11. UNCERTAINTY ANALYSIS (ERAGS STEP 7)

There are several sources of uncertainties associated with ecological risk estimates, and uncertainties are present at the various Ecological Risk Assessment Guidance for Superfund (ERAGS) steps, as discussed in this chapter. Sources of uncertainty in this BERA include:

- Sampling representativeness and analysis and quantitation error.
- Onondaga Lake conditions.
- Chemical of concern (COC)/stressor of concern (SOC) selection.
- Background and reference concentrations.
- Sediment effect concentrations (SECs) and macroinvertebrate analyses.
- The conceptual model.
- Natural variation and parameter error.
- Food-web model error.
- Toxicological studies used as measures of effect.

The following sections identify the strengths and limitations of the various components of this assessment.

11.1 Sampling Representativeness and Analysis and Quantitation Error

This section discusses the potential impact of sampling representativeness (locations and frequency) and analysis and quantitation error on the uncertainty inherent in the BERA.

11.1.1 Representativeness of Sampling Locations

The representativeness of the sampling locations selected for chemical analyses of surface water, surficial sediment, sediment porewater, surface soils, benthic macroinvertebrates, fish, and other media are discussed in this section.

11.1.1.1 Surface Water Sampling

The 1992 Remedial Investigation (RI) sampling program conducted by Honeywell/PTI analyzed lake water from the centers of the southern and northern basins and water from near the mouths of each tributary from April to November. At each of the two lake stations, samples were collected at 1-meter (m) intervals from the water surface to the lake bottom. Based on findings from the Onondaga County Department of Water Environment Protection (OCDWEP) monitoring program, it is known that the centers of the two basins are generally representative of water quality conditions throughout most of the deep portions of the lake due to lake circulation patterns and wind-induced mixing processes. The 1-m vertical sampling interval is commonly used in limnological studies (including the OCDWEP monitoring program for Onondaga Lake) and is considered sufficiently small to capture all major limnological features of the water column in Onondaga Lake. The sampling locations near the mouth of each tributary are considered representative because they were generally located upstream of the influence of the lake, and there were no major sources of inflow between each sampling station and the lake.

A limited amount of supplemental water column sampling was also conducted by Honeywell in 1999 to evaluate conditions during fall turnover at stations in the centers of both basins of the lake and to evaluate water quality from a human health perspective at nine nearshore stations. The 1999 data were not intended to provide a representative overview of the lake. The combined surface water database provides sufficient coverage of the lake, and little uncertainty is associated with sampling locations.

11.1.1.2 Surface Sediment Sampling

For the RI sampling, Honeywell/PTI evaluated surface sediment in 1992 at 114 stations distributed throughout the lake, with the densest coverage in areas near major known sources (i.e., between Tributary 5A and Ley Creek and at the mouths of other tributaries). Nineteen samples were fully characterized and the remaining 95 were partially characterized (see Chapter 7, Table 7-1 for details).

In 2000, Honeywell/PTI sampled an additional 84 stations in areas where the 1992 data indicated that additional information was needed (see Chapter 7, Table 7-2). Given the comprehensive and stratified nature of the station-location scheme used for the RI, most potential areas of concern are likely to have been sampled. The selection of stations to evaluate areas of concern may bias any Onondaga Lake-wide mean values for individual COCs or SOCs.

The surface sediments analyzed from the 1992 sampling were collected at a depth of 0 to 2 cm, while the 2000 surface sediment samples were collected at a depth of 0 to 15 cm. The variation in sampling location and depth adds uncertainty to comparisons between sampling years. Sediment samples from 0 to 2 cm do not accurately characterize the entire horizon of sediment that ecological receptors are exposed to (i.e., the biologically active zone). The overall affect of sampling from only 0 to 2 cm rather than the 0 to 15 cm is unknown. However, because the concentration of contaminants in Onondaga Lake sediments tends to increase with depth, it is possible that results from sampling 0 to 2 cm could underestimate the overall exposure of ecological receptors to contaminants.

11.1.1.3 Sediment Porewater Sampling

Sediment cores for porewater were collected at four locations in 1992 and seven locations in 2000. The chloride profiles from the 1992 porewater samples obtained in August and November are strikingly different throughout the length of the cores. The August concentrations are consistently and substantially lower, which could indicate a rapid mechanism for movement through the sediments; however, it most likely indicates that the lake water and porewater were allowed to mix during collection of the August samples, which would invalidate the samples. Because of the significant change in chloride concentrations, the August and possibly all of the porewater results from 1992 are suspect and unusable and, therefore, were not used in this analysis.

Porewater samples were also collected for analysis of mercury and methylmercury from three depths at seven locations in the lake in 2000 (see Chapter 8, Section 8.1.2.10). These data were considered acceptable for use in the RI and BERA.

11.1.1.4 Wetland Soils/Sediment and Surface Soil Sampling

Four wetland soil/sediment samples (i.e., 0 to 15 cm) were collected from each of four wetland areas around Onondaga Lake (i.e., New York State-regulated Wetlands SYW-6, SYW-10, SYW-12, and SYW-19) in 2000. Although these samples cover only a small portion of the wetland areas, they are considered adequate for a general characterization of wetlands in the context of the overall BERA. Because of the high mercury levels in 2000 at one station in Wetland SYW-6 (Station S375), NYSDEC sampled five additional locations in this wetland area in May 2002 to determine if the mercury was indicative of widespread contamination from the lake. These supplemental wetland SYW-6 data are also used in this BERA. Wetland SYW-10 will be further evaluated as part of the RI/FS for the Geddes Brook/Ninemile Creek site and Wetland SYW-19 will be further evaluated as part of the RI/FS for the Wastebed B/Harbor Brook site.

Eight surface soil samples were collected in the dredge spoils area in 2000 to characterize the quality of the fill material that exists in varying thicknesses over the more contaminated dredge spoils material. The dredge spoils samples did not have a depth distribution corresponding to depth intervals preferred for use in ecological assessments (i.e., 0 to 15 cm). As a result, the depth intervals used for assessing ecological exposure to surface soils included data from depths ranging from the surface to 15 to 107 cm. Most of the surface samples were collected from the relatively uniform cover material above the more-contaminated spoils; thus, no significant bias is expected from using the thicker surface soil profile and the surface soil data are deemed suitable for assessing ecological exposure. The dredge spoils area will be further evaluated as part of a separate site with its own investigation.

11.1.1.5 Benthic Macroinvertebrate Sampling

Benthic macroinvertebrates were analyzed for total and methylmercury in 1992 and 2000. The combined data set and its coverage of the lake are considered adequate for a general estimate of mercury concentrations in Onondaga Lake macroinvertebrates. Six macroinvertebrate samples were analyzed for polychlorinated biphenyls (PCBs) on an Aroclor basis in 1992. Macroinvertebrates were sampled from areas of the lake with the highest concentrations of contamination and are not representative of overall lake conditions, particularly if uptake rates vary depending on PCB concentration. Due to the small sample size, unrepresentative sampling, and analytical uncertainty, data from the US Army Corps of Engineers (USACE) database (2002) were used to derive a biota-sediment accumulation factor (BSAF) for PCBs. No other contaminants were analyzed in macroinvertebrates from the lake. Consequently, BSAFs were required to estimate most COC concentrations (see Section 11.8.1).

Sediment toxicity to macroinvertebrates was evaluated in 1992 and 2000. In 1992, sediment toxicity was analyzed at 79 stations in Onondaga Lake and five stations in Otisco Lake (which was used as the reference location), while in 2000 sediment toxicity was tested at 15 stations in Onondaga Lake and two stations in Otisco Lake. Comparison of the results from the 1992 macroinvertebrate toxicity testing locations to the 1992 sediment sampling locations indicates that most potential areas of concern were sampled. Therefore, these tests are considered to be representative of lake bottom conditions.

The purpose of the 2000 sediment toxicity testing was to evaluate potential differences in toxicity through the use of chronic test protocols versus the 1992 short-term protocols. Based on the results of the 2000 toxicity tests and associated sediment chemistry as compared to the 1992 test results, these analyses met their intended goals.

11.1.1.6 Fish Sampling

Hundreds of fish were analyzed by Honeywell and NYSDEC between 1992 and 2000 (Appendix I, Tables I-6, I-13, and I-15). The coverage of combined fish collected from the lake is considered good, but fish data sets may not be adequate to evaluate species-specific uptake or movement of COCs through the food web.

However, few fish samples were analyzed for the full suite of volatile organic compounds (VOCs), semivolatile organic compounds (SVOCs) (including polycyclic aromatic hydrocarbons [PAHs]), and Target Analyte List (TAL) metals. Estimates of some of these contaminants may not accurately reflect concentrations of contaminants found in the lake. There is no systematic bias in the direction of these estimates. The larger fish data sets for mercury, pesticides, and PCBs are considered to accurately represent fish-flesh concentrations under current lake conditions.

11.1.1.7 Sampling of Other Media

A limited number of samples of other media (phytoplankton, zooplankton, and muskrat [*Ondatra zibethicus*]) were sampled (Chapter 7, Tables 7-1 and 7-2). In general, data from other media were inadequate to be considered representative of the lake. The mercury analyses of phytoplankton were used to estimate macrophyte intake in the mallard duck (*Anas platyrhynchos*) food model. There was some uncertainty associated with these data because concentrations were provided on a per-sample basis, rather than as parts per billion (ppb) (or another unit). Despite the uncertainty associated with these data they were used due to the absence of other plant data and a reliable uptake equation.

Representativeness of sampling of other media was not evaluated because of other issues that precluded data use. Wildlife receptors selected in this BERA do not feed directly on zooplankton (with the possible exception of the mallard duck, for which phytoplankton data were used) and, therefore, those data were not used in food-web modeling. Zooplankton data were also not needed to model fish concentrations, as fish body burdens were measured directly. As discussed in Chapter 8, Section 8.2.6.5, the muskrat data were not used because of questions related to the reliability of the transfer factors and the inappropriate use of a herbivorous mammal to represent small mammals, inclusive of insectivores, based on differences in bioaccumulation related to feeding strategies.

11.1.2 Representativeness of Sampling Frequency

For the 1992 RI sampling, both lake and tributary water were analyzed monthly between April and December 1992, when the lake was free from ice. In addition, the monthly sampling events for the tributaries included separate sampling events for different flow conditions. The monthly sampling frequency is considered sufficient for characterizing chronic exposure conditions in the lake and tributaries. The sampling of variable flow regimes in the tributaries allowed a broader range of water quality conditions to be assessed.

Other media sampling occurred primarily during the summer (see Chapter 7, Table 7-1), which is the period of greatest biological activity and considered an appropriate sampling time for BERA data. The 1999 sampling took place from September to December and the 2000 sampling occurred from July to September (Chapter 7, Table 7-2). Although this sampling covered a narrower time frame, the sampling frequency is considered to be acceptable as the data were collected primarily to fill in data gaps.

11.1.3 Analysis and Quantitation Uncertainties

The analysis and quantitation of analytical parameters was minimized by following quality assurance/quality control (QA/QC) protocols and using USEPA Contract Laboratory Program (CLP) laboratories. Data were validated prior to being entered in the Onondaga Lake database.

The exceptions to these protocols were field variables (i.e., pH, specific conductivity, temperature, and dissolved oxygen [DO]) that were determined in the field, and geotechnical analyses, such as grain size and density and solids, which were performed by a geotechnical laboratory.

All data provided by Honeywell were used with the exception of the 1992 PCB biota data. As part of NYSDEC's rewrite of the Honeywell-generated risk assessments, the quality of the historical data, including the 1992 data set, was revisited. As a result of this reevaluation of the fish data, NYSDEC decided to exclude the 1992 Honeywell/PTI PCB data set because it may have underestimated the actual total PCB concentration in Onondaga Lake fish fillets. This determination was based on a review of the Onondaga Lake RI/FS Bioaccumulation Investigation Data Report (PTI, 1993b) prepared by PTI for Honeywell. The report describes the sampling and analysis methods for the 1992 fish collection and includes the analytical data and the quality assurance reviews.

The report indicates that PCBs were not detected in 973 of 1,232 (79 percent) of the samples. Of those samples in which PCBs were not detected, 793 (82 percent) were qualified for various factors of non-compliance with data quality objectives, including the possibility of false negatives. Surrogate recovery was poor, averaging around 43 percent. NYSDEC's data quality review indicated that the Honeywell 1992 data set consistently underestimated fish PCB concentrations, based on the laboratory's failure to meet data quality objectives (specifically, poor surrogate spike recovery) and the high percentage of samples that were reported as not detected for PCBs.

Despite the fact that a large amount of earlier (i.e., 1992) Honeywell PCB data were considered unusable, there are almost 200 NYSDEC (1992 to 2000) and Honeywell (2000) PCB analyses used in this BERA, covering both low molecular weight (either Aroclor 1016 or Aroclor 1242) and high molecular weight (reported as Aroclor 1254/1260) PCBs for fish tissue, which is considered sufficient for the BERA.

11.1.3.1 Mercury Methylation in Wetlands

Owing to the absence of site-specific data, mercury methylation in wetlands was estimated to be 1 percent, based on data from the LCP Bridge Street site (NYSDEC/TAMS, 1998a) and the literature (see Chapter 6, Section 6.3.1). This is considered to be a realistic estimate and resulted in risks up to an order-of-magnitude greater than one to the short-tailed shrew in all wetland areas. Even if mercury methylation rates are lower, for example the 0.25 percent average seen at the LCP Bridge Street site (Chapter 6, Table 6-3), the NOAEL for methylmercury would still be exceeded at all four wetland areas, indicating that methylmercury in wetlands poses a risk to ecological receptors. If methylmercury comprises more than 1 percent of the mercury found in wetland soils, hazard quotients (HQs) would be even higher.

11.1.4 Summary of Sampling and Analysis Uncertainties

The 1992 and 1999/2000 RI sampling programs provided sufficient sediment, soil, water, and fish data for overall coverage of the site, which includes the lake and Wetlands SYW-6 and SYW-12. Honeywell data were supplemented with NYSDEC fish data, NYSDEC Wetland SYW-6 data, and Onondaga County stressor data. All data were considered acceptable for use, with the exception of Honeywell's 1992 PCB fish data. The level of uncertainty associated with surface water, sediment, and fish mercury sampling frequency and locations is considered low. The data sets for other media were not as extensive, providing a moderate level of uncertainty, but were considered acceptable for use in the BERA.

11.2 Onondaga Lake Conditions – Uncertainties

There are a number of uncertainties associated with conditions other than COCs (e.g., stressor concentrations, effects of natural disturbances, etc.) in Onondaga Lake. For example, stressors may impact exposure or exclusion of biota from exposure (e.g., due to low levels of DO), or result in increased eutrophication, affecting the composition of ecological communities. Effects on organisms that provide food or habitat at the bottom of the food chain can affect upper trophic level receptors. In addition, stress on receptors may affect the toxicity of COCs. This section discusses major uncertainties in the Onondaga Lake ecosystem, such as macrophyte distribution and eutrophication-related stressors.

11.2.1 Factors Limiting the Distribution and Abundance of Macrophytes

Factors limiting the distribution and abundance of aquatic plants include the following:

- Light (availability, transparency, and depth).
- Water chemistry.

- Sediment chemistry.
- Sediment texture and composition.
- Temperature.
- Competition.
- Abiotic disturbance (e.g., wave action, water level).
- Biotic disturbance (e.g., herbivory).
- Stressors such as salinity (Madsen and Owens, 2000).

All these factors influence the species of macrophytes found in Onondaga Lake and their coverage. As discussed below, wave disturbance is an additional, significant factor for aquatic plant life.

Wave Disturbance

Sediment disturbance by waves can be a major limiting factor for macrophytes. However, the presence of macrophytes can reduce the action of waves. Therefore, the influence of wave disturbance can be greater in a damaged aquatic system than in a healthy unstressed system. Madsen et al. (1993) indicated that the combination of low transparency and moderate fluctuations of water level limits plant colonization in Onondaga Lake. Even though the level of water in Onondaga Lake is regulated by the dam (on the Oswego River at Phoenix, New York, downstream from the lake), variations in lake level can affect macrophyte communities. However, the rarity of floating-leaved aquatic plants forming defined wetlands along the shoreline of Onondaga Lake indicates that other factors (e.g., salinity, substrate, nutrient loadings, reduced water transparency) besides wave disturbance are also affecting plant growth and establishment.

Macrophytes and oncolites may exhibit a negative relationship with respect to their field distributions in Onondaga Lake. This negative relationship may be due to the direct or indirect effects of the oncolites on the macrophytes or an independent variable, such as wave-induced sediment disturbance. Wave disturbance can cause direct harm to the plants or it can have an adverse effect in combination with the coarse, nutrient-poor sediments found in Onondaga Lake. Laboratory studies showed macrophyte growth to be lower on all Onondaga Lake sediments than on reference sediments (Madsen et al., 1993).

The dominant macrophyte occurring at 391 of the 3,497 Onondaga Lake quadrats surveyed in 1991 was *Potamogeton pectinatus* (sago pondweed) (Madsen et al., 1993). *P. pectinatus* is a pioneering species and quickly inhabits newly flooded areas and invades shallow waters with relatively strong wave action or those that are polluted (Kantrud, 1990). It often shows mass development in areas where the environment became temporarily unsuitable for other species and belongs to a group of plants tolerant of, and able to maintain dominance in, altered ecosystems. These plants are also able to withstand rapid and considerable fluctuations in the salt content of the waters they inhabit and are able to tolerate a wide range of nutrient (nitrogen, phosphorus) concentrations (Kantrud, 1990). The small size of *P. pectinatus* beds indicates that this species and other macrophytes have recently reestablished themselves in the lake or extended their range due to water quality improvements in Onondaga Lake since the late 1980s (e.g., reduction of the high levels of salinity).

The recent reestablishment of macrophytes in the lake indicates that, although wave action may be a factor affecting the species and abundance of aquatic macrophytes found in Onondaga Lake, other stressors (e.g., reduced water transparency, salinity, oncolites, and calcium carbonate deposition) and COCs discussed in Chapter 8, Section 8.1 may be more significant in influencing the Onondaga Lake macrophyte community.

11.2.2 Effects of Calcium and Oncolites on the Macroinvertebrate Community

The effects of calcium and oncolites on benthic macroinvertebrate communities in Onondaga Lake are evaluated in this section. As noted previously, there is a strong association between the calcium carbonate content of sediments and the density of oncolites in the nearshore zone of Onondaga Lake. To evaluate whether the calcium carbonate content of sediments or oncolites are limiting to benthic macroinvertebrate communities in Onondaga Lake, correlation analyses were conducted using the Spearman rank correlation coefficient.

The species richness of benthic macroinvertebrate communities at depths up to 4.5 m exhibited positive significant correlations with the calcium carbonate content of sediment ($P \leq 0.05$) (Figure11-1). Chironomid abundance showed no significant correlation ($P>0.05$) with the calcium carbonate content of sediments, while amphipod abundance exhibited a significant correlation (Figure 11-2). These results indicate that with the increase of calcium carbonate content of sediment in Onondaga Lake there is a corresponding increase in taxa richness and amphipod abundance. However, there is no evidence linking this relationship directly to levels of calcium carbonate in the sediments, levels of contamination, or other physical properties of the sediments.

With respect to oncolites, the species richness of benthic communities and amphipod abundance both correlated significantly ($P \leq 0.05$) with the oncolite volume of the sediment at the depths evaluated (Figures 11-3 and 11-4). However, the amount of scatter in these plots makes any direct quantitative prediction relatively uncertain. Chironomid abundance was also correlated significantly ($P \leq 0.05$) with oncolite volume at the 1.5 m depth, but was not correlated significantly ($P>0.05$) at the 4.5 m depth (Figure 11-4). These results indicate that with the increase of oncolites in Onondaga Lake sediment there is a corresponding increase in taxa richness and amphipod abundances. It is unknown whether this relationship is directly due to the levels of calcium carbonate in the sediments, the levels of contamination, or other physical properties of the sediments, such as providing increased numbers of microhabitats and refuges from predation.

Based on these uncertainties, it is difficult to define the potential direct effects of oncolites on the benthic community in Onondaga Lake. However, the benthic community should be considered in conjunction with the macrophyte community. If the presence of oncolites reduces macrophyte coverage, the onocolites themselves may provide an alternative microhabitat for benthic invertebrates to use. The oncolite habitat is likely to be of lower quality than the macrophyte habitat because of its lower productivity and complexity.

11.2.3 Oxic Hypolimnion

An evaluation was made of the future conditions that may occur in Onondaga Lake if its eutrophic conditions improve to the point that anoxic conditions in the hypolimnion are eliminated. Such changes would result in a series of chemical and biological responses in both the water column and sediment.

Oxic conditions could reduce the concentrations of some substances, such as ammonia, in the water column. The concentrations of at least some metals, including mercury, iron, and manganese, would decrease in the water column, as was evident during fall turnover. Currently there is a large production of methylmercury in the water column in the anoxic hypolimnion, which would be greatly reduced if the hypolimnion remained oxic year-round and there was an increase in the concentration of DO, both of which would make it possible for biota to inhabit the hypolimnion and profundal sediments throughout the year. However, depending on the oxygen demand of the sediments, oxygenating the water column could also merely shift the location of some oxidation reduction boundaries from the water column to the sediment, leaving contaminants available at the sediment-water interface.

The current absence of oxygen and the presence of acutely toxic substances precludes higher (eukaryotic) life and the direct uptake of contaminants in the hypolimnion, although there is some evidence that pelagic biota are presently exposed to the elevated methylmercury concentrations across the thermocline. However, if the decrease in acutely toxic contaminants and increased oxygen leads to the reestablishment of aquatic life in the hypolimnion and surface sediments, then those biota would have the opportunity to interact more extensively with the profundal sediments. These interactions would include direct contact with contaminated surface sediments, and would involve the potential for more deeply buried sediments with higher levels of contaminants to be brought to the surface by bioturbation. Mean concentrations of some bioaccumulative contaminants, including mercury, are currently found at higher levels in deep sediments than in shallow sediments (see Appendix I, Tables I-3 and I-4). This inventory of contaminants, particularly mercury, could be taken up by these organisms and introduced into the food chain. The absence of life in the hypolimnion currently precludes uptake of these contaminants.

In summary, oxic conditions in the hypolimnion are likely to:

- Substantially reduce the water column methylation in the hypolimnion and the subsequent transport of the methylmercury to the epilimnion via mixing.
- Shift the oxic/anoxic boundary from the water column to the sediment.
- Increase bioavailability of contaminants to the food chain via the interaction of benthic invertebrates with the sediments.

It is, however, unclear what the net effect of a shift to oxic conditions in the lake would be. While the concentrations of certain contaminants might decrease under oxic conditions in the hypolimnion, they would not be eliminated from the sediments. As a result, risks to benthic invertebrates, fish, and wildlife could increase substantially due to increased uptake and bioavailability of contaminants.

11.2.4 Eutrophication

The eutrophic nature of Onondaga Lake is due to elevated concentrations of ammonia, nitrite, phosphorus, and sulfide, resulting in depleted DO and reduced water transparency. However, measures are currently underway to improve wastewater treatment. Upgrades to the Metropolitan Syracuse Sewage Treatment Plant (Metro) are being guided by an Amended Consent Judgment (ACJ) from 1998 and decreases in effluent concentrations have been made in the last several years (e.g., Matthews et al., 2001). Under the ACJ, Onondaga County is to reduce stressors in Metro effluent over two intervals by December 2012.

Although future improvements are expected to lessen eutrophication in the lake, some COCs may become more bioavailable and could have a larger impact on the overall health of the ecosystem, as metals and synthetic organic chemicals in the sediments have much greater environmental persistence than stressors. Thus, the relative importance of COCs as compared to stressors of concern (SOCs) could increase in the future. In addition, unlike bioaccumulative metals and organic chemicals, nutrients may only affect the organisms that are directly exposed to them, and not wildlife at higher trophic levels.

11.3 Selection of Chemicals and Stressors of Concern

COC selection followed the procedures laid out in USEPA ERAGS (1997a) and subsequent guidance (USEPA, 2001). Screening-level exposure estimates and risk calculations were used to select COCs (see Chapter 6 and Appendix D). Stressors such as ionic waste (i.e., calcium, chloride, sodium), nutrients (i.e., nitrite, phosphorous, sulfide), salinity, ammonia, depleted DO, and reduced water transparency were used generally for qualitative evaluations and therefore were not screened in the same manner as COCs.

The COCs for water in Onondaga Lake and its tributaries were based on a screening evaluation applied to the results of chemical analyses conducted on water samples collected in the lake and its tributaries during the RI sampling in 1992 and 1999 and a review of the recent (1997 to 2001) OCDWEP water quality data. Given the amount of data available for review, as well as the conservativeness of the water quality standards, criteria, and guidance used for the screening evaluation (Chapter 4, Tables 4-3 and 4-4), it is probable that the list of COCs for water is complete.

COCs selected for the surface sediments of Onondaga Lake were based on a screening evaluation applied to the 1992 and 2000 lake sediment sampling data (see Appendix D, Tables D-14 to D-17 and D-52 to D-55). Given the relatively large amounts of data from multiple sources, the lakewide coverage of the sediment data sets, and the conservativeness of the sediment screening values used for the screening evaluation (Chapter 4, Tables 4-5 and 4-6), it is highly likely that the list of COCs for lake sediment is complete.

Surface soil/sediment COCs for the wetlands around Onondaga Lake and the dredge spoils area were based on a screening evaluation using the 2000 RI sampling data. Full TAL/Target Compound List (TCL) analyses were performed on samples from all areas using both soil and sediment screening values (Chapter 4, Table 4-7).

Plant screening values were available for only a subset of contaminants (mainly metals; Chapter 4, Table 4-8), so that although it is likely that the list of COCs for wildlife receptors exposed via terrestrial pathways is complete, the list of plant COCs may be incomplete.

The COCs identified for Onondaga Lake fish were based on a screening evaluation using the 1992 to 2000 Honeywell and NYSDEC sampling data. In the absence of fish screening values, wildlife (consumer) screening values were used (Chapter 4, Table 4-9), which are generally more protective. Given the relatively large amount of data from multiple sources, the range of species covering different trophic levels, and the conservative screening values used, it is highly likely that the list of COCs for fish is complete.

Food-web exposures were calculated using measured and modeled contaminant concentrations in surface water, sediment, and prey to select COCs on a receptor-specific basis. Maximum concentrations were used and conservative receptor parameters, such as minimum weight and maximum ingestion rates, were selected to maximize exposure. Therefore, it is highly likely that the list of COCs selected for avian and mammalian receptors is complete. A small subset of the COCs selected may be attributable to background or reference levels of contaminants, as discussed in the following section.

11.4 Background and Reference Concentrations

In keeping with USEPA policy (USEPA, 2002), this BERA retained constituents that exceeded risk-based screening concentrations. This evaluation of background and references concentrations evaluates the uncertainty associated with whether COCs are likely to be site-related. The contribution of background concentrations to risks is discussed separately for each medium in this section, based on available data for water, sediment, soil, and fish.

Estimations of background COC concentrations were based on the most appropriate data available in the Onondaga Lake database. Reference sampling stations associated with Geddes Brook and Ninemile Creek (i.e., Stations GB-2, NM-2, TN-17 [excluding TN-17-1A], and TN-18 [excluding TN-18-1A] [Exponent, 2001e; under revision]) were selected for comparison of contaminant concentrations to water, sediment, soil, and fish measured at Onondaga Lake. These stations are upstream of the Honeywell LCP Bridge Street site. Samples from Otisco Lake were also used as reference samples for sediment comparisons.

11.4.1 Reference Water Concentrations

Onondaga Lake surface water samples were compared to surface water samples collected from Geddes Brook/Ninemile Creek reference stations (i.e., GB-2 and NM-2). Concentrations of COCs in lake and reference water are provided in Table 11-1.

Most of the COCs selected for Onondaga Lake water were detected at substantially higher concentrations in lake water than reference surface water using both 95 percent upper confidence limit (UCL) or maximum concentrations and mean concentrations (Table 11-1). For example:

- Methylmercury concentrations were up to 17 times greater than reference concentrations.

- Total mercury concentrations were up to nine times greater than reference levels.

- Lead concentrations were up to seven times higher in lake water than reference samples.

- Organic COCs, zinc, and dissolved mercury were not detected in reference samples, but were detected in Onondaga Lake water.

- Barium had mean and upper-bound ratios ranging from 0.9 to 1.2, and is considered to be a reference COC in water.

- Manganese mean and upper-bound ratios were 2.3 and 1.8, respectively.

Based on these comparisons, all surface water COCs selected for Onondaga Lake, with the exception of barium and possibly manganese, can be considered site-related.

11.4.2 Reference Sediment Concentrations

Onondaga Lake surface sediment samples were compared to surface sediments collected from Otisco Lake and the upper Geddes Brook/Ninemile Creek reference stations (i.e., GB-2, NM-2, TN-17 [excluding TN-17-1A], and TN-18 [excluding TN-18-1A]). Concentrations of COCs in lake and reference sediments are provided in Table 11-2. Onondaga Lake and Otisco Lake sediment concentrations are based on the combined 1992 (0 to 2 cm) and 2000 (0 to 15 cm) data set, while the Geddes Brook/Ninemile Creek concentrations are based on the 1998 and 2001 (0 to 15 cm) sampling.

Two depths of Onondaga Lake sediments were compared to reference levels: the 1 m depth contour and the 9 m depth contour. Only receptors feeding on adult forms of aquatic invertebrates (i.e., tree swallow [*Tachycineta bicolor*] and little brown bat [*Myotis lucifugus*]) were modeled using the 9 m depth contour. Ratios of Onondaga Lake sediment concentrations to reference concentrations are provided in Table 11-3. Contaminants were considered elevated if either the upper bound or mean concentration was more than twice that of any of the reference locations. The use of a factor of two screening level does not mean that there is no toxic effect associated with those concentrations. The purpose of this factor is to concentrate on those COCs that are the most significant contributors to the ecological risk and are most likely site-related.

Concentrations of arsenic, copper, manganese, and vanadium in the 1 m and 9 m contours of Onondaga Lake sediments were within a factor of two for both Geddes Brook/Ninemile Creek and Otisco Lake sediments (Table 11-3). Zinc was within a factor of two for all comparisons except the Onondaga Lake-Otisco Lake 9 m contour comparison, where it was 2.1.

Levels of antimony, selenium, and DDT and metabolites were within a factor of two of Otisco Lake stations and were not detected in Geddes Brook/Ninemile Creek.

Silver, benzene, dichlorobenzenes, ethylbenzene, toluene, trichlorobenzenes, total xylenes, hexachlorobenzene, phenol, and dieldrin were not detected at either of the reference locations. Ratios of the remaining COCs that were greater than those of background levels by at least an order-of-magnitude are as follows:

- Total PCBs were detected at more than an order-of-magnitude higher in Onondaga Lake than at reference stations.

- Mercury, methylmercury, and dioxin/furans (avian toxicity equivalent [TEQ]) were detected at more than two orders-of-magnitude higher than at reference stations.

- Total PAHs in Onondaga Lake sediments were detected at more than three orders-of-magnitude higher than at reference stations.

Based on these comparisons, sediment COCs considered to be site-related are: barium, cadmium, chromium, lead, mercury, methylmercury, nickel, and all organic COCs (with the exception of DDT and metabolites).

Sediment concentrations were used for some receptors to model benthic invertebrate concentrations in prey. Receptors with a major sediment-based component of their diet are the tree swallow, mallard, and little brown bat. Receptor COCs with HQs above 1.0 that may be attributable to background risks are:

- Tree swallow (Table 10-5) – selenium and zinc.
- Mallard (Table 10-6) – zinc.
- Little brown bat (Table 10-11) – arsenic, copper, and vanadium.

The remaining HQs are considered to be site-related. Receptors deriving the majority of their food from prey other than aquatic invertebrates are discussed in the following sections.

11.4.3 Background and Reference Soil Concentrations

Mean and 95 percent UCL surface soil concentrations in wetlands and dredge spoils were compared to upper Geddes Brook/Ninemile Creek reference stations (i.e., GB-2 and NM-2) (Table 11-4). The most suitable background numbers for the wetlands would be samples from similar type wetlands. In the absence of that data, the reference station and background soil values were used, but are not ideal. Ratios comparing Onondaga Lake wetland and dredge spoils area concentrations to reference station concentrations were calculated (Table 11-5). Contaminants were considered elevated if either the upper bound or mean concentration was more than twice that of reference locations. Naturally occurring inorganic elements detected in Onondaga Lake soil that were not detected at the reference stations were also compared to the average concentrations provided in "Background of 20 Elements in Soils with Special Regard to New York State" (McGovern, nd).

The results of the reference location and background comparisons were considered together to determine whether inorganic COCs were site-related. For the four wetland areas and the dredge spoils area, the results were as follows:

- Wetland SYW-6 (northwest end of the lake) had elevated concentrations of mercury, cadmium, chromium, lead, nickel, selenium, zinc, cyanide, total PCBs, and dioxins/furans, compared to upper Geddes Brook/Ninemile Creek reference stations and background levels. The elevated concentrations of PAHs detected in the NYSDEC/TAMS 2002 Wetland SYW-6 sample were in the 15 to 30 cm interval, which is below the interval used in this BERA (0 to 15 cm).

- Wetland SYW-10 (near the mouth of Ninemile Creek) had elevated concentrations of arsenic, cadmium, lead, mercury, hexachlorobenzene, total PCBs, and dioxins/furans as compared to upper Geddes Brook/Ninemile Creek reference stations and background levels. Wetland SYW-10 will be evaluated further as part of the Geddes Brook/Ninemile Creek site RI/FS.

- Wetland SYW-12 (southeast end of the lake) had elevated concentrations of cadmium, chromium, lead, mercury, chlorobenzene, dichlorobenzenes, hexachlorobenzene, selenium, and total PCBs as compared to upper Geddes Brook/Ninemile Creek reference stations and background levels.

- Wetland SYW-19 (southwest end of the lake) had concentrations of mercury and PCBs over an order-of-magnitude above levels at the reference stations. Concentrations of barium, cadmium, lead, selenium, silver, and all organic contaminants (i.e., benzene, chlorobenzene, dichlorobenzenes, trichlorobenzenes, hexachlorobenzene, phenol, aldrin, dieldrin, total PAHs, total PCBs, and dioxins/furans) were also elevated compared to upper Geddes Brook/Ninemile Creek reference stations and background levels. Wetland SYW-19 will be evaluated further as part of the Wastebed B/Harbor Brook site RI/FS.

- In the surface soils of the dredge spoils area ,concentrations of all COCs, with the exception of mercury, silver, dichlorobenzenes, and hexachlorobenzene were comparable to upper Geddes Brook/Ninemile Creek reference stations and/or average background concentrations found in the literature. The dredge spoils area will be evaluated further as a separate site with its own investigation.

Based on this comparison, soil COCs considered to be site-related at one or more wetland area include: arsenic, barium, cadmium, chromium, lead, mercury, nickel, selenium, silver, zinc, cyanide, and all organic COCs (i.e., benzene, chlorobenzene, dichlorobenzenes, trichlorobenzenes, hexachlorobenzene, phenol, aldrin, dieldrin, total PAHs, and total PCBs). All mercury was considered to be site-related, based on known releases to the lake.

Soil concentrations were used for some receptors for direct comparisons (plants) or to model concentrations in terrestrial invertebrate prey (short-tailed shrew [(*Blarina brevida*)]) or small mammals (red-tailed hawk [*Buteo jamaicensis*] and mink [*Mustela vison*]). Receptor COCs with HQs above 1.0 that may be attributable to background risks are:

- Plants (Table 10-1) – selenium (Wetland SYW-10 and dredge spoils), vanadium (all locations), and zinc (all locations except Wetland SYW-6).
- Red-tailed hawk (Table 10-10) – no COCs attributable to background risks.
- Short-tailed shrew (Table 10-12) – selenium (Wetland SYW-10 and dredge spoils)and vanadium (all locations).
- Mink (Table 10-13) – no COCs attributable to background risks.

11.4.4 Reference Fish Concentrations

Reference sampling stations used to estimate reference fish concentrations were based on selected sampling stations (i.e., GB-2 and NM-2) associated with the Geddes Brook and Ninemile Creek RI (Table 11-6). Six white sucker (*Catostomus commersoni*), one creek chub (*Semotilus atromaculatus*), and one tessellated darter (*Etheostoma nigrum*) were collected at these stations.

The white sucker feeds on a variety of organisms occurring in the mud, including aquatic insect larvae, small mollusks, and crustaceans in stream and lake bottoms. The white sucker serves as prey for other fish, including walleye (*Stizostedion vitreum*), largemouth bass (*Micropterus salmoides*), smallmouth bass (*Micropterus dolomieui*), and other game fish. Because of similar feeding strategies, contaminant concentrations in the white sucker were compared to carp (*Cyprinus carpio*) and catfish contaminant concentrations measured in Onondaga Lake.

The white sucker reference station samples had HQs above 1.0 for arsenic, chromium, selenium, vanadium, and zinc when the maximum concentration was compared to the no observable adverse effect level (NOAEL) (Table 11-6). Selenium and vanadium exceeded the NOAEL at both mean and maximum concentrations. All exceedances were within one order-of-magnitude.

Concentrations of contaminants in carp and catfish from Onondaga Lake fish were up to two orders-of-magnitude greater than concentrations of contaminants measured in reference station fish (Table 11-7). Ratios of site concentrations to reference concentrations were highest for bioaccumulative organic contaminants, such as endrin and DDT and metabolites. Concentrations of mercury, total PCBs, chromium, selenium, and vanadium were also substantially higher in Onondaga Lake fish.

The creek chub and tessellated darter were compared to the bluegill (*Lepomis macrochirus*) to evaluate reference concentrations, as both the creek chub and tessellated darter feed on benthic invertebrates (Table 11-7). Creek chub and tessellated darter reference samples were analyzed for only a limited number of contaminants (mercury, PCBs, dioxins/furans), none of which had HQs greater than one (Table 11-6). Concentrations of mercury in lake fish were over an order-of-magnitude greater than in reference fish and concentrations of dioxins/furans were also higher in Onondaga Lake fish (Table 11-7).

Based on these ratios of COC HQs, all COCs in fish and receptors feeding primarily on fish (belted kingfisher [*Ceryle alcyon*], great blue heron [*Ardea herodias*], osprey [*Pandion haliaetus*], mink [*Mustela vison*], and otter [*Lutra canadensis*]) are considered to be site-related.

11.5 Sediment Effect Concentrations and Macroinvertebrate Uncertainties

11.5.1 Representativeness of Toxicity Tests

The two test species used to assess sediment toxicity during the 1992 and 2000 RI sampling were the amphipod *Hyalella azteca* and the chironomid *Chironomus tentans*. Both species are standard test organisms, but their true sensitivity to sediment contamination levels is uncertain. In addition, the two species represent two different major taxonomic groups (crustaceans and insects, respectively) and occupy different positions in the sediment. *H. azteca* tends to live on the sediment surface, whereas *C. tentans* lives in a case it constructs within the sediment. The joint use of the these two test species ensured that a range of taxa and exposure scenarios was evaluated.

In addition to using two different test species, a variety of exposure conditions and toxicity endpoints were used to assess sediment toxicity in Onondaga Lake. In 1992, the test species were exposed to the top 2 cm of field-collected sediments for ten days under static conditions, with toxicity endpoints of survival and biomass. In 2000, the test species were exposed to the top 15 cm of field-collected sediment for 42 days with renewal of overlying water, with toxicity endpoints of survival and biomass for both species, number of young for the amphipods, and emergence for the chironomids. Comparisons of the 1992 and 2000 results showed that the spatial patterns of toxicity identified by both sets of data were similar, indicating that sediment toxicity in the lake was adequately characterized.

The principal uncertainties surrounding toxicity tests are that while they are a good measure of the potential for adverse environmental effects by providing indications of whether conditions are toxic enough to kill or otherwise impact test species, they do not exactly mimic natural exposure. As a consequence, there is some degree of uncertainty in relating the toxicity test results directly to the potential for actual responses in Onondaga Lake. Laboratory toxicity tests were used to measure the acute or chronic toxicity of site sediment samples to test organisms. They were unable to definitively determine if the sediment was also toxic to lake biota in-situ or to determine which COCs or SOCs were the specific cause of the toxicity. The rationale for conducting toxicity tests on Onondaga Lake sediments was to provide a quantifiable measure of the potential for the occurrence of effects.

Sediment toxicity tests automatically take into account the relative toxicity of a mixture of chemicals, including any synergistic or antagonistic effect between chemicals. However, the ability to determine a direct causative link to one or more contaminant in the sediment may be highly uncertain due to the presence of many co-occurring contaminants. If toxic effects are found and there is no correlation between the effects and the contamination levels, the measured toxicity could be the result of an unanalyzed substance or other substances such as ammonia or sulfides. Because the sediment toxicity in much of Onondaga Lake is a result of the exposure of organisms to a very complex mixture of metals and organic contaminants, it is generally difficult to correlate the toxicity at any given site to any particular contaminant.

11.5.2 Selection of Sediment Effect Concentrations

The sediment screening values used for the general screening evaluation were based on the lowest available values. The site-specific probable effect concentrations (PECs) used for the evaluations of potential risk posed by individual sediment samples were based on an approach developed by Ingersoll et al. (1996 and 2000). Because this approach was applied to the extensive amount of data collected during the RI on sediment chemical concentrations and associated sediment toxicity in Onondaga Lake, it has considerably more site-specific relevance than do generic sediment screening values. As shown in Table 11-8, Onondaga Lake PECs are generally more conservative than those previously published (Ingersoll et al., 2000), with PEC values anywhere from one-half the value to ten times lower than values previously published. However, the Onondaga Lake PEC for mercury (2.2 mg/kg) is twice the Ingersoll value (1.1 mg/kg).

Despite the site-specific applicability of the PECs, uncertainties exist with respect to their ability to identify which COCs may be responsible for causing any observed toxicity. This uncertainty is present with most kinds of sediment quality values found in the literature and is the result of potential confounding factors that are often encountered in environmental samples (e.g., co-occurring chemicals, site-specific factors that modify bioavailability, heterogeneous sediment matrices). In addition, no PECs based on field information can conclusively identify causal relationships. Instead, associations between chemicals and biological effects are used to infer potential causative relationships. This lack of conclusive causality is a source of uncertainty in using PECs for sediment assessment.

11.5.3 Uncertainties Associated with the Use of Benthic Metrics

The benthic data were evaluated using five metrics (i.e., species richness, dominance index, abundance of indicator species, community composition, and species diversity). The cumulative review of the five metrics (referred to as a "multi-metric approach") was used to coalesce the metrics into a single overall assessment of each station. There are uncertainties associated with the use of the benthic metrics in this manner, as follows:

- The level of impact, or "impairment" (i.e., non-impaired, slightly impaired, moderately impaired, or severely impaired), that is derived from the metrics cannot be attributed directly to specific COCs.

- A level of uncertainty exists due to the fact that all five metrics rarely show the same impairment level at a given station. The assessment determination for each station was made on the basis that three or more of the five metrics exhibited the same impairment.

- Benthic invertebrate communities typically occur in patches in the natural environment. To account for this when sampling, replicate samples were collected to improve sampling precision. In this study, five replicates were collected at each station. While this is a good effort (three replicates are considered the absolute minimum for macroinvertebrate characterization), there is an inherent uncertainty about whether enough replicates were taken to obtain meaningful estimates.

11.5.4 Uncertainties Associated with the Simultaneously Extracted Metals/Acid-Volatile Sulfide Ratios

Although it has been shown through numerous laboratory experiments that consideration of simultaneously extracted metals/acid-volatile sulfide (SEM/AVS) ratios can improve predictions of sediment toxicity due to divalent metals, there are uncertainties associated with the approach. The approach has been largely tested for acute toxicity and, therefore, has uncertain applicability to chronic toxicity. In addition, the predictive ability of this approach for sediments in a stratified lake such as Onondaga Lake is uncertain because it has been demonstrated that sediment AVS concentrations can vary temporally and spatially.

In general, AVS concentrations tend to increase during periods of stratification and decrease at times of the year when the water column becomes oxygenated. The AVS may, therefore, limit the bioavailability of divalent metals for only part of the year. Use of AVS data collected from the time of year when AVS levels are expected to be highest could bias the data interpretation for the remaining part of the year. This is due to the seasonal oxidation conditions at the sediment-water interface, so that metals sequestered during one season may be released during another. In addition, the SEM/AVS approach should not be used to characterize mercury toxicity, even though mercury forms sulfide complexes, because the organic form of mercury is the most bioavailable and toxic form and does not complex with sediment sulfides. However, it should also be pointed out that AVS may limit the availability of inorganic mercury to methylate in lake sediments.

11.6 Conceptual Model Uncertainties

The conceptual model links COC sources, likely exposure pathways, and potential ecological receptors. It is intended to provide broad linkages from various receptor groups found in and around Onondaga Lake to contamination in Onondaga Lake water, sediments, soils, and prey. The conceptual model has been refined since its initial presentation in the Onondaga Lake Work Plan (PTI, 1991). Based on changes made to the model as more was learned about the Onondaga Lake ecosystem, there is considered to be a low level of uncertainty associated with the conceptual model. However, since it is a generalized model, it is not intended to represent specific individuals currently living around Onondaga Lake. The actual linkages between the biotic levels depend on the seasonal availability of various prey and food items.

The results of the risk characterization show that the majority of risk is due to exposure to contaminated prey, which is consistent with other studies. Specific uncertainties in the exposure and food-web modeling are discussed in the following section.

11.7 Natural Variation and Parameter Error

Natural variation represents known variation in parameters based on observed heterogeneity in the characteristics of a particular receptor species. Variability can often be reduced with additional data collection, whereas uncertainty can be reduced directly through the confirmation of applied assumptions or inferences through direct measurement. Parameter error includes both uncertainty in estimating specific parameters related to exposure or the specific exposure point concentrations (EPCs) being applied in the exposure models (e.g., sediment, water, and fish concentrations, etc.) as well as variability (e.g., ingestion rate, body weight, temporal and spatial habitat use, etc.). Some parameters can be both uncertain and variable.

11.7.1 Receptor Exposure Parameters

11.7.1.1 Body Mass

Body mass plays a quantitative role in the water, dietary, and incidental sediment ingestion pathways as part of the average daily dosage term for each pathway on a per-kilogram body weight basis. Body masses for adult birds were generally based on mean or median body masses provided in references such as Dunning (1993) and USEPA (1993b), in contrast to the screening-level risk assessment where minimum body weights were used. Representative mammalian body masses were taken from North American populations. Measurements for regional populations, such as New York populations of tree swallows and little brown bats, were used when available.

On a cumulative dosage basis, a higher body mass estimate would reduce the estimated cumulative daily dosage fraction of COCs on a per-kilogram body weight basis. Likewise, a lower body mass estimate would result in a higher average daily dosage estimate. Since it is not known if typical body masses for Onondaga Lake populations are indicative of either extreme in the range of body masses, no systematic bias is associated with these estimates. Therefore, body masses employed in the exposure pathway modeling for avian and mammalian receptors are considered reliable and representative of Onondaga Lake populations.

11.7.1.2 Ingestion Rates

Food Ingestion Rates

Estimates of food ingestion rates (FIRs) for all receptors, with the exception of the short-tailed shrew, were derived using the bioenergetic allometric scaling function of Nagy (1987). This function relates field metabolic rates to body mass across receptors within a given class (birds or mammals). The bioenergetic algorithm of Nagy (1987) did not include data for very small, very active eutherian mammals, such as the shrew. Since the field metabolic rate is strongly correlated with body size, it was considered inappropriate to use Nagy's equation to calculate a metabolic rate for shrews, and literature ingestion rates were used instead to estimate intake. The little brown bat is also a small, active mammal; however, as it spends part of the year in hibernation, the Nagy equation was used to estimate a year-round intake rate.

Use of allometric scaling incorporates some degree of uncertainty in the absence of field verification. To reduce this uncertainty, diet-normalized metabolic rates and the metabolizable energy contents of specific foods consumed were used. Ingestion rates were calculated as the quotient of the species-specific normalized metabolic rate and the average metabolizable energy content of the diet. Estimation of the average gross energy content in wildlife foods is limited to a number of select broad phylogenic groups and is rarely available for species-level evaluations of prey included in the diet. Reliance upon the gross energy estimates for representative taxa groups introduces some uncertainty in derivation of the ingestion rates, as it is assumed that the gross energy content and assimilative efficiency of select groups of invertebrates and fish taxa are equivalent to other freshwater benthic invertebrate and fish taxa. This assumption in the energy content of the diet can influence the ingestion rate estimate, if under- or overestimated. An overestimate of the average metabolizable energy in the diet will decrease the ingestion rate (i.e., the actual metabolic average is lower than estimated), while an underestimate of the metabolic average results in an overestimate of the ingestion rate. There was no systematic bias inherent in the FIRs used in this BERA.

Water Ingestion Rates

Water ingestion rates (WIRs) for avian and mammalian receptors were estimated based upon allometric relationships developed for mammals and birds by Calder and Braun (1983). For this pathway, it was assumed that avian and mammalian receptors use Onondaga Lake as their exclusive drinking water source. The dosage estimate for water ingestion did not account for metabolic- or dietary-derived sources of water for the individual receptors. Consequently, the allometric methods assumed that hydration demands in the receptors are solely accounted for by direct ingestion of surface water. This assumption may result in a slight overestimate of surface water-derived COC exposure through the drinking water pathway by exclusion of metabolic and dietary sources.

Incidental Sediment Ingestion Rates

Of the receptors evaluated, only the mallard and short-tailed shrew have published estimates for ingestion of soil/sediment. The value of 3.3 percent for the mallard (Beyer et al., 1994) and 13 percent for the short-tailed shrew (Talmage and Walton, 1993) are quantified estimates and are considered reliable for application to Onondaga Lake populations.

Estimates of incidental sediment ingestion for other receptors were made based upon feeding behavior used for prey capture and consumption and nesting/resting habitats of each species. Both the tree swallow and little brown bat feed primarily on flying insects that are captured and consumed in flight. The tree swallow nests in trees, while the little brown bat roosts in sheltered locations, such as caves and abandoned buildings. These feeding and roosting preferences result in incomplete pathways for incidental sediment ingestion. Therefore, a zero percent incidental sediment ingestion rate (SIR) was used for both receptors.

The great blue heron, belted kingfisher, and osprey were characterized as primarily piscivorous in diet. All three receptor species visually follow their prey and seize the specific prey item using their bill (great blue heron and belted kingfisher) or talons (osprey). An SIR of 1 percent was used for the great blue heron as it may ingest some incidental sediment during prey capture, prey consumption, and grooming. This rate was also applied to the belted kingfisher, which has little contact with sediments during feeding, but may ingest some sediments during grooming because it nests in riverbanks. An SIR of zero percent was applied to the osprey based on its feeding and nesting habits.

Stomach content and scat analyses on mink from New York State have shown trace quantities (i.e., less than or equal to 1 percent of the diet) of sand present (Hamilton, 1940). Based upon this study and the potential for the mink to also ingest sediments during grooming, a 1 percent incidental ingestion composition in the diet of the mink was applied.

No quantitative dietary information regarding the occurrence of soils/sediments in the diet of the river otter was available, but based on the potential to ingest sediments during feeding and grooming, a 1 percent ingestion rate was also applied to the river otter.

The 1 percent SIR does not consider sediment contained in the digestive system of prey. A study evaluating the stomach contents of bluegills reported that an average of 9.6 percent of the bluegill's diet consisted of detritus and sediment (Kolehmainen, 1974). The majority of the fish data used in this BERA were based on whole-fish samples, which include sediment contained in the stomach. Whole fish to fillet conversion factors were applied to fish that were analyzed as fillet samples, when appropriate. Therefore, the incidental SIR of piscivorous receptors is considered appropriate, as incidental sediment contained in fish prey is included in fish exposure concentrations.

11.7.2 Temporal and Spatial Parameters

11.7.2.1 Uncertainty in Temporal Parameters

A year-round exposure time was used for avian and mammalian receptors in this BERA. Although the avian receptors considered in this BERA are migratory in nature, there have been year-round sightings of them at Onondaga Lake (see Chapter 8, Section 8.2). Even if receptors migrate, they are likely to breed and raise their young at Onondaga Lake during warmer periods of the year. As reproductive effects were generally selected as toxicity endpoints, full-time residency may slightly overestimate exposure; however this is considered to be appropriate.

The mammalian receptors selected in this assessment are year-round residents of Onondaga Lake. However, the little brown bat hibernates during the winter. During hibernation it relies on food reserves obtained during the summer and fall. As food reserves are obtained from Onondaga Lake, it is considered to have year-round exposure.

11.7.2.2 Uncertainty in Spatial Parameters

The conceptual model assumes that receptors modeled belong to closed populations that forage exclusively in and around Onondaga Lake. While this may be accurate for receptors with small home ranges (e.g., belted kingfisher, short-tailed shrew), exposure may be overestimated for receptors with larger home ranges (e.g., osprey, river otter). Prey availability plays a major role in the home range and location of receptors. During years with low prey availability, some receptors may obtain a portion of their food off-site, while during years with high prey abundance all food may come from Onondaga Lake. Therefore, the uncertainty in the spatial use of the site may introduce a conservative bias in some years.

11.8 Model Error

A food-web model was used to approximate relationships between site-specific environmental conditions (i.e., exposure sources) and receptors. Relationships between trophic levels and food-web components are well understood, but available models are generally simplistic. Potential sources of error in the model are discussed below.

11.8.1 Prey Contaminant Exposure Concentrations

The evaluation of the 95 percent UCL exposure assumes that exposure via multiple routes is at the 95 percent UCL for all media. In the calculation of the total exposure, the summation of the exposure route concentrations at the 95 percent UCL of the mean could result in estimates that may be higher than the actual 95 percent UCL exposure concentrations. However, as the majority of risk in food-chain models is derived from dietary intake (e.g., fish or invertebrates as prey), the use of multiple 95 percent UCL exposure concentrations is not considered overly conservative. In addition, mean exposure concentrations are calculated to provide a range of HQs for each receptor.

11.8.1.1 Uncertainty in Chemical of Concern Exposure Concentrations in Fish

Estimates of COC concentrations in fish (as both prey and receptor) were derived from direct measurements. There were a limited amount of data for TAL metals and some organic contaminants, particularly in the 3 to 18 cm range, but data were generally sufficient to perform the statistical tests (i.e., determination of data distribution type and calculation of the 95 percent UCL on the mean). The use of maximum concentrations as the upper-bound estimate for contaminants with few size-class or species-specific samples available may introduce a conservative bias into these estimates, specifically in the 95 percent UCL estimate for receptors that eat small fish (see Appendix I, Tables I-5 to I-14 for upper-bound exposure concentrations used). However, sufficient mercury data were available in both the 3 to 18 cm and 18 to 60 cm fish size ranges to calculate a 95 percent UCL.

Fillet to Whole Fish Conversion Factors

Conversion factors were used to adjust mercury, DDT and metabolites, PCBs, and dioxin/furan concentrations in fillets to whole-body concentrations. The factors used for mercury and PCBs have a high degree of confidence based on the large number of site-specific data and comparison to literature values. The DDT and dioxin/furan values are based on smaller data sets and have a moderate degree of uncertainty associated with them, based upon the generalized assumption that fillet to whole-body relationships were independent of species and age. The remaining conversion factors calculated (see Table 8-4) were considered to have a high degree of uncertainty associated with them, and were, therefore, not used in this assessment. There is no systematic bias in the fillet to whole-fish conversion factors.

11.8.1.2 Uncertainty in Chemical of Concern Exposure Concentrations in Plant and Non-Fish Prey Sources

Concentrations of COCs were not measured in terrestrial plants, terrestrial invertebrates, birds (including eggs), mammals, or benthic macroinvertebrates, with the exception of mercury and PCBs in benthic macroinvertebrates. Therefore, COC concentrations in these prey types were estimated based on assumed media-transfer relationships. The uncertainty in media-transfer ratios and functional relationships between COC concentrations in sediment (or soil) and in tissue is greater than that of using measured prey concentrations.

Biota-sediment accumulation factors for aquatic invertebrates and uptake factors (UFs) or equations for earthworms and small mammals were taken from reports published by Oak Ridge National Laboratory (ORNL) (Sample et al. 1998a,b; US Department of Energy [USDOE], 1998). General, rather than conservative, estimates were applied to reduce the level of conservatism associated with these estimates. The COC-specific factors in these publications are based on linear or transformed functions or point estimates, of transfer coefficients derived from a survey of available literature data. There were situations where no COC-specific transfer relationship could be found and the estimate of the concentrations in tissue relative to a medium had to be based on a surrogate contaminant for which a transfer coefficient was available.

For functional relationships that represent contaminant transfer from the environment to tissue, such as those applied in Sample et al. (1998a,b) and USDOE (1998), the regression relationships are derived from a database pooled from various studies. The underlying assumption is that observations from independent studies represent random and unbiased estimates of the same relationship, and that all variances between such observations are experimental in nature and not the result of differences in experimental design and/or approach. Transfer of contaminants to body tissue is a multi-functional and dynamic process dependent on factors such as duration of exposure, availability from the medium, depuration rates, receptor species, health status, and habitat type. The uncertainty resulting from this approach cannot be quantified based on the data available from either the literature or Onondaga Lake sampling, but is not considered to be biased in either direction. Omitting pathways that lack site-specific data could substantially underestimate risk.

11.9 Toxicological Uncertainties

Uncertainties in toxicological studies may result from the use of laboratory or field studies that may differ from the actual toxicity present at Onondaga Lake due to:

- Site-specific conditions.
- Interspecies differences in sensitivity to contaminants.
- Extrapolating between lowest observed adverse effect level (LOAEL) and the NOAEL, and vice-versa.
- Extrapolating from subchronic to chronic exposures.
- Actual bioavailability of contaminants.

Risk prediction is dependent upon the assumption that daily exposure to COC doses greater than the toxicity reference value (TRV) will result in an adverse effect. Toxicological studies showing reproductive effects were preferred when available, as reproductive effects are considered to be a sensitive endpoint. However, the actual impact of reproductive effects on receptor populations possesses some uncertainty with regard to magnitude of ecological impact relative to predicted risk. Because the level of impact is based upon a physiological rather than an ecological TRV, the uncertainty tends to be conservative. The range of toxicity thresholds reported in the literature is large, even among those studies deemed suitable for extrapolation to the receptor species of interest. The range may be due to test species, life stage, exposure dosage and duration, the form of a contaminant, or other factors.

11.9.1 Laboratory Versus Field Studies

Both laboratory and field studies have advantages and disadvantages with respect to use in the development of TRVs. Laboratory experiments offer the advantage of being able to control exposure conditions, while field studies may more closely represent actual exposure conditions. For example, the concentrations of contaminants in environmental media, especially tissue, may be strongly influenced by differential rates of transport, uptake, metabolism, and elimination. Contaminants that are resistant to metabolism are more persistent and tend to be present at higher concentrations in environmental media than in a commercial mixture (e.g., Aroclor 1254). Therefore, forms of contaminants in environmental media (e.g., fish tissue or bird eggs) may be more toxic than laboratory mixtures, and TRVs based on laboratory dietary doses may underestimate the toxicity of the dietary dose received by a receptor in the field.

Laboratory studies are often designed to test the effect of a single contaminant on a test species in the absence of other co-occurring contaminants, and, thus, observed effects are clearly related to exposure to the test compound. In field studies, organisms are typically exposed to other co-occurring contaminants. The presence of co-occurring contaminants may be a disadvantage to the use of field studies for development of TRVs, since observed effects may not be solely attributable to exposure to a specific contaminant. Laboratory studies were used to derive most TRVs in this BERA, except when a more appropriate field study was available.

11.9.2 Interspecies Sensitivity

Species often vary in their sensitivity to contaminants. An interspecies uncertainty factor estimates differences in sensitivity, but the test species could be either more or less sensitive than the receptor of concern. Certain taxonomic groups of animals, such as salmonid fish, gallinaceous birds, and mink have been shown to be highly sensitive to the reproductive effects of certain contaminants, such as PCBs (e.g., Beyer et al., 1996). To minimize this source of uncertainty in the BERA, studies on sensitive receptors were only selected when alternative studies were not available, excluding receptors classified as sensitive species. Analysis of the available literature provided no reason to assume that the receptors evaluated in this investigation would be more or less sensitive to the COCs than those tested in the respective toxicity studies selected, unless noted in the text. Therefore, any variance in the sensitivity of the receptor relative to the test species used to develop the TRV would most likely be evenly distributed around the estimated TRV, and no interspecies uncertainty factors were applied.

11.9.3 Application of Conversion Factors

Additional areas of uncertainty are encountered when the best available study for the development of a final TRV uses a subchronic, rather than a chronic, exposure. A conversion factor of 0.1 is used to estimate a chronic TRV from a subchronic TRV. A conversion factor differs from an uncertainty factor in that the direction of the uncertainty is known. For example, the chronic TRV is expected to be lower than the subchronic TRV. Use of a subchronic-to-chronic conversion factor of 0.1 is supported by the results of a study that compared subchronic to chronic NOAELs and LOAELs (Dourson and Stara, 1983). For more than half of the chemicals studied, the ratio of subchronic to chronic endpoints was 2.0 or less, and for 96 percent of the chemicals the ratios were below 10. Therefore, application of a conversion factor of 10 was considered protective and may result in a slight conservative bias for the following TRVs where a subchronic-to-chronic conversion factor was applied:

- Chromium – Mammalian NOAEL and LOAEL.
- Dichlorobenzenes – Avian NOAEL and LOAEL.

Uncertainty also exists when conversion factors are used to estimate NOAELs from LOAELs. Data on the ratio of LOAEL to NOAEL indicates that all chemicals examined have a LOAEL to NOAEL ratio of 10 or less and 96 percent have a ratio of five or less (Dourson and Stara, 1983). Therefore, a factor of 10 was used to convert between NOAELs and LOAELs. The direction of uncertainty associated with the use of a LOAEL to NOAEL conversion factor is known, since NOAELs are always expected to be lower than LOAELs. LOAEL to NOAEL conversion factors were used for the following TRVs:

- Arsenic – Mammalian NOAEL.
- Total mercury – Mammalian LOAEL.
- Methylmercury – Avian and mammalian NOAELs.
- Thallium – Mammalian NOAEL.
- Vanadium – Avian LOAEL and mammalian NOAEL.
- Total PCBs – Avian and mink/otter NOAELs.
- Total PAHs – Avian and mammalian NOAELs.
- Zinc – Fish NOAEL.

11.9.4 Uncertainty in Relative Bioavailability

The bioaccumulation and response models (for both plants and animals) assumed that the form of the chemical present in the environment was absorbed with the same efficiency as the chemical form used in the laboratory toxicity study. Chemical solubility is an important factor in absorption efficiency, and for many chemicals, laboratory toxicity studies are performed using the most soluble form. This is particularly true of the metal COCs, which are themselves natural but often biologically unavailable constituents of abiotic media such as soils and sediments.

Concentrations of COCs in the soils surrounding Onondaga Lake were analyzed using USEPA Method 3050b extractions, which rely on digestion using nitric acid and hydrogen peroxide under high temperature to solubilize the metal constituents. Metals may be more available using this method than in their natural state, where they are usually covalently bound within the soil matrix. The assumption that the concentrations measured from matrices that have undergone strong acid digestion represents the fraction available for uptake by plants or absorption by animals is a conservative assumption. Although this method provides a conservative estimate of plant risk, the resulting data are the only available estimate of contaminants in soil and are therefore used to determine whether concentrations of COCs measured in soil may pose a risk to plants.

11.9.5 Uncertainty Due to Lack of Appropriate Toxicity Data

Appropriate toxicity studies were not available for avian receptors for thallium, trichlorobenzenes, and xylenes. It was therefore not possible to calculate risks for these COCs to avian receptors, potentially underestimating risks.

Other toxicity studies measured endpoints, such as mortality, which are generally less sensitive than reproductive endpoints and may underestimate risks to receptors. Survival of eggs to juvenile life stages were grouped together with reproductive effects. Studies used to derive TRVs for this BERA based on adult survival endpoints are:

- Antimony – fish and birds.
- Chromium – fish.

11.10 Summary

Uncertainty is an inherent component of risk assessments. Elements of uncertainty in this BERA have been identified and efforts have been made to minimize them. For components in which a moderate degree of uncertainty is unavoidable (e.g., sampling data), efforts have been made, to the extent possible, to minimize any systematic bias associated with the data.