Chapter 6
Risk Characterization

What's Covered in Chapter 6:

- Risk Estimation
- Risk Description
- Uncertainty and Limitations of the Screening Level Risk Assessment

Risk characterization includes risk estimation and risk description (U.S. EPA 1992b). Risk estimation is an integration of the exposure assessment (see Section 5.1) and the toxicity assessment (see Section 5.4) to determine the potential risk to a community or guild from exposure to a COPC. Risk estimation is quantified using the quotient method to calculate an ecological screening quotient (*ESQ*) (Suter 1993). 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 *ESQ* values. Risk description also discusses the significance of the default assumptions used to assess exposure, because they affect the magnitude and certainty of the calculated *ESQ* value. The resultant risk characterization should consider any major uncertainties and limitations associated with results generated in performing the screening level risk assessment.

Section 6.1 discusses using the quotient method and calculation of *ESQ*s to estimate potential ecological risk. Section 6.2 discusses various aspects of the risk description. Section 6.3 discusses consideration of uncertainties and limitations.

6.1 RISK ESTIMATION

To estimate potential ecological risk, an *ESQ* should be calculated specific to each measurement receptor, COPC, and exposure scenario location evaluated in the risk assessment. Also, dietary-variable *ESQ*s should be computed for class-specific guild measurement receptors based on "equal diet" dose and "exclusive diet" dose, as discussed in Section 5.3. As expressed in Equation 6-1, an *ESQ* is the quotient of the COPC estimated exposure level (*EEL*) divided by the COPC and measurement receptor specific toxicity reference value (*TRV*), as follows:

$$ESQ = \frac{EEL}{TRV}$$ Equation 6-1

where

ESQ	=	Ecological screening quotient (unitless)
EEL	=	COPC estimated exposure level (mass COPC/mass media [communities] or mass daily dose COPC ingested/mass body weight-day [class-specific guilds])
TRV	=	COPC toxicity reference value (mass COPC/mass media [communities] or mass daily dose COPC ingested/mass body weight-day [class-specific guilds])

Care should be made to ensure that the units for the *EEL* value and the *TRV* are consistent, including correct use of corresponding wet and dry weights. *TRV*s specific to organic and inorganic compounds are typically expressed in units of µg/kg and mg/kg, respectively. General guidance for determining *TRV*s is provided in Chapter 5. Also, Appendix E provides compound specific *TRV*s for the example measurement receptors identified in the food webs in Chapter 4.

*ESQ*s for community measurement receptors are calculated using *EEL*s specific to the COPC concentration in the corresponding media. A COPC specific *ESQ* should be calculated for each community measurement receptor at each location evaluated, as appropriate for the food web being analyzed in the risk assessment. For calculating *ESQ*s for class-specific guild measurement receptors, the *EEL* is the daily dose of COPC ingested. A COPC specific *ESQ* should also be calculated for each class-specific guild measurement receptor at each location evaluated, as appropriate for the food web being analyzed in the risk assessment. For class-specific guild measurement receptors, *ESQ*s should be calculated specific to equal and exclusive diets (see Chapter 5).

To evaluate potential risk resulting from exposure of a measurement receptor to multiple COPCs at a specific location, each of the COPC-specific *ESQ* values should be summed to determine a total *ESQ*.

$$ESQ_{ReceptorTotal} = \Sigma\ ESQ_{COPC\ Specific}$$ Equation 6-2

where

$ESQ_{Receptor\ Total}$ = Total ecological screening quotient for receptor (unitless)
$ESQ_{COPC\ Specific}$ = COPC specific ecological screening quotient (unitless)

As for COPC-specific *ESQ*s, total *ESQ*s for class-specific guild measurement receptors should be calculated specific to equal and exclusive diets (see Chapter 5).

6.2 RISK DESCRIPTION

Risk description considers the magnitude and nature of potential risk for community and class-specific guild measurement receptors evaluated, and provides information for the risk manager and permitting authority to evaluate the significance of an *ESQ* value. Also, Section 6.2.2 recognizes some of the default exposure assumptions that may affect the magnitude of an *ESQ* value.

6.2.1 Magnitude and Nature of Ecological Risk

The magnitude and nature of potential risk should be further considered for each measurement receptor with a COPC-specific *ESQ* value equal to or above risk target levels specified by the appropriate permitting authority. Interaction between the risk assessor and the risk manager and permitting authority has been noted throughout the process (See Figure 1 for Scientific Management Decision Points). At the risk characterization phase of the risk assessment, most of the interaction between the risk assessor and the risk manager and permitting authority is through description of the certainty of the resulting risk estimates. Consistent with the NCP and current U.S. EPA guidance (1998c), the risk manager and permitting authority with input from the risk assessor should also consider the need to collect additional information to refine risk estimates and/or implement permit requirements (i.e., operating conditions, use of APCDs, waste feed conditions, or environmental monitoring) at combustion facilities where an *ESQ* exceeds risk target levels for ecological communities or guilds that may reasonably be expected to be exposed.

The magnitude and nature of potential risk should also be further considered for each measurement receptor with a total *ESQ* value greater than or equal to the target risk levels. While the total *ESQ* provides the risk manager and permitting authority with useful information regarding potential risk resulting from exposure of a measurement receptor to multiple COPCs at a specific location, potential limitations and uncertainties

associated with the calculation of the total *ESQ* should be considered before its use. Specifically, the resulting total *ESQ* is determined by summing COPC-specific *ESQ*s that will usually be calculated utilizing *TRV*s (see Chapter 5) based on different effects (e.g. growth, reproduction), toxicity endpoints (e.g., NOAEL, LOAEL) and/or exposure durations (e.g., chronic, acute). In considering usability of total *ESQ*s, U.S. EPA OSW recommends that the risk manager and permitting authority focus on the highest contributing COPCs, or classes of COPCs which can appropriately be added across effects, toxicity endpoints and exposure durations, in further evaluating potential risks due to exposure to multiple COPCs.

Broad assessment endpoints rather than toxicologically-specific endpoints are recommended for performing a screening level ecological risk assessment (see Chapter 5). Therefore, the potential risk to each community and guild evaluated in the risk assessment should be described. Specifically, potential adverse effects should be described for each community and guild with a COPC-specific or total *ESQ* value equal to or above risk target levels. This should be performed for each selected food web and receptor location evaluated, and specific to equal and exclusive diets for applicable class-specific guilds. The description should characterize potential risk to the selected assessment endpoints, based on the measures of effect and measurement receptors. U.S. EPA OSW recommends that the risk description specific to a measurement receptor include, at a minimum, the contributing COPCs, emission sources, exposure pathways, and significant uncertainties.

6.2.1.1 Target Levels

Target levels are risk management based and set by the regulatory authority. Target values are not a discrete indicator of observed adverse effect. If a calculated risk falls within target values, a regulatory authority may, without further investigation, conclude that a proposed action does not present an unacceptable risk. A calculated risk that exceeds these targets, however, would not, in and of itself, indicate that the proposed action is not safe or that it presents an unacceptable risk. Rather, a risk calculation that exceeds a target value triggers further careful consideration of the underlying scientific basis for the calculation.

6.2.2 Fate and Exposure Assumptions

As noted throughout this guidance, the screening level ecological risk assessment is based on numerous conservative assumptions affecting the potential for a receptor to be exposed to a compound emitted from a facility and the numeric magnitude of the resulting estimated risk. These fate and exposure assumptions are required as a result of current data gaps and uncertainties associated with available scientific information and data required for risk evaluation. However, U.S. EPA OSW recommends that as information is available to address data gaps and reduce uncertainties specific to ecological risks identified at a facility by the screening level risk assessment, it should be provided to the permitting authority for approval to be incorporated into evaluation of risk. Some of the fate and exposure assumptions utilized in this guidance to conduct a screening level risk assessment are listed below:

- The estimated COPC concentration in soil and sediment is 100 percent bioavailable. This includes a COPC that is weakly or strongly adsorbed to particles and a COPC that is dissolved in interstitial water.
- The estimated dissolved COPC concentration in the water column is 100 percent bioavailable. For ingestion of water by wildlife, this includes a COPC that is freely dissolved as an ion or compound, and a COPC that may be adsorbed to another matrix, such as dissolved organic carbon.
- The total COPC mass estimated to be ingested by a measurement receptor is taken up across the gut and reaches the site of toxic action. This includes COPC concentrations in food items and abiotic media. This assumes that no fraction of the COPC mass is metabolized or otherwise depurated by an ecological receptor, and that there is no competition for available sites where the toxic action occurs.
- The chemical species present is the most toxic form, and is the form represented by the *TRV*.
- Community measurement receptors inhabiting an abiotic medium take up 100 percent of the COPC concentration to which they are exposed. All COPC mass taken up by a plant or animal food item of a measurement receptor is assimilated into edible biomass.
- An ecological receptor is continuously exposed during its entire life, including critical life stage(s).
- A measurement receptor's home range is 100 percent within the assessment area being evaluated in the risk assessment.
- A measurement receptor's food is 100 percent contaminated.

The relevance of fate and exposure assumptions specific to COPCs at a site, and their numerical bias to resulting *ESQ* values should be considered before application of results. Also, to facilitate the qualitative assessment of toxicokinetic and toxicodynamic factors (e.g., bioavailability, metabolism), toxicological profiles of numerous compounds often considered in combustion risk assessments (see Section 2.3) are included in Appendix H. U.S. EPA OSW prepared these profiles because it believes that these compounds (1) will be the principal compounds of ecological concern at combustion facilities, and (2) to promote consistency in presenting and evaluating relevant COPC-specific toxicity information.

6.3 UNCERTAINTY AND LIMITATIONS OF THE RISK ASSESSMENT PROCESS

This section describes how to interpret uncertainties associated with the risk assessment. The discussion of uncertainties in this section and in Section 6.3.1 was adopted from the U.S. EPA *1996 Risk Assessment Support to the Development of Technical Standards for Emissions from Combustion Units Burning Hazardous Waste* (EPA Contract Number 68-W3-0028), dated February 20, 1996.

Uncertainty can be introduced into a risk assessment at every step of the process outlined in this document. Uncertainty occurs, because risk assessment is a complex process, requiring the integration of the following:

- Release of pollutants into the environment
- Fate and transport of pollutants, in a variety of different and variable environments, by processes that are often poorly understood or too complex to quantify accurately
- Potential for adverse effects in receptors, as extrapolated from studies of differing species
- Probability of adverse effects in functionality of food web that is made up of species that are highly variable

Uncertainty is inherent in the process even if the most accurate data with the most sophisticated models are used. The methodology outlined in this document relies on a combination of point values—some conservative and some typical—yielding a point estimate of exposure and risk that falls at an unknown percentile of the full distributions of exposure and risk. For this reason, the degree of conservatism in risk estimates cannot be known; instead, it is known that the values combine many conservative factors and are likely to overstate actual risk (Hattis and Burmaster 1994). Therefore, a formal uncertainty analysis is

required to determine the degree of conservatism. This section discusses the types of uncertainty and the areas in which uncertainty can be introduced into an assessment. In addition, this section discusses methods for qualitatively and quantitatively addressing uncertainty in risk assessments.

It should also be noted, variability is often used interchangeably with the term "uncertainty," but this is not strictly correct. Variability may be tied to variations in physical and biological processes, and cannot be reduced with additional research or information, although it may be known with greater certainty (for example, the weight distribution of a species may be known and represented by the mean weight and its standard deviation). "Uncertainty" is a description of the imperfect knowledge of the true value of a particular variable or its real variability in an individual or a group. In general, uncertainty is reducible by additional information-gathering or analysis activities (that is, better data or better models), whereas real variability will not change (although it may be more accurately known) as a result of better or more extensive measurements (Hattis and Burmaster 1994).

6.3.1 Types of Uncertainty

Finkel (1990) classified all uncertainty into four types: (1) variable uncertainty, (2) model uncertainty, (3) decision-rule uncertainty, and (4) variability. Variable uncertainty and model uncertainty are generally recognized by risk assessors as major sources of uncertainty; decision rule is of greatest concern to the risk manager.

6.3.1.1 Variable Uncertainty

Variable uncertainty occurs when variables appearing in equations cannot be measured precisely or accurately, because of either (1) equipment limitations, or (2) spatial or temporal variances between the quantities being measured. Random, or sample, errors are common sources of variable uncertainty that are especially critical for small sample sizes. It is more difficult to recognize nonrandom, or systematic, errors that result from the basis for sampling, experimental design, or choice of assumptions. As stated in Section 6.3, true variability is something we can not do much about (except to know that it exists).

6.3.1.2 Model Uncertainty

Model uncertainty is associated with all models used in all phases of a risk assessment. For example, the use of a single species to represent several will introduce uncertainty into the risk assessment because of the considerable amount of interspecies variability in sensitivity to a COPC. Computer models are simplifications of reality, requiring exclusion of some variables that influence predictions but cannot be included in models because of (1) increased complexity, or (2) a lack of data for these variables. The risk assessor needs to consider the importance, in consultation with the modeler, of excluded variables on a case-by-case basis. In addition, a model which was developed to use "average" conditions as its inputs, could result in a large amount of uncertainty when "specific" conditions are used. Finally, choosing the correct model form is often difficult, because conflicting theories appear to explain a phenomenon equally well.

The models specified for use in this document were selected on the basis of scientific policy. Therefore, the air dispersion and deposition model (ISCST3) and the indirect exposure models (IEM) were selected, because they provide the information needed to conduct indirect assessments and are considered by U.S. EPA to be state-of-the-science models. This choice of models could also be considered under decision rule uncertainty. ISCST3—the air dispersion model recommended for use—has not been widely applied in its present form. Few data are available on atmospheric deposition rates for chemicals other than criteria pollutants, thereby making it difficult to (1) select input variables related to deposition, and (2) validate modeled deposition rates. Because dry deposition of vapor phase materials is evaluated external to the air dispersion model, the plume is not depleted and, as a result, mass balance is not maintained. The effect of this would be to overestimate deposition, but the magnitude of the overestimation is unknown. Mass balance is maintained for other forms of deposition (such as wet deposition and particle phase dry deposition). Long-range transport of pollutants into and out of the areas considered was not modeled, resulting in an underestimation of risk attributable to each facility.

In addition to air dispersion modeling, the use of other fate and transport models recommended by this guidance can also result in some uncertainty. For example, the models which estimate COPC concentrations in waterbodies may be particularly conservative for waterbodies located in estuarine environments with tidal influence. Because tidal influence is not considered in the models presented in Chapter 3, the resultant dilution of COPC concentrations in water and sediments likely caused by tidal

influence will not be considered in the risk assessment. Thus, the risk assessment results will likely be more conservative for tidally influenced waterbodies than for those waterbodies that are not tidally influenced. Permitting decisions based on risk estimates for estuarine environments should consider this uncertainty. The delineation of this uncertainty may be one area that could be addressed in a more refined site-specific risk assessment, if warranted.

6.3.1.3 Decision-rule Uncertainty

Decision-rule uncertainty is probably of greatest concern to risk managers. This type of uncertainty arises, for example, out of the need to balance different social concerns when determining an acceptable level of risk. The uncertainty associated with risk analysis influences many policy and risk management decisions. Possibly the most important aspect for the risk estimates is the selection of constituents to be included in the analysis. Constituents identified by this guidance will include compounds that have the potential to pose the greatest risk to ecological receptors through exposure. For example, many PICs are highly lipophilic and tend to bioaccumulate, thereby presenting a potentially high risk to upper trophic level receptors through the consumption of contaminated food items.

6.3.2 Description of Qualitative Uncertainty

Often, sources of uncertainty in a risk assessment can be determined but cannot be quantified. For example, this can occur when a factor is known or expected to be variable, but no data are available (e.g., presence of COPCs without toxicity data). In this case, default data may be available that can be useful in estimating a possible range of values. Uncertainty also often arises out of a complete lack of data. A process may be so poorly understood that the uncertainty cannot be quantified with any confidence. In addition, some sources of uncertainty (such as uncertainty in theories used to deduce models) are inherent qualifications reflecting subjective modes of confidence rather than probabilistic arguments. When uncertainty can be presented only qualitatively, the possible direction and orders of magnitude of the potential error should be considered.

6.3.3 Description of Quantitative Uncertainty

Knowledge of experimental or measurement errors can also be used to introduce a degree of quantitative information into a qualitative presentation of uncertainty. For example, standard laboratory procedures or field sampling methods may have a known error level that can be used to quantify uncertainty. In many cases, uncertainty associated with particular variable values or estimated risks can be expressed quantitatively and further evaluated with variations of sensitivity analyses. Finkel (1990) identified a six-step process for producing a quantitative uncertainty estimate:

- Define the measure of risk (i.e., assessment endpoint). More than one measure of risk may result from a particular risk assessment: however, the uncertainty should be quantified or reached individually.
- Specify "risk equations" that present mathematical relationships that express the risk measure in terms of its components. This step is used to identify the important variables in the risk estimation process.
- Generate an uncertainty distribution for each variable or equation component. These uncertainty distributions may be generated by using analogy, statistical inference techniques, expert opinion, or a combination of these.
- Combine the individual distributions into a composite uncertainty distribution.
- Recalibrate the uncertainty distributions. Inferential analysis could be used to "tighten" or "broaden" particular distributions to account for dependencies among the variables and to truncate the distributions to exclude extreme values.
- Summarize the output clearly, highlighting the important risk management implications. Address specific critical factors.
    - Implication of supporting a point estimate produced without considering uncertainty
    - Balance of the costs of under- or over-estimating risks
    - Unresolved scientific controversies, and their implications for research

When a detailed quantitative treatment of uncertainty is required, statistical methods are employed. Two approaches to a statistical treatment of uncertainty with regard to variable values are described here and were used in this analysis where appropriate. The first is to use an appropriate statistic to express all variables for which uncertainty is a major concern. For example, if a value used is from a sample (such as

yearly emissions from a stack), the mean and standard deviation should both be presented. If the sample size is very small, it may be appropriate to (1) give the range of sample values and use a midpoint as a best estimate in the model, or (2) use the smallest and largest measured value to obtain two estimates that bound the expected true value. Selection of the appropriate statistic depends on the amount of data available and the degree of detail required. Uncertainties can be propagated by using analytical or numerical methods.

A second approach is to use the probability distributions of major variables to propagate variable value uncertainties through the equations used in a risk analysis. A probability distribution of expected values is then developed for each variable value. These probability distributions are typically expressed as either probability density functions (*PDF*) or cumulative probability density functions (*CPF*). The *PDF* presents the relative probability for discrete variable values, whereas the *CPF* presents the cumulative probability that a value is less than or equal to a specific value.

A composite uncertainty distribution is created by combining the individual distributions with the equations used to calculate the probability of particular adverse effects and points. Numerical or statistical methods are often used. In Monte Carlo simulations, for example, a computer program is used to repeatedly solve the model equations, under different selections of variable values, to calculate a distribution of exposure (or risk) values. Each time the equations are calculated, values are randomly sampled from the specified distributions for each variable. The end result is a distribution of exposure (or risk). These can again be expressed as *PDF*s or, more appropriately, as *CPF*s. The distribution enables the risk assessor to choose the value corresponding to the appropriate percentile in the overall distribution. For example, the risk assessor can select an exposure level or risk level that corresponds to the 95th percentile of the overall risk distribution rather than a point estimate of risk that is based on the 95th percentile values for each variable.

6.3.4 Risk Assessment Uncertainty Discussion

The science of risk assessment is evolving; where the science base is incomplete and uncertainties exist, science policy assumptions must me made. It is important for risk assessments of facilities that burn hazardous waste to fully explain the areas of uncertainty in the assessments and to identify the key assumptions used in conducting the assessments. Toward that end, a table should be added to the end of each section (e.g., stack emissions, air modeling, exposure assessment, risk characterization) which lists the

key assumptions in that section, the rationale for those assumptions, their effect on estimates of risk (overestimation, underestimation, neutral), and the magnitude of the effect (high, medium, low). For example, it could explain that using a particular input variable, such as exit gas temperature, will under- or overestimate long-term emissions, and the resulting risks, by a factor of *x*. These tables can be used to evaluate the extent to which protective assumptions were used in the risk assessments. They can also help determine the nature of the uncertainty analysis to be performed. The assumptions listed in the risk characterization section, which synthesizes the data outputs from the exposure and toxicity analyses, should be the most significant assumptions from each of the previous sections.

Within this guidance, identification of uncertainties and limitations are also included with the discussion of specific technical issues (e.g., TOE, estimates of emission rates, COPC selection process, quantification of non-detects) as they are presented in their respective sections. Limitations associated with parameter values and inputs to equations are presented in the Appendices.

As an example discussion, the following summarizes some of the uncertainty involved in the air dispersion modeling component of the risk assessment process.

Although dispersion modeling is a valuable tool for estimating concentration and deposition impacts, it has many limitations. The accuracy of the models is limited by (1) the ability of the model algorithms to depict atmospheric transport and dispersion of contaminants, and (2) the accuracy and validity of the input data. For example, most refined models require input of representative meteorological data from a single measuring station. In reality, a release will encounter highly variable meteorological conditions that are constantly changing as it moves downwind. U.S. EPA's *Guideline on Air Quality Models—Revised* (Title 51 CFR Appendix W) describes two types of model uncertainty. Inherent uncertainty involves deviations in concentrations that occur even if all of the model input is accurate. Reducible uncertainty is associated with the model and the uncertain input values that will affect the results. Although it is important to accurately represent actual conditions by selecting the right model, and using accurate and representative input data, all model results are subject to uncertainty. Nevertheless, models are generally considered reasonably reliable in estimating the magnitude of highest concentrations resulting from a release, although they may not necessarily be time-and space-specific (Title 51 CFR Appendix W). When applied properly, air dispersion models are typically accurate to ± 10 to 40 percent and can be used to yield a "best estimate" of air concentrations (Title 51 CFR Appendix W).

Uncertainties specific to other technical components (e.g., TOE, quantification of non-detects) of the risk assessment process are further described in their respective chapters or sections of this guidance.

6.3.5 Limitations and Uncertainties Specific to a Screening Level Ecological Risk Assessment

As a screening-level tool, the screening level ecological risk assessment has several inherent limitations. Some of these limitations are discussed in Section 6.3.5.1. After computing the *ESQ*s and analyzing the risk assessment results, the risk assessor should evaluate the uncertainty associated with the screening level risk assessment. Section 6.3.5.2 provides a list of uncertainties that U.S. EPA OSW recommends should typically be evaluated, at least qualitatively, in a screening level risk assessment.

6.3.5.1 Limitations Typical of a Screening Level Ecological Risk Assessment

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. Although this approach is reasonable given the nature of the analysis and the availability of the data, protection of measurement receptors may not always adequately protect all ecologically significant assessment endpoints. Similarly, the selection process for ecological receptors relies on a modified trophic element approach. 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 natural history information.

COPCs were selected to provide a conservative representation of those compounds in hazardous waste combustion stack and fugitive emissions that have the highest potential to result in adverse ecological effects. Due to a lack of data on adverse ecological effects associated with combustion emissions through all exposure pathways, this list may not be all inclusive.

The toxicity of compounds varies with the measurement receptors and with the availability and form of a given compound. If a compound is more bioavailable to an organism for absorption or uptake (such as through increased solubility in the surface soil, surface water, or sediment), then the toxic potential of the compound 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

compounds. In the risk assessment, bioavailability of COPCs 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 compound bioavailability differs from those in studies reported in the literature.

Attempts to quantify and correct for uncertainty resulting from the use of surrogate species is 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. 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 a LOAEL to a NOAEL). The net effect of uncertainty factors on the accuracy of the risk assessment depends on the accuracy of the assumptions that underlie the factors themselves.

6.3.5.2 Uncertainties Typical of a Screening Level Ecological Risk Assessment

A screening level risk assessment is typically performed using at least some default parameter values in place of site-specific measured data (see Sections 3.12 and 6.2.2), and incorporating assumptions (see Section 6.2) as a result of data gaps. The absence of site-specific information and the need to use these assumptions may result in uncertainty associated with the calculation of *ESQ*s. An understanding of the uncertainties associated with the *ESQ*s is necessary for understanding the significance of the *ESQ*s. After identifying the major uncertainties associated with the risk assessment results, their significance should be evaluated with respect to the computed *ESQ*s. Uncertainties that generally should be evaluated in a screening level ecological risk assessment for a combustion facility are listed below:

- Changes in future COPC emissions compared with modeled emission rates used in the risk assessment.
- Quantification of emissions and evaluation of non-detects used in the risk assessment.
- The site-specific representativeness of food web(s) used in the risk assessment.

- The exposure potential of the measurement receptors.
- The representativeness of equal and exclusive diet assumptions for measurement receptors.
- The effect of COPC physicochemical properties on estimates of fate and bioavailability.
- The effect of site-specific environmental conditions affecting the fate, transport, and bioavailability of the COPCs.
- The assumption that once exposed, a measurement receptor does not metabolize or eliminate a COPC.
- The potential risk to measurement receptors of COPCs with no *TRV*s.