects (Exhibit 25-2):

- **Median effect concentrations or doses** are those levels that result in effects that occur in 50 percent of the test organisms exposed to a stressor. The median effect level is always associated with a time parameter (e.g., 24 hours, 48 hours). Because the tests used to derive median effects levels seldom exceed 96 hours, these values are used primarily to assess acute (short-term) exposures.

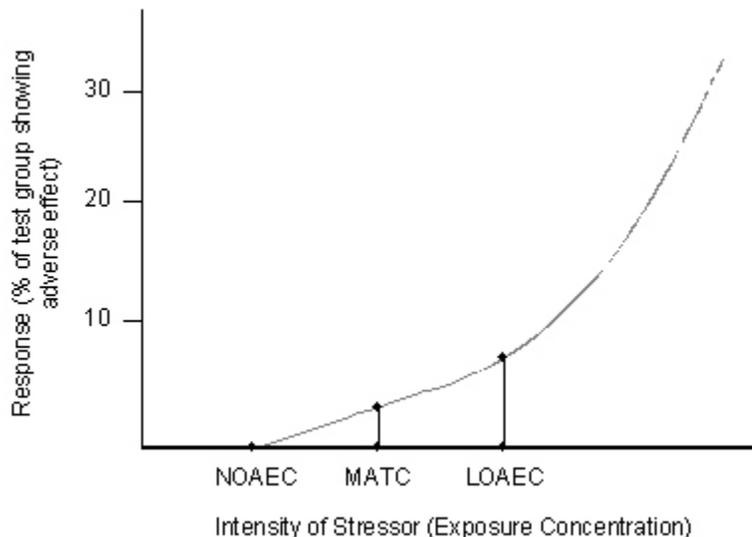
Exhibit 25-2. Commonly Used Point Estimates

Median effect concentrations or doses (acute exposures)

LC ₅₀	Concentration (food or water) resulting in mortality in 50 percent of the exposed organisms
LD ₅₀	Dose (usually in dietary studies) resulting in mortality in 50 percent of the exposed organisms
EC ₅₀	Concentration resulting in a non-lethal effect (e.g., growth, reproduction) in 50 percent of the exposed organisms
ED ₅₀	Dose resulting in a non-lethal effect (e.g., growth, reproduction) in 50 percent of the exposed organisms

Low- or no-effect concentrations or doses (chronic exposures)

NOAEL	no-observed-adverse-effect-level, the highest dose for which adverse effects are not statistically different from controls
LOAEL	lowest-observed-adverse-effect level, the lowest dose at which adverse effects are statistically different from controls
NOEC	no-observed-effect-concentration, the highest ambient concentration for which adverse effects are not statistically different from controls
LOEC	lowest-observed-effect concentration, the lowest ambient concentration at which adverse effects are statistically different from controls
MATC	maximum acceptable toxicant concentration, the range of concentrations between the LOEC and NOEC
GMATC	geometric mean of the MATC, the geometric mean of the LOEC and NOEC



- **Low- or no-effect concentrations or doses** are derived from experimental data using statistical estimates. The no-effect level is determined by experimental conditions as well as the variability inherent in the experimental data. Thus, depending on experimental conditions (e.g., the range of concentrations tested), two separate tests using the same chemical and the same organism could result in different no-effect levels. Low- or no-effect levels are used primarily to assess chronic (longer-term) exposures.

A variety of different types of studies can be used to develop ecological stressor-response relationships, including field studies, laboratory studies, and microcosm studies (Exhibit 25-3).

For air toxics, stress-response analysis can include both primary and secondary effects.

- **Primary effects** (e.g., lethality, reduced growth, neurological/behavioral deficits, impaired reproduction) result from exposure of aquatic and terrestrial organisms to air toxics. An example of a chronic effect would be reduced reproduction in a fish species exposed to air toxics in a surface water body or in a terrestrial bird eating contaminated fish from a small pond. An extreme example of an acute primary effect might be deaths of birds caused by inhalation of a particular toxin. Toxic effects on survival, growth, development, and reproduction might have population-level consequences for a species (e.g., result in local population extinction over time) and are widely accepted as endpoints for characterizing ecological risks. In recent years, more subtle effects have been investigated, including those pertaining to clinical signs of poisoning, immunotoxicity, and even behavioral changes that might influence survival, growth, development, or reproduction.
- **Secondary effects** (e.g., loss of prey species in the community) result from the action of air toxics on supporting components of the ecosystem. These secondary effects occur through biological interaction of one or more species' populations with individuals or populations that have been primarily affected. For example, exposure to an air toxic may adversely affect one or more species of microscopic algae, bacteria, or fungus, which can adversely affect an ecosystem's nutrient cycling and primary production. This can lead to an alteration in the abundance, distribution, and age structure of a species or population dependent on these microscopic organisms, which can then lead to changes in competition and food web interactions in other species. These ecosystem effects can be propagated to still other populations, affecting their presence or representation within the ecosystem. A relatively simple example of secondary effects involves the aerial application of pesticides that dramatically reduced the population of an aquatic insect. This impact to the insect population indirectly affects wild ducklings in the ecosystem, which depend on the insects as a food supply.⁽²⁾ Although it often is possible to identify the potential for secondary effects, developing stressor-response functions for secondary effects (e.g., in a manner analogous to that illustrated in Exhibit 25-2) is not an easy task. A recent paper provides one example of the evaluation of secondary effects in ecological risk assessment.⁽³⁾

Point Estimates, TRVs, and Benchmarks

The terms **Toxicity Reference Values (TRVs)** and **Ecological Benchmarks** are used to describe those **Point Estimates** identified or derived for use in ecological risk assessments. These particular point estimates may be derived from a single study (e.g., an NOEC or EC₅₀) or from the integration of multiple studies (e.g., water quality criteria). When TRVs or benchmarks are drawn from a single study, they are usually set in consideration of multiple studies (e.g., from the study most relevant to the purposes and specifics of the assessment has been selected, or the most sensitive result among the relevant studies)

The use of the point estimate approach has some potential limitations. The most important is that the point estimate established by a given study depends on both the range of doses tested and the statistical power of the study (e.g., the ability to detect an effect if it occurs). For example, studies with low power (e.g., those with only a few test animals per dose group) tend to yield NOAEL or NOEC values that are higher than studies with good power (those with many animals

per dose group). In addition, the choice of some point estimates (e.g., NOEC and LOEC) is restricted to concentrations that were tested, which may or may not be close to the environmentally relevant concentrations, and this uncertainty increases as the interval between doses increases. Finally, it is not always easy to interpret the significance of an exposure that exceeds some particular point estimate, since the severity and incidence of response depends on the shape and slope of the exposure response curve (information that is not captured in a point estimate).

Exhibit 25-3. Types of Ecological Stressor-Response Studies

- **Laboratory Studies.** Most information on ecological stressor-response comes from laboratory ecotoxicology studies using a generic set of species to represent different components of terrestrial or aquatic ecosystems. For example, the freshwater crustacean *Daphnia*, is often used as a surrogate for all small invertebrates that inhabit surface waters, and various species of minnows are used as surrogates for fish. Laboratory studies are relatively easy and inexpensive to conduct, and effects can be directly linked to exposure to a single air toxic. There is uncertainty, however, in extrapolating the results from standard laboratory species to the wide array of species in the environment or from the controlled laboratory conditions to the complex conditions that occur in nature. Additionally, in most cases, laboratory studies are not designed to assess effects on populations, communities, and ecosystems.
- **Field Studies.** Studies of wildlife, populations, communities, and ecosystems exposed to air toxics in natural settings can provide valuable information on stressor-response effects. Field data can be valuable in demonstrating the presence or absence of a cause-effect relationship that can provide a basis for prioritization or for recognizing the efficacy of a risk reduction action. These studies also can be used to assess stressor-response relationships for the site-specific mixtures of concern. However, the study organisms may be exposed to numerous types of stressors (chemical and non-chemical), and the effects of individual air toxics (and sometimes site-specific mixtures) may be difficult to isolate. In addition, field studies are conducted infrequently due to the significant time and resources required. Comparison of the study area to a control area is necessary to evaluate the potential impact of the chemical release.
- **Microcosm Studies.** Microcosm studies use assemblages of several different taxa and environmental media in an enclosed experimental system as a surrogate for natural ecosystems. Such studies can control for some of the uncertainty associated with multiple stressor exposure in field studies. These studies also may provide information about food web dynamics and the interactions of populations or organisms. As with field studies, microcosm studies are time and resource intensive and, therefore, may be relatively uncommon for air toxic studies.

A variety of point estimates are used in ecological risk assessments. Some are developed from acute (short-term) exposures; others are developed from chronic (long-term) exposures. Three general types of point estimates are available for use in ecological risk assessments:

- **Community-level criteria.** EPA has developed ambient water quality criteria (AWQC) and sediment quality criteria for the protection of aquatic communities. These values are based on consideration of a cumulative distribution function (see Section 25.2.1.4). For example, AWQC are designed to protect 95 percent of all aquatic species in freshwater or marine environments. Criteria have been developed for both acute and chronic exposures, although for a limited number of chemicals.

- **Effect levels from laboratory toxicity tests.** A variety of aquatic species are routinely used in ecological toxicity tests, including fathead minnows (a small fish species) and *Daphnia* (a tiny freshwater crustacean). Effects of concern can include acute effects such as mortality (e.g., LD₅₀) as well as chronic effects such as reproduction. Toxicity tests also are available for terrestrial organisms (e.g., earthworms) and occasionally involve vertebrate species of wildlife (e.g., the effects of polychlorinated biphenyls (PCBs) have been studied extensively in mink).
- **Effect levels from field bioassays.** In some cases, ecological effects are evaluated directly by exposing test organisms to ambient conditions. This most often is done where complex mixtures of chemicals are present (e.g., in soils or sediments).

The point estimates employed in ecological risk assessments may be generally termed toxicity reference values (TRVs).^(a) They may be values taken from individual toxicity studies (e.g., NOECs or EC₅₀s) or the result of integration of multiple studies (e.g., water quality criteria). TRVs may be developed for site-specific ecological receptors, depending on the importance of those receptors to the local ecosystem, or for an endpoint not previously evaluated. For example, while some TRVs may be based on survival, growth, and reproductive success of a population, TRVs protective of a threatened or endangered species, a valuable game species (e.g., trout), or an ecologically key species (e.g., wolf) might be based on an endpoint that is relevant to individual organism health (e.g., a neurological deficit) rather than to population maintenance. On the other hand, TRVs based on higher effect levels (e.g., 20 to 50 percent or higher of the population is affected) might be appropriate for species for which great functional redundancy exists in the ecosystem (e.g., different herbaceous plants).⁽⁴⁾

Derivation of TRVs for pathways involving wildlife ingestion would require information on food ingestion rates for sensitive and highly exposed animal species and information on the degree of bioaccumulation in appropriate trophic components. Examples of these derivations for aquatic systems can be found in the Great Lakes Water Quality Initiative (GLWQI) for mercury, dichlorodiphenyltrichloroethane (DDT), PCBs, and dioxin (2,3,7,8-TCDD)⁽⁵⁾ and for terrestrial systems in the EPA methods of assessing exposures to combustor emissions.⁽⁶⁾ EPA's *Wildlife Exposure Factors Handbook*⁽⁷⁾ also provides data, references, and guidance for conducting exposure assessments for wildlife species exposed to toxic chemicals in their environment.

EPA and other organizations have developed a number of types of TRVs based on data for a chemical's toxicity to freshwater or saltwater organisms (see Exhibit 25-4). Toxicity data for longer term or chronic exposures generally will be more useful for an air toxics risk assessment; however, short term or acute toxicity data may be used for chemicals that lack or have incomplete chronic data. EPA has in the past used acute values in conjunction with conversion factors (i.e., acute-to-chronic ratios) to estimate chronic toxicity values, specifically for the derivation of chronic Ambient Water Quality Criteria and Great Lakes Water Quality Initiative criteria for aquatic life.

^aNote that some ecological risk assessment guidance refers to the point estimates of ecological effects selected for a given assessment as Toxicity Reference Values (TRVs), while others use the term ecological benchmarks.

25.2.1.2 Selection of TRVs for a Particular Assessment

In reviewing toxicity studies for potential use in identifying or developing specific TRVs to use in a given assessment, the following questions should be considered:

- What taxa are used in the study?
- Did the study present any significant methodological difficulties?
- Did the study identify a LOAEL?
- Were the adverse effects seen possibly related to growth and survival, or reproduction and development?
- Did the study identify a NOAEL?
- Was the study duration appropriate to assess potential effects of chronic exposure?

If the test species are not within the taxonomic group of the ecological receptors of concern, the study may need to be rejected because the test species are too distantly related to assume similar physiological responses to a toxic agent.

Many studies may be of limited use in selection of TRVs. Potential deficiencies include:

- No control group was analyzed, or there was a high incidence of effects in the control group (applies to laboratory studies);
- No reference area was analyzed, or there was a high incidence of effects in the reference area (applies to field studies);
- No statistical analysis of results was conducted;
- In the case of fish/shellfish, body burdens were estimated, not measured;
- In the case of fish/shellfish, only fillet, carcass (guts, gills, and scales removed), or other body part concentrations were measured, not the whole body;
- In the case of wildlife, insufficient data were provided to calculate the dose to the animal; and
- Multiple contaminants were present in the experimental studies.

Most environmental contamination concerns for air toxics that persist and bioaccumulate will tend to be long-term and relatively low-level. As such, the most appropriate toxicity studies are those evaluating chronic (long-term) toxicity or, if chronic studies are not available, subchronic (medium-term) exposure durations. Although no one definition of “chronic” is accepted by human or ecological toxicologists, the general concept is that the duration encompasses a significant portion of the species life span (e.g., ten weeks for birds and one year for mammals). “Subchronic” is commonly defined as a 90-day or longer study for mammals and 10 weeks or fewer for birds. For aquatic bioassays, chronic tests may span multiple generations and assess sensitive growth or reproductive endpoints. In mammalian and avian tests, the term average daily dietary dose (e.g., expressed as mg/kg-day) generally implies chronic or subchronic exposure.⁽⁸⁾

In order to develop TRVs (sometimes termed benchmarks) for avian and mammalian receptors, Oak Ridge National Laboratory’s Toxicological Benchmarks for Wildlife,⁽¹¹⁾ and some information from EPA’s Integrated Risk Information System⁽⁹⁾ can be used (in a more limited fashion). Information provided in these sources has to be modified using allometric information available in EPA’s *Wildlife Exposure Factors Handbook*⁽⁷⁾ to better represent potential wildlife species sensitivity.

Exhibit 25-4. Sources of Ecological TRVs or Benchmarks

Data Source	Available Toxicity Reference Value(s)	Overview of Data Source and Values
EPA Office of Water Ambient Water Quality Criteria (AWQC)	<ul style="list-style-type: none"> • AWQC Chronic Criteria • AWQC Acute Criteria <p>Note: many state water quality standards are based on AWQC</p>	<p>EPA has developed national recommended water quality criteria for the protection of aquatic life for approximately 150 pollutants. These criteria are published pursuant to Section 304(a) of the Clean Water Act (CWA) and provide guidance for States and Tribes to use in adopting water quality standards under Section 303(c) of the CWA.</p> <p>Source: http://www.epa.gov/waterscience/criteria/aqlife.html</p>
Great Lakes Water Quality Initiative (GLWQI) Criteria Documents	<ul style="list-style-type: none"> • GLWQI Tier I Criteria • Final Chronic Values (FCVs) 	<p>GLWQI Tier I criteria and final chronic values (FCVs) are calculated under the same guidelines as the Sediment Quality Criteria (SQC). Draft GLWQI criteria documents were released for public review and were revised as necessary before they were published as “final.”</p> <ul style="list-style-type: none"> • Tier I Criteria are designed to be protective of aquatic communities • FCVs are designed to measure chronic toxicity to aquatic organisms <p>Source: <i>Final Water Quality Guidance for the Great Lakes System</i>. Federal Register, Mar. 23, 1995, vol. 60, no. 56, p. 15365-15424</p>
EPA Soil Screening Levels	<ul style="list-style-type: none"> • Soil screening levels 	<p>EPA has developed a methodology and initial soil screening levels protective of ecological receptors.</p> <p>Source: U.S. Environmental Protection Agency. 2000. <i>Ecological Soil Screening Guidance (Draft)</i>. Office of Emergency and Remedial Response, Washington, D.C., July 2000.</p> <p>http://www.epa.gov/superfund/programs/risk/ecorisk/ecossl.htm.</p>
EPA Region 4 Soil Screening Levels	<ul style="list-style-type: none"> • Soil screening levels 	<p>Source: U.S. Environmental Protection Agency. 1995. Supplemental Guidance to RAGS: Region 4 Bulletins No. 2. Ecological Risk Assessment. Region IV, Waste Management Division.</p> <p>http://www.epa.gov/region04/waste/ots/ecolbul.htm</p>

Exhibit 25-4. Sources of Ecological TRVs or Benchmarks

Data Source	Available Toxicity Reference Value(s)	Overview of Data Source and Values
<p>Ecotox Thresholds ECO Update and EPA's Hazardous Waste Identification Rule (HWIR) documents</p>	<ul style="list-style-type: none"> • GLWQI Tier II Criteria • Secondary Chronic Values (SCVs) 	<p>The GLWQI Tier II criteria and SCVs have received some peer review prior to publication, and 12 of them are included in the HWIR, which underwent public comment before promulgation. The GLWQI Tier II methodology calculates SCVs in a similar way to FCVs, but uses statistically derived "adjustment factors" and has less rigorous data requirements.</p> <ul style="list-style-type: none"> • Tier II Criteria are designed to be protective of aquatic communities • SCVs are designed to measure chronic toxicity to aquatic organisms <p>Source: <i>Ecotox Thresholds ECO Update</i> (volume 3, No. 2, January 1996, EPA/540/F-95/038).</p>
<p>ECOTOXicology database (ECOTOX)</p>	<ul style="list-style-type: none"> • Point Estimates from Chronic Tests (e.g., EC₅₀, EC₁₀, LC₅₀, or GMATC) • Point Estimates from Acute Tests (e.g., LC₅₀) 	<p>ECOTOX is a source for locating single chemical toxicity data for aquatic life, terrestrial plants, and wildlife. ECOTOX was created and is maintained by EPA's Office of Research and Development and the National Health and Environmental Effects Research Laboratory's Mid-Continent Ecology Division. ECOTOX is a source for locating single chemical toxicity data from three EPA ecological effects databases: AQUIRE, TERRETOX, and PHYTOTOX. AQUIRE and TERRETOX contain information on lethal, sublethal, and residue effects. AQUIRE includes toxic effects data on all aquatic species including plants and animals and freshwater and saltwater species. TERRETOX is the terrestrial animal database. It primarily focuses on wildlife species but the database does include information on domestic species. PHYTOTOX is a terrestrial plant database that includes lethal and sublethal toxic effects data. Source: http://www.epa.gov/ecotox.</p>
<p>Sediment Quality Criteria</p>	<ul style="list-style-type: none"> • Varies 	<p>EPA and other agencies have developed sediment quality criteria for the protection of benthic communities. These criteria are highly specific to regions and bodies of water in the U.S. Regional experts are the recommended source for appropriate site-specific criteria.</p>

Exhibit 25-4. Sources of Ecological TRVs or Benchmarks

Data Source	Available Toxicity Reference Value(s)	Overview of Data Source and Values
Ecological Structure Activity Relationships (ECOSAR)	<ul style="list-style-type: none"> • Estimated Chronic GMATC • Estimated Acute Data (LC₅₀ or EC₅₀) 	<p>ECOSAR is a computer program that uses structure-activity relationships (based on available data) to predict the acute and chronic toxicity of organic chemicals to aquatic organisms. ECOSAR provides quantitative estimates of chronic values (e.g., GMATC), acute LC₅₀ values, and acute EC₅₀ values for industrial chemicals for several aquatic species (e.g., fish, daphnia, green algae, mysids). When the estimated aquatic toxicity value exceeds the water solubility of the compound, the estimated value is flagged; this situation generally is interpreted to mean that the chemical has no toxic effects in a saturated solution. Source: http://www.epa.gov/oppt/newchems/21ecosar.htm</p>
Exposure-Related Effects Database (ERED)	Tissue-based effects values for fish and benthic invertebrates	<p>The U.S. Army Corps of Engineers Exposure-Related Effects Database (ERED) lists toxicity information for a large number and wide taxonomic range of fish and shellfish. ERED is constantly being updated. Source: http://www.wes.army.mil/el/ered/</p>
Jarvinen and Ankley database	Fish and shellfish exposure and effects information	<p>The authors assembled a database of fish and shellfish exposure and effect information. Source: Jarvinen and Ankley (1999)⁽⁹⁾</p>
Oak Ridge National Laboratory (ORNL) Soil Invertebrate toxicity database	Acute and chronic TRVs for soil invertebrates and microbial processes	<p>This report focuses on chemicals found at U.S. Department of Energy (DOE) sites; however there are overlaps with air toxics (metals and organics). Source: Efroymson et al. (1997);⁽⁸⁾ http://www.esd.ornl.gov/programs/ecorisk/documents/tm126r21.pdf</p>
ORNL Plant toxicity database	Acute and chronic TRVs for terrestrial plants	<p>This report presents a standard method for deriving TRVs, a set of data concerning effects of chemicals in soil or soil solution on plants, and a set of phytotoxicity TRVs for 38 chemicals potentially associated with DOE sites. Source: Efroymson et al. (1997)⁽⁸⁾</p>
ORNL Wildlife toxicity database	Wildlife NOAEL and LOAELs	<p>This report presents both NOAEL- and LOAEL-based TRVs for assessment of effects of 85 chemicals on 9 representative mammalian wildlife species and 11 avian wildlife species. Source: Sample et al. (1996)⁽¹⁰⁾</p>

25.2.1.3 Stressor-Response Curves

One way to resolve some of the limitations in the TRV approach is to fit a mathematical equation to the available exposure-response data and describe the entire **stressor-response curve**. Data from individual experiments may be used to develop curves and point estimates both with and without associated uncertainty estimates. The advantages of curve-fitting approaches include using all of the available experimental data, the ability to interpolate to values other than the data points measured, and an improved ability to extrapolate to values outside the range of experimental data (e.g., for a low- or no-effect level). Curve-fitting often is used to extrapolate from observed effects levels to develop estimates of NOAELs, NOECs, and/or GMATCs. Stressor-response curves can be developed using any convenient data fitting software, but EPA has developed a software package specifically designed for this type of effort. This software is referred to as the Benchmark Dose Software (BMDS). More information on this software can be found on the National Center for Environmental Assessment's webpage.⁽¹¹⁾ A disadvantage of curve fitting is that the number of data points required may not always be available (e.g., especially for toxicity tests with wildlife species)

25.2.1.4 Species Sensitivity Distribution

In some cases, risk management decisions may also consider community-level effects as well as population-level or sub-population effects (one example is the Ambient Water Quality Criteria for the protection of aquatic life discussed in Section 25.2.1.1). That is, a stressor might be considered to be below a level of concern for the sustainability of a community if only a small fraction of the total number of exposed species are affected. In this case, toxicological responses may be best characterized by the distribution of toxicity values across species. This is called a **Species Sensitivity Distribution (SSD)**. The SSD approach is generally used for communities of aquatic receptors, since all of the different species that make up the community (e.g., all fish, benthic invertebrates, aquatic plants, and amphibians that reside in a stream) will be exposed to approximately the same concentration of contaminant in the water.

The process for generating an SSD consists of the following steps:

- (1) Select an appropriate type of endpoint (e.g., lethality, growth, reproduction), and select an appropriate type of point estimate from the exposure-response curve for each species. For example, the TRV might be the LC_{50} for lethality or the EC_{20} for growth. The key requirement is that the SSD be composed of TRVs that are all of the same type, not a mixture.
- (2) Collect all reliable values for that type of TRV from the literature for as many relevant species as possible. When more than one value is available for a particular species, either select the value that is judged to be of highest quality and/or highest relevance, or combine the values across studies to derive a single composite value for each species. It is important to have only one value per species to maintain equal weighting across species.
- (3) Characterize the distribution of values across species with an appropriate SSD. Note that there is no *a priori* reason to expect that an SSD will be well characterized by a parametric distribution, so both parametric and empirical distributions should be considered.

Once an SSD has been developed, the fraction of species in the exposed community that may be affected at some specified concentration may be determined either from the empirical distribution or from the fitted distribution. These distributions can help identify stressor levels that affect a minority or majority of species.

A limiting factor in the use of SSDs is the amount of data needed as inputs. SSDs also can be derived from models that use Monte Carlo or other methods to generate distributions based on measured or estimated variation in input parameters for the models.

25.2.2 Linking Measures of Effects to Assessment Endpoints

As noted in Chapter 23, assessment endpoints express the environmental values of concern for the risk assessment; however they cannot always be measured directly. For example, the assessment endpoint may be maintaining a healthy population of trout in a lake, but measures of effect (e.g., toxicity tests) were conducted on different species (e.g., fathead minnows). Where there is a lack of time, monetary resources, or practical means to acquire more data, extrapolations may be the only way to bridge the gap in available data. Two general approaches are used for such extrapolations:

Examples of Extrapolations

- Between taxa (e.g., minnow to rainbow trout)
- Between responses (e.g., mortality to growth or reproduction)
- From laboratory to field
- Between geographic areas
- Between spatial scales
- From data collected over a short time frame to longer-term effects

- **Empirical extrapolations or process models.** Empirical extrapolations use experimental or observational data; process-based approaches rely on some level of understanding of the underlying operations of the system of interest.
- **Professional judgment.** This is not as desirable as empirical or process-based approaches, but it is the only option when data are lacking. However, professional judgment can be credible, provided it has a sound scientific basis.

One of the most common types of extrapolations is that of effects observed in the laboratory (e.g., toxicity tests) to those observed in the field. Exhibit 25-5 highlights the general questions to consider when performing such an extrapolation.

When conducting field sampling or other monitoring studies, it sometimes is difficult to identify exposure-response relationships. However, there are a number of reasons why a relationship between a chemical and a toxic response in a natural system may not be apparent (Exhibit 25-6). Therefore, the lack of an observed exposure-response relationship does not disprove that one or more air toxics caused an apparent toxic effect. These sources of variation should be considered during planning and scoping, but may not become apparent until field studies have begun.

Exhibit 25-5. Questions to Consider When Extrapolating from Effects Observed in the Laboratory to Potential Effects in Natural Systems

Exposure Factors

- How will environmental fate and transformation of the air toxic affect exposure in the field?
- How comparable are exposure conditions and the timing of exposure?
- How comparable are the routes of exposure?
- How do abiotic factors influence bioavailability and exposure?
- How likely are preference and avoidance behaviors in the receptors of concern?
- How does life-stage affect exposure?

Effects factors

- What is known about the biotic and abiotic factors controlling populations of the receptors of concern?
- To what degree are critical life-stage data available?
- How may exposure to the same or other stressors in the field have altered organism sensitivity?

Empirical approaches are derived from experimental data or observations. They commonly are used when adequate effects data are available, but the understanding of the underlying mechanisms, action, or ecological principles is limited. Two types of empirical approaches are generally used:

- **Uncertainty factors** are derived numbers that are divided into measure of effects values to derive an estimated level of stressor that should not cause adverse effects to the assessment endpoint. An example might be an uncertainty factor of 10 to convert an acute LC₅₀ value into a presumed NOAEL. Uncertainty factors should be used with caution, especially when used in an overly conservative fashion, as when chains of factors are multiplied together without sufficient justification.
- **Allometric scaling** is used to extrapolate the effects of a chemical stressor on one species to another species. Allometry is the study of change in the proportions of various parts of an organism as a consequence of growth and development. Processes that influence toxicokinetics (e.g., renal clearance, basal metabolic rate, food consumption) tend to vary across species according to allometric scaling factors that can be expressed as a nonlinear function of body weight. Allometric scaling factors are commonly used for human health toxicity assessments (see for example Chapter 12), but have not been applied as extensively to ecological effects.

When sufficient information on stressors and receptors is available, process-based approaches such as population or ecosystem process models may be used. Process models allow information on individual effects (e.g., mortality, growth, reproduction) to be extrapolated to potential alterations in specific populations, communities, or ecosystems. Such models are particularly useful in evaluating hypotheses about the duration and severity of impacts from a stressor on an assessment endpoint (e.g., species diversity) that cannot be tested readily in a laboratory. Two types of process-based models are commonly used:

**Exhibit 25-6. Reasons Why Contaminant Concentrations in Ambient Media
May Not Be Correlated with Toxicity of Those Media**

Variation in bioavailability

- Due to variance in medium characteristics
- Due to variance in contaminant age among locations (contaminants deposited to soil and sediments may become less bioavailable over time due to sequestration)
- Due to variance in transformation or sequestration rates among locations

Variation in the form of the chemical (e.g., ionization state)

Variation in the concentration over time or space (i.e., samples for analysis may not be the same as those tested)

- Spatial heterogeneity
- Temporal variability (e.g., aqueous toxicity tests last for several days but typically water from only one day is analyzed)

Variation in the composition of releases (concentrations of components of releases other than the individual air toxic that is believed to be the principal toxicant may vary over space and time, thereby obscuring the relationship)

Variation in co-occurring contaminants (concentrations of contaminants from upgradient [background] sources may vary over time)

Inadequate detection limits (if detection limits are too high, gradients of toxic effect may be observed even when the chemicals are at the “not detected” levels)

Variation in toxicity tests

- Inherent variation
- Variation due to variance in medium characteristics (e.g., hardness, organic matter content, pH)

Source: Guidelines for Ecological Risk Assessment⁽¹⁾

- **Single-species population models** describe the dynamics of a finite group of individuals through time. They have been used extensively in ecology and fisheries management to assess the impacts of power plants and toxic chemicals on specific fish populations.
- **Community and ecosystem models** are particularly useful when the assessment endpoint involves structural (e.g., community composition) or functional (e.g., primary productivity) elements or when secondary effects are of concern.

Exhibit 25-7 provides further discussion of process-based models, highlighting a few models that have been applied in ecological risk assessment.

Exhibit 25-7. Process-based Model Applications in Ecological Risk Assessment

Process-based models can help the assessor understand the potential significance of toxicant effects to the population structure, and ecosystem models can help determine whether the effect may result in secondary effects on other species in the system that are linked in the food web or on overall ecosystem functions. Pastorok et al.⁽¹²⁾ review a number of population, and community and ecosystem models, as well as software that implement these models.

Population models typically deal with the dynamics of the abundance or distribution of a single species, sometimes with explicit descriptions of endpoints in time and space. These models can be categorized as scalar abundance, life history, individual-based, and metapopulation models. The first two types of models are highlighted here:

- *Scalar abundance models*, which represent populations as a single scalar dimension without a breakdown of population age structure, are frequently used in screening assessments. These models include Malthusian population growth models and logistic population growth models.
- *Life history models* estimate population characteristics such as survival rates and fecundity as a function of age or size/morphological state. These models are important because toxicants can have a differential impact on different demographic sections of the same species. These models include deterministic and stochastic age- or stage-based models, which are implemented in software by programs such as *RAMAS-Age*[®], *-Stage*[®], *-Metapop*[®], or *-Ecotoxicology*[®]; and *ULM*[®].

Community and Ecosystem models are intended to describe ecological systems composed of interacting species. These models incorporate species dynamics and specific biological interactions (predator-prey, competition, dependence) to predict ecosystem endpoints such as species richness or the productivity of a multi-species assemblage. Pastorok et al. categorize these models as food web, aquatic, and terrestrial models.

- *Food web models* capture feeding relationships between all or some species in an ecological community, thus determining population dynamics as well as identifying key exposure pathways for bioaccumulative chemicals. These models include predator-prey models and population-dynamic food chain models, which are implemented in software such as *RAMAS Ecosystem*[®], *Populus*[®], and *Ecotox*.
- *Aquatic ecosystem models* are spatially aggregated models that represent biotic and abiotic structures in combination with physical, chemical, biological, and ecological processes in rivers, lakes, reservoirs, estuaries, or coastal ecosystems. A number of models exist for each type of aquatic ecosystem. The standard water column model or *SWACOM*[®] requires the use of laboratory data to predict changes in the parameters of an entire ecosystem. The extrapolation is accomplished with knowledge of toxicological modes of action, and by simulation of the effects of a toxic substance across different trophic levels according to the relationship between nutrients, phytoplankton, zooplankton, and fish. *AQUATOX* (<http://www.epa.gov/ost/models/aquatox/>) predicts the fate of various pollutants, such as nutrients and organic chemicals, and their effects on the aquatic ecosystem, including fish, invertebrates, and aquatic plants. The Comprehensive Aquatic Simulation Model (*CASM*) is a bioenergetics-based food web model that includes phytoplankton, periphyton, macrophytes, zooplankton, benthic invertebrates, fish, bacteria, and cyanobacteria.
- *Terrestrial ecosystem models* represent biotic and abiotic components in deserts, forests, grasslands, or other terrestrial environments, and often include physical, chemical, biological, and ecological processes. The primary endpoints of these models include the abundance of individuals within species or guilds, biomass, productivity, and food-web endpoints such as species richness or trophic structure.

25.3 Stressor-Response Profile

The final product of an ecological response analysis is a summary profile in the form of a written document or a component of a larger process model. The stressor-response profile should address the following questions:

- What ecological entities are affected? These may include single species, populations, general trophic levels, communities, ecosystems, or landscapes.
- What are the nature of the effects? The nature of effects should be germane to the assessment endpoints. For example, if a single species is affected, the effects should represent parameters (e.g., growth, reproduction) appropriate for that level of organization.
- Where appropriate, what is the time scale for recovery? Short- and long-term effects should be reported as appropriate.
- How do changes in measures of effects relate to changes in assessment endpoints (see Section 25.2.2 above)?
- What is the uncertainty associated with the analysis (see Section 25.4)?

25.4 Evaluating Variability and Uncertainty

The stressor-response profile described in the previous section should include an explicit description of any uncertainties associated with the ecological response analysis. If it was necessary to extrapolate from measures of effect to the assessment endpoint, both the extrapolation and its basis should be described. Similarly, if a TRV was calculated, the extrapolations, assumptions, and uncertainties associated with its development should be described. The discussion also should include any information about known or potential variability in a stressor-response profile (e.g., among different species or taxa).

Professional judgment often is needed to determine the uncertainty associated with information taken from the literature and any extrapolations used in developing a parameter to estimate stressor-response. All assumptions used to develop stressor-response relationships and TRVs should be stated, including some description of the degree of bias possible in each. Where literature values are used, an indication of the range of values that could be considered appropriate also should be indicated. A more thorough description of how to deal with variability and uncertainty in the risk assessment process is provided in Chapter 31.

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